Centre for Alternative Wastewater Treatment

Tundra Wetlands: the treatment of municipal wastewaters
RBC Blue Water Project: performance & operational tools
(manual only)

Chouinard, Balch, Jørgensen, Yates & Wootton

May 2014

RBC Blue Water Project™

TRENT UNIVERSITY

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Performance Review & Operational Tools

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Forward

Funding provided by the federal government of Canada through the International Polar Year programme and Environment Canada was awarded to the Centre for Alternative Wastewater Treatment (CAWT) to conduct site investigation at 13 tundra treatment wetland locations in Nunavut and the Northwest Territories. Funding from the Royal Bank of Canada (RBC) Blue Water Project was used to consolidate and analyze this information and to develop a guidance document for wastewater stakeholders within indigenous communities in Canada's far north. This guidance document "Tundra Wetlands: the treatment of municipal wastewaters – performance and operational tools, 374 pages", along with a smaller summary "companion" report (e.g., 34 pages) outlines the major treatment processes occurring within wetlands, provides information on the performance of 13 northern treatment wetlands and presents operational tools (e.g., SubWet 2.0) that can be used by wastewater managers and regulators. This information can be used to predictively

assess how wetlands may respond to changing conditions (e.g., wastewater strength, flow rates, etc.) and how wetlands could be used in a hybridized treatment train that includes both waste stabilization ponds and wetlands as part of a wastewater strategy for northern Canada. The RBC Blue Water Project funding was awarded to the Institute for Watershed Science (IWS), Trent University who in turn subcontracted the CAWT to produce the performance and operational tools to serve as a guidance document for wastewater treatment within indigenous communities of Canada's far north. The primary focus of the RBC award to the IWS was to develop teaching materials and tools dedicated to the protection of drinking water within indigenous communities of Canada's north. To this end, the IWS has developed source water protection guidance information generated for a northern indigenous audience. The CAWT's contribution to this work was focussed on the treatment of domestic sewage in the belief that proper treatment is an important component in the overall protection of source waters used for drinking purposes.

The genesis for this wetland work began during the International Polar Year (IPY) in 2007. Once every 50 years, the attention of researchers from around the world is focussed on both the north and south poles. At this time, the Centre for Alternative Wastewater Treatment (CAWT), an applied research facility located at Fleming College, Lindsay, Ontario, Canada was awarded a grant from the federal government of Canada through the IPY programme to examine the performance of tundra wetlands that had been used for several decades in Canada's far north for the treatment of domestic sewage. During this initial IPY study, the CAWT investigated the performance of six wetlands in the Kivalliq region of Nunavut, Canada. This work was further extended to include the study of seven additional wetlands with funding provided by Environment Canada.

During the course of this work it became apparent that most of the wetlands examined do provide an effective wastewater treatment service to the northern communities utilizing them. The natural treatment wetlands provide an economical, passive system that is well suited to the needs of northern communities in that these systems are not reliant on constant supervision by highly skilled operators nor do they require support from the installation and operation of expensive infrastructure. In fact, until recent decades, natural wetlands provided the only viable wastewater treatment option available to some of these communities.

Most northern, isolated communities of Canada's far north now utilize wastewater stabilization ponds (lagoons) as their primary mode of treatment. Despite a historic reliance on tundra wetlands, these areas are typically not formally recognized as being part of the wastewater treatment strategy in the far north. The finding of the CAWT wetland studies suggests that the reasons why wetlands are not formally recognized could be due to:

- an overall lack of study documenting the treatment performance of wetlands,
- a lack of understanding in how wetlands are best managed in order to optimize treatment performance,
- a lack of a standardized sampling protocol, and
- the absence of predictive tools required by wastewater managers and regulatory agencies in order to assess operational management options and future capacities.

The RBC funding has enabled the CAWT to develop management tools (such as SubWet 2.0) and to summarize these tools and findings into one document that is being made available to northern wastewater stakeholders. This document outlines how natural tundra treatment wetlands differ from constructed or engineered wetland systems. It also provides a basic overview of the main treatment processes operative in treatment wetlands. The data and finding generated from the study of 13 natural treatment wetlands within Nunavut and the Northwest Territories are also contained in this document. To our knowledge, this work represents one of the largest and most comprehensive data sets currently existing for northern Canada. The data from these studies has been summarized into tabular form; however, the raw data table are appended along with interpolated mapping for seven of the eleven wetland sites.

During the IPY study, the CAWT joined forces with Dr. Sven Jørgensen, the originator of the SubWet wetland model, to modify this predictive tool for natural tundra wetlands. The grant provided by RCB enabled the CAWT to subsequently develop a user's manual for this model and to calibrate SubWet 2.0 to the eleven wetlands studied. This document provides an overview along with the calibrated rate constants for all eleven wetlands. The SubWet 2.0 wetland user's manual is found in the appendix.

It is believed that the material presented in this document provides valuable insight and tools for wastewater managers and regulators and addresses many of the unknowns that have hampered the formal inclusion of natural tundra wetlands into wastewater

management strategies. It is hoped that the information presented will help to facilitate future discussions concerning the development of a northern wastewater management strategy that formally recognizes the critical services that tundra wetland are currently providing. It is suggested that a reclassification of the treatment wetlands from being part of the "receiving environment" to being an integral part of a hybridized treatment system could provide a treatment strategy with the capacity to meet CCME guidelines for the north. This hybridized system would include the wastewater stabilization pond as the provider for primary treatment and the wetlands as the provider to secondary treatment. It is also suggested that the SubWet 2.0 model could also provide managers and regulators the predictive capacity to determine the best management approaches and the system's capacity to accommodate future growth.

For readers wanting additional information regarding study sites and applications of the SubWet 2.0 program you are in directed to the following published manuscripts that have arisen from this work:

Chouinard, A., Balch, G.B., Wootton, B.C., Jørgensen, S.E. and Anderson, B.C., in press. Modelling the performance of treatment wetlands in a cold climate. In *Advances in the Ecological Modelling and Ecological Engineering applied on Lakes and Wetlands*. Jørgensen, S.E.; Chang, N.B.; Fuliu, X., Eds. Elsevier: Amsterdam, Netherlands

Chouinard, A., Yates, C.N., Balch, G.C., Jørgensen, S.E., Wootton, B.C., Anderson, B.C., 2014. Management of Tundra Wastewater Treatment Wetlands within a Lagoon/Wetland Hybridized Treatment System Using the SubWet 2.0 Wetland Model. *Water*, 6(3):439-454

Yates, C. N., Wootton, B. C., and Murphy, S. D., 2012. Performance assessment of Arctic tundra municipal wastewater treatment wetlands through an Arctic summer. *Ecological Engineering,* 44(0), 160-173

Yates, C.N., Wootton, B.C., Jørgensen, S.E., Murphy, S.D., 2013. Wastewater Treatment: Wetlands Use in Arctic Regions. *In* Encyclopedia of Environmental Management. Taylor and Francis: New York

Yates, C., Balch, G.B., Wootton, B.C., Jørgensen, S.E., in press. Practical Aspects, Logistical Challenges, and Regulatory Considerations for Modeling and Managing

Treatment Wetlands in the Canadian Arctic. In: *Advances in the Ecological Modeling and Ecological Engineering applied on Lakes and Wetlands.* Eds., Jørgensen, S.E., Chang, N. B. and Fuliu, X. Elsevier, Amsterdam, The Netherlands, 560 pages

Yates, C.N., Balch, G.C., Wootton, B.C., Jørgensen, S.E., in press. Exploratory Performance Testing of a Pilot Scale HSSF wetland in the Canadian Arctic. In *Advances in the Ecological Modelling and Ecological Engineering applied on Lakes and Wetlands.* Jørgensen, S.E.; Chang, N.B.; Fuliu, X., Eds. Elsevier: Amsterdam, Netherlands

Yates, C.N., Balch, G.C., Wootton, B.C., Jørgensen, S.E., in press. Framing the Need for Application of Ecological Engineering in Arctic Environments. In *Advances in the Ecological Modelling and Ecological Engineering applied on Lakes and Wetlands.* Jørgensen, S.E.; Chang, N.B.; Fuliu, X., Eds. Elsevier: Amsterdam, Netherlands

Yates, C.N., in press. A Review of Wastewater Treatment in the Canadian Arctic: Comments and Recommendations for New Municipal Effluent Performance Standards. *Arctic*

In summary, it is hoped that this work will prove to be a valued resource for wastewater stakeholders of Canada's far north.

Contents

List of Figure

Chapter 5

Chapter 6

Appendix A

Appendix C

Appendix E

List of Tables

Chapter 5

Chapter 7

Appendix B

Appendix D

Appendix E

Appendix F

Appendix G

1.0 Treatment Wetlands

1.1 Introduction

Treatment wetlands are either natural or constructed systems managed in a specific manner for the treatment of wastewaters. Although traditionally applied for the treatment of domestic and municipal sewage from both separate and combined sewage, treatment wetlands have been applied globally since the late 1980s to treat various types of wastewaters, including agricultural wastewaters (cattle, swine, poultry, dairy), mine drainage, food processing wastewaters (winery, abattoir, fish, potato, vegetable, meat, cheese, milk, sugar production), heavy industry wastewaters (polymers, fertilizers, chemicals, oil refineries, pulp and paper mills), landfill leachate and runoff waters (urban, highway, field, airport, nursery, greenhouse) (Babatunde *et al.*, 2010; Kadlec and Wallace, 2009; Mander and Jenssen, 2003; Vymazal, 1998; Hammer, 1989). Treatment is achieved by a variety of physical, chemical and biological processes, including sedimentation, filtration, precipitation, sorption, plant uptake, microbial decomposition and nitrogen transformations (Kadlec and Wallace, 2009; Scholtz, 2005). An overview of basic treatment processes within wetlands is presented in Appendix A.

The literature related to mechanisms of action published in recent years mainly contains references to the efficient and effective use of constructed wetlands (CWs) or engineered wetlands (EWs) for the treatment of municipal effluents and specific wastewaters from a variety of sources (e.g., mining, industry, etc.). Less information has been published regarding the use of natural wetlands. The term CWs is associated with manmade structures designed to control many of the treatment processes within defined operational conditions, spatial dimensions and process parameters. Similarly, EWs generally refer to wetlands designed to optimize specific treatment processes necessary for the effective treatment of a specific waste stream such as the removal of particular trace elements or organic constituents. By contrast, natural wetlands have developed through natural or spontaneous processes; therefore, many of the characteristics regarding the biochemical,

chemical, physical processes and spatial conditions are unknown and/or unregulated. Because of this, natural wetland tends to have highly variable functional components which make them site specific. Many of the characteristics of the treatment process remain undocumented (Hayward et al., 2012).

Harnessing natural processes or systems for wastewater quality improvement has several advantages over conventional technologies. Cost-reduction is one of them, and a prime reason for the increased interest in CWs and EWs since the high costs associated with the implementation of advanced wastewater technologies is cost prohibitive in many regions of the world. Rural areas with low population densities do not have the financial capital or tax base, and are not able to raise the public funds necessary for such an investment. Another advantage of these passive systems is their low maintenance since it is often difficult for small communities to attract qualified individuals with the technical expertise necessary to oversee large conventional treatment facilities. Some additional ecosystem services include biomass production, carbon sequestration, seasonal agriculture, reusable water supply, regional climate regulation, habitat conservation, and educational and recreational usage (Liu *et al.*, 2008).

Based on hydrological flow patterns, constructed wetlands can be divided into surface flow (SF) and sub-surface flow (SSF) systems (Fonder and Headley, 2011; Kadlec and Wallace, 2009; Kadlec *et al.*, 2000; Suthersan, 1999; Cooper et al., 1998; Cooper, et al., 1996). Sub-surface flow can, in turn, be further divided into horizontal flow (HF) and vertical flow (VF) types, and vertical flow can be either vertical downflow or vertical upflow. Table G-1 in Appendix G elaborates on the classification derived from functional definitions coupled with brief descriptions as well as relevant references. In order to achieve higher treatment efficiencies, especially for nitrogen, various types of systems may be combined. Hybrid systems most frequently combine VF and HF systems arranged in a staged manner, where the advantages of the HF and VF systems can be maximized in a complimentary manner and thus overcome some limitations inherent in each type. With HF and VF hybridized systems it is possible to produce an effluent low in BOD, which is fully nitrified and partly denitrified with an overall reduction in total-nitrogen concentrations (Cooper, 2001).

Natural wetlands can be characterized with similar terminology as the effluent flow in natural wetlands can vary from SF to SSF, or a combination of both.

1.2 CW or EW systems vs natural tundra wetlands

Natural tundra wetlands used for the treatment of municipal wastewaters are fundamentally different from CW or EW systems that are used for the same purposes. Constructed wetlands or engineered wetlands, as their name implies, refers to wetlands that are man-made and designed to specific dimensions, porosity, flow paths, hydraulic retention times, and related design features for the intended purpose of achieving predetermined levels of treatment (Akratos and Tsihrintzis, 2007; Kadlec and Wallace, 2009; Vymazal, 2005). The science regarding treatment processes, reaction rate constants, soil porosity, hydraulics, design options and management practices has been thoroughly investigated in the last two decades and is well documented (Buchberger and Shaw, 1995; Campbell and Ogden, 1999; Cooper, 2009; Hammer, 1989; Babatunde *et al.*, 2010; Kadlec and Wallace, 2009; Mander and Jenssen, 2003; Vymazal, 2011).

In contrast, tundra wetlands are significantly different in several aspects. First, tundra wetlands have developed through natural processes and have not been specifically designed to meet a desired performance characteristic. Therefore there is no control on design, and little options for alteration to enhanced treatment. Each tundra wetland is unique and very little is known about site specific hydrology, porosity, soil types and depth, flow paths and other key parameters influencing wastewater treatment. Treatment potential varies widely from site to site, and is thus site specific. Vegetative boundaries are relatively easy to identify, however it is difficult to know the subsurface flow paths that the wastewater travels and how these may change seasonally or annually and thus it is difficult to determine what portions of the wetland are actually involved in the treatment process. Likewise, soil types and depths are not homogeneous and unlike CW or EW systems it is difficult to gather information on many of the physical parameters required to make predictions regarding treatment performance.

Lastly, there are several aspects inherent within tundra wetlands used for the treatment of wastewaters that can make them logistically challenging to work with. Some of these aspects include the difficulty in identifying single or representative "point of release" where the effluent leaves the wetland. Wetlands can have more than one point of release and it is possible that the location and relative volumes delivered can change seasonally and from year to year. This makes it difficult for regulatory agencies when trying to identify the best location to obtain an effluent sample representative of treatment efficiencies. In addition, unlike engineered lagoons, it is often not possible to control the flow leaving the wetland and thus if treatment is lower than desired, then there is no way of controlling the flow from the wetland until better treatment is achieved. The remoteness of tundra treatment wetlands presents challenges for not only sample collection but also sample analysis. In most cases the nearest laboratories capable of providing the analytical services needed are located further south in Ottawa, Winnipeg or further west in Yellowknife. Some wastewater parameters such as BOD_5 and *E. coli* have relatively short (e.g. 48h) holding times to ensure sample integrity.

Further challenges include the fact that most of the research on processes and reaction rates has been generated from CW or EW systems established in warmer temperate or subtropical climates which are quite different from the harsh conditions of northern Canada were tundra wetlands freeze solid for a significant portion of the year. Because of the logistical challenges in gathering the type of information described above, most regulatory agencies have tended to view the tundra wetlands as unknowable and unpredictable and therefore of little use as part of a formally recognized wastewater treatment strategy.

1.3 Natural tundra wetlands for wastewater treatment in the Canadian Arctic

Communities in the Canadian Arctic and sub-Arctic have small populations of 100 to 2,000 people and many can only be accessed by air, or by sea during the brief summer season, making them dependent upon self-supported infrastructure to deliver community services, such as wastewater treatment, solid waste disposal and provision of potable water

(Jamieson and Krkosek, 2013; Yates *et al.*, in press). The extreme climate, the logistical challenges of bedrock and/or permafrost together with the lack of financial and human resources represent significant impediments to the development and operation of mechanized wastewater treatment infrastructure commonly used in more southern locations of Canada. Therefore, people living in the Arctic often have to rely on a trucked system for water delivery and wastewater collection. Historically, domestic sewage within these communities was trucked outside of the community and deposited into natural depressions which were typically better classified as natural tundra wetlands. Current practices have evolved somewhat with domestic sewage being initially contained within heated storage tanks of individual dwellings only to be pumped and trucked to a holding lagoon or lake with eventual release of the effluent often to a natural tundra wetland.

The Canadian Council of Ministers of the Environment (CCME) recently developed a Canada-wide strategy for the management of municipal wastewater effluent. This strategy was endorsed by the Council of Ministers on February 17, 2009. It is a strategy that sets out a harmonized framework to manage discharges from more than 3,500 wastewater facilities in Canada, many of which currently need repair and upgrading. The intent is to develop performance standards which will increase the protection for human health and the environment on a national basis. The CCME is comprised of the 14 environment ministers from federal, provincial and territorial governments. The CCME recognizes that the Canada-wide performance standards may not be appropriate for some regions of Canada because of regional differences, such those within portions of Canada's Far North (Nunavut, Northwest Territories, and regions located north of the 54th parallel in Quebec and Newfoundland and Labrador). As such, the CCME has provided a 5 year window during which time a strategy will be developed to prepare regional specific performance standards. Regardless of the outcomes, communities north of 60° N latitude will have to ensure that the domestic wastewater effluent meets the performance standards for the far north.

Tundra wetlands located downstream of waste stabilization ponds or those connected with facultative lakes are generally considered as part of the receiving environment and not part

of the treatment process. The exploratory research by Yates *et al.* (2012) and Yates *et al.* (2013) has demonstrated that although tundra wetlands are not formally recognized as part of the treatment process they do in fact provide significant additional treatment benefit. Yates *et al.* (2012) and Yates *et al.* (2013) assessed the wastewater treatment potential of several tundra wetlands located downstream of primary treatment facilities. Apart from these investigations there are relatively few studies that match the scope of their tundra specific work. Most of the existing literature provides little guidance regarding treatment process reaction rates, management strategies or predictive tools for assessing the capacity of existing tundra wetlands to meet the needs of expanding populations (Doku and Heinke, 1995). Natural tundra wetlands are by nature open and diffuse systems, often with poorly defined boundaries, flow patterns and permeable boarders (Kadlec, 2009). These conditions present challenges for wastewater regulators who require well defined points of control. This manual is therefore an attempt to address some aspects of the knowledge gap and provide help to those tasked with the management and regulation of wastewaters in Northern Canada by providing background information related to how tundra wetlands process wastewaters and to provide a summary of the most recent data that assesses the efficacy of using tundra wetlands to treat municipal wastewaters along with some operational tools.

In 2008 a team of researchers, environmental educators, and national aboriginal organizations collaboratively presented a proposal to the Royal Bank of Canada under the bank's Blue Water Trust Fund that outlined a path forward to address water issues in aboriginal communities of Canada's north. The proposal addressed both source water protection and the treatment of domestic sewage. This manual has developed out of the work focussed on the development of resource tools for the management and operation of tundra wetlands for the treatment of municipal wastewaters. The manual in its current form is limited to the discussion of wetlands and does not address the operation and maintenance of sewage lagoons, facultative lakes or ponds other than to recognize that they are an important component in the effective treatment of sewage and are required as a pre-treatment step for the sewage prior to it being released to the treatment wetland. In many

ways the sewage lagoons perform much of the primary treatment while the wetlands perform functions more characteristic of secondary treatment.

2.0 Tundra wetland as part of the treatment chain

2.1 Natural tundra wetland for wastewater treatment

Focusing on waste disposal as a management technique, rather than wastewater treatment was the original intent of many Arctic systems (Yates *et al.*, in press). As described by Gunnarsdottir et al. (2013) and Ritter (2007) the focus on waste disposal has also been common in many other Polar Regions. Because of the remote nature of these communities, small population sizes, harsh climates and the technical and logistical barriers that hinder the application of mechanical treatment systems more typical of developed regions in southern Canada, the use of waste stabilization ponds (sewage lagoons) and facultative lakes arose. When compared to lagoons/lakes, tundra wetlands have been generally viewed as providing little to no treatment benefit. There has also been concern about the potential release of untreated or partially-treated wastewaters into a natural environment may pose a human health risk if tundra wetlands were the sole treatment option (Doku and Heinke, 1995). In many cases, the genesis of wetlands located downstream of the waste stabilization ponds has been in response to the release of nutrients and organic matter exiting the lagoons. This in turn provided the conditions conducive to the growth and establishment of natural vegetation (Yates et al., in press; Doku and Heinke, 1995). Consequently, many tundra wetlands did not arise because of any intentional design on the part of waste managers and thus cannot be considered akin to constructed wetlands in terms of design features.

Two major tundra wetland assessment studies funded as Canada's contribution to the International Polar Year (2007) and Environment Canada have been summarized and published by Chouinard et al., in press; Chouinard *et al*, 2014; Yates *et al.*, 2012; Yates *et*

al., 2013; Yates et al., in press; Yates *et al.*, in press and Yates *et al.*, in press. These studies have demonstrated that even though tundra wetlands are not formally recognized as part of the treatment process they do provide a significant additional benefit to the treatment of municipal wastewater effluents. Apart from these investigations there are relatively few studies that have matched the scope of this tundra specific work. Most of the previously collected information related to the anticipated cold climate treatment performance and predictive aspects of wetland size is found in the unpublished literature and much of this work provides little guidance regarding management strategies, treatment process reaction rates or predictive tools required to assess the capacity of existing tundra wetlands to meet the needs of expanding populations (Doku and Heinke, 1995).

Wastewaters intentionally released from waste stabilization ponds (e.g., decanted), unintentionally released as leakage from the detention berms of the ponds (exfiltration) or the natural releases from facultative lakes typically exhibited levels of carbonaceous biochemical oxygen demand (cBOD), total ammonia nitrogen (TAN) and microbial indicator organism *E. coli* and fecal coliforms that were higher than desired by territorial regulatory authorities (Yates *et al.*, 2012; Yates *et al.*, in press; Yates *et al.*, in press; Chouinard *et al.*, 2014; Challen-Urbanic, 2009). Duko and Heinke (1995) have described wetlands as an energy-efficient and low-cost treatment option for municipal wastewater in the Northwest Territories, Canada. However, they did identify that that the lack of adequate design criteria and performance data presented significant challenges to their use in the Canadian Arctic. Because of these reasons, wetlands are overlooked by some as a viable treatment option, particularly until recently when there were only a few sources from non-peer-reviewed literature that provided only limited guidance regarding their use or potential use (Kadlec and Johnson, 2008).

2.2 Logistical challenges

Because of the remote nature of the communities in the Canadian Arctic, the management and treatment of wastewater are being confronted with a number of logistical challenges. Logistical challenges related to wastewater management by communities in the Arctic at the

time of their formation in the early 1960's remain (Yates et al., in press). Dawson and Grainge (1969) had suggested that heated piped systems for Arctic communities was better than haulage systems, because of a reduced chance of accidental spillage, lower chance of contamination and lower maintenance cost. However, the logistical challenges with respect to designing piped systems in Arctic communities because of permafrost, topography, poor town layout and the high construction costs remain impediments to utilodor systems.

The logistical challenges of disposal and management have largely remained unchanged to this day due to the complex nature of Arctic communities. Haulage systems remain the most common form of wastewater collection, despite periods of no service because of service repairs to vehicles or inclement weather (Yates et al., 2012). It is difficult to directly quantify volumes of wastewater discharged into the treatment systems because of the way wastewater is managed. Wastewater volumes can however be indirectly estimated by knowing the volume of potable water delivered to each building. The volume of potable water is monitored more closely since municipalities are mandated under the territorial water taking license agreements to record drinking water withdrawals (Government of Nunavut, 2002).

Access to trained personnel required for the management and operation of municipal infrastructure including wastewater facilities is a major logistical issue for Arctic communities. Johnson (2010) explained that this administrative challenge can be found at multiple levels and is often expressed as a lack of resources for hiring, and even more so a lack of resources for training and retaining qualified personnel. Poor management of records and consequently an erosion of the community's knowledge-base are results of constant shuffling of staff. For the majority of Canadian Arctic communities, this has become an endemic problem (Johnson, 2010).

The fact that accredited laboratories close to Arctic communities are lacking creates a logistical hurdle for compliance testing required for wastewater facilities. Many wastewater parameters are time sensitive and require analysis within defined time lines. The shipment of samples often requires more than twenty-four hours to reach the closest laboratory for

analysis: therefore, when samples are collected for regulatory purposes, the quality of the sample arriving at the laboratory is often poor (Yates et al., in press).

2.3 Natural tundra wetland sampling

Natural tundra treatment wetlands present many logistical challenges to sampling and despite the recent studies, much still remains to be known regarding seasonal influences on treatment efficiencies and year-to-year variability in performance. The main challenges to a formal recognition of wetlands as part of a treatment strategy can likely be summarized as:

- No point of control
- No standardized protocol for the establishment of representative sampling points for wetland influent and effluent
- Seasonal and year-to-year influences on treatment performance are poorly understood
- Logistical challenges in sample analysis

2.3.1 Point of control

In most installations, waste stabilization ponds (lagoons) are engineered in a manner that provides the manager control over the release of the treated effluent. In theory, the manager has the ability to retain the wastewater until a desired level of treatment is attained, or to halt the release of wastewater if desired. In the case of natural tundra treatment wetlands, managers do not have the same control over the release of wastewater leaving the wetland. Thus if a manager determined that the wetland was under performing, they would have few options for stopping the flow and retain the wastewater in the wetland until the desired level of treatment was obtained. Therefore without an ability to manage the release of effluent, the regulatory control is undermined.

One option to gain better control over the treatment and release of effluent from the wetland would be to provide good control on what enters the wetland in terms of both effluent quality and quantity. The work outlined earlier in this document demonstrated that treatment performance of the wetland decreased when effluents were decanted (released) into the wetland in late summer over a relatively short time period of a few days (e.g., Fort Providence) as opposed to a continue slow release from an exfiltrating berm of the lagoon. Therefore controlling the volume and timing of the release is one way to better control what is potentially released from the wetland.

2.3.2 Inlet / Outlet sampling points

Site conditions vary greatly amongst individual treatment wetlands. For example, the flow patterns of wastewater released from lagoon systems into the wetland can vary widely within and amongst wetlands. In some situations the flows released from exfiltrating berms travels underground while within the same wetland, some of that flow may travel on the surface in braided streams. Determining which waters (ground waters, surface waters) to sample and in what locations in order to get a representative sample of the influent into the wetland becomes challenging. Likewise it can be just as challenging to determine sample locations of waters exiting the wetland that would provide a good representation of the treatment performance. In addition Mitsch and Gosselink, (1986) concluded that in diffuse and dynamic systems like wetlands, it was unrealistic to establish static formal sampling points since flow patterns alter on an annual basis. What is clearly needed is the establishment of a formalized sampling protocol that provides a standardized methodology for determining sampling locations for the collection of both influent and effluent samples. The sampling protocol described earlier in this document by Yates et al. (2012) provides a good starting point. However, broader input is needed by regulatory agencies and other stakeholders in order to develop a standardized protocol that is acceptable by all major players.

2.3.4 Seasonal and yearly variability

More study is needed to better understand the how treatment performance may vary over seasonal changes and amongst years. One of the major areas that need to be better defined is the period of spring freshet. Wetlands can vary significantly in how they are managed during winter months. In some cases there can be a buildup of wastewater over the winter months that rapidly melt during the freshet. Most of the recent investigations have missed monitoring this time due to logistical challenges associated with travel and sampling. Better understanding the performance dynamics at this time will help in the establishment of better management practices. Likewise, monitoring wetland performance over several years and correlating climatic conditions with treatment performance will help managers better quantify uncertainties in predicted treatment efficiencies.

In an attempt to better understand treatment processes within these Arctic tundra wetlands, the timing of sample collection is important to ensure key treatment periods are captured. Yates et al., (in press) mentioned that sampling should take place at a minimum three times within the year. Accessing these communities more frequently is recognized to not be feasible given the limited availability of resources to do so. Yates *et al.*, (in press) stated that samples should be conducted during thaw (to capture any freshet event - if logistically possible), the active summer period of the wetland, and just before freeze up. The authors explained that these shoulder periods are important, as they will represent the period of poorest performance in the wetland.

2.3.5 Logistical analysis in sample analysis

Due to the fact that the geographical location of the investigated wetlands by the CAWT and Yates *et al.,* (2012) spanned a large geographic area and because of shipping logistics, the laboratory used for sample analysis varied from location to location and between years. Appendix B provides a list of contact information regarding the specific laboratories at which the samples were analyzed.

Natural tundra wetlands have been a reliable technology to manage wastewater to-date despite the lack of knowledge regarding performance, or acceptance as part of the regulated treatment chain. Even though the climatic conditions in the Arctic are harsh wetlands have still demonstrated the capacity for wastewater treatment via natural processes; albeit at slower rates. Evidence of ecosystem response to nutrients in the Arctic wet tundra provides further detail that these systems can quickly assimilate small additions of nutrients and organic matter (Mack *et al.*, 2004; Shaver and Chapin, 1980; Shaver and Chapin, 1995; Gough *et al.*, 2002). Because of the logistical challenges in gathering the type of information described above, most regulatory agencies have tended to view the tundra wetlands as unknowable and unpredictable and therefore of little use as part of a formally recognized wastewater treatment strategy. The need and or merit of formally recognizing these lands in land use planning documents should be revisited in light of the most recent evidence regarding the efficacy of tundra wetlands for the treatment of municipal wastewater effluents. Formal recognition may include the reclassification of these tundra treatment wetlands from their current designation as the receiving environment to being classified as part of the treatment train.

3.0 Current wastewater treatment strategies in Northern Canada

3.1 Current treatment practices employed in the Canadian Arctic

The logistical challenges of bedrock and/or permafrost together with the lack of financial and human resources and within the cold arctic climate represent significant impediments to the development of mechanized wastewater treatment infrastructures commonly used in more southern locations within Canada. As such wastewater disposal to lagoons, engineered lagoons, facultative lakes and/or direct discharge to land have been seen as the most feasible historical options available. The release of primarily treated municipal effluent to the land

occurs through the intentional decanting of effluent from a lagoon or via exfiltration (intentional or unavoidable leakage) of the effluent through the wall of the lagoon berm on to the land. Until recently a few communities discharged their hauled wastewater directly to natural depressions or surface water bodies termed facultative lakes without prior treatment. Anecdotal information suggests that the release of municipal wastewaters or primarily treated wastewater effluents into natural depressions appears to have either enhanced vegetative growth or in some cases may have even facilitated vegetative growth in areas that were naturally devoid of vegetation.

In most locations within Canada's Far North, municipal wastewater is first held within sewage lagoons where the wastewater undergoes primary treatment for the reduction of suspended solids and biochemical oxygen demand. In most locations, the retention of the wastewaters is not sufficient to produce municipal effluents that would meet the proposed NPS of 25 mg L^{-1} for the parameters of cBOD and TSS. Tundra wetlands that have either serendipitously or by design developed downstream of sewage lagoons may have the potential to become part of a hybridized treatment system that includes not only the lagoon but also the associated wetland

3.2 Wastewater handling

Yates *et al.*, (2012) stated that wastewater streams in Arctic communities are often quite homogenous and that in most communities it only contains blackwater and greywater. Greywater is discharged directly on to the land from the residence in some communities. Wastewater from commercial sources is typically limited to a few service providers in the community, generally consisting of a hotel, grocery and hardware stores (Yates *et al.*, 2012). In the majority of Arctic communities, industrial wastewater is not a component of the waste stream. Communities will occasionally have process/packaging plants for fish or other locally harvested foodstuff. The only other source of industrial wastewaters is provided by airports and fueling depots, but most of the waste generated in these systems is not incorporated into the municipal waste stream, as they are often diffused across the landscape (Yates *et al.*, 2012).

As stated earlier, wastewater is managed at the source by binging temporary held in insulated holding tanks located just outside or under raised buildings. Tanks are pumped out and sewage hauled via pumper trucks to the disposal area. Drinking water is also distributed via tanker trucks. Consequently, water use in these communities is significantly less than the Canadian average (Yates *et al.*, 2012). Wootton and Yates (2010) reported that the average for many communities in Nunavut is approximately 90 Liters/day/per, but some communities have usage averages below 70 Liters/day/per. Low usage in these communities is often the result of logistical issues described previously.

3.3 Long and short-term holding lagoons

Long and short-term holding (discontinuous and continuous discharge) lagoons are the most common treatment system in Canadian Arctic communities (Heinke *et al.*, 1991). Long and short-term lagoon systems are often engineered using aspects of the natural landscape. The use of small lakes, with additional berms to prevent spring overflow and engineered berms in a natural depression, are common methods of creating lagoons to treat wastewater in the Arctic. Wastewater that is directly discharged into a lake is referred to as facultative lakes or ponds. Facultative lakes may be contained (retention) or experience percolation (continuous discharge or detention) of wastewater through the berm sides. The engineered and facultative lake lagoon systems rely on algae-bacterial populations to breakdown organic matter in aerobic and anaerobic zones of the lake. Discontinuous or intermittent discharge lagoons are also common. In the past these systems have been designed in the same manner as lagoons in more temperate regions, but are often much larger to accommodate for deeper winter ice depths, lower bacterial-algae biomass and longer residency time (Dawson and Grainge, 1969). Because lower bacteria-algae populations limit the metabolism of organic particles, mechanical aeration has been recommended for northern regions (Dawson and Grainge, 1969). Although mechanical aeration is a common solution in temperate regions, in remote northern communities the availability of infrastructure to power those systems is not feasible due to operational

limitations. Despite extensive use of lagoons in the Arctic there remains little peer-reviewed literature on their performance (Wootton *et al.*, 2008c; Yates *et al.*, 2012).

3.4 Mechanical and connected central serviced systems

The use of mechanical and connected centrally serviced wastewater treatment facilities in the Canadian Arctic is minimal. Some large communities utilize the Hudson Bay or Arctic Ocean as a receiving environment, with primary treatment connected on line prior to discharge into the receiving environment. These communities have municipal services (piping) that serve many of the residents, commercial buildings and any industry present. The wastewater passes through a pumping or lift station to the receiving environment. The pumping station may contain primary treatment systems, such as screening and/or communitors to remove or break down grit and large organic debris (Johnson, 2008). This form of wastewater treatment is uncommon in communities of the Canadian Arctic. The only community that is recorded to have anything more advanced than primary treatment is Pangnirtung, Nunavut on Baffin Island. Pangnirtung is reported to have a secondary treatment facility using a rotating biological contactor and activated sludge system (Wootton *et al.*, 2008a). Most Arctic communities remain without mechanical systems since earlier attempts have shown that these systems under Arctic conditions regularly fail to produce effluent that meet regulatory standards, or because of operation costs, or the lack of a skilled labor pool to maintain them (Johnson and Wilson, 1999). Initial attempts to use mechanical treatment could be considered an oversight by planners to appropriately address community needs, as many communities have returned to using simpler technologies such as lagoons (Johnson, 2008). This evidence demonstrates the need for alternative low cost, simple, yet efficient techniques for wastewater treatment in the Arctic.

3.5 Land treatment and wetlands

Land disposal or land treatment is another common method of wastewater treatment or disposal in the Arctic (Wootton *et al.*, 2008a). In Arctic Canada, wastewater disposed onto

the land is done so at some distance away from the community and drinking water sources, although there are examples where the receiving environment is indirectly connected to the community water supply, as in Baker Lake (Wootton *et al.*, 2008a; Yates *et al.*, 2012). Although overland flow is present, such as in Coral Harbour, many of the land treatment locations are actually in natural wet-sedge tundra wetlands. However it is not known whether these systems existed as wetlands before receiving increased water and nutrient loads, or whether they are a result of the anthropogenic influence. Evidence from fertilization studies show that nitrophilous and hydrophilic plants have been found to colonize these environments following long periods of increased water and nutrient loading. Kadlec and Johnson (2008) suggested that the wetlands may not have been present prior to sewage being disposed at these sites.

Whether or not these landscapes have been altered, the use of wetlands is extensive secondary and, in some locations, primary treatment in Arctic Canada. They have shown excellent ability to treat wastewater in the past in more temperate locations (Kadlec and Wallace, 2009; Mander and Jenssen, 2002). However, similarly to lagoon systems in the Canadian Arctic, there is very little data from peer-reviewed literature on wetland performance.

3.6 Current and future wastewater regulations in the Canadian Arctic

Performance standards for wastewater effluents are currently in transition within Canada as the federal government is developing national performance standards (NPS) for municipal wastewater effluent. In 2009, the Canadian Council of Ministers of the Environment (CCME) released the final draft of the *Canada-wide Strategy for the Management of Municipal Wastewater Effluent* which details regulatory changes to be implemented through the Canadian *Fisheries Act.* The intent of the strategy is to ensure there are no deleterious effects to the water bodies receiving the treated effluent, particularly with regard to fish health and or fish habitat. This strategy has identified specific national performance standards for effluent of Canadian wastewater treatment

facilities at 25 mg L^{-1} for the parameters of cBOD and TSS, 1.25 mg L^{-1} for un-ionized ammonia expressed as NH$_3$-N at 15°C±1°C and a standard of 0.02 mg L^{-1} of total residual chlorine (TRC) (CCME, 2009). The Federal Government recognizes that conditions in portions of Canada's Far North (Nunavut, Northwest Territories, and regions located north of the 54th parallel in Quebec and Newfoundland and Labrador) are unique and as such NPS have not yet been determined for these areas. A five-year research period was initiated in 2009 to determine what NPS (treated effluent concentration levels) would be appropriate in the Canadian north (CCME, 2009).

4.0 Cold Climate Treatment Wetland Studies: Overview from the 1970s to present day

This section 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 operating specifically in cold climates. This information is intended to summarize the current performance of tundra treatment wetlands in Canada's far north and to identify knowledge gaps that need to be addressed to improve the application.

Our knowledge of wastewater treatment in remote Canadian Arctic communities has grown very little since the 1970s, despite a half-century of operation. Current knowledge of treatment performance of natural treatment wetland systems in the Canadian Arctic is largely restricted to site-specific governmental and consultant reports (Dillon Consulting Ltd., 2004; Environment Canada, 1985), and other sources of unpublished literature. Only a few peer-reviewed documents, as well as conference proceedings (Miyamoto and Heinke, 1979; Johnson and Wilson, 1999) exist to contribute to our current understanding of performance with most investigations confined to the performance of lagoons. In the Canadian Arctic, wastewater treatment facilities such as lagoons and wetlands are generally designed and managed using southern engineering standards, adopting design models to

reflect Arctic temperature (Kadlec and Johnson, 2008; Prince *et al.*, 1995; Heinke *et al.*, 1991).

Most of the intentional use of natural wetlands for the treatment of municipal wastewater has been primarily for the provision of tertiary treatment (Breaux *et al.*, 1995; Cooke, 1994; Kadlec *et al.*, 1979). In the 1990s, it was estimated that approximately half of the 200 surface flow wetlands used for wastewater treatment in North America were natural wetland systems (Brix, 1994). There has been a general reluctance to use natural wetlands for treatment purposes since the long term impacts upon the natural ecology of these areas is generally uncertain (Kadlec and Wallace, 2009). Despite these uncertainties, it has been demonstrated on numerous occasions that natural wetlands are effective in the treatment of municipal wastewater. Previous studies and reviews on this topic have demonstrated that wastewater treatment can be effective even in cold climate regions (Mander and Jenssen, 2002; Kennedy and Mayer, 2002; Wittgren and Mæhlum, 1997; Jenssen *et al.*, 1993). The following paragraphs in this section summarize the research conducted to date on cold climate treatment wetlands in Canada, particularly the arctic and sub-arctic.

Wright (1974) conducted one of the first natural treatment wetland studies in northern Canada. This site was a 32 ha natural swamp in Hay River, NWT that received primary treated municipal wastewater from three wastewater stabilization ponds WSPs. The author reported that the effluent discharge area was estimated at 110 m^2 per capita per year. In this study, hydrological context for the swamp was not quantified; therefore, dilutive effects could not be estimated. The swamp was sampled monthly from August 1972 to September 1973 (Wright, 1974). Favorable percent reductions in concentrations of treatment performance parameters were observed in Wright's study, including: 97.7% for BOD; 96.8% for TSS; 98.0% for VSS; 96.2% for Un-ionized Ammonia Nitrogen (NH$_3$-N); 97.6% for TP; and 98.7% for Total Coliform (TC). The author observed an ecological alteration to the swamp in terms of decreased benthic organism diversity, zooplankton, nekton, and alterations in the overall composition of the ecological community. Flows were observed to be transient in the swamp over the treatment season with a high spring freshet flow. Increased bacterial concentrations were observed at the swamp's outlet during the

spring freshet. Wright (1974) recommended guidelines for the use of similar natural swamps as part of the wastewater treatment process. These recommendations stipulated that the: 1) wetland treatment areas be sited away from groundwater recharge zones; 2) alteration of the natural wetlands should be deemed acceptable; and 3) wetland morphology should be such that the HRT is relatively long. Doku and Heinke (1993) reported that the Hay River, NWT swamp was studied for over twenty years and during that time it consistently met the regulatory compliance requirements for the treatment system.

The work of Dubuc *et al.* (1986) is one of very few papers to investigate long-term performance of treatment wetlands in Northern Canada. The authors demonstrated that a natural peat wetland in Northern Quebec was highly effective at treating domestic wastewater. The study site was located at the James Bay Energy Company's Fontanges construction camp for the James Bay Energy Company in Northern Québec and was a mid- to high boreal wetland area near the 55th parallel. The camp wastewater was discharged into a peatland that was approximately 1.5 km in length. For most treatment performance parameters, average percent reductions were greater than 90%, indicating that the natural peatland system was highly effective at treating the wastewater. Dubuc *et al.* (1986) noted that the study failed to quantify the hydraulic parameters of this area and thus the proportion of treatment related to dilution could not be assessed.

Doku and Heinke (1993) reported that Canada's northern communities are small and remote, experience extreme climates, have limited infrastructure funds for construction, design and maintenance of facilities, and have a chronic shortage of skilled labour. Natural tundra wetland areas in northern territories are generally nutrient deficient, and abundant. 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 Doku and Heinke (1993), the primarily domestic origin of the wastewater generated and the relatively small size of northern communities render the use of natural tundra treatment wetlands an appropriate option for use in the North, particularly for communities that have sufficient land available for the application. 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 were recommended by the authors.

The use of natural wetlands in the Yukon was also discussed by Doku and Heinke (1995). The Yukon Water Board had at that time 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 treatment performance and site-specific hydraulic assessments be conducted before use of the systems as part of the wastewater treatment train. Primary treatment of wastewater, at a minimum, prior to discharge into the wetlands was a regulatory requirement. The demonstration of long-term treatment of effluent discharge in the absence of ecological impacts was a stipulation of acceptance by the regulatory body. Doku and Heinke (1993) suggested, as part of their study, 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 treatment wetland 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 natural wetland; and 4) that an areal BOD loading rate not exceed 8 kg BOD_5/ha·d, and that a HLR of between 100 and 200 m^3/ha·d be maintained. In a subsequent publication (Doku and Heinke, 1995), the authors provided additional recommendations: 5) conduct further research on the use of natural treatment wetlands in order to assist in the establishment of design criteria; 6) that the NWT government encourage 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.

A study by Kadlec and Johnson (2008) addresses some mechanistic function in a Canadian Arctic treatment wetland but did not provide significant background data. Much of the current knowledge on plant and microbial influence on wastewater treatment in the Arctic has been derived from smaller-scale fertilizations and carbon cycling studies in different

Arctic environments (Shaver and Chapin, 1995; Arens *et al.*, 2008; Edwards and Jefferies, 2010). Edwards and Jeffries (2010), in a study on the winter treatment abilities of arctic wetlands, disproved the common assumption that arctic wetland treatment mechanisms cease in the winter. Their study located in a low arctic meadow in Churchill, Manitoba, focused on nitrogen uptake by *Carex aquatilis*. The authors determined that plant uptake still occurs at temperatures below 0°C. They suggested that decomposition processes occur year-round in tundra ecosystems, and mentioned that vegetation nitrogen uptake was elevated during spring melt. These studies demonstrate the effective use of natural arctic and sub-arctic wetlands for use in municipal wastewater treatment.

Recently, researchers and graduate students from Dalhousie University's Centre for Water Resources Studies have established environmental monitoring programs in six Nunavut communities: Grise Fiord, Pond Inlet, Clyde River, Coral Harbour, Pangnirtung, and Kugaaruk. Hayward *et al.*, (2012) conducted hydrological characterization and assessed treatment performance of a natural tundra wetland receiving effluent from a single-cell wastewater treatment exfiltration lagoon in Coral Harbour, NU. The authors reported that the hydraulic loading rate of effluent on the wetland was highly dynamic, depending on seasonal factors with greater loading occurring during the spring melt period. The HRT of the natural treatment wetland was also highly variable depending on the period of observation; generally, the HRT was much shorter during the spring melt when flows into the wetland were high. Treatment performance (in terms of concentration reductions) of the natural treatment wetland was observed to be reduced in June compared to September, attributable to decreased retention time and higher amounts of dilution. Hayward *et al.*, (2012) concluded that the incorporation of natural tundra wetland areas in wastewater treatment may be a viable option in Canada's North; however, appropriate rate constants must be applied when determining appropriate hydraulic loading rates, and sizing the required wetland treatment area.

The Centre for Alternative Wastewater Treatment (CAWT), Fleming College, Lindsay, Ontario has also been investigating the treatment performance of Arctic wetlands since 2008. The majority of this work was funded by the Federal government through the

International Polar Year (IPY) program and from Environment Canada (EC). Much of this work has been published in Yates *et al.*, (2012, 2013 and in press) and Chouinard *et al.*, (2014 and in press). The IPY study assessed the treatment performance of six natural tundra wetlands that received municipal wastewater from the hamlet communities of Arviat, Baker Lake, Chesterfield Inlet, Coral Harbour, Repulse Bay, and Whale Cove in the Nunavut region of Kivalliq. During the EC study the treatment performance of seven additional northern treatment wetlands located in both Nunavut and Northwest Territories were investigated. The focus of the EC study was to monitor treatment as the wastewater traversed the wetland in order to provide Environment Canada background information needed to help in the determination of appropriate NPS specific to arctic conditions within Canada. The overall goal of both studies was to evaluate the efficacy of using northern tundra wetlands to treat municipal wastewaters. The results are presented in section 6.1.2 and 6.2.2 of Chapter 6. The treatment performance assessments 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.

5.0 Natural tundra wetlands for wastewater treatment

The following represents the culminating summary for two separately funded studies investigating the performance of northern treatment wetlands undertaken by the Centre for Alternative Wastewater Treatment (CAWT), Fleming College, Lindsay, Ontario. Data from these two studies likely represent the most extensive data base currently existing. This data base assesses the capacity of 13 tundra treatment wetlands situated in Nunavut and the Northwest Territories of Canada. The first study was funded by the Federal government through the IPY program. During the IPY study (2008-2011) the treatment wetlands associated with six Nunavut communities (Kilvalliq region) were investigated by monitoring

key water quality parameters within the influent and effluent (e.g., two sample locations) over the course of an Arctic summer (e.g., several weeks of sampling per wetland). A summary of the IPY investigation has been published in Yates *et al.*, (2012, 2013 and in press) and Chouinard et al., (2014 and in press). Subsequent funding was provided by Environment Canada (EC) through the EC Aquatic Ecosystem Management Research Division to investigate the treatment performance of seven additional northern treatment wetlands located in both Nunavut and Northwest Territories (2009-2011). The focus of the Environment Canada study was to monitor treatment as the wastewater traversed the wetland. This meant that each of the EC wetlands was studied more intensely than the IPY wetlands (e.g., multiple sites between inlet and outlet of wetland). The duration of time spent at each of the EC wetlands was generally only a few days per year (unlike the IPY study); however, several of the EC wetlands were revisited in subsequent years. The overall scope of both studies was to evaluate the efficacy of using northern tundra wetlands to treat municipal wastewaters. This involved monitoring the change in chemical, biochemical and physical characteristics of municipal wastewater effluent as it traveled through wetland areas and to assess major influencers within the wetland that affect wetland performance. The EC study was undertaken to provide Environment Canada background information needed to help in the determination of appropriate NPS specific to arctic conditions within Canada.

5.1 International Polar Year Wetland Study

The CAWT was awarded a research grant from the Federal government through IPY to investigate the efficacy of northern treatment wetlands and to document efficiencies in the treatment of municipal sewage received from six adjacent hamlets in the Kivalliq region of Nunavut. Systems in the Hamlets of Arviat, Baker Lake, Chesterfield Inlet, Coral Harbour, Repulse Bay and Whale Cove were studied (Figure 5.1). Three of the six treatment wetlands, located in the communities of Arviat, Coral Harbour, and Whale Cove received effluent after pre-treatment in waste stabilization ponds or lakes. The other wetlands in

Figure 5.1   Map of Canada showing location of communities studied (Map Credit: Noreen Goodliff).

Baker Lake, Chesterfield Inlet, and Repulse Bay received either wastewater that had received minimal pre-treatment or raw wastewater from trucks directly. This section provides a summary the study results. Those wishing more detail regarding this work are referred to the published work of Yates *et al.* (2012 and in press). All IPY wetlands sites studied in these six communities were all physiographically distinct, with varying cover and composition of vegetation communities, presence of surface water and treatment area (Yates *et al.*, 2012).

5.1.1 Methodology

Samples were collected from six treatment wetlands between June 21st and September 24th, 2008 which approximates the historical ice-free period of the year (9-12 weeks); June 10-15 to September 5-20 (Mack, 2004; Yates *et al.*, 2012). Samples were transported in coolers to a laboratory in Rankin Inlet and analyzed within 24 h of collection for time sensitive analysis of parameter (e.g. cBOD$_5$, and pathogens) following Standard Methods for Wastewater (Yates *et al.*, 2012). At each of the six wetlands, samples of 500 mL each from the point of influence and effluence were obtained. The samples gathered weekly were used to evaluate the temporal variation associated with treatment efficacy of the tundra wetlands (Yates *et al.*, 2012). Biological, chemical and physical water quality parameters were assessed; particularly cBOD$_5$, TSS, and NH$_3$-N which are regulatory parameters of the Fishery Act regulations (Government of Canada, 2010).

Other sampled parameters include DO, TC, *E. Coli.*, TP and COD. Temperature was recorded continuously over the ice-free period, with Onset Temperature logging tidbits situated in the surface water of the influent and effluent streams, obtaining readings at 0.5 h intervals (Yates *et al.*, 2012). The focus of this study was to assess treatment efficiencies over the duration of one Arctic summer (June – September). This was accomplished by monitoring selected wastewater quality parameters in both the influent entering the wetland and the in the effluent exiting the wetland. No attempts were made to monitor the progression of treatment at sample locations between the influent and effluent sites. Sampling at the influent and effluent is considered the minimum required sampling for wastewater treatment facilities (Kadlec and Wallace, 2009). Yates *et al.*, (2012) reported that sampling more than once per week was not logistically possible, given restrictions of flight schedules in the Arctic to transport samples within a 24-h period. A second season of data was collected in 2009 only for Baker Lake.

5.1.2 Results

Within these communities, the collected wastewater was place into either wastewater stabilization lagoons (sewage lagoons), into facultative lakes or discharged directly into the

wetland without pre-treatment. Yates *et al.*, (2012) found that the concentration of the $cBOD_5$ entering these systems ranged between 550-1000 mg/L, and noted that influent wastewater entering wetlands following pre-treatment in either facultative lakes or lagoons was significantly less than that of direct discharge into the wetland. Influent values observed in Whale Cove (facultative lake pre-treatment) as compared to Chesterfield Inlet (direct discharge) (Table 5.1) shows this difference. For each community, the performance of wetland measured varied for different wastewater parameters; some wetlands were much more efficient in the treatment of TP or NH_3-N than other wetlands, while TSS was particularly variable (Yates *et al.*, 2012). TSS reductions were generally high in systems where wastewater was diluted in stream and small water bodies; believing that sedimentation was a primary treatment process. This was especially noticeable in Repulse Bay and Baker Lake.

$cBOD_5$ and COD removal was observed to vary between 47-94% and 57-96%, respectively amongst the wetlands. The percent removal of COD and $cBOD_5$ and TSS were generally lower in the wetlands that received wastewater that was first pre-treated in either a facultative lake (e.g., Whale Cove) or an engineered lagoon (e.g., Coral Harbour). The reason for the lower treatment rates in these wetlands likely relates to the fact that much of the oxidative treatment and settling of TSS was occurring within the pre-treatment phase (Yates *et al.*, 2012). In all wetlands the effluent was below 25 mg/L for $cBOD_5$ and TSS, which are the effluent standards for municipal wastewater for $cBOD_5$ and TSS in southern Canada. Yates *et al.*, (2012) found that at the time of study, treatment facilities with minimal holding capacity during the winter months observed increases in $cBOD_5$ effluent concentrations during the spring freshet, such as in Chesterfield Inlet. Amongst the wetlands removal was observed to vary between 80-99% and 85-100% for TP and NH_3-N, respectively. Systems with more surface water flow obtained high levels of DO, although all achieved concentrations of greater than 8 mg/L on average in the effluent. Pathogen concentrations were generally reduced significantly, although results were variable and likely influenced by natural sources such as snow geese (*Chen caerulescens L.*) which were commonly present throughout some of the wetlands (Yates *et al.*, 2012).

Air temperature and soil temperature can significantly influence the treatment of wastewater in the Arctic and during winter time temperatures (e.g. -17° C to -32° C between November and May) no significant biological treatment occurs (Yates *et al.*, 2012). The authors also mentioned that wastewater treatment would be expected to be minimal during the spring freshet during the release of thawing wastewater that had accumulated over the winter in communities that did not have the capacity of long term storage and thus discharged to the wetland during these frozen periods. The sampling conducted in this study captured a portion of the spring freshet, which likely accounted for variation in effluent concentration of many of the parameters tested. These variations are the most prominent at the end of June during final snow melt and at the end of September following senescence and short periods of freezing temperatures.

Yates *et al.*, (2012) reported that some treatment wetlands such as Arviat and Cambridge Bay, were modified through the construction of berms to help both direct and detain the flow of wastewater through the wetland in an attempt to increase the hydraulic residence time (HRT) and thus allow more time for the microbial uptake/transformation of nutrients in the wastewater (Yates *et al.*, 2012).

Table 5.1   Mean influent and effluent data, with percent concentration change, and wetland characteristics from six tundra treatment wetlands studied in Nunavut.

Performance measured & Wetland characteristics	Coral Harbour	Arviat	Repulse Bay	Whale Cove	Baker Lake	Chesterfield
Wetland – Conc. In (cBOD$_5$ $_{mg\,L}$$^{-1}$)	181	103	385	40.3	466	221
Wetland – Conc. Out (cBOD$_5$ $_{mg\,L}$$^{-1}$)	14	16	25	21	6	14
% Reduction between in & out	92	85	93	47	99	94
Wetland – Conc. In (COD $_{mg\,L}$$^{-1}$)	308	236	450	133	798	300
Wetland – Conc. Out (COD $_{mg\,L}$$^{-1}$)	66.3	100	64.4	39.5	24	64.3
% Reduction between in & out	79	58	86	70	97	79
Wetland – Conc. In (TSS $_{mg\,L}$$^{-1}$)	93.2	55.7	197	29.4	314	74.9
Wetland – Conc. Out (TSS $_{mg\,L}$$^{-1}$)	10.5	19.1	34.8	18.0	3.2	10.3
% Reduction between in & out	88	66	82	39	99	86
Wetland – Conc. In (TP $_{mg\,L}$$^{-1}$)	5.5	11.3	9.2	4.1	13.9	5.6
Wetland – Conc. Out (TP $_{mg\,L}$$^{-1}$)	0.8	2.3	1.4	0.1	0.2	0.4
% Reduction between in & out	86	80	85	97	99	92
Wetland – Conc. In (NH$_3$-N $_{mg\,L}$$^{-1}$)	21.8	73.2	70.0	9.0	82.5	39.6
Wetland – Conc. Out (NH$_3$-N $_{mg\,L}$$^{-1}$)	2.8	11.0	2.8	0.0	0.1	0.1
% Reduction between in & out	87	85	96	100	>99	99>
Wetland – Conc. In (*E Coli* $_{mg\,L}$$^{-1}$)	3.7E4	2.9E4	14.6E6	7.5E3	16.4E6	1.3E6
Wetland – Conc. Out (*E Coli* $_{mg\,L}$$^{-1}$)	1.7E2	9.0E2	1.6E2	3.6E1	1.4E1	8.7E1
% Reduction between in & out	100	97	100	100	>99	>99
Wetland – Conc. In (TC $_{mg\,L}$$^{-1}$)	4.9E6	6.3E5	2.1E9	1.2E5	30.6E6	57.1E6
Wetland – Conc. Out (TC $_{mg\,L}$$^{-1}$)	6.9E3	4.7E3	1.9E3	2.0E2	1.1E3	7.7E2
% Reduction between in & out	100	99	100	100	>99	99>

Wetland – Conc. In (DO $_{mg\,L}^{-1}$)	3.3	1.9	1.3	10.0	0.7	1.7
Wetland – Conc. Out (DO $_{mg\,L}^{-1}$)	10.6	9.1	10.1	10.9	8.9	11.0
% Reduction between in & out	68	79	87	8	92	84
Temperature In (°C)	11.7	9.2	6.1	8.2	14.2	6.6
Temperature Out (°C)	9.7	6.3	6.2	10.7	3.2	6.2
Volume discharged (m^3/day)	96	235	66	82	167	36
Size of wetland (ha)	10	7.8	9.5	3.7	1.1	55
Approximate length (m)	650	500	1,400	900	70	700
Approximate width (m)	130	140	70	35	150	200
Water holding capacity (m^3)						

5.1.3 Discussion

The study results obtained in the harsh climatic conditions and low biomass producing ecosystems of the Canadian Arctic exemplifies the ability of natural wetlands to act as sinks and transformers of nutrients, organic material and pathogens. Yates *et al.*, (2012) noted that the exact mechanisms and processes of transformation and removal have not been identified in this study and should be examined further. The wetlands surpassed expectations for the removal of organic matter in the form of cBOD$_5$/COD, pathogens, NH$_3$-N, TP and had reasonable TSS removal, despite the lack of knowledge in processes. In all cases, removals for cBOD$_5$ were below regulatory standards for effluent in southern Canada (CCME, 2009). TSS was also below regulatory standards in southern Canada; only the Coral Harbour wetland was the exception. Pathogen concentrations were variable, which may be attributed to local wildlife populations, a common variable in natural wetlands (Yates *et al.*, 2012).

Treatment efficiencies are negatively impacted during the spring freshet in areas where large volumes of frozen wastewater have accumulated during the winter months. An

appropriate management strategy to improve treatment during the spring freshet would be to store wastewater generated over the winter period in lagoons or facultative lakes (Yates *et al.*, 2012). These lagoons should be designed as continuous flow exfiltrating systems, which slowly decant into throughout the summer months. This way, the wetlands would be able to sustain performance with lower and longer sustained loading rates, than with an annual rapid release of frozen wastewater during spring freshet or the annual end of summer decant when most plants have already begun to senesce (Yates *et al.*, 2012). Chesterfield Inlet and Baker Lake have both received larger lagoons as part of their treatment systems since the time of study by Yates *et al.*, (2012).

5.2 Environment Canada Wetland Study (2009-2011)

During the 2009 field season the CAWT investigated the treatment wetlands associated with Paulatuk (NT) and Pond Inlet (NU). The additional sites of Edzo (NT), Fort Providence (NT), Gjoa Haven (NU), and Ulukhaktok (NT) were studied in the 2010 field season. During the summer of 2011 Edzo (NT) was revisited and Taloyoak (NU) was studied for the first time. Figure 5.2 illustrates the generalized locations of these wetlands and Table 5.2 provides specific information concerning the dates of the on-site visits.

This study was undertaken to:
- Better understand the range of treatment efficiencies currently achieved at existing tundra wetlands to inform the development of a NPS for the Far North in the regulations of the Fisheries Act;
- To present these data in an interpolated format to provide a visual overview of treatment performance within the wetland, and
- Better understand correlations between effluent strength and influential factors that modulate treatment efficiencies in order to determine if there are some common insights / principles that could be used to develop better operational (design) standards for the use of tundra wetlands.

5.2.1 Methodology

The wetland surveys conducted for Environment Canada were performed in midsummer (e.g., late July to early September) for each of the three years of investigation. Each wetland was visited over a one to six days duration depending on whether the investigation undertaken was ether: i) a full survey or a reconnaissance survey and ii) the physical size of the wetland with larger wetlands requiring more time to survey. Reconnaissance surveys were employed only for Fort Providence and Edzo, both surveyed in 2010. Full surveys were conducted on all other Environment Canada wetlands including Edzo which was revisited for a second time in 2011. Reconnaissance surveys were intended to provide a rapid assessment of wetland performance and differed from full surveys in that the number of sample collections sites was reduced, along with a reduction in the number of water quality parameters investigated. The water quality parameters eliminated from the reconnaissance survey were typically those parameters such as cBOD5 or microbial samples whose analysis was time sensitive. Both survey methods provide a one-time "snap-shot" analysis of wetland performance, with most water quality parameter samples collected on one day. Thus the data generated for Environment Canada are discrete and do not represent time series data. The following provides a detailed explanation of what constituted a full survey and a reconnaissance survey.

Figure 5.2   Generalized location of treatment wetlands investigated by the CAWT for Environment Canada during the summer months of 2009 - 2011.

Table 5.2   Summary of wetland studied, by year and intensity of study.

Location	Territory	lat / long	2009	2010	2011
Paulatuk	NT		full		
Pond Inlet	NU		full		
Edzo	NT			RC	full
Fort Providence	NT			RC	
Gjoa Haven	NU			full	
Uluhaktok	NT			full	
Taloyoak	NU				full

Legend: full means full study protocol; RC means reconnaissance study protocol

Full Survey

Each wetland was surveyed using a standardized methodology developed for this investigation. In brief, transects were established to provide full coverage of the portion of the wetland believed to be actively involved in the treatment of municipal effluents. Sample locations were established at various points along each transect in a standardized manner in order to achieve the desired intensity of sample collection. The location of each collection site, along with numerous other points was geo-spatially referenced with the use of a TopCon 3105W reflectorless total station that provided not only referenced site locations but also elevation differences and the boundary of the active portion of the wetland involved in the treatment of municipal effluents. The full survey was designed to assess key water quality parameters of the effluent as it traversed the length of the wetland in order to determine the effectiveness of the wetland treatment process. In a similar manner, soil samples were collected to monitor key parameters influenced by municipal effluents in order to better assess effluent / soil interactions such as nutrient concentrations and trace elements within the soil matrix. Physical and hydrological parameters of the wetland's soil matrix, such as, grain size, porosity, hydraulic conductivity and percent moisture were also assessed to better understand the primary factors influencing subsurface flow and preferential flow of the effluent within the wetland. Lastly, the full survey also included information regarding the vegetative community structure within the wetland. The information presented in this document will summarize the major parameters such as $cBOD_5$, TSS, TAN and TP, and the hydrological and hydraulic context of the wetlands.

The establishment of transects and sample locations and the mapping of micro-topographic changes within the wetland

A thorough ground-truthing of the wetland was first undertaken by an initial walk around the site to determine point(s) where influent entered the wetland and the effluent exited the wetland and major preferential flow pathways through the wetland complex. During this time, effort was also taken to determine the approximate boundaries within the wetland believed to embody the landscape actively involved in the effluent treatment process. This information was then used to establish a series of transects at right angles to the general

flow path; beginning near the point of influence and ending near the point of effluence. The number of transects established longitudinally through the wetland was dependent on the length of the wetland assessed. The placement of transects was done in a manner that captured the primary treatment areas identified during the ground-truthing exercise. The number of sample locations along each transect was dependent upon the width of wetland that a particular transect traversed. It should be noted however, that a transect may or may not contain a surface water/ preferential flow sample point depending on the physical makeup of the wetland. Groundwater sampling locations were established approximately every 15 m.

A TopCon 3105W reflectorless total station was used in Paulatuk, Gjoa Haven, Ulukhaktok, and Taloyoak, to map wetland boundaries, elevation, and microtopographic variation within the wetland. Similar data was collected in Edzo, Pond Inlet and Fort Providence via a global positioning system (GPS, Garmin eTrex Vista HCx) since factors such as thick vegetative understory interfered with line of sight for the total station or the lack of human resources available did not permit the use of the total station. For the collection of total station data, a single base station was established to which bench marks were tied together spatially using back sighting. An individual with the prism and rod walked along transects throughout the wetland in order to accurately reference the sample locations. Surveying was conducted by running cross-sectional transects across the entire width of the wetland, making sure to capture surface and groundwater sampling locations. Transects traversed the length of the wetland and spaced approximately 30 m apart or closer in some locations. Measurements were recorded every 10-15 m across the wetland. Each point was also referenced with a global positioning latitude-longitude coordinate. At surface and groundwater sampling points used in the water quality analysis, measurements were also taken 0.25 m and 0.75 m away from the sampling point. This was conducted to help map micro-topographic changes within the wetland.

Surface Water Sampling

The surface water samples were generally collected from surface flow or preferential flow locations. Surface samples were often collected into the sample bottle by submerging the

sampling container below the water surface. However, in cases where water depth was shallow, syringes were used to minimize the disturbance of sediment that could have been collected inadvertently using the other method. Surface water samples were obtained from a minimum of 40 sample points throughout active treatment zone but could be more depending on the dimensions of the wetland. The chemical and biochemical water quality parameters surveyed in surface waters are summarized in Table 5.3 Likewise physical and ionic parameters of surface water samples are summarized in Table 5.4 with the trace elemental parameters identified in Table 5.5.

In addition, surface water temperature was recorded during the site visit with the placement of a hobo tidbit data logger into the surface water of the wetland in one location. Likewise, air temperature was also logged during this time by the suspension of one hobo tidbit data logger at one location in the wetland.

Table 5.3   Chemical and biochemical water quality parameters surveyed in surface waters collected from tundra wetlands

Ammonia (NH_3-N)	Total Phosphorus (TP)
Nitrite (NO_2-N)	Phosphate (PO_4)
Nitrate (NO_3-N)	Dissolved Organic Carbon (DOC)
Total Kjeldahl Nitrogen (TKN-N)	Dissolved Oxyen (DO)
Total Coliforms (TC)	Chemical Oxygen Demand (COD
Escherichia coli (EC)	Carbonaceous Biological Oxygen Demand - 5 Day (cBOD5)

Table 5.4   Physical chemistry and ionic parameters surveyed in surface water samples collected from tundra wetlands

Temperature	Conductivity
Total Alkalinity	pH
Hardness	Total Solids (TS)
Sulfate (SO_4^-)	Total Suspendid Solids (TSS)
Chloride (Cl^-)	Volatile Solids (VS)
Flouride (F^-)	Volatile Suspended Solids (VSS)

Table 5.5   Trace elements surveyed in surface water samples collected from tundra wetlands

Aluminum (Al)	Copper (Cu)	Rubidium (Rb)
Antimony (Sb)	Iron (Fe)	Selenium (Se)
Arsenic (As)	Lead (Pb)	Silver (Ag)
Barium (Ba)	Lithium(Li)	Sodium (Na)
Beryllium (Be)	Magnesium (Mg)	Strontium (Sr)
Calcium (Ca)	Manganese (Mn)	Thallium (Ti)
Cadmium (Cd)	Mercury (Hg)	Uranium (U)
Cesium (Cs)	Molybdenum (Mo)	Vanadium (V)
Chromium (Cr)	Nickel (Ni)	Zinc (Zn)
Cobalt (Co)	Potassium (K)	

Subsurface Water Sampling

In locations where surface waters were not present, subsurface water samples were collected with the use of a lysimeter (0.05 m diameter) constructed from polyvinyl chloride (pvc) piping which was placed into a bore hole at a maximum depth of 0.25 m or less where soils were shallow. A series of 1/8 inch holes were drilled into the lower 10 cm portion of the lysimeter which were then covered with a microfilter sheath to allow the infiltration of groundwater into the tube without the intrusion of sediment or organic matter. Sample water was collected with the aid of a sterile 60 mL syringe fitted with peristaltic pump tubing that allowed the extraction of water from deep within the lysimeter. A single

syringe/tube system was designated to a specific subsurface sampling point and corresponding lysimeter. Lysimeters were purged prior to sampling followed by the collection of a 500 mL water sample after recharge. Once completed syringes and tubing were discarded. If a new sampling period was required, then a new syringe and tube was prepared for that lysimeter.

Groundwater flow

Groundwater flow was surveyed only at the Ulukhaktok, Edzo and Taloyoak treatment wetlands. This work was undertaken to provide information regarding the subsurface flow of wastewater through the wetland and to provide in a generalized manner an ability to compare one treatment wetland with another in order to gain insight to the relative flow rates and hydraulic retention times. To accomplish this, piezometers were installed at each groundwater water quality sampling station. Piezometers were constructed from 0.02 m diameter polyvinyl chloride (PVC) pipe, with perforations for the inflow of water in the bottom 0.15 m of the piezometer tip. The length of the piezometer piping was 0.53 m. Two piezometers were installed at each collection site, with up to 50 sites per wetland. Tundra wastewater treatment wetlands have been found to be rarely greater than 0.30 m in depth. The piezometers were installed at the two different depth ranges of 0.10-0.20 m depth and 0.20-0.30 m depth. A subsurface well (lysimeter), for subsurface water quality samples was included in the cluster with the piezometers. A description of the subsurface well or lysimeter is provided in the water quality sampling section (above).

The characterization of the wetland's hydrology was further augmented with the use of soil moisture probes. This provided a rapid infield assessment of the relative closeness of the water table to the surface. At each topographic sample location, soil moisture content was assessed. A TDR soil moisture probe was used to take readings of percent soil moisture of the upper 12cm of soil. These data were used to conduct a spatial analysis of moisture gradients throughout the wetland, potentially indicating primary flow subsurface paths of wastewater passing through the wetland.

Vegetative Community Structure

A digital photograph of a 1 m^2 plot centered on each water quality sample location was taken. The dominant plant species were later determined for each plot with each of the dominant plants expressed as a percent cover of the entire vegetative cover in the photograph.

Quality Assurance / Quality Control

For all samples, HDPE bottles were used. Each filled sample bottle was labeled with sample location and Fleming College and packed in appropriate coolers together with a chain of custody form identifying each bottle and the analyses required. Coolers were packed with ice and Onset Hoboware temperature logging tidbits to record temperature variances during sample shipment. The shipment of samples may have been 24 to 48 hours in duration and therefore it was necessary to monitor the temperature to ensure sample integrity was known. The desired temperature was 4±3°C.

A series of measures were adopted to ensure that all water samples collected in the wetland had not been contaminated by poor handling, or pre-assessment contamination of sampling bottles. Nutrient parameters were also preserved with acid at the site of collection prior to being shipped for analysis.

5.2.2 Results

The results generated from the work on the seven Environment Canada wetlands are summarized below. In brief, the following text describes in a generalized manner the similarities and differences amongst the wetlands in terms of cBOD$_5$, TSS, Ammonia, microbial presence, and the hydrology of the sites. The first portion of the results section focuses on treatment performance differences amongst the wetland site. The later portion of the results section provides a summary of the major findings for each individual wetland. Many of the water quality parameters of the analyzed wetland effluent samples have also been expressed as an interpolated map for ease in visualizing the trends noticed at each wetland site. A collection of selected interpolated maps can be found in the Appendix.

Review of overall treatment performance amongst wetlands

This section summarizes the generalized treatment performance patterns evident in each of the wetlands at the time of study. The values expressed here are intended to provide an overview of the generalized performance with the understanding that these results provide only a "snap shot" overview that can vary seasonally and from year to year. However, this high level overview is helpful in understanding basic information regarding the level of treatment afforded by the wetlands and major differences amongst the wetlands.

Wetland Characteristics

Each wetland is unique in both its natural characteristics and how municipal effluents are discharged to the wetland. In some locations such as Paulatuk and Taloyoak the municipal sewage is pre-treated by disposal to a facultative lake while pre-treatment occurs within engineered lagoons at Pond Inlet, Edzo, Fort Providence, Ulukhaktok, Gjoa Haven. In addition, the effluent from the pre-treatment lagoons at Fort Providence and Pond Inlet sites are decanted, meaning that a large volume of effluent is discharged over a short period of time. This is different from the other sites where the effluent either exfiltrates through the berm wall of the lagoon or overflows through a natural drainage channel from the facultative lake with the result that the effluent is released at these sites in generally a continuous manner with smaller daily volumes than what is experienced at the sites where decanting takes place. Most wetland sites are relatively flat, allowing a slow progression of effluent over the length of the wetland. Pond Inlet is different in that the slope of the site is much steeper.

Apart from the descriptors above, wetlands can also be characterized by the associated hydraulic loading rate (e.g., depth of water applied per unit area), organic loading rate (e.g., mass of organic material applied per unit area), hydraulic conductivity (flow rate through the substrate) and hydraulic retention time (turn over time of the water volume retained in the substrate). Each of these parameters will affect treatment efficiencies by influencing how long the effluent is retained in the wetland which in dictates the length of treatment (often microbial and/or transformation and mechanical filtration). Treatment can also be

influenced by many other parameters such as the inflow of new water from melt or precipitation events (e.g., dilution), temperature, vegetation cover (areal extent and speciation), substrate grain size, composition of the effluent and so on.

Some of the more prominent wetland characteristics and hydrological parameters have been summarized in Table 5.6. These values can be used to provide an overall indication as to the major similarities and differences amongst the wetlands. In this table, the depth of the active wetland substrate has been arbitrarily set at 0.4 m in order to provide an estimate of the relative groundwater holding capacity of the wetland. Survey results generated from this study suggests that the depth of the saturated substrate level is close to this value (e.g., 0.4 m), however, it should be understood that a standardized depth has been chosen to facilitate comparisons amongst wetlands. Although the porosity of each wetland was measured, some of the measurements appear high (particularly for Pond Inlet) and may have been influenced

Table 5.6   Wetland characteristics: daily volume applied equals the annual effluent released in a 122 day period; organic loading based on cBOD5 influent concentrations.

	wetland size m^2	arbitary depth m	average porosity	water holding capacity m^3	daily vol applied m^3/d	cBOD5 influent mgL^{-1}	Hydraulic Loading cm / d	Organic Loading kg/ha·d	Hydraulic Retention d
Paulatuk	14600	0.4	0.48	2803	102	40	0.69	2.8	27
Pond Inlet	5800	0.4	0.82	1902	312	70	5.4	38	6.1
Edzo	21300	0.4	0.50	4260	325	26	0.51	1.3	13
Fort Providence	8700	0.4	n.a.	n.a.	n.a.	60	2.6	16	n.a.
Gjoa Haven	169000	0.4	0.46	31096	356	113	0.21	2.4	87
Ulukhatok	72900	0.4	0.62	18079	121	94	0.17	1.6	149
Taloyoak	61200	0.4	0.48	11750	257	80	0.42	3.4	46

by the high organic content of the soil which absorbed water and artificially inflated the corresponding pore space volume. The water holding capacity of the wetland has been generated by multiplying the wetland size by substrate depth and associated porosity. The daily volume of effluent entering the wetland is based on the annual volume of wastewater

generated by the community and dividing this value by 122 days or the number of days in June, July, August and September. It is believed that effluent enters the wetland only during periods above freezing and hence the reason for this calculation. It is not known however, how evenly this volume is distributed over the 122 day period and it is anticipated that greater flows are experienced early in the season when the thaw first begins. The hydraulic loading rate is an expression of the depth of water entering the wetland on a daily basis if this water was evenly distributed and it is one way of visualizing how flows differ amongst wetlands. The organic loading rate is based on the concentration of the $cBOD_5$ in the influent entering the wetland. The hydraulic retention time (HRT) provides a measure of how quickly the volume of water contained in the substrate of the wetland is replaced with the associated volume of influent discharged to the wetland per day. It should be noted that the HRT expressed in Table 5.6 is based on the assumption that the hydraulic conductivity of the substrate is large enough to not impede the subsurface flow rate needed to accommodate this volume. However, an evaluation of the hydraulic conductivity of the wetland substrates was found to be slow enough to like impede the total infiltration of these volumes thus resulting in overland flow. The extent and duration of overland flow is still unknown.

From the data contained in Table 5.6 it can be seen that the wetland at Gjoa Haven is the largest while the Pond Inlet site is the smallest. Pond Inlet also has the highest hydraulic loading rate with Fort Providence being having the second highest HLR. Likewise the greatest organic loading rates are occurring at the Pond Inlet and Fort Providence wetland sites. The flow of ground water through the subsurface substrate was studied more intensely at the Ulukhaktok, Taloyoak and Edzo wetland sites. Groundwater flow through these wetlands was estimated using the Darcy equation.

$$Q=K_h d_s (dh/dx)$$

Where Kh is the hydraulic conductivity, which was estimated by conducting pumping tests at each of the piezometers using the methods described by Luthin (1966).

The hydraulic conductivities for the Ulukhaktok site ranged from 1.20 m/d to 1.24×10^{-3} m/d. The term *dh/dx* is the hydraulic gradient estimated by the elevation of the water table at a transect in relation to the adjacent transects. The d_s is the cross sectional area of the saturated zone, which is the elevation difference between the permafrost layer and the water table. The depth of the permafrost layer was considered the deepest point at which the drivepoint piezometers could penetrate to.

The hydraulic conductivity of the Ulukhaktok wetland is highest in the vicinity of the lagoon berm. The average hydraulic conductivity of the soils in Transect 1, closest to the lagoon, is 7.98×10^{-2} m/d, considerably higher than the 3.9×10^{-2} m/d averages of the remaining 7 transects. As expected, the hydraulic conductivity of the wetland decreases with depth. The average hydraulic conductivity of the soils surrounding the shallow piezometers was 6.38×10^{-2} m/d, whereas hydraulic conductivity of the soils surrounding the deep piezometers was 2.34×10^{-2} m/d. The result is that flow rates are substantially higher in the upper portion of the saturated layer. The piezometer data shows that the water table is perched in at multiple locations in the south-eastern part of the wetland. The areas where this perching occurs are adjacent to locations with extremely low hydraulic conductivities, which is to be expected. At sites where there were both deep and shallow piezometers (and both contained water), it was possible to estimate if water was recharging or discharging at that particular location. All but three nests locations showed hydraulic gradients that indicated a downward movement of water. Three nests (2D, 5C and 6C) had gradients that would indicate upward discharge of water indicating that the flow of water through the wetland subsurface matrix was complex. Overall water flow through the subsurface saturated layer of the wetland is limited. The low hydraulic conductivities and low hydraulic gradients in the wetland result in estimated groundwater flows ranging from 0.00001 m^3/day per unit width to 0.0004 m^3/day per unit width.

The hydraulic conductivity values for Taloyoak were similar to Ulukhaktok in that the upper portion of the substrate (shallow) was slightly faster flowing (0.0035 m/d) in comparison to the deeper layers of the substrate (0.00082 m/d). Overall the hydraulic

conductivity range within the Taloyoak substrate varied between 0.013 m/d to 9.76 X 10^{-6} m/d. The hydraulic conductivity at the Edzo wetland was found to be slightly faster flowing in the deeper portions of the substrate (0.0054 m/d) than the upper portions (0.0035 m/d). Overall the hydraulic conductivity values at the Edzo site ranged from 0.016 m/d to 3.46 X 10^{-5} m/d. Hydraulic conductivity measurements were conducted on composite soil samples taken from the Paulatuk and Pond Inlet wetland sites. These soils were first oven dried and later the soil clod were gently broken and placed into a static head permeameter. The resulting values were 1.81 m/d for Paulatuk and 1.04 m/d for Pond Inlet indicating that in general terms the hydraulic conductivity of Paulatuk was greater than that of Pond Inlet.

Carbonaceous Biochemical Oxygen Demand (5 day)

An overview of the general treatment performance has been summarized in Table 5.7. For comparative purposes, this table indicates the $cBOD_5$ target identified in the hamlet's water licence (at the time of study) and an approximate $cBOD_5$ concentration entering the wetland and exiting the wetland in order to provide an approximate indication of overall treatment expressed as a "% reduction". For example if the $cBOD_5$ concentration entering the wetland is 40 mg L^{-1} and exiting the wetland this concentration has decreased to 2 mg L^{-1}, then this is expressed as a 95% reduction, or the $cBOD_5$ concentration has been reduced by 20 times.

The concentration of the $cBOD_5$ leaving the wetland is then compared to the CCME national performance standard of 25 mg L^{-1} set for southern municipalities. It is understood that national performance standards have not yet been determined for northern communities; however, this southern standard is being used for comparative purposes in order to assess the relative ability of the wetlands in the treatment of this parameter. This comparison is expressed as a percent value. For example if the $cBOD_5$ exiting the wetland is 2 mg L^{-1} then this value is lower than the NPS of 25 mg L^{-1} and is expressed as approximately 8% of the NPS. If however, the $cBOD_5$ exiting the wetland is 50 mg L^{-1}, then the percentage is 200% meaning that the value is twice as great as the NPS of 25 mg L^{-1}.

The treatment performance is further summarized by providing a generalized indication as to where in the flow path of wastewater through the wetland the incoming $cBOD_5$ concentration is reduced by half (e.g., 50% reduction) and when the $cBOD_5$ reaches an approximate steady state. These generalized zones of reduction are expressed as a percentage of the total distance the effluent travels through the wetland. For example, if the total length of the wetland is approximately 100 m and the $cBOD_5$ concentration is reduced by half (e.g. 50% reduction) after traveling the first 30 m into the wetland then this values is expressed as "30% of the total distance of the wetland's length is required to achieve a reduction of 50% in the strength of the wastewater parameter". Likewise, if the $cBOD_5$ values are generally stable for the last quarter of the wetland's length, then this is expressed as "75% of the total distance of the wetland's length is required to achieve steady state". Note that the steady state can vary amongst wetlands and therefore it does not represent a standardized performance value.

Table 5.7   Overview of wetland treatment performance in the reduction of cBOD5

Performance Measure	Paulatuk	Pond Inlet	Edzo	Fort Providence	Gjoa Haven	Ulukhaktok	Taloyoak
Water Licence Conc. (BOD5 mg L^{-1})	100	100	30	100	80	120	unknown
Wetland - Conc. In (cBOD5 mg L^{-1})	40	70	26	60	113	94	80
Wetland - Conc. Out (cBOD5 mg L^{-1})	2	50	2	32	2	5	25
% Reduction between in & out	95	29	92	47	98	95	69
% of NPS for cBOD5 (25 mg L^{-1})	8	200	8	128	8	20	100
% Wetland length to achive 50% reduction	25%	***	50%	***	20%	30%	50%
Wetland length to achieve 50% reduction (m)	70	***	150	***	150	160	240
% Wetland length to achive steady state	50%	***	85%	***	60%	80%	70%
Wetland length to achieve steady state (m)	140	***	260	***	450	425	335
Size of Wetland (m^2)	14600	5800	21300	8700	169000	72900	61200
Approximate length of wetland (m)	275	250	300	160	750	530	480

Legend: *** means that <50% reduction was achieved

In all wetlands, except for Pond Inlet and Fort Providence, a 50% reduction in $cBOD_5$ was achieved within the first half (e.g., inlet side) of the wetland and in the case of larger wetlands such as Gjoa Haven and Ulukhaktok this reduction was achieved within the first 20 to 30 percent of the wetland. All wetlands met or exceeded the NPS of 25 mg L^{-1} 'for $cBOD_5$ except for Pond Inlet and Fort Providence. The $cBOD_5$ removal rates were even greater in Taloyoak than what is shown in Table 5.7 when sampled from the stream exiting the Taloyoak wetland. The $cBOD_5$ concentrations of approximately 2 - 3 mg L^{-1} are observable within the appended data tables for this stream location. However, this stream area was not considered in the mapping of the Taloyoak wetland and is therefore not incorporated into the interpolated maps for Taloyoak. The $cBOD_5$ treatment appears to be poorer in areas associated with high moisture content, or in areas of standing water, particularly in the Taloyoak wetland. The underlying cause for this association is unknown, but may be related to a greater influence of surface water flow and less subsurface flow which may in turn be related to shorter HTR at those wetter locations. This is however, only speculation at this point.

The poor performance of both Pond Inlet and Fort Providence may be related to their relatively smaller size. The steep slope of the Pond Inlet wetland facilitates a rapid travel of the effluent down the slope leading to the assumption that the HRT at Pond Inlet is very short. In the case of the Fort Providence wetland it was assessed in 2010 during the decant period and likely a major factor as to why the treatment efficiency was poorer. In addition it should be noted that the the surface water of the adjacent *Typha* marsh was not included as part of the wetland boundary due to the depth of the water which made sample collection difficult. However, this wetland marsh should have likely been included as part of the wetland. It is anticipated that the samples collected from this site would likely have had significantly lower $cBOD_5$ concentrations. Future investigations of the Fort Providence wetland should include the *Typha* marshland. These results also suggest that better treatment may be achieved if the rate of the lagoon decant was slower, allowing for a greater time for wetland treatment.

Total Suspended Solids

  The current water licence agreements for total suspended solids amongst the eight sites varies from a low of 35 mg L^{-1} for Edzo to a high of 180 mg L^{-1} for Ulukhaktok. Individual limits are summarized in Table 5.8. The pattern of TSS is variable within and amongst most wetland sites and it is therefore difficult to assess how well individual wetlands are performing in the removal of TSS. In some locations there appears to be an association between higher TSS values and higher relative soil moisture content. This association may be due in part to how the samples were collected. Water samples in drier areas were, for the most part, collected from sampling wells inserted into the wetland that provided access to subsurface water samples and hence may naturally contain less TSS because of the filtering process occurring with subsurface flow. In wetter locations surface waters were often collected. Wetter locations tended to be in lower points of elevation and located nearer the outflow of the wetland. It is also suspected that the wetter locations may be more influenced by surface flow and thus could be one reason why in some wetlands the concentration of TSS increases rather than decreases as the effluent traverses this area.

  A further subdivision of TSS into its components of Fixed Suspended Solids (FSS) and Volatile Suspended Solids (VSS) can be used to provide an indication of the organic component of TSS. Municipal effluents tend to be high in organic content in comparison to the inorganic fraction. Graphing the ratio of VSS to FSS and monitoring the change in this ratio can at times provide a better distinction between the portion of TSS originating from municipal effluents and the portion that could be a natural constituent of the site. Volatile suspended solids were analyzed for all sites except for Paulatuk and Fort Providence.

  A review of the ratio of VSS to FSS for the Pond Inlet wetland (Figure 5.3) reveals that the VSS remains relatively constant and thus suggests that this wetland was poor at removing suspended solids from the wastewater effluent. The poor removal of $cBOD_5$ supports the conclusion that relatively little treatment of the effluent in terms of carbon removal was occurring.

A similar graph of VSS to FSS for the Taloyoak wetland shows a changing ratio where the percentage of VSS (the organic portion) decreases with travel length through the wetland suggesting that the wetland is effective in the removal of TSS originating from municipal effluents (Figure 5.4). The trend for the ratio of VSS to FSS in the remaining wetlands is more variable and difficult to interpret as evident in Gjoa Haven and Ulukhaktok (Figure 5.5 and Figure 5.6, respectively). There does not appear to be a clear trend of decreasing VSS with increased travel through the wetland. It should be understood however that VSS is determined by measuring the mass loss after ignition at 550°C.

Table 5.8   Summary of current water licence compliance targets for total suspended solids

Performance Measure	Wetland Site	Paulatuk	Pond Inlet	Edzo	Fort Providence	Gjoa Haven	Ulukhaktok	Taloyoak
Water Licence Conc. (TSS mg L^{-1})		120	120	35	120	100	180	unknown

Overall, the use of TSS as an indicator of treatment performance in wetlands is likely a poor choice for compliance testing. It appears that in some cases, wetlands can be a generator (not a sink) for TSS. Generation of TSS can occur through erosional forces within the wetland, particularly during melt events where the velocities of surface water flows may be strong enough to transport inorganic fines. In other situations wetland may contribute organic constituents to surface flows from the decomposition of plant matter. If

TSS is to be used as a treatment indictor, then it is suggested that the sample be analyzed to determine the proportional composition of organic to inorganic matter (e.g., the ratio of VSS to FSS) in order to gain better insight into how the wetland is functioning in regards to the removal and or addition of these constituents.
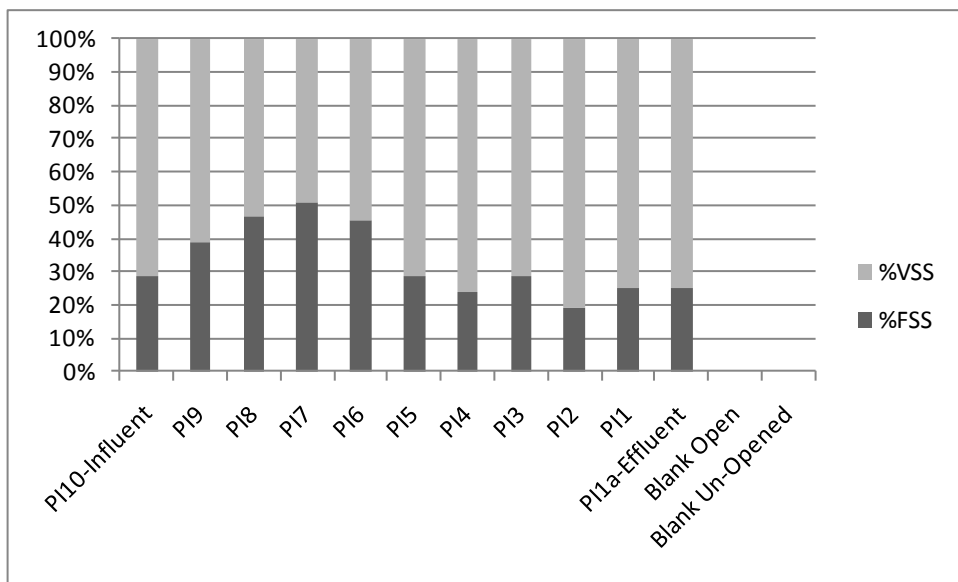
Figure 5.3   Ratio of VSS to FSS at the Pond Inlet wetland. Effluent direction through the wetland is from left to right.
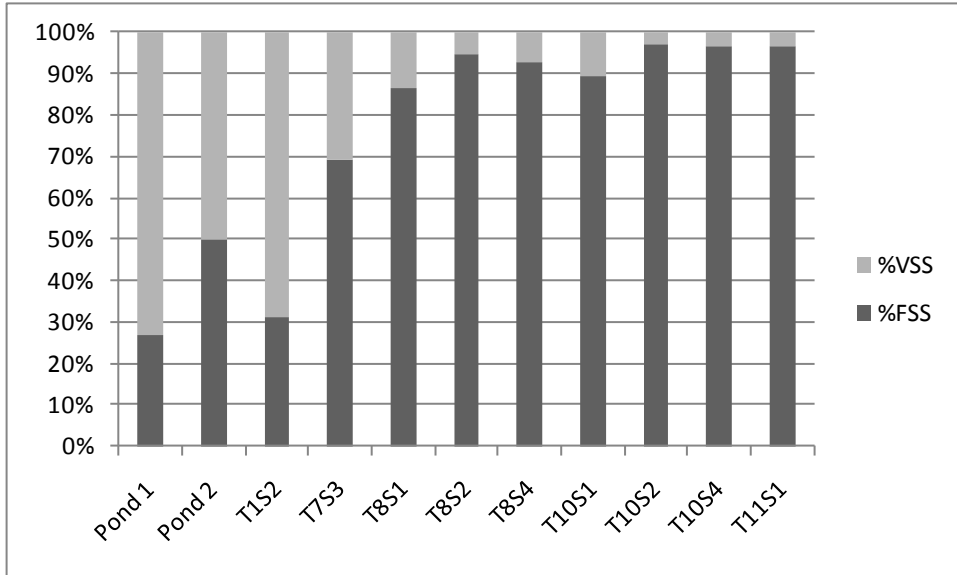
Figure 5.4   Ratio of VSS to FSS at the Taloyoak wetland. Effluent direction through the wetland is from left to right.
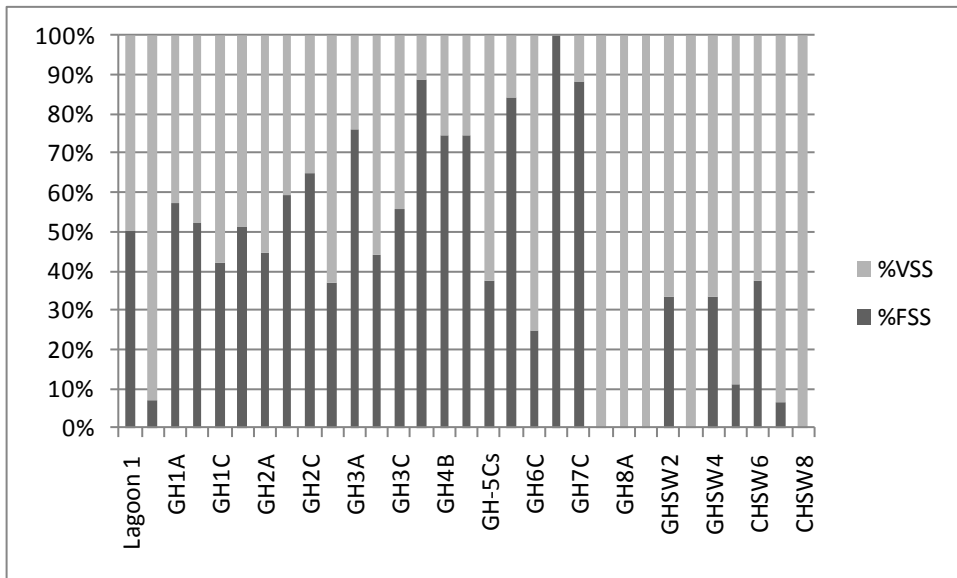
Figure 5.5   Ratio of VSS to FSS at the Gjoa Haven wetland. Effluent direction through the wetland is from left to right.
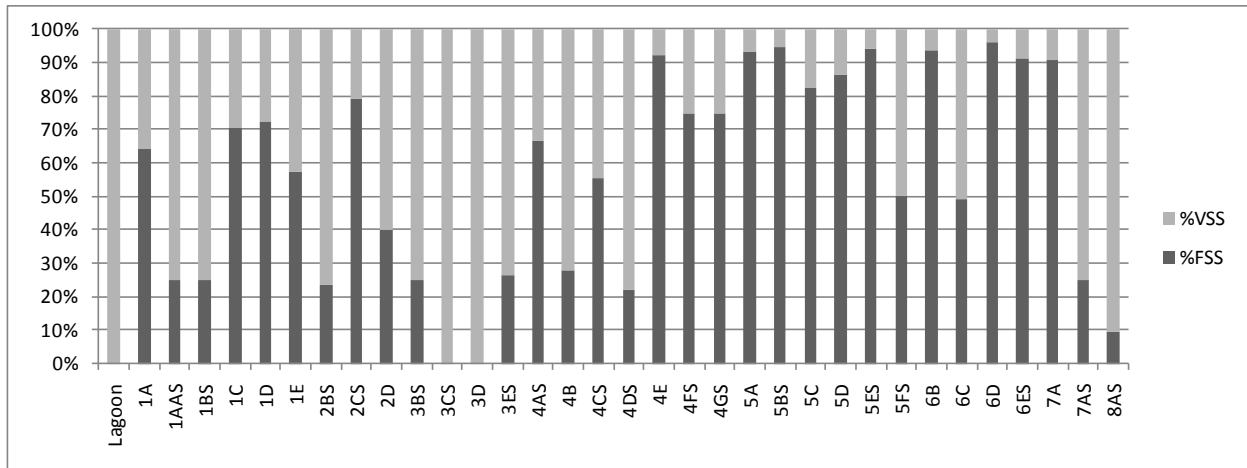
Figure 5.6 Ratio of VSS to FSS at the Ulukhaktok wetland. Effluent direction through the wetland is from left to right.

Ammonia

Ammonia can exist in both an un-ionized form (NH_3) and an ionized form (ammonium NH_4^+). The proportion of these two forms is both pH and temperature dependant with higher percentages of NH_3 favoured with higher pH values. The un-ionized form (NH_3) is toxic to aquatic life forms and as such CCME has set a national performance standard for the concentration NH_3 (measured as N) at 1.25 mg L^{-1} for southern treatment plants. A NSP guideline for northern communities is currently under review.

The ammonia concentrations expressed in this report are expressed as the concentration of nitrogen measured in NH_3. This is written as NH_3-N. However the nitrogen measure from the NH_3 form does not accurately represent the toxic form of NH_3 found in the original environmental sample. The effluent sample in its natural state would contain a fraction of both the un-ionized form (NH_3) and the ionized form (NH_4^+). The effluent sample is analyzed under a basic environment which forces all of the NH_4^+ into the NH_3 form. Thus what is expressed in the value NH_3-N is actually the nitrogen from both the un-ionized and

ionized forms. Therefore a better expression of this value is a term called "total nitrogen ammonia" or TAN for short. In order to approach the NPS of 1.25 mg L^{-1} of the toxic un-ionized form (NH$_3$), a "total nitrogen ammonia" (TAN) concentration of approximately 100 mg L^{-1} in an environment with a pH of 8 and a temperature of 5°C would be needed.

All TAN values (expressed in this manual as NH$_3$-N) were well below the 100 mg L^{-1} example provided above. The highest concentration of TAN (e.g., 76 mg L^{-1}) was found entering the wetland at Gjoa Haven. The pH of the wetland effluent was less than 8 at all sites except for some locations within the Taloyoak wetland (pH range: 7.1 to 8.6), however, the TAN concentrations at the Taloyoak were below 5 mg L^{-1}. Because of either the relatively non-basic pH values in the wetland or the low TAN concentrations it can be surmised that the concentration of the un-ionized toxic NH$_3$ would be well below the threshold of 1.25 mg L^{-1} in all areas of the wetland including the effluent exiting the wetland to the receiving water body.

The removal of ammonia was efficient in all wetlands with the exception of Pond Inlet and Fort Providence. Once again, the steep slope and the anticipated short hydraulic retention time at Pond Inlet likely contributed significantly to the poor ammonia removal. The poor performance at Fort Providence may have been related to the fact that this wetland was surveyed during the lagoon decant period and so flows would have been higher than normal with the inflow of fresh effluent. Ammonia removal in all other wetlands was generally ninety percent or greater (Table 5.9) and strongly suggests that the release of NH$_3$ would be well below the 1.25 mg L^{-1} NPS established for southern Canada.

Table 5.9   Overview of wetland treatment performance in the reduction of total ammonia nitrogen (TAN) expressed as NH3-N

Performance Measure	Wetland Site	Paulatuk	Pond Inlet	Edzo	Fort Providence	Gjoa Haven	Ulukhaktok	Taloyoak
Water Licence Conc. (TAN mg L^{-1})		not set	not set	not set	not set	not set	not set	not set
Wetland - Conc. In (TAN mg L^{-1})		3.2	75.4	16.1	26	76.4	9.6	4.6
Wetland - Conc. Out (TAN mg L^{-1})		0.01	31.6	0.31	18	1	0.1	0.13
% Reduction between in & out		100	58	98	31	99	99	97
% Wetland length to achive 50% reduction		33%	100	50%	***	30%	20%	30%
Wetland length to achieve 50% reduction (m)		90	250	150	***	230	100	150
% Wetland length to achive steady state		70%	***	60%	***	60%	50%	70%
Wetland length to achieve steady state (m)		190	***	180	***	450	270	335
Size of Wetland (m^2)		14600	5800	21300	8700	169000	72900	61200
Approximate length of wetland (m)		275	250	300	160	750	530	350

Legend: *** means that <50% reduction was achieved

Microbial Indicators

When microbial indicators are stipulated for compliance purposes, most water licences base treatment on the reduction of fecal coliforms. For this investigation, total coliforms and E. *coli* were the primary microbial indicators surveyed in all wetlands except for Pond Inlet were microbial parameters were not monitored and in Paulatuk where fecal coliforms were monitored in addition to total coliforms and *E. coli*. Microbial organisms by nature are very heterogeneous in their distribution and thus sample results often reflect high variability. As such, any results generated from this study must be understood as providing only a snap shot of the conditions on that particular sample day with the realization that no attempt was made to quantify the variability in microbial densities at a particular sample locations. Thus the results can at best provide only a generalized impression of treatment efficiencies. The expression of these data into interpolated maps has been challenging since outliers caused from either natural variability or perhaps sample contamination at the time of collection can skew the resulting maps. It should also be understood that *E. coli* is a

subset of the microbial density that is normally captured when monitoring the larger body of organisms categorized as fecal coliforms. However, the *E. coli* data does provide an indication of trends in the overall reduction of *E. coli* that likely reflect similar trends that would have been noticed in fecal coliforms should these organisms had been monitored. An overall comparison of the microbial removal by wetland has been provided in Table 5.10.

The information presented in Table 5.10 is intended to provide a high level overview of treatment performance that will allow some generalized trends to be identified amongst the treatment wetlands. This information should not be used for generating specific values since all values presented are rough approximates, particularly when describing approximate lengths within the wetlands associated with a percent reduction in bacterial counts.

  Overall, the densities of *E. coli* exiting the wetlands are below the densities stipulated in the water licences for fecal coliforms within all wetlands. It is understood that *E. coli* is only a subset of fecal coliforms and had fecal coliforms been measured, then the microbial densities exiting the wetlands would likely have been higher. In most cases the log reduction of *E. coli* was approximately 2 or greater except for Fort Providence where it was less than 1 log unit.

Table 5.10   Overview of wetland treatment performance in the removal of microbial organisms

Performance Measure	Paulatuk	Pond Inlet	Edzo	Fort Providence	Gjoa Haven	Ulukhaktok	Taloyoak
Water Licence (Fecal Coliforms - CFU/100mL)	$1x10^4$	$1x10^6$	$1x10^3$	$1x10^5$	$1X10^3$	$1X10^5$	unknown
Wetland - In (E. *coli* - CFU/100mL)	2850		9090	2480	408000	9210	1300
Wetland - Out (E. *coli* - CFU/100mL)	1		1	990	300	1	24
Approximate log reduction	3		4	<1	3	4	1.5
% Reduction between in & out	100		100	60	100	100	98
% Wetland length to achive 50% reduction	20%		50%	80	10%	50%	75%
Wetland length to achieve 50% reduction (m)	50		150	130	50	270	360
% Wetland length to achive steady state	40%		90%	***	20%	60%	85%
Wetland length to achieve steady state (m)	110		270	***	100	320	400
Size of Wetland (m^2)	14600	5800	21300	8700	169000	72900	61200
Approximate length of wetland (m)	275	250	300	160	750	530	480

Legend: *** means that <50% reduction was achieved

Trace elements

The concentration of trace elements in the effluent samples exiting from the wetland is for the most part below the Canadian Water Quality guidelines for the protection of aquatic health. Where elevations are occurring within the wetlands, the primary elements seen above water quality guidelines include iron > copper > zinc. The elevated occurrence of arsenic, chromium and cadmium are less frequent. At the Paulatuk and Edzo sites, iron and copper where the only two elements found in the effluent exiting the wetland.

5.2.3 Discussion

The water quality data for each of the seven sites studied for Environment Canada are presented in the form of interpolated maps in Appendix C of this manual. Note that the raw

data files for the interpolated maps are contained in Appendix D. The interpolated maps for most wetlands represent only the more prominent parameters.

  The findings from the study funded by Environment Canada indicate that at the time of investigation all wetlands, with the exception of the Pond Inlet and Fort Providence sites, were reducing $cBOD_5$ sufficiently to meet the CCME NPS of 25 mg L^{-1}. Some wetlands like Paulatuk, Edzo, Gjoa Haven and Ulukhaktok were able to lower the values to less than 10 mg L^{-1}. This likely was influenced by both the larger size of the wetlands and the correspondingly lower organic loads entering these sites. Likewise all wetlands with the exception of Fort Providence and Taloyoak (note: Pond Inlet not assessed) were able to achieve a 2 log or greater reduction in *E. coli* counts. All the wetlands studied were, however relatively poor in the removal of total suspended solids. This may be due in part to the phenomenon that wetlands can both remove and generate their own suspended solids, and in particular the organic portion (VSS). Thus it becomes difficult to interpret the TSS findings because the current assessment performed by laboratory methods do not distinguish the portion of TSS originating external to the wetland (e.g., municipal influent) and the TSS generated by the wetland itself. Although the study monitored total ammonia nitrogen (TAN) and not the toxic un-ionized NH_3-N, the values of TAN were low enough in all effluents exiting the wetland to ensure that the concentration of the un-ionized fraction would be well below the NPS of 1.25 mg L^{-1} NH_3-N. Trace elements within effluent samples taken from the wetland were generally below the Canadian water quality guidelines for the protection of aquatic health. Iron, copper and zinc were slightly elevated at or near the discharge of some wetlands. At this stage it is not known if the concentration seen in the effluent samples is a reflection of metal concentration within the sediment portion of the wetland and reflects a buildup of these trace elements above what would be found in nearby reference locations. More study would be needed to be able to better understand the long term impact of land disposal of municipal effluents to these areas.

  The data were analyzed to determine if there was an association between the concentration of COD and BOD_5 and $cBOD_5$. If a relationship was found it was speculated that COD may be able to serve as a proxy parameter for the evaluation of BOD_5 and/or

cBOD$_5$. The ability to use a proxy parameter such as COD would prove valuable since the COD sample can be preserved in the field and thus does not have the sample time constraint regarding sample shipment to an analytical laboratory. However, no observable relationship was observed for any of the wetland sites.

Analysis of the data indicated that performance of Pond Inlet and Fort Providence was distinctly poorer than the other five sites. Interestingly the effluent transfer from the pre-treatment lagoons to the wetlands was through decanting. The release of effluent into all other wetlands was via a slower but continuous exfiltration through leaky berms or continuous release from facultative lakes during the frost free season suggesting that better treatment can be achieved when releases are slow and continuous. The impact to treatment process caused by a sudden discharge related to a decant event or spring freshet is unknown. It is anticipated that higher flow volumes can decrease HRTs and increase organic loading to the point that the treatment system is overwhelmed and treatment efficiency decreases. Decanting of lagoons is also typically done at the end of the summer period therefore wetlands have less time to assimilate the nutrients and other pollutants.

A four month summer period (e.g. 122 days) was arbitrarily chosen to represent the average period of time when wetlands could be expected to be unfrozen and when effluents would be expected to be flowing through the wetlands. It is anticipated that the extreme cold during winter months would freeze the municipal effluent exiting the pre-treatment lagoons and effectively stop all treatment by the wetland. The study was for the most part conducted in late summer and so little is known about the effluent flow volumes and concentrations occurring in the early part of summer shortly after the thaw and when effluents started to once again flow to the wetlands. In some cases it may be that the flow volumes overwhelmed the wetland's treatment capacity and yet it could also be that there is such a dilution effect occurring from the melt of winter ice and snow that effluent concentrations were lower than normal. More study is needed in the early part of summer shortly after the thaw in order to better understand the influential conditions occurring at that time.

The mechanisms of action operative in the treatment of effluent are different for waters travelling subsurface and that portion flowing overland. This study attempted to understand the hydraulic dynamics within the wetlands by investigating the hydraulic conductivity and pore size of the sediments as an indirect method to determine the hydraulic retention time for the effluent. The result from these efforts had indicated that flow through the subsurface was quite slow and suggested that a significant portion of the effluent may be traveling overland in preferential flow paths. The proportion of effluent flowing subsurface in comparison to surface has been difficult to assess. For example, the piezometer work undertaken at the Ulukhaktok site would suggest that the vast majority of the effluent volume should be traveling as overland flow, however, at this site, no overland flow or preferential pathways were observed as there was also no observable outflow point where effluent was exiting the wetland. This led to the speculation that most of the effluent loss from the site might be occurring through an evaporation process. However, some debris associated with wastewater was found well into the middle of the wetland area suggesting that higher flow must have occurred at some point in time to have carried the debris this far. This observation once again indicates that more study is needed to better understand the seasonality of wastewater flow.

One of the challenges encountered during the interpretation of the data was identifying a representative exit point for the effluent. In many cases the inflow of effluent to the wetland was diffuse. Exfiltration from the lagoon berm is often variable both in location and the volume of flow. It is common to find preferential channelling or ponding occurring while some of the inflow may also be entering the wetland subsurface and therefore not easily seen. This makes it difficult to directly measure the daily hydraulic loading of the wetlands. Likewise monitoring the outflow can also be challenging. In some situations such as Taloyoak, the wetland flow (at least the overland flow portion) was funnelled into a small stream exiting the wetland; however in other wetlands the exit of surface waters can be more diffuse making it difficult to determine if the sample location chosen is the best site to represent treatment efficiencies. The lack of a clearly defined effluent exit point can make it challenging from a regulatory perspective unless clearly marked static sample locations are chosen and agreed upon in advance.

Despite the challenges and the unknowns associated with wetland treatment sites, it appears that wetlands are significantly improving the water quality of the effluent beyond what is being achieved during the primary treatment process. Wetlands appear to play a critical role as a key component within a hybridized approach to the treatment of municipal effluents in the north that utilizes lagoons or facultative lakes for pre-treatment and storage, and wetlands for secondary treatment during the frost-free season.

5.3 Summary and future research directions

Despite the global wealth of knowledge regarding the use of constructed wetlands for the treatment of municipal wastewater, a review of the published literature suggests that there is a considerable lack of knowledge regarding the overall understanding of natural treatment wetlands in the Arctic (Yates *et al*., 2012; Kadlec and Wallace, 2009; Vymazal, 2011). Current understanding of Arctic systems generated from research conducted by the CAWT and published by Yates *et al.,* (2012) has shown that wetlands which received continuous exfiltrates from lagoons or facultative lakes, despite cold ambient air temperature, permafrost soils, minimal soil depth, and a growing season of as little as two months achieved or exceeded performance standards set for southern Canada. However, there are a number of specific research needs that need to be fulfilled as we try to determine appropriate performance standards for wetland treatment systems in the Canadian Arctic.

A greater understanding of the complex roles that the hydrological conditions and biogeochemical interactions play in the overall treatment performance of tundra wetlands is needed in order to better apply this technology to cold climate natural tundra regions. For most natural wetland sites, site specific information regarding subsurface and surface flow is generally lacking. The volume of wastewater entering the wetland can be estimated from the volume of waste hauled to the site; however, determining flow volumes exfiltrating from the lagoon berm and how much of this flow travels overland and what portion travels subsurface is difficult. Determining the volume of new water entering the wetland either via

surface or subsurface flow and how this might influence wastewater strength through dilution is also difficult.

The exact location of the wetland outlet is not always obvious for natural wetlands. Water quality can change dramatically in short distances from dilution from non-effluent watershed contributions and other factors. Additionally, the wetland outlet point location can 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.

The limited amount of information generated from these studies suggest that the level of treatment may vary seasonally, and particularly during the spring freshet when subsurface soils are still frozen and the wastewater that has accumulated over the winter time on top of the wetland surface begins to melt. At this point, there is a lack of understanding regarding early season variability and ability to identify which wetlands are at risk to being overwhelmed by high organic loadings and which wetlands have the capacity to assimilate high spring time loadings. Similarly, the effects of dilution from spring meltwater are unknown. Dilution from meltwater may mitigate concentration based effects.

Tracer testing to determine site-specific HRTs should be conducted at strategic times during the spring freshet or in conjunction with the highest effluent discharge period for decanted systems. Furthermore, the permafrost melt rates and depths of active layers of the subsurface areas receiving effluent would be an important parameter to characterize, especially in relation to HRT.
Further research on the effluent effects on arctic vegetation in the wetland treatment areas is required. Eventually, determination of the threshold HLRs to avoid detrimental effects to the native vegetation would be useful for design purposes.

Since monitoring treatment performance data collected in wetlands is limited, it is rare to have comparable data from one year to the next. As a result, little is known regarding how treatment efficiencies may vary and what factors influence this variability. In the Arctic, this is especially true where climate change is expected, and already is experiencing drastic changes. With increases in mineralization rates of organic matter and nutrients and increases in plant biomass, treatment periods would likely become longer, and performance would only improve (Yates *et al.*, 2012). However, such changes would also require changes in the management strategies, because of changes in the hydrological regime, eutrophication downstream and prolonged increases in pathogens that may have human and ecosystem consequences given the current management of several treatment systems (Yates *et al.*, 2012).

The fulfillment of these research needs would help significantly in both interpreting the results and predicting how the wetland would perform under different organic loading regimes. This would facilitate informed, ecologically responsible and safe incorporation of natural tundra wetlands into the overall northern wastewater management. 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 programs and associated funds.

6.0 Predictive tools

6.1 Treatment wetland design models

The increase use of wetlands for wastewater treatment together with increasingly stricter water quality standards is an ever growing motive for the development of numerical models

to be used as predictive process design tools. The main objective of the modeling effort was to increase the predictive insight into the functioning of complex treatment wetlands through the use of process or mechanistic based models that describe in detail transformation and degradation processes (Langergraber et al, 2009). Once reliable numerical models are developed and validated against experimental data, they can be used for evaluating and improving existing design criteria. Most of the literature on models refers to simple first-order decay models (e.g. Stein *et al.,* 2006; Rousseau et al., 2004) or describes the treatment wetland as a black box (e.g. Tomemko *et al.,* 2007; Pastor et al., 2003) acknowledging only a limited understanding of the studied facility. The number of mechanistic or process based models is limited.

There are currently a variety of approaches that can be applied to predict the future capacity of constructed wetlands. However, the options available for modeling the performance of natural tundra wetlands are limited. This is primarily due to the fact that these natural wetlands are not engineered and because of this much less is known regarding media depth, flow rates, the influence of preferential flow paths and infiltration of surface or ground waters and many other characteristics required for model input. The options currently available generally include relatively simple design models like "rules of thumb" and regression equations along with first-order kinetic models or sophisticated 2-dimensional or 3-dimentional models. The best approaches are likely those that incorporate site specific performance data into the model in an attempt to calibrate the model to an individual wetland.

Regardless of whether the wetland is constructed or natural, these tools are needed by on-site managers as well as consulting engineers, regulatory agencies, municipal planners and territorial water boards to allow them the ability to validate a technology and predict future needs as communities expand and regulations change. This chapter discusses some of the more common methods used to determine what the optimal wetland size is for the current and future volume of sewage to be treated.

6.2 Sizing calculations for existing and new sites

There are a variety of methods that can be used to determine the approximate size a wetland must be in order to effectively improve the water quality of the influent (e.g., Domestic sewage) before it is released to the environment. The Canada - wide strategy for municipal effluents prepared by the Canadian Council of Ministers of the Environment (CCME) addresses four water quality parameters (BOD, TSS, ammonia and total residual chlorine).

Sizing tools can be generally categorized under the following headings:

- Rules of thumb (sometimes also called scaling factors)
- Regression equations and loading charts
- Simple first order kinetic models (e.g., $k – C^*$ model)
- Variable - order, mechanistic or compartmental models (e.g., SubWet 2.0) and sophisticated 2D and 3D models (e.g., HYDRUS, WASP, TABS-2, STELLA)

In general terms, the rules of thumb methods contain the greatest amount of uncertainty and thus are often used primarily as a "first-cut" estimate of wetland size. The variable - order and compartmental models can provide the most precise measurements, but their use is often hampered by the need for a large data set of site specific information which often does not exist or is not easily obtained. Without the calibration of these models to the specific conditions of the site, the results can be quite inaccurate. Figure 6.1 provides an overview of the strength and weaknesses for each major predictive tools category.

6.2.1 Rules of Thumb

Rules of thumb, which are sometimes referred to as "scaling factors", are based on observations from wetlands that exhibit a wide range of climatic, vegetative and physical conditions and water quality types. These generalized observations can be used to predict the behaviour of certain water quality parameters in relation to different physical components of the wetland. From an engineering perspective, rules of thumb are the easiest and fastest method for determining the approximate size needed for a wetland in order to

achieve generalized water quality parameters or treatment. Since the rules of thumbs have evolved from a wide range of conditions, they can only be used at best as a very rough approximation. In fact, Rousseau *et al.* (2004) has suggested that they are best used as a method for validation of other more sophisticated sizing methods such as the first order kinetic model k-C* or other variable-order or compartmental type models.

Rules of thumb methods are generally based on either the speed at which the sewage traverses the wetland (e.g., hydraulic retention time), the volume of water entering the wetland per unit size of wetlands (e.g., hydraulic loading rate), the mass of organic loading that is being applied per unit of wetland (e.g., organic loading rate) or are composed of a generalized set of observations that have been compiled over the years from a wide range of wetland conditions.
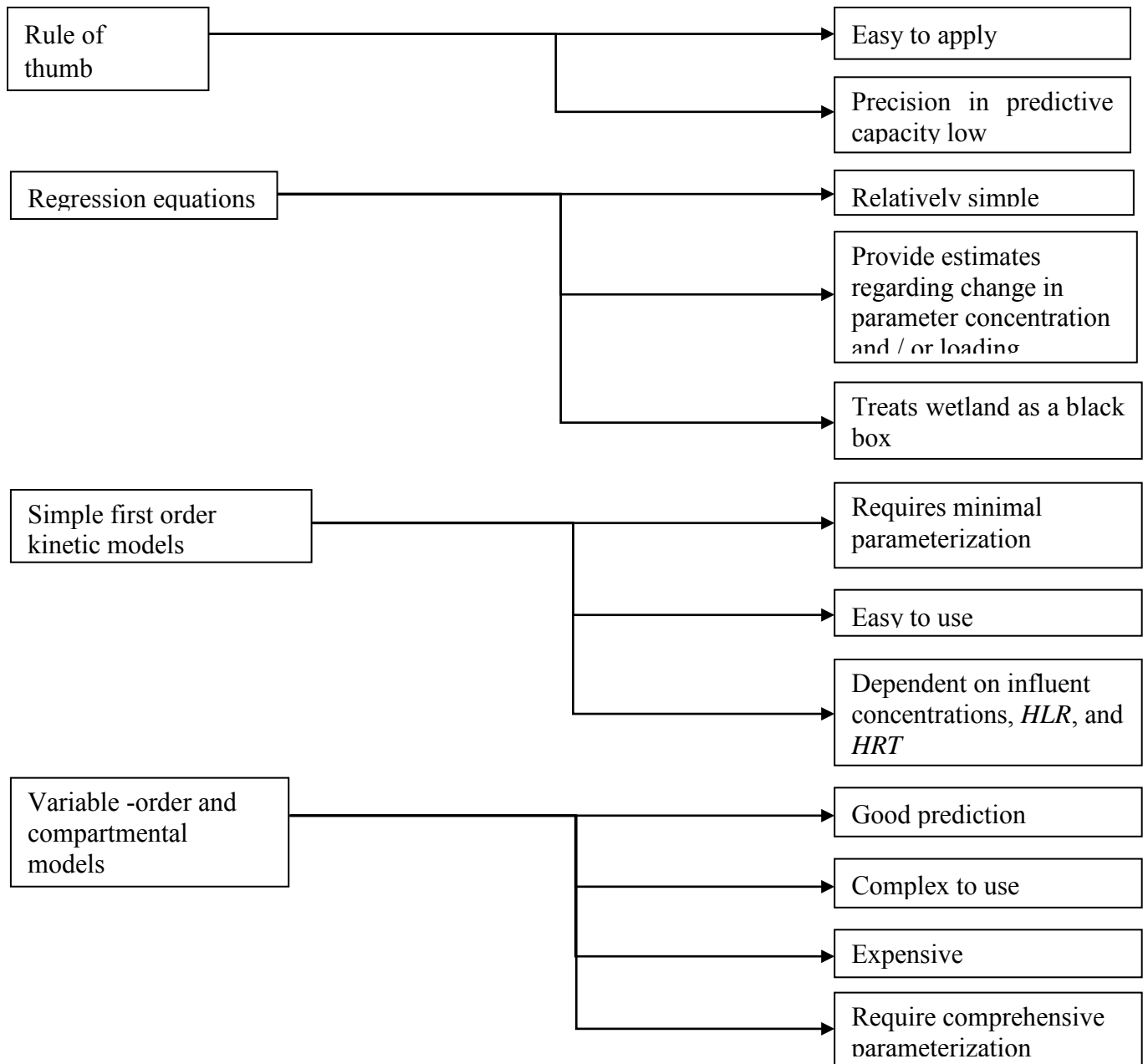
Figure 6.1   Overview of the strength and weaknesses for each major predictive tools category.

6.2.2 Hydraulic Retention Time (HRT)

The hydraulic retention time provides an estimate predicting how long it will take for the water entering the wetland to exit the wetland. Many of the treatment mechanisms within the wetland are biologically and or chemically driven and as such take time to complete. It has been generally accepted that most of the biological and chemical treatment can occur within a 2 to 7 day residency time within wetland. Many of these observations however have been made for temperate wetlands further south than those found in Nunavut and thus some have suggested a more realistic HRT for those wetlands north of 60° latitude might be closer to two weeks.

In many northern situations where natural wetlands are being used to treat domestic sewage, the size of the wetland is fixed by the surrounding landscape and cannot be easily adjusted. In these situations, it will still be important to determine what the actual HRT is for a specific wetland. This can be used to determine if the existing HRT falls within the generally accepted range of 2 to 7 days or preferable longer. If however, the HTR is less than 2 to 3 days then there is a significant chance that the wetland is not large enough to effectively treat the volume of sewage entering the wetland.

The HRT is basically a measure of volume / flow. For example, if the wetland has a volume of 1000 cubic metres and the inflow is 200 cubic metres per day, then the HRT is 1000 m^3 / 200 m^3 per day = 5 days. This is based on the rate of water entering the wetland together with the volumetric capacity of the wetlands, or in other words, how much water can the wetland hold (similar in principle to determining how much volume a pond or lake can hold and how often that water volume is replenished or exchanged).

Determining water volume of the wetland

The water (sewage) holding capacity of a wetland is determined by knowing the wetlands width, length and depth (to the bottom of the permafrost or bedrock). In most tundra wetlands, the water volume above the ground is minimal compared to the volume of water contained in the mineral and organic soils of the wetland. It is important to remember that the saturated portion of the wetland is much like a sponge and as such is composed of both

water and sediments. In order to determine the portion of the saturated zone occupied by only the water, the average porosity of the soil needs to be known. The porosity provides an estimate of the pore space between soil particles that is occupied by water. In some wetlands, the porosity is 30% meaning that within 1 cubic metre of soil, 70 percent of the volume would be occupied by soil and 30 percent would be occupied by water which in this case would mean that 300 L of water could be stored with a 1 cubic metre of soil that had a porosity of 30%.

If it were a perfect world, then measuring the water volume of the wetland would be as easy as it is described above, however, in tundra wetlands, the wetland size is often irregular in shape and the depth of the unfrozen zone of soil and rocks can be quite variable, as can be porosity. It is therefore often quite difficult to get accurate measurements, however, the intent here is to get a reasonable approximation of the wetland volume; remembering that rule of thumb methods are by nature imprecise.

Determining the flow of sewage through the wetland

In order to determine the HRT, the flow of water (sewage) into the wetland must be estimated. Typically this is measured as litres of sewage entering per minute or hour, but generally expressed as cubic metres of sewage per day. One cubic metre contains 1000 L and one day contains 1440 minutes per day (e.g., 60 minutes per hour X 24 hours per day = 1440 minutes per day). Depending on the wetland the flow can be estimated in different ways. For those wetlands that have scheduled periods of decanting, the flow can often be measured to provide an estimated flow per day or week. In some communities, the lagoon berm may leak. If leakage is confined to one region, then it may be possible to estimate the approximate discharge rate to the wetland, in other situations, particularly when leakage through the berm is more diffuse, estimates are made from knowing the number of trucks per day discharging sewage to the lagoon and by monitoring water levels within the lagoon itself.

An indirect way to determine flow, but one that is just as valid as those mentioned above is to estimate the volume of sewage generated by the community. In communities with

trucked service the volume of drinking water delivered within the community is tracked (recorded) and as such can be used as an indirect measure of sewage produced. The volume of sewage trucked for treatment is generally not tracked. It is also realized that the raw sewage for most communities is temporarily stored in a lagoon prior to disposal to the wetland. The underlying assumption here is that evaporation from the lagoon is minimal and that the discharge to the wetland from the lagoon is consistent in terms of flow volume, even if the decant occurs over an established period of time rather than being continuous.

In communities where it is difficult to get a reasonable estimate of sewage generation or in situations where the future volume of sewage generated by a community is of interest, an estimate can be determined by knowing the population and an estimated volume of water used per individual per day. This estimated volume is often referred to as "personal equivalence or PE". The volume of water consumed per day for an individual varies greatly depending on their location and the availability of water. Note that the PE method does not provide any information regarding what strength of effluent can be treated, nor does it provide any information concerning the quality of treatment. The values generated by the PE method provide information only in regards to the amount of effluent predicted to be produced by a certain population size. The Canadian average (for southern Canada) is 454 L/d per person which is a noticeably high estimate that averages in personal consumption and use by industry. In the USA, the personal equivalents (PE) determined for common households is 190 L per day per person. In developing countries it ranges between 60 to 80 L per day per person. In the Canadian north, the Department of Municipal and Community Affairs (MACA), Government of Northwest territories has developed the following formula to determine the volume of sewage generated from a known population base. The equation is:

Water Usage (L/community/d) = 90 L/c/d X (1.0 + 0.00023 * population)

Once the flow rate and hydraulic retention time are known, then an aerial estimate of the wetland size can be determined. A HRT of 1 day would estimate the size of the wetland needed to contain the volume of water (sewage) generated from one day within a

community. A HRT of 5 days would estimate the size of the wetland that would have a capacity to hold 5 times the volume generated from one day within a community. Thus it would take 5 days for this sewage to leave the wetland. Likewise a HRT of 7 days would mean that the size of the wetland needed to retain (hold) the sewage for 7 days would have to be 7 times the volume produced from a community during one day.

The following provide some example calculations when determining the HRT.

Step 1: Converting a flow rate of litres per minute to cubic meters per day

$$\frac{m^3}{d} = \frac{L}{min} X \frac{m^3}{1000\ L} X \frac{60\ min}{h} X \frac{24h}{d}$$

For an example of 2.7 L per minute the flow rate expressed in m³/d is

$$\frac{m^3}{d} = \frac{2.7L}{min} X \frac{m^3}{1000\ L} X \frac{60\ min}{h} X \frac{24h}{d} = 3.88\ cubic\ metres\ per\ day$$

Step 2: Factoring in soil porosity to determine the size of the wetland needed for a one-day HRT

- Using the flow value of 3.88 m³ above, and a porosity of 30 percent then 3.88 m³ represents only 30 percent of the wetland size that is needed. In order to determine the total size of the wetland needed the following formula is used:

$$\frac{3.88\ m^3}{X} = \frac{30\ \%}{100\ \%}$$

Solving for X you determine that X = 12.9 m^3

This means that a wetland with 30 % porosity must be 12.9 m^3 in size in order to hold a daily flow of 3.88 m^3.

Step 3: Determining the size of the wetland needed for a 5-day HRT is as follows

$$\frac{1d\ HTR}{5d\ HTR} = \frac{12.9\ \text{m}^3}{X}$$

Solving for X you determine that X = 64.7 m^3

Step 4: Determining the aerial size of the wetland taking into account the depth of the soil

- If the depth of the wetland soil is 0.2 m then the surface area of the wetland can be determined by:

$$\frac{64.7\ \text{m}^3}{0.2\ m} = 323\ \text{m}^2$$

Converting to hectares

$$323\ m^2 = \frac{1\ \text{ha}}{10000\ m^2} = 0.0323\ ha$$

6.2.3 Hydraulic Loading Rate (HLR)

The hydraulic loading rate is a quick method to determine if the flow through the wetland (distance / time) is within a broad range of values generally considered suitable for the

treatment of sewage within wetlands. The range of values for HLR are broad and thus must be interpreted cautiously and considered as only one course indicator to determine if the wetland is receiving an appropriate load. HLR is a measure of flow divided by area. For example an 8000 m² wetland which receives a flow of 200 m³ per day has a HLR of 200 m³ per d / 8000 m² = 0.025 m/d or 2.5 cm / d. The HLR provides a measure of flow velocity and the lower the flow velocity, the greater the chance for solids to settle out.

The HLR is determined as:

HLR = flow (m³ per day) divided by wetland size (m²)

$$HLR = \frac{m^3}{d} X \frac{1}{m^2}$$

$$HLR = \frac{200m^3}{d} X \frac{1}{8000m^2} = 0.025 \frac{m}{d}$$

Often HLR will be expressed in cm/d which can be achieved by multiplying m/d by 100, thus 0.025 m/d = 2.5 cm / d

Commonly accepted ranges for HLR range greatly. Typically, a normal HLR is considered to range between 0.2 to 3.0 cm/day (Wood, 1995) and in colder climates it has been suggested that a more appropriate range is 1 to 2 cm/day (Doku and Heinke, 1993), but others such as Kadlec and Knight (1996) suggest the range to be somewhere between 8 to 30 cm per day.

6.2.4 Organic Loading Rate

Dillon Consulting Limited prepared an assessment of the Kugaaruk, Nunavut treatment wetland in 2009. In that report they cite the work of Doku and Heinke (1993) who states that northern wetlands should not receive an organic loading of greater than 8 kg

BOD$_5$/ha/d in order to ensure adequate aerobic conditions exist within the wetland. Other investigators have used BOD loadings but more from a water quality perspective. Most uses suggest that BOD loadings should not exceed certain levels in order to ensure defined water quality parameters are not exceeded. For example Wallace and Knight (2006) indicate that BOD loadings should not exceed 80 kg BOD/ha/d to ensure that the water quality of the effluent exiting the wetland has a BOD concentration of 30 mg/L or less. Most of these rule of thumb values have be derived from loading charts which plot the BOD concentration of the water exiting the wetland as a function of the aerial loading rate of the BOD (e.g., kg BOD/ha/d) entering the wetland. The scatter around these regression curves is often large since the loading charts are often developed from the inclusion of data gathered from a wide range of wetland sizes, shapes, flow rates and climatic conditions.

An example of how to calculate the organic loading rate is as follows:

Equations

$$\text{Organic Matter} \left(BOD5 \frac{kg}{m3} \right) = \frac{BOD5 \frac{mg}{L} \; X \; 1000 \frac{L}{m3}}{1,000,000 \frac{mg}{kg}}$$

$$\text{Organic Loading Rate} \left(\frac{kg \; BOD5}{ha \cdot d} \right) = \frac{\text{Organic Matter} \left(BOD5 \frac{kg}{m3} \right) \; X \; \text{Flow} \left(\frac{m3}{d} \right)}{\text{Area } (ha)}$$

Calculations

$$\text{Organic Matter} \left(BOD5 \frac{kg}{m3} \right) = \frac{120 \frac{mg}{L} \; X \; 1000 \frac{L}{m3}}{1,000,000 \frac{mg}{kg}}$$

BOD$_5$ = 0.12 kg/m^3

$$\text{Organic Loading Rate} \left(\frac{kg \; BOD5}{ha \cdot d} \right) = \frac{\text{Organic Matter} \left(0.12 \frac{kg}{m3} \right) \; X \; 314.6 \left(\frac{m3}{d} \right)}{1.6 \; (ha)}$$

Organic Loading Rate = 23.6 kg BOD$_5$/ ha per day

Once again, these prescriptive rule of thumb methods can at best provide only a rough approximation for anticipated results. The loading rates such as those discussed above will provide rough guidelines for anticipated results, but it should be remembered that most of the rule of thumb approaches have been generated for warmer climates where higher temperature rates are likely more reflective of faster biological and chemical reaction rates. Loading rates such as the above for BOD cannot be used to provide an estimate of the wetland size needed to ensure the desired water quality targets are met. Furthermore, rule of thumb approaches are based on constructed wetlands, not natural wetlands.

Some common scaling factors for BOD are as follows:

Rousseau *et al.,* (2004)
- Max BOD loading rate of 75 kg BOD ha^{-1} d^{-1}

EC/EWPCA Emergent Hydrophyte Treatment System Expert Contact Group and Water Research Centre, (1990) [cf Kadlec and Wallace, 2009]
- 80 kg BOD ha^{-1} d^{-1} to produce an effluent BOD of less than 30 mg/L for primary-treated domestic wastewater

U.S. EPA, (2000a) [cf Kadlec and Wallace, 2009]
- 60 kg BOD ha^{-1} d^{-1} at inlet to produce an effluent BOD of less than 30 mg/L

Wallace and Knight, (2006) [cf Kadlec and Wallace, 2009]
- 80 kg BOD ha^{-1} d^{-1} at inlet to produce an effluent BOD of 30 mg/L
- 50 kg BOD ha^{-1} d^{-1} at inlet to produce an effluent BOD of less than 25 mg/L

Doku and Heinke, (1993) [cf Dillion 2009 KUG]
- 8 kg BOD$_5$ ha^{-1} d^{-1}

Campbell and Ogden, (1999)

Good estimates for BOD removal in a temperate climate (2.5 kg BOD / m^2 / year)

- In slightly warmer climates, this changes to 3.3 kg / m^2 / year @ 15°C and 4.4 kg / m^2 / year at 20°C

6.2.5 Regression Equations

The majority of the wetland studies appear to measure input and output concentrations, with some measuring input and output loadings. From this limited information many have developed regression equations in an attempt to mathematically describe the processes occurring within the wetland. This overly simplified approach treats the wetland as a black box and does not allow for the input of other influential parameters such as climate, bed material or physical dimensions of the wetland (length, width, depth, etc.). A lack of knowledge concerning these parameters hinders the appropriateness and closeness of fit to site specific wetlands. Wetlands, particularly those classified as natural wetlands being used for treatment, are quite variable in many of the influential parameters such as HLR, HRT, soil type and porosity, wetland dimensions, influent concentrations, climate, etc. and as such the application of generic scaling factors can only be used as a rough estimate of anticipated performance.

Regression equations can provide estimates regarding the change in parameter (e.g., BOD, COD, TSS, etc.) concentration and / or loading but are limited in the ability to be used to determine an estimate of the wetland size needed to meet certain effluent standards unless the regression equation accommodates the input of the hydraulic loading rate which when inputted into the equation Area (m^2) = flow (m^3 / d) / HLR (m / d) allows for the determination of wetland size, or, when a rule of thumb value is available for estimating the removal rate of the parameter of interest (e.g., a good estimate for BOD removal is 2.5 kg BOD m^{-2} y^{-1}).

Area based on HLR

$$A = \frac{Q}{q}$$

Where:

    Q = flow (m³/d)

    q = hydraulic loading rate (m/d) [note: the equation used to calculate q is provided above]

Example:

$$A = \frac{200\,\frac{m3}{d}}{0.2\,\frac{m}{d}}$$

A = 1000 m²

The following provide some regression equations for HSSF wetlands:

Biological Oxygen Demand

Regression models with concentration or loading only (no HLR) as input parameters
Brix, (1994) [c.f. Rousseau *et al.*, 2004] System = Danish and UK soil-based HSSF
Equation: C_{out} = (0.11 * C_{in}) + 1.87
Input Range = 1<C_{in}< 330 mg L⁻¹; Output Range 1<C_{in}<50 mg L⁻¹; q = 0.8<q<22 cm d⁻¹; R^2 = 0.74

Vymazal, (1998) [c. f. Rousseau *et al.*, 2004] System = HSSF in Czech Republic
Equation: C_{out} = (0.099 * C_{in}) + 3.24
Input Range = 5.8<C_{in}< 328 mg L⁻¹; Output Range 1.3<C_{in}<51 mg L⁻¹; q = 0.6<q<14.2 cm d⁻¹; R^2 = 0.33
Reed and Brown, (1995) [c.f. Rousseau *et al.*, 2004] System = 14 USA HSSF
Equation: $L_{removed}$ = (0.653 * L_{in}) + 0.292

Input Range = 4<L_{in}< 145 kg ha^{-1} · d^{-1}; Output Range 4<$L_{removed}$<88 kg ha^{-1} · d^{-1}; q = not given; R^2 = 0.97

Vymazal, (1998) [c.f. Rousseau et al., 2004] System = HSSF in Czech Republic
Equation: L_{out} = (0.145 * L_{in}) – 0.06
Input Range = 6<L_{in}< 76 kg ha^{-1} d^{-1}; Output Range 0.3<L_{out}<11 kg ha^{-1} · d^{-1}; q = not given; R^2 = 0.85

Chemical Oxygen Demand

Regression models with concentration or loading only (no HLR) as input parameters
Vymazal, (1998) [c.f. Rousseau et al., 2004] System = HSSF in Czech Republic
Equation: L_{out} = (0.17 * L_{in}) + 5.78
Input Range = 15<L_{in}< 180 kg ha^{-1} · d^{-1}; Output Range 0.3<L_{out}<11 kg ha^{-1} · d^{-1}; q = not given; R^2 = 0.85

Total Suspended Solids

Regression models with concentration and HLR (q) as input parameters
Reed and Brown, (1995) [c.f. Rousseau et al., 2004] System = 14 USA HSSF
Equation: C_{out} = C_{in} * (0.1058 + 0.0011 * q)
Input Range = 22<C_{in}< 118 mg L^{-1}; Output Range 3<C_{in}<23 mg L^{-1}; q = not given; R^2 = not given

Regression models with concentration or loading only (no HLR) as input parameters
Vymazal, (1998) [c. f. Rousseau et al., 2004] System = HSSF in Czech Republic
Equation: C_{out} = (0.021 * C_{in}) + 9.17
Input Range = 13<C_{in}< 179 mg L^{-1}; Output Range 1.7<C_{in}<30 mg L^{-1}; q = 0.6<q<14.2 cm d^{-1}; R^2 = 0.02

Brix, (1994) [c. f. Rousseau et al., 2004] System = Danish and UK soil-based HSSF
Equation: C_{out} = (0.09 * C_{in}) + 4.7
Input Range = 0<C_{in}< 330 mg L^{-1}; Output Range 0<C_{in}<60 mg L^{-1}; q = not given; R^2 = 0.67

Total Nitrogen

Regression models with concentration and HLR (q) as input parameters
Kadlec and Knight, (1996) [c.f. Rousseau *et al.*, 2004] System = NADB + others
Equation: $C_{out} = 2.6 + (0.46 * C_{in}) + (0.124 * q)$
Input Range = $5.1 < C_{in} < 58.6$ mg L^{-1}; Output Range $2.3 < C_{in} < 37.5$ mg L^{-1}; $q = 0.7 < q < 48.5$ cm d^{-1}; R^2 = 0.45

Regression models with concentration or loading only (no HLR) as input parameters
Kadlec *et al.*, (2000) [c.f. Rousseau *et al.*, 2004] System = Danish soil-based HSSF
Equation: $C_{out} = (0.52 * C_{in}) + 3.1$
Input Range = $4 < C_{in} < 142$ mg L^{-1}; Output Range $5 < C_{in} < 69$ mg L^{-1}; $q = 0.8 < q < 22$ cm d^{-1}; R^2 = 0.63

Vymazal, (1998) [c.f. Rousseau *et al.*, 2004] System = HSSF in Czech Republic
Equation: $C_{out} = (0.42 * C_{in}) + 7.68$
Input Range = $16.4 < C_{in} < 93$ mg L^{-1}; Output Range $10.7 < C_{in} < 49$ mg L^{-1}; $q = 1.7 < q < 14.2$ cm d^{-1}; R^2 = 0.72

Vymazal, (1998) [c.f. Rousseau *et al.*, 2004] System = HSSF in Czech Republic
Equation: $L_{out} = (0.68 * L_{in}) + 0.27$
Input Range = $145 < L_{in} < 1894$ kg ha^{-1} · d^{-1}; Output Range $134 < L_{out} < 1330$ kg ha^{-1} · d^{-1}; $q = 1.7 < q < 14.2$ cm d^{-1}; R^2 = 0.96

Total Phosphorus

Regression models with concentration and HLR (q) as input parameters
Kadlec and Knight, (1996) [c.f. Rousseau *et al.*, 2004] System = USA HSSF
Equation: $C_{out} = 0.23 * (q^{0.6} * C^{0.76}_{in})$
Input Range = $2.3 < C_{in} < 7.3$ mg L^{-1}; Output Range $0.1 < C_{in} < 6$ mg L^{-1}; $q = 2.2 < q < 44$ cm d^{-1}; R^2 = 0.60

Regression models with concentration or loading only (no HLR) as input parameters

Brix, (1994) [c.f. Rousseau *et al.*, 2004] System = Danish soil-based HSSF

Equation: C_{out} = (0.65 * C_{in}) + 0.71

Input Range = 0.5<C_{in}< 19 mg L^{-1}; Output Range 0.1<C_{in}<14 mg L^{-1}; q = 0.8<q<22 cm d^{-1}; R^2 = 0.75

Vymazal, (1998) [c.f. Rousseau *et al.*, 2004] System = HSSF in Czech Republic

Equation: C_{out} = (0.26 * C_{in}) + 1.52

Input Range = 0.77<C_{in}< 14.3 mg L^{-1}; Output Range 0.4<C_{in}<8.4 mg L^{-1}; q = 1.7<q<14.2 cm d^{-1}; R^2 = 0.23

Regression equation examples for sizing of wetlands

Example 1: When a rule of thumb removal rate is known

This is an example of BOD removal rates being used along with a regression equation to determine the approximate wetland size that is needed in order to achieve a desired removal rate.

Given:
- BOD concentration entering wetland (120 mg L^{-1})
- BOD regression equation: C_{out} = (0.11 * C_{in}) + 1.87
- Volume of effluent entering the wetland (314 m^3 · d^{-1})
- Number of days per year the wetland is functioning (90 d · y^{-1})
- Rule of thumb BOD removal rate (2.5 kg BOD m^{-2} y^{-1})

Step 1: Calculate the expected concentration of BOD exiting the wetland

C_{out} = (0.11 * C_{in}) + 1.87

C_{out} = (0.11 * 120) + 1.87

C_{out} = 15 mg L^{-1}

Step 2: Calculate the mass of BOD removed per year (Note: active portion is only 90 days)

BOD removed (kg/yr) = (BOD loss mg/L) * (Flow m^3/d) * (1 g / 1000 mg) * (1 kg / 1000 g) * (1000 L / m^3) * 90 d/y)

BOD removed = (120-15 mg / L) * (314 m^3/d) * (1 g / 1000 mg) * (1 kg / 1000 g) * (1000 L / m^3) * 90 d/y)

BOD removed = (105 mg / L) * (314 m^3/d) * (1 g / 1000 mg) * (1 kg / 1000 g) * (1000 L / m^3) * 90 d/y)

BOD removed = 2967 kg BOD per the 90 day active period of the wetland

Step 3: Calculate the area required based on the rule of thumb (e.g., 2.5 kg/m^2/y). NOTE: in this case a year represents 90 days

Area Required = BOD mass removed / rule of thumb

Area Required = 2967 kg BOD per year / 2.5 kg BOD / m^2 / y

Area Required = 1187 m^2 or 0.1187 ha

Example 2: Regression equation when both influent concentration and HLR are known

This is an example illustrating the use of a regression equation for TP which will allow the determination of the HLR if the concentration of TP is known at both the inlet and outlet. In this example, two equations are being used. One regression equation is used to predict the TP concentration exiting the wetland for a known TP influent concentration. Once the input and output of TP concentrations are known, then a second equation is used to

estimate the HLR. A third equation is then used to determine the Area based on knowing the flow entering the wetland and the HLR.

Given:

- Equation used to model TP loss: $C_{out} = (0.65 * C_{in}) + 0.71$
- Equation used to estimate HLR (once TP in and out are known): $C_{out} = 0.23 * (q^{0.6} * C^{0.76}_{in})$
- Equation used to estimate wetland size based on flow and HLR: A = flow / HLR
- Concentration of TP entering the wetland is: 5 mg/L

Step 1: Calculate the concentration of TP exiting the wetland

$$C_{out} = (0.65 * C_{in}) + 0.71$$

$$C_{out} = (0.65 * 5) + 0.71$$
$$C_{out} = 3.96 \text{ mg / L}$$

Step 2: solve for HLR (q)

$$C_{out} = 0.23 * (q^{0.6} * C^{0.76}_{in})$$

$$3.96 \text{ mg / L} = 0.23 * (q^{0.6} * 5^{0.76})$$

$$3.96 = 0.23 * (q^{0.6} * 3.40)$$

$$3.96 / 0.23 = (q^{0.6} * 3.40)$$

$$17.2 / 3.40 = q^{0.6}$$

$$5.06 = q^{0.6}$$

$$5.06^{1/0.6} = q$$

$$q = 14.6 \text{ cm} / \text{d}$$

Step 3: determine size of wetland based on formula:

$$\text{Area (m}^2) = \text{flow (m}^3/\text{d}) / \text{HLR (cm} / \text{d})$$

$$\text{Area (m}^2) = 314 \text{ (m}^3/\text{d}) / 0.146 \text{ m} / \text{d} \quad \text{(Note: HLR expressed in m/d not cm/d)}$$

$$\text{Area} = 2151 \text{ m}^2 \text{ or } 0.21 \text{ ha}$$

6.2.6 First -Order Kinetic Models

The first-order k-C* models are based on areal rate constants (k), flow rates, and wastewater concentrations entering the wetland. They consist of first-order equations which under the influence of ideal plug-flow behaviour and constant conditions (e.g. influent, flow and concentrations) predict an exponential profile between inlet and outlet (Kadlec and Wallace, 2009). The parameters k, C* and θ group a large number of other characteristics representing a complex matrix of interactions in a treatment wetland as well as external influences like weather conditions. Therefore, there can be high variability in reported values for k_A, k_V, C* and θ (Rousseau *et al.*, 2004). Many if not most of the areal rate constants used for these models have been developed in more southern locations under warmer climatic conditions and with data generated from constructed wetlands, not natural wetlands similar to those found in the Arctic.

The first-order kinetic models have their own set of limitations and care must be used to acknowledge the underlying assumptions that are being made by the user and to understand the limitations regarding many of the unknowns within a wetland, particularly natural

wetlands. Such unknowns can include factors such as preferential flow paths and hydraulic dead zones, inconsistencies in bed medium, hydraulic conductivity, porosity, etc. Many of these uncertainties can also be present in the more sophisticated models when site specific parameters are substituted with generalized parameters gathered from other sites.

Alberta Model

There are many variations of the first - order kinetic model. One that has gained popularity for use in the Canadian north is what is typically called the Alberta model (2000) which was prepared by the Alberta Environment ministry with the help of CH2M Gore and Storries Limited and an Alberta Environment Advisory / Working group. The basic expression of the model is a variant of the $k - C^*$ model described by Kadlec and Knight (1996).

This model has been rearranged to allow for the estimation of wetland size using the following expression.

$$A = \left[\!\left[\frac{0.0365Q}{k}\right]\!\right] x ln \left[\!\left[\frac{Ci - C^*}{Ce - C^*}\right]\!\right]$$

Where:

A = area (ha)

k = aerial rate constant @ 20°C, m/yr

Q = design flow (m^3/d)

C_i = influent concentration (mg/L)

C_e = effluent concentration (mg/L)

C^* = wetland background limit (mg/L)

This equation can be re-written to determine if the predicted size actually produces the target effluent concentration (C_o). This step is done primarily as a check to make sure the equation is consistent in giving the same answer. The rearrangement of the equation for determining if the size meets the target concentration is:

Where:

C_o = effluent concentration in mg/L

$$Co = C^* + (Ci - C^*) \exp\left[\!\!\left[-\frac{kAmax}{0.0365\ Q}\right]\!\!\right]$$

The Alberta model provides k - values for TSS, BOD, TP, TN, NH_4-N and Org - N, thus allowing size to be based on all of the above water quality parameters.

The Alberta model requires an estimate of the background concentration for the parameter of interest. The model developed for the Alberta Environment Ministry provides regression equations that will allow the background concentrations for TSS and BOD to be estimated. These equations are as follows:

TSS $C^* = 7.8 + 0.063\ C_i$

BOD $\quad\quad C^* = 3.5 + 0.053 C_i$

Major drawbacks to the Alberta model appear to be related to the aerial rate constants which do not account for the influence of temperature. Likewise there is no ability to adjust for wetland depth or for differences in bed porosity.

Campbell and Ogden 1999

The first - order kinetics model presented by Campbell and Ogden (1999) enables the size of the wetland to be predicted based on the concentration of the BOD entering and exiting the wetland. It appears to have greater utility in that it will accommodate the influence of temperature and porosity, and yet the model as presented below is for use with BOD only. The Campbell and Ogden (1999) equation is as follows:

$$As = \frac{Q(\ln Co - \ln Ce)}{Kt \cdot d \cdot n}$$

Where:

    As = surface area of the wetland

    Q = flow, in m^3/day

    Co = influent BOD (mg/L)

    Ce = effluent BOD (mg/L)

    Kt = temperature – dependent rate constant

    d = depth of bed medium

    n = porosity of bed medium

The influence of temperature can be accommodated by modifying K_t, the temperature - dependent rate constant using the modified Arrhenius equation (Kadlec and Wallace, 2009, p. 643):

$$K_t = K_{20} \theta^{(T-20)}$$

Where:

    K_{20} = rate constant at 20°C

    Θ = theta, the temperature correction factor set at 1.06

    T = temperature of the water in °C

[NOTE: Kadlec and Wallace (2009) provide K_{20} and theta values for BOD_5, Ammonia, TKN, T Nitrogen, T Phosphorus. However, these were given for FWS systems (e.g., free water surface wetlands), and it is not known if they can be used for HSSF systems (e.g., horizontal subsurface flow wetlands)]

[Note: cf Reed, Crites and Middlebrooks 1995, p. 226: K_{20} increases linearly as the organic loading increases, up to an organic loading of 100 kg/ha / d (e.g., at a loading of 100 kg/ha/d

the K_{20} is 1.104/d). This is a value for a HSSF wetland. Values for Surface flow wetlands are different and generally lower most likely because surface area is less (e.g., at a loading of 60 kg/ha/d the K_{20} in surface flow wetlands is approx. 0.66/d)]

Campbell and Ogden (1999) also provide sizing equations based on i) TKN and HN_4 and ii) NO_3:

- **TKN / HN$_4$**

$$ln \, (TKN)/NH_{4eff}) = K_t*HRT$$

or

$$HRT = ln \, (TKN)/NH_{4eff})/ \, K_t$$

where:

TKN = influent Kjeldahl nitrogen in mg/L

NH_{4eff} = ammonia concentration in the effluent in mg/L

$K_{NH} = 0.01854 + 0.3922(rz)^{2.6077}$

rz = percent of bed depth occupied by roots (a range between 0 and 1; use 1 for 100% occupied)

$K_t = K_{NH} * (1.048)^{(T-20)}$

HRT = hydraulic retention time in days

Example:

Determine the HRT required to drop TKN from 45 mg/L to 4 NH_4 mg /L at a temperature of 5°C

$$HRT = ln \, (TKN/NH_{4eff})/ \, K_t$$

1st Step: determine K_t

$$K_t = K_{NH} * (1.048)^{(T-20)}$$

Where

$$K_{NH} = 0.01854 + 0.3922(rz)^{2.6077}$$

$$K_{NH} = 0.01854 + 0.3922(1)^{2.6077}$$ (assuming that the root zone penetrates 100% of the bed depth)

$$K_{NH} = 0.01854 + 0.3922(1)$$ (Note: $(1)^{2.6077} = 1$

$$K_{NH} = 0.4107$$

Now determine K_t

$$K_t = K_{NH} * (1.048)^{(T-20)}$$

$$K_t = 0.4107 * (1.048)^{(5-20)}$$ (at temp = 5°C)

$$K_t = 0.4107 * (0.495)$$

$$K_t = 0.4107 * (0.495)$$

$$K_t = 0.20$$ (note: K_t will range between 0.2 and 0.25 when temp varies between 5 to 10°C)

2nd Step: solve for HRT

$$HRT = ln\ (TKN/NH_{4eff})/\ K_t$$

$$HRT = ln\ (45/4)/\ 0.2$$

$$HRT = ln\ (11.25)/\ 0.2$$

HRT = 2.42 / 0.2

HRT = 12 days

3ʳᵈ Step: determine the surface area of the wetland needed

NOTE: use the volume based 1ˢᵗ order kinetic equation to determine volume of wetland needed:

Volume = Flow (Q) * HRT (where Q is known. In this example I am using Q = 60.5 m³/d)

$V (m^3) = Q(m^3/d) * HRT(d)$

V = 60.5 m³/d * HRT

Re-writing the equation where V = A$_s$ * d * p   (where A$_s$ = surface area, d = bed depth, p = porosity)

Solving for Area:

A$_s$ = [Q(m³/d) * HRT(d)]/[depth (m) * porosity (unit less)]   (where bed depth is given as 0.6 and porosity is given as 0.4)

A$_s$ = [60.5 * 12] / [0.6 * 0.4]

A$_s$ = 3,025 m²

- **NO$_3$**

The following formula can be used to determine the surface area of the wetland needed based on a targeted wetland NO3 value:

$ln \, (NO_{3 \, inf} / NO_{3 \, eff}) = K_t * HRT$

or

$HTR = [ln(NO_{3 \, inf} / NO_{3 \, eff})] / K_t$

Where

$K_t = 1.15^{(T-20)}$
$NO_{3 \, inf}$ = influent nitrate in mg/L
$NO_{3 \, eff}$ = effluent nitrate in mg/L

1st Step: determine Kt at 5°C

$K_t = 1.15^{(T-20)}$
$K_t = 1.15^{(-15)}$
$K_t = 1.15^{(T-20)}$
$K_t = 0.1229$

2nd Step: determine HRT

$HTR = [ln(NO_{3 \, inf} / NO_{3 \, eff})] / K_t$
$HTR = [ln(40 / 23)] / 0.12$
$HTR = [ln(NO_{3 \, inf} / NO_{3 \, eff})] / K_t$
$HTR = 0.55 / 0.12$
$HTR = 4.6$

3rd Step: determine the surface area of the wetland needed

NOTE: use the volume based 1st order kinetic equation to determine volume of wetland needed:

Volume = Flow (Q) * HRT (where Q is known. In this example Q = 60.5 m^3/d)

V (m^3) = Q(m^3/d) * HRT(d)

V = 60.5 m^3/d * HRT

Re-writing the equation where V = A_s * d * p (where A_s = surface area, d = bed depth, p = porosity)

Solving for Area:

A_s = [Q(m^3/d) * HRT(d)]/[depth (m) * porosity (unit less)]   (where bed depth is given as 0.6 and porosity is given as 0.4)

A_s = [60.5 * 4.6] / [0.6 * 0.4]

A_s = 1,160 m^2

Rousseau *et al.*, (2004) concluded that the first - order kinetic models provided the best method for sizing wetlands. These authors found that rule of thumb methods were very generalized and therefore could not be relied on other than to provide a very rough estimate that could be used to confirm the measurements generated from the first-order kinetic models (used to verify that the model was being employed correctly). Rule of thumb methods were generally found to be overly conservative and overestimated the size of the wetland needed. This may in fact be an advantage in cold climate regions were space constraints are often of little concern and where microbial action during treatment is

inhibited by the cold climate which often demands a larger treatment area than typically needed in more temperate regions.

Rousseau *et al.* (2004) also concluded that regression equations were often of limited value, stating that there is a wide range of variation amongst the different equations which is reflective of the site specific conditions under which they were generated. Thus it becomes difficult to find a regression equation that would model a specific wetland, and in particular one in the Canadian arctic. Compounding the problem is the fact that most regression equations model only input and output concentrations without taking into consideration important parameters such as the hydraulic loading rate (HLR). These authors found only a few regression equations which allowed investigators to model both concentrations and HLR in the same equation, thus allowing an estimate of the wetland size to be calculated using the formula: Area (m^2) = flow (m^3/d) / HLR (m/d). Kadlec and Wallace (2009) provides additional comment on regression models (e.g., loading charts) outlining the limitations to these methods.

6.2.7 Sophisticated 2D and 3D models

Other sizing methods exist, such as variable-order, monod-type, mechanistic and compartmentalized models, and more sophisticated 2D and 3D models such as HYDRUS, WASP, TABS-2, STELLA, ANN, BASINS and NPS-WET. However, Rousseau *et al.* (2004) suggest that the main limitations to these models relates to the inherent complexity and parameter requirement which often does not exist for most sites. The lack of site specific parameter data often leads to the use of more generalized parameter data to satisfy the demands of the model which often creates greater uncertainty without much more gain in precision than could be achieved with the simpler first - order kinetic models.

SubWet 2.0

The SubWet 2.0 model is a horizontal subsurface flow modeling program designed to predict the level of treatment that can be expected based on the characteristics of several parameters known to influence treatment (e.g., wetland size, loading rates, etc.). This model

has been modified for use within natural wetlands of northern Canada, and is believed to be a good compromise between first-order kinetic models and the more sophisticated 2D, 3D models. SubWet utilizes 16 rate constants in an integrated manner to predict the treatment of BOD, organic nitrogen, ammonium, nitrate and total phosphorus. SubWet also provides the user the ability to calibrate these rate constants to site conditions in order to better reflect actual measured values. The calibration method to some extent accommodates for some of the influential processes that could be occurring within the wetland for which input data do not exist. For example, in most northern tundra wetlands, the area involved in the actual treatment process is likely smaller than the physical borders of the wetland. Modification (calibration) of specific rate constants within SubWet can therefore be used to ensure that simulated results closely match measured results as illustrated in the examples of Chapter 7. Obviously, variability between seasons and years may require more frequent calibrations although it is anticipated that the accuracy of the SubWet predictions will only increase as the data set for the wetland increases, thus providing greater insight into seasonal and yearly variability.

7.0 SubWet 2.0

7.1 General considerations

The SubWet model is a software program package used to simulate the treatment of wastewater in subsurface horizontal flow artificial wetlands. This model was originally developed by the United-Nations Environment Programme-Division of Technology, Industry and Economics-International Environmental Technology Centre (UNEP-DTIE-IETC). The model is distributed by the United-Nations as free-ware and can be found on the home web page for UNEP-IETC. Initially developed for warm climate applications and after being successfully used as a design tool in 15 cases in Tanzania,

SubWet was upgraded for use within cold climates for both artificial and natural treatment wetlands. The Centre for Alternative Wastewater Treatment of Fleming College worked in collaboration with UNEP-DTIE-IETC and Sven Jørgensen (the originator of the model) to developed SubWet 2.0, a new version to accommodate temperate and cold climatic conditions including summer Arctic and temperate winter conditions. SubWet was modified for use in cold climates by calibrating the model with data originating from the natural tundra wetlands investigated during the International Polar Year study detailed earlier in this manual. The application of this software to natural tundra wetlands is beyond the original purpose it was designed for. However, the calibration of SubWet with Arctic data has demonstrated its ability to model treatment performance within natural tundra wetlands and thus provide an additional predictive tool to aid northern stakeholders in the treatment of municipal effluents.

7.2 Model structure

SubWet incorporates the influence of several factors at one time while empirical equations are generally not able to consider more than two factors at one time and usually in isolation of the other influential parameters. The model has causality: this means that the process behind the model is known and can therefore be modelled by a mathematical equation. The model employs 25 differential process equations and 16 parameters (e.g., rate coefficients such as the temperature coefficient of nitrification). Readers desiring to know more about the design parameters of the SubWet model are directed to Foundations of Ecological Modelling (4th Ed.) edited by Sven Erik Jørgensen and Brian D. Fath (2011). Chapter 7.6 of this edition (Jørgensen and Gromiec, 2011) profiles the SubWet model and provides an in-depth description of differential process equations, default parameters, forcing functions and output parameters. The SubWet model was originally designed by Sven Jørgensen and colleagues as part of the Danida project, promoting cooperation between Copenhagen and Dar es Salaan University in Tanzania. Software for this model was later developed by the United Nations Environmental Programme, International Environmental Technology Centre (UNEP-IETC), so that it could be used in developing countries to design subsurface flow constructed wetlands for the treatment of domestic wastewaters. In 2009, the SubWet model

was further developed by Sven Jørgensen and the Centre for Alternative Wastewater Treatment, Fleming College, Canada for use with natural tundra wetlands of the Canadian arctic.

SubWet suggests default parameters for both warm climate and cold climate scenarios; however each parameter can be modified to improve the simulation for site specific conditions. The design input values of the model are used to specify the wetland width, length, depth, slope, % particulate matter, precipitation factor, hydraulic conductivity and selected flow rate (in cubic meters per day). The forcing functions outlining the operational parameters include wetland volume, flow of wastewater, porosity, average oxygen concentration, average temperature, the input of $cBOD_5$, ammonium, nitrate, total phosphorus, and organic nitrogen along with the fraction of $cBOD_5$, phosphorus, and organic N as suspended matter. The model calculates the simulated output values for $cBOD_5$, nitrate, ammonium, total phosphorus and organic nitrogen in milligrams per liter and the corresponding removal efficiencies in percentage.

7.3 Model calibration

The procedure used to calibrate SubWet 2.0 to site conditions has been outlined by Chouinard *et al.* (in press). In brief, the calibration is achieved by comparing wastewater effluent concentrations measured exiting the wetland site against the simulated concentrations generated by the SubWet 2.0 model. Rate coefficients, referred to as parameters in the SubWet model, are then selectively adjusted within defined limits to bring simulated values closer to measured values. Thus the model is calibrated to each individual wetland; however, calibration can take place only when site specific measured data exists. The use of measured concentrations to calibrate SubWet integrates, in a limited manner, some of the unknown processes influencing treatment performance; lessening the need to know specific details concerning individual influencing processes. Therefore the burden to know precise details regarding factors such as soil depth and the influence of melt waters becomes less demanding since the model is comparing a simulated integrated treatment response to a measured integrated treatment response. It should be noted that the

cold climate default rate coefficients embedded within the cold climate operations mode of SubWet do provide simulated output values that are generally with approximately 25 % of the measured values for most wetlands we have investigated. The calibration procedure as outlined by Chouinard *et al.*, (in press) generally reduces the difference between measured and simulated values to approximately 10 % or less. The work by Chouinard *et al.* (in press) and the SubWet user manual (Appendix E) provides a step by step overview in how the SubWet model is operated.

7.4 Predictive tool based on different scenarios

SubWet can be used to allow managers to predict the impact to treatment efficiency based on different scenarios involving an alteration to the HRT, aerial loading rates and the desired level of influent treatment. Furthermore, the model can be used as a predictive tool to help managers determine the size of a wetland needed to meet treatment objectives. This will assist managers in determining if the current wetland size can accommodate projected growth in population and anticipated effluent volumes. The model can be used to predict treatment performance anticipated from alterations to the size of the treatment area that could be increased through the construction of infiltration/dispersion ditches and structures that divert flow to other parts of the wetland that are not currently involved in treatment of the influent. Ultimately, SubWet can be used by resource managers to demonstrate the treatment benefit acquired from the use of designated treatment wetlands and can also be used as a predictive tool to forecast the potential these areas could provide from the application of selected management operations. This will help resource managers in cost benefit analysis when planning for future needs. Chouinard *et al.*, (2014) present the analysis of five different hypothetical scenarios to demonstrate how SubWet 2.0 can provide Arctic municipal wastewater managers with a tool to adapt to changing treatment conditions as well as the impact to treatment when wetland systems are altered. The simulated scenarios show that despite reducing wetland size, or increasing discharge volumes, as well as reducing temperature regimes the tundra wetlands provide excellent treatment potential, both on their own, or as an integrated/hybridized system with either a lagoon or facultative lake.

7.5 Field trials from eleven natural tundra wetlands in Arctic Canada

In addition to the work described by Chouinard *et al.*, (in press), the collected data sets in this chapter are used to illustrate how SubWet 2.0 can be calibrated to model the performance of eleven individual northern municipal treatment tundra wetlands in the Canadian Arctic. As stated in Chapter 6, treatment wetlands in Whale Cove, NU, Coral Harbour, NU, Arviat, NU, Repulse Bay, NU, Paulatuk, NT, Pond Inlet, NU, Edzo, NT, Fort Providence, NT, Gjoa Haven, NU, Ulukhaktok, NT and Taloyoak, NU were monitored by the Center for Alternative Wastewater Treatment (CAWT), Fleming College, Lindsay, Ontario under contract with Environment Canada and through the IPY study. The data that were generated during the surveys are used to refine the calibration of SubWet.

7.5.1 Whale Cove, NU data set

The following example with the Whale Cove data set illustrates how SubWet 2.0 c an be calibrated to this tundra wetland. Table 7.1 c ompares the values simulated by SubWet to the measured treatment values observed in the field, and provides the percent deviation of concentration values before calibration. It is generally accepted that the standard deviation around sampling and analytical procedures typically is between 10-12% and thus the standard deviation to be expected for comparisons between measured values and model simulated values can generally be expected to be in the range of 15 to 20 %. In the following comparisons, we will try to target a percent deviation of concentration values below 5% by altering the input variables in the model (a sample calculation of the percent deviation of concentration values is presented for this data set in Appendix F).

An examination of the values in Table 7.1 reveals that there is discrepancy between the simulated to the observed results for all parameters. The values indicate that SubWet may be overestimating the decomposition of organic matter (e.g., removal of BOD_5) while

underestimating the phosphorus adsorption capacity and underestimating the rate of nitrification. This suggests that the coefficients for the decomposition of organic matter and for the inverse phosphorus adsorption capacity are too high and should be lowered, and that the coefficient for the rate of nitrification is too low and should be increased.

Table 7.1: Comparison of simulated and observed values before calibration for the Whale Cove, NU data set

	Unit	Simulation Results	Observed Values	% deviation of concentration values
BOD$_5$	mg O$_2$/L	8.60	21	64
Ammonium-N	mg N/L	0.90	0	10
Phosphorus	mg P/L	1.46	0.1	34

The magnitude by which these coefficients are altered was approached in a trial and error manner where one coefficient at a time is altered and the simulation re-run and the graphical expression of the simulated to observed values re-examined, such as described by Chouinard *et al.*, (in press). It has been determined that for this data set a change in decomposition rate of organic matter (OC) from 0.25 to 0.05, a change in the inverse phosphorus adsorption capacity (AF) from 0.36 to 0.2 and a change in the nitrification rate (NC) from 0.9 to 2.5 produces simulation values for BOD$_5$ ammonium-N and total phosphorus that are much closer to the observed values (see the Glossary of symbols applied in SubWet 2.0 for a description of these coefficients in Appendix E). Table 7.2 shows the simulation results and the percent deviation of concentration values after calibration. The simulated parameters values are now closer to the observed values, and the percent deviations of concentration value all are below 5%.

Table 7.2: Simulation results after calibration for the Whale Cove, NU data set

	Unit	Simulation Results	Observed Values	% deviation of concentration values
BOD$_5$	mg O$_2$/L	20.9	21	0.52
Ammonium-N	mg N/L	0.14	0	1.56
Phosphorus	mg P/L	0.23	0.10	3.25

7.5.2 Coral Harbour, NU data set

Table 7.3 compares the values simulated by SubWet to the measured values observed in the field, and provides the percent deviation of concentration values before calibration. An examination of the values in Table 7.3 reveals that for all parameters monitored there is discrepancy between the simulated and the observed results. The values indicate that SubWet may be overestimating the decomposition of organic matter (e.g., removal of BOD$_5$) and the rate of nitrification (e.g., conversion of ammonium to nitrate), and underestimating the rate of ammonification and the inverse phosphorus adsorption capacity. This suggests that the coefficients for the decomposition of organic matter and nitrification rate are too high and should be lowered, and the coefficients for the ammonification rate and the inverse phosphorus adsorption capacity should be increased.

With a percent deviation of concentration value of 4.52 %, calibration of BOD$_5$ was not necessary, but was performed to demonstrate how close it is possible to get the simulated value to the observed value with the model. It has been determined that for this data set a change in OC from 0.25 to 0.17 and a change in NC from 0.9 to 0.1, a change in AC from 0.9 to 1.5, and a change in AF from 0.36 to 0.38 produces simulation values for BOD$_5$, ammonium-N and TP that are much closer to the observed values. Table 7.4 shows the simulation results and the percent deviation of concentration values after calibration.

Table 7.3: Comparison of simulated and observed values before calibration for the Coral Harbour, NU data set

	Unit	Simulation Results	Observed Values	% deviation of concentration values
BOD$_5$	mg O$_2$/L	6.45	14	4.52
Ammonium-N	mg N/L	0.09	2.8	14.3
Phosphorus	mg P/L	0.43	0.8	7.90

Table 7.4: Simulation results after calibration for the Coral Harbour, NU data set

	Unit	Simulation Results	Observed Values	% deviation of concentration values
BOD$_5$	mg O$_2$/L	14.6	14	0.35
Ammonium-N	mg N/L	3.12	2.8	1.68
Phosphorus	mg P/L	1.04	0.8	0.20

The simulated parameters values are now closer to the observed values, and the percent deviations of concentration value all are very low. The default values gave acceptable results that could be used for wetland design and predictive management needs; however, as shown, calibration can improve the model's ability to produce simulated values that are very close to the observed values, and by extension, very close to the real values and therefore producing a model that is more realistic in its predictive capabilities.

7.5.3 Arviat, NU data set

The data set from Arviat, NU is interesting in that the ammonium concentration of the effluent entering the wetland are much higher (11 mg/L) than normally encountered in municipal wastewater effluents (similar to the Baker Lake case study presented in Chouinard et al., in press). As shown in Table 7.5, the simulated total phosphorus values are relatively close to the observed values for this parameter. However, the values for BOD5 and ammonium-N are not acceptable, but can be improved when SubWet is calibrated for this specific site. Table 8-5 summarizes the differences between the values from the effluent after the wetland treatment observed in the field and the simulated results and provides the percent deviation of concentration values before calibration. The values in Table 7.5 indicate that SubWet may be overestimating the decomposition of organic matter (e.g., removal of BOD5), the rate of nitrification (e.g., conversion of ammonium to nitrate) and the rate of denitrification (for an explanation of the modification of the rates, see the Baker Lake data set under section 19.3.1.2 in Chouinard et al., in press). This suggests that the coefficients for the decomposition of organic matter, nitrification and denitrification rate are too high and should be lowered. In the Canadian Arctic, denitrification is in all likelihood a combination of bacterial conversion and plant uptake.

Table 7.5: Comparison of simulated and observed values before calibration for the Arviat, NU data set

	Unit	Simulation Results	Observed Values	% deviation of concentration values
BOD$_5$	mg O$_2$/L	0	16	18.4
Ammonium-N	mg N/L	6.42	11	7.40
Phosphorus	mg P/L	2.5	2.3	2.22

It has been determined that for this data set a change in OC from 0.25 to 0.05, a change in NC from 0.9 to 0.6 and a change in the denitrification rate coefficient (DC) from 3.5 to 0.2 produces simulation values for BOD_5 and ammonium-N that are much closer to the observed values. The unusually high ammonium present in the wastewater resulted in the need for the calibration of the denitrification rate constant. This type of wastewater typically forms high nitrate content in the anaerobic zone, which can induce higher denitrification under the proper conditions (mainly temperature and presence of sufficient organic carbon as substrate). In the case of the Arviat's system, the lagoon is massive and very anaerobic; treatment is minimal, and the wetland is undersized. Table 7.6 shows the simulation results and the percent deviation of concentration values after calibration.

The Arviat, NU data set provides an example of one of the more challenging calibration exercises. Despite the unusually high strength of the waste stream, the calibration of SubWet demonstrated that this model can provide a reasonable approximation of treatment efficiencies. The calibration efforts significantly improved the BOD_5 and ammonium-N values, and the percent deviations of concentration value are now well within the acceptable limit for the model. Furthermore, since this wetland has variable flow paths, the model is capable of accommodating these differences in tundra wetland operation, despite the variability in the wetland type and operation.

Table 7.6: Simulation results after calibration for the Arviat, NU data set

	Unit	Simulation Results	Observed Values	% deviation of concentration values
BOD_5	mg O_2/L	12.2	16	4.32
Ammonium-N	mg N/L	11.8	11	1.30
Phosphorus	mg P/L	2.5	2.3	2.22

7.5.4 Repulse Bay, NU data set

Table 7.7 compares the values simulated by SubWet to the values from the effluent after the wetland treatment observed in the field for Repulse Bay, NU, and provides the percent deviation of concentration values before calibration. An examination of the values in Table 7.7 reveals that for all parameters there is discrepancy between the simulated and the observed results. However, the percent deviations of concentration values are relatively low (all below 5%). Nevertheless, we can utilize this example to see how closely the observed and simulated values can be matched. The values indicate that SubWet may be overestimating the decomposition of organic matter (e.g., removal of BOD_5) and the rate of nitrification (e.g., conversion of ammonium to nitrate), and underestimating the inverse phosphorus adsorption capacity. This suggests that the coefficients for the decomposition of organic matter and nitrification rates are too high and should be lowered, and the coefficients for the inverse phosphorus adsorption capacity should be increased.

It has been determined that for this data set a change in OC from 0.25 to 0.15, a change in NC from 0.9 to 0.23 and a change in AF from 0.36 to 0.45 produces simulation values for BOD_5, ammonium-N and TP that are much closer to the observed values. Table 7.8 shows the simulation results and the percent deviation of concentration values after calibration.

The simulated parameters values are now closer to the observed values, and the percent deviations of concentration value all are extremely low, and show that by improving the precision of the calibration, simulation is in turn improved.

Table 7.7: Comparison of simulated and observed values before calibration for the Repulse Bay, NU data set

	Unit	Simulation Results	Observed Values	% deviation of concentration values
BOD$_5$	mg O$_2$/L	7.34	25	5
Ammonium-N	mg N/L	0.17	2.8	3.9
Phosphorus	mg P/L	1.07	1.4	4.23

Table 7.8: Simulation results after calibration for the Repulse Bay, NU data set

	Unit	Simulation Results	Observed Values	% deviation of concentration values
BOD$_5$	mg O$_2$/L	25.3	25	0.10
Ammonium-N	mg N/L	2.8	2.8	0.00
Phosphorus	mg P/L	1.31	1.4	1.15

7.5.5 Paulatuk, NT data set

Table 7.9 compares the values simulated by SubWet to the values from the effluent after the wetland treatment observed in the field in Paulatuk, NT, and provides the percent deviation of concentration values before calibration. An examination of the values in Table 7.9 reveals that there is discrepancy between the simulated and the observed results for BOD$_5$ and ammonium-N. The values indicate that SubWet may be underestimating the decomposition of organic matter (e.g., removal of BOD$_5$), the rate of nitrification (e.g., conversion of ammonium to nitrate), and the rate of ammonification. This suggests that the coefficient for the decomposition of organic matter, the nitrification rate and the coefficient for the ammonification rate are too low and should be increased.

Table 7.9: Comparison of simulated and observed values before calibration for the Paulatuk, NT data set

	Unit	Simulation Results	Observed Values	% deviation of concentration values
BOD$_5$	mg O$_2$/L	13.2	2	29.5
Ammonium-N	mg N/L	0.36	0.01	9.75
Phosphorus	mg P/L	0.03	0.04	0.41

It has been determined that for this data set a change in OC from 0.25 to 0.88, a change in NC from 0.9 to 2.5 and a change in AC from 0.9 to 2 produces simulation values for BOD$_5$ and ammonium-N that are much closer to the observed values, and percent deviation concentration values which are reasonable. Table 7.10 shows the simulation results and the percent deviation of concentration values after calibration.

Table 7.10: Simulation results after calibration for the Paulatuk, NT data set

	Unit	Simulation Results	Observed Values	% deviation of concentration values
BOD$_5$	mg O$_2$/L	1.9	2	0.26
Ammonium-N	mg N/L	0.08	0.01	2.20
Phosphorus	mg P/L	0.03	0.04	0.41

7.5.6 Pond Inlet, NU data set

Table 7.11 compares the values simulated by SubWet to the values from the effluent after the wetland treatment observed in the field at Pond Inlet, NU and provides the percent deviation of concentration values before calibration. An examination of the values in Table 7.11 reveals that there is discrepancy between the simulated and the observed results for BOD_5, ammonium-N and total phosphorus. As with Repulse Bay, NU, the percent deviations of concentration values for these parameters are relatively low (below 5%). Nevertheless, we will also utilize this example to see how closely the observed and simulated values can be matched through calibration. The values indicate that SubWet may be overestimating the decomposition of organic matter (e.g., removal of BOD_5), and the rate of nitrification (e.g., conversion of ammonium to nitrate), and underestimating the inverse phosphorus adsorption capacity. This suggests that the coefficients for the decomposition of organic matter and nitrification rate are too high and should be lowered, and that the coefficient for the inverse phosphorus adsorption capacity should be increased.

Table 7.11: Comparison of simulated and observed values before calibration for the Pond Inlet, NU data set

	Unit	Simulation Results	Observed Values	% deviation of concentration values
BOD_5	mg O_2/L	7.34	25	4.90
Ammonium-N	mg N/L	0.17	2.8	3.90
Phosphorus	mg P/L	1.07	1.4	4.23

It has been determined that for this data set a change in OC from 0.25 to 0.15, a change in NC from 0.9 to 0.23 and a change in AF from 0.36 to 0.45 produces simulation values for BOD$_5$, ammonium-N and total phosphorus that are much closer to the observed values. Table 7.12 shows the simulation results and the percent deviation of concentration values after calibrations; which are now extremely low and therefore quite acceptable.

Table 7.12: Simulation results after calibration for the Pond Inlet, NU data set

	Unit	Simulation Results	Observed Values	% deviation of concentration values
BOD$_5$	mg O$_2$/L	25.35	25	0.10
Ammonium-N	mg N/L	2.79	2.8	0.01
Phosphorus	mg P/L	1.31	1.4	1.15

7.5.7 Edzo, NT data set

Table 7.13 compares the values simulated by SubWet to the measured values observed in Edzo, NT, and provides the percent deviation of concentration values before calibration. An examination of the values in Table 7.13 reveals that for all parameters there is discrepancy between the simulated and the observed results. The values indicate that SubWet may be underestimating the rate of nitrification (e.g., conversion of ammonium to nitrate) and the inverse phosphorus adsorption capacity and either overestimating or underestimating the rate of denitrification. This suggests that the coefficients for the nitrification and the inverse phosphorus adsorption capacity are too low and should be increased, and that the denitrification rate is too high or too low and should be modified. The value for the decomposition of organic matter (e.g., removal of BOD$_5$) suggests that the model overestimated the coefficient for the decomposition of organic matter. Through calibration of the other parameters, it has been determined that no calibration of this specific

coefficient was necessary; calibration of the other parameters was sufficient to improve the discrepancy between the simulated and observed values for BOD_5.

Table 7.13: Comparison of simulated and observed values before calibration for the Edzo, NT data set

	Unit	Simulation Results	Observed Values	% deviation of concentration values
BOD_5	mg O_2/L	0	2	8.30
Ammonium-N	mg N/L	2.67	0.31	15
Phosphorus	mg P/L	0.08	0.16	9.20

It has been determined that for this data set a change in NC from 0.9 to 2.5, a change in DC from 3.5 to 0.1, and a change in AF from 0.36 to 0.55 produces simulation values for BOD_5, ammonium-N and TP that are much closer to the observed values. Table 7.14 shows the simulation results and the percent deviation of concentration values after calibration.

The calibration efforts significantly improved the parameters values, and the percent deviations of concentration values are now within the acceptable limit for the modeling for these parameters.

Table 7.14: Simulation results after calibration for the Edzo, NT data set

	Unit	Simulation Results	Observed Values	% deviation of concentration values
BOD_5	mg O_2/L	2.41	2	1.71
Ammonium-N	mg N/L	0.41	0.31	0.63
Phosphorus	mg P/L	0.16	0.16	0

7.5.8 Fort Providence, NT data set

Table 7.15 compares the values simulated by SubWet to the values from the effluent after the wetland treatment observed in the field for Fort Providence, NT, and provides the percent deviation of concentration values before calibration. An examination of the values in Table 7.15 reveals that for all parameters there is discrepancy between the simulated and the observed results.

Table 7.15: Comparison of simulated and observed values before calibration for the Fort Providence, NT data set

	Unit	Simulation Results	Observed Values	% deviation of concentration values
BOD$_5$	mg O$_2$/L	9.77	32	79.39
Ammonium-N	mg N/L	5.28	18.5	56.73
Phosphorus	mg P/L	5.10	8.94	56.25

The values indicate that SubWet may be overestimating the decomposition of organic matter (e.g., removal of BOD$_5$), and the rate of nitrification (e.g., conversion of ammonium to nitrate), and underestimating the inverse phosphorus adsorption capacity. This suggests that the coefficients for the decomposition of organic matter and nitrification rate are too high and should be lowered, and the coefficient for the inverse phosphorus adsorption capacity should be increased. It has been determined that for this data set a change in OC from 0.25 to 0.2, a change in NC from 0.9 to 0.1, and a change in AF from 0.36 to 2.5 produces simulation values for BOD$_5$, ammonium-N and total phosphorus that are much closer to the observed values. Table 7.16 shows the simulation results and the percent deviation of concentration values after calibration.

Table 7.16: Simulation results after calibration for the Fort Providence, NT data set

	Unit	Simulation Results	Observed Values	% deviation of concentration values
BOD$_5$	mg O$_2$/L	33.8	32	6.40
Ammonium-N	mg N/L	19.2	18.5	9.73
Phosphorus	mg P/L	8.88	8.94	2.80

The simulated parameters values are now closer to the observed values; however, in this case BOD and ammonium-N are not within the 5% range targeted, especially ammonium-N. Considering the unusually high ammonium-N content in the wastewater, this is the best percent deviation concentration value that could be obtained with the model for both cases. Furthermore, these values are still within the acceptable 15% of the modeling standards.

7.5.9 Gjoa Haven, NU data set

Table 7.17 compares the values simulated by SubWet to the values from the effluent after the wetland treatment observed in the field for Gjoa Haven, NU, and provides the percent deviation of concentration values before calibration. An examination of the values in Table 7.17 reveals that there is a discrepancy between the simulated and the observed results for total phosphorus. The values indicate that SubWet may be overestimating the inverse phosphorus adsorption capacity. This suggests that the coefficient for the inverse phosphorus adsorption capacity is too high and should be lowered.

With a percent deviation of concentration value of 3.31% for ammonium-N, calibration was not necessary, but was performed to demonstrate how close it is possible to get the simulated value to the observed value with the model. It has been determined that for this data set a change in AF from 0.36 to 0.3 and a change in NC from 0.9 to 2.5 produces simulation values for total phosphorus and ammonium-N that are closer to the observed

values. Table 7.18 shows the simulation results and the percent deviation of concentration values after calibration.

Table 7.17: Comparison of simulated and observed values before calibration for the Gjoa Haven, NU data set

	Unit	Simulation Results	Observed Values	% deviation of concentration values
BOD$_5$	mg O$_2$/L	9.60	7	1.82
Ammonium-N	mg N/L	1.79	0.12	3.31
Phosphorus	mg P/L	1.31	0.78	12

Table 7.18: Simulation results after calibration for the Gjoa Haven, NU data set

	Unit	Simulation Results	Observed Values	% deviation of concentration values
BOD$_5$	mg O$_2$/L	9.60	7	1.82
Ammonium-N	mg N/L	0.16	0.12	0.08
Phosphorus	mg P/L	0.55	0.78	5.20

The simulated parameters values are now closer to the observed values, and the percent deviations of concentration value are well within acceptable limits for the model.

7.5.10 Ulukhaktok, NT data set

Table 7.19 compares the values simulated by SubWet to the values from the effluent after the wetland treatment observed in the field for Ulukhaktok, NT, and provides the percent deviation of concentration values before calibration. An examination of the values in Table 7.19 reveals that for all parameters there is discrepancy between the simulated and the observed results. The values indicate that SubWet may be underestimating the decomposition rate of organic matter, the rate of nitrification (e.g., conversion of ammonium to nitrate), and the rate of ammonification and overestimating the inverse phosphorus adsorption capacity. This suggests that the coefficients for the decomposition rate of organic matter, nitrification rate and ammonification are too low and should be increased, and the coefficients for the inverse phosphorus adsorption capacity should be decreased.

Table 7.19: Comparison of simulated and observed values before calibration for the Ulukhaktok, NT data set

	Unit	Simulation Results	Observed Values	% deviation of concentration values
BOD$_5$	mg O$_2$/L	8.03	5	4.50
Ammonium-N	mg N/L	1.09	0.09	16
Phosphorus	mg P/L	1.47	0.2	10.5

It has been determined that for this data set that a change in OC from 0.25 to 0.31, NC from 0.9 to 2.5, a change in AC from 0.9 to 1.5, and a change in AF from 0.36 to 0.24

produces simulation values for BOD$_5$, ammonium-N and total phosphorus that are much closer to the observed values. Table 7.20 shows the simulation results and the percent deviation of concentration values after calibration.

   With a percent deviation of concentration value of 4.50 %, calibration of BOD$_5$ was not necessary, but was performed to demonstrate how close it is possible to get the simulated value to the observed value with the model. The simulated parameters values are now closer to the observed values, and the percent deviations of concentration value all are within acceptable limits for the model.

Table 7.20: Simulation results after calibration for the Ulukhaktok, NT data set

	Unit	Simulation Results	Observed Values	% deviation of concentration values
BOD$_5$	mg O$_2$/L	5.03	5	1.14
Ammonium-N	mg N/L	0.12	0.09	0.31
Phosphorus	mg P/L	0.19	0.2	0.12

7.5.11 Taloyoak, NU data set

   Table 7.21 compares the values simulated by SubWet to the values from the effluent after the wetland treatment observed in the field for Taloyoak, NU, and provides the percent deviation of concentration values before calibration. An examination of the values in Table 7.21 reveals there is discrepancy between the simulated and the observed results for BOD$_5$ and phosphorus. The values indicate that SubWet may be overestimating the decomposition of organic matter (e.g., removal of BOD$_5$.) and underestimating the inverse phosphorus adsorption capacity. This suggests that the coefficients for the decomposition of organic matter is too high and should be lowered, and that the coefficient for the inverse phosphorus adsorption capacity is too low and should be increased. With a percent deviation of concentration value of 1.80 %, calibration of ammonium-N was not necessary, but was again performed to demonstrate how close it is possible to get the simulated value

to the observed value with the model. It has been determined that for this data set a change in OC from 0.25 to 0.17, a change in AF from 0.36 to 0.83 and a change in NC from 0.9 to 1.2 produces simulation values that are much closer to the observed values. Table 7.22 shows the simulation results and the percent deviation of concentration values after calibration.

Table 7.21: Comparison of simulated and observed values before calibration for the Taloyoak, NU data set

	Unit	Simulation Results	Observed Values	% deviation of concentration values
BOD$_5$	mg O$_2$/L	16.7	25	15.1
Ammonium-N	mg N/L	0.21	0.13	1.80
Phosphorus	mg P/L	0.00	0.32	8.94
Nitrate	mg N/L	0.03	0.02	0.89

Table 7.22: Simulation results after calibration for the Taloyoak, NU data set

	Unit	Simulation Results	Observed Values	% deviation of concentration values
OD$_5$	mg O$_2$/L	25.4	25	0.75
Ammonium-N	mg N/L	0.13	0.13	0
Phosphorus	mg P/L	0.32	0.32	0
Nitrate	mg N/L	0.03	0.02	0.89

The simulated parameters values are now closer to the observed values, and the percent deviations of concentration value all are within acceptable limits for the model.

7.5.12 Summary of the coefficient changes

Table 7.23 summarizes the changes that were made to the cold climate default parameters of the SubWet 2.0 model. The coefficient changes column reflects which coefficient has been modified, and provides the new value after calibration. The coefficient range and default coefficient values refer to the coefficients required by the differential equations that SubWet uses to model wetland processes. The range for each coefficient has been identified from published literature. The default parameters fall within the normal range and are embedded within the SubWet model before calibration. The range of coefficient changes column reflects the proposed new range found after calibration of the 11 sites: some of the ranges have stayed the same, others have changed (i.e. the initial coefficient range for DC is: 0.00-5 and after calibration of all 11 sites, the proposed new range for DC is: 0.1-0.2).

Even though a 15% difference between simulated and measured values is generally considered acceptable, it was shown in the previous calibration models that most simulated values could be calibrated to approximately 5% deviation. A review of the SubWet 2.0 simulations revealed that the cold water default parameters were poorly suited for the Fort Providence wetland. This may be in part related to the fact that the wastewater is decanted from the lagoon to the wetland in a concentrated period of time, unlike the slow continuous release that occurs from the exfiltration of leaky lagoon berms or the release from facultative lakes. The impact to treatment process caused by a sudden discharge related to a decant event or spring freshet is unknown. It is anticipated that higher flow volumes can decrease HRTs and increase organic loading to the point that the treatment system is overwhelmed and treatment efficiency decreases. Decanting of lagoons is also typically done at the end of the frost free period; therefore wetlands have less time to assimilate the nutrients and other pollutants. This could have an effect on the model simulations and may be why there is discrepancy between observed and simulated values. The simulated values (before calibration) from Whale Cove, NU and Paulatuk, NT are also moderately poor. These two sites are the only locations where pre-treatment of the wastewater is through a facultative lake, and thus it appears that the default cold climate default parameters of SubWet 2.0 may

always need calibration to measured values for wetlands receiving wastewaters pre-treated in these lake systems.

Table 7.23: Summary of the coefficient changes for the 11 tundra treatment wetlands from the Canadian Arctic presented

11 Tundra Treatment Wetlands from the Canadian Arctic	Coefficient changes	Coefficient range and default coefficient values used in calibration	Range of coefficient changes
Whale Cove, NU	OC: 0.05 AF: 0.2		
Coral Harbour, NU	OC: 0.17 AF: 0.38 NC: 0.1 AC: 1.5		
Arviat, NU	OC: 0.05 NC: 0.6 DC: 0.2	AC: 0.05- 2.0 0.9 (1/24h)	AC: 0.5- 2.0
Repulse Bay, NU	OC: 0.15 AF: 0.45 NC: 0.23	NC: 0.1- 2.5 0.9 (1/24h)	NC: 0.1- 2.5
Paulatuk, NT	OC: 0.88 NC: 2.5 AC: 2	OC: 0.05- 2.0 0.25 (1/24h) DC: 0.00-5 3.5 (1/24h)	OC: 0.05- 1.7 DC: 0.1-0.2
Pond Inlet, NU	OC: 0.15 NC: 0.23	AF: 0-100 0.36	AF: 0.2-2.5
Edzo, NT	AF: 0.55 NC: 2.5 DC: 0.1		
Fort Providence, NT	OC: 0.2 AF: 2.5 NC: 0.1		
Gjoa Haven, NU	AF: 0.3		
Ulukhaktok, NT	OC: 0.31 AF: 0.34 NC: 2.5 AC: 1.5		
Taloyoak, NU	OC: 0.17 NC: 1.2		

The main challenges in modeling tundra treatment wetlands in the far north is that they are extremely heterogeneous (patchy systems, with both preferential flow and dead zones), have fast and slow retention times and good and poor conductivity. Furthermore, there is the problem of measuring these variables and parameters. In some cases hidden dilution from groundwater might be a problem. Because permafrost exists at nearly all wetland sites it is assumed that melt water may be the primary source of any subsurface flow that could cause wastewater dilution with the treatment zone. Each wetland system is unique and therefore it is always best to calibrate SubWet to measured values when possible. The measured water quality parameter values along with the calibration values reflect the conditions for only one short time period. These values do not reflect the seasonal variation that can occur throughout the frost free period, nor do they capture the year to year variability that likely occurs. The values do however; provide a relative indication of the type of treatment that may be expected. It is anticipated that with the inclusion of subsequent years of measured data the uncertainties regarding season and year to year influences will be better understood, thus providing greater confidence in modelled results.

It should be remembered that some of the modelled water quality parameters such as ammonium are actually influenced by more than one process. For example the treatment of ammonium reflects the ammonification of organic nitrogen and the nitrification of ammonium and can also be influenced by the oxygen demand related to BOD. Thus, the calibration of SubWet may require changing more than one rate coefficient.

7.6 Summary and possible future modifications

A variety of approaches can currently be applied to predict the performance of constructed wetlands, however, only and a few numerical models are available as a predictive tool for horizontal subsurface flow treatment wetlands. All numeric models can be problematic when applying to natural wetlands; which nature are often open and diffuse systems, with poorly defined boundaries, flow patterns and permeable borders (Kadlec and Wallace, 2009). These conditions present challenges for wastewater regulators who require well

defined points of control. The best approach is to incorporate site specific performance data (when available) into the model in an attempt to calibrate the model to the unique characteristics of the particular wetland. This was the approach taken with the application of the SubWet model to the natural tundra wetlands.

The calibration of SubWet with the eleven northern wetlands in the Canadian Arctic clearly demonstrates its ability to model treatment performance within natural tundra wetlands and thus provide an additional predictive tool to aid northern stakeholders in the treatment of municipal effluents. It was demonstrated that even during periods of wastewater decanting, SubWet was robust enough to provide moderately good treatment predictions. Likewise it has also been demonstrated that SubWet is able to provide good predictions for those municipalities that produce a high strength wastewater (e.g., Baker Lake and Arviat).

It is recognized however, that our present knowledge regarding year to year variability in wetland performance and seasonal influences is lacking and future monitoring will be needed to improve temporal predictions.

8.0 Summary

The study results outlined in the preceding chapters demonstrates the wastewater treatment services that are provided by the eleven natural tundra wetlands investigated. For the most part, the concentration of cBOD5, ammonia, phosphorus and microbial pathogens were significantly reduced with final treated effluent values well below the CCME NPS for southern Canada. Treatment performance was poorer at some wetland sites such as Whale Cove, Pond Inlet and Fort Providence where the wetlands were either undersized or the slope and hydraulic retention times were too steep or short. Total suspended solids did not appear to be a good indicator of wastewater treatment since some wetlands generate TSS

that does not originate in the wastewater influent. It is suggested that a monitoring of fixed suspended solids (FSS) and volatile suspended solids (VSS) and a comparison of their contribution to TSS may provide more insight into treatment performance. In some wetlands that generated TSS it was found that the VSS component of the wastewater decreased as it traversed the wetland, suggesting that the organic fraction of TSS was being removed by sedimentation, filtration and or microbial oxidation while the FSS (inorganic portion) increased, suggestive that erosional forces were mobilizing inorganic particles into the effluent.

The IPY study investigated treatment performance over the course of an entire arctic summer. Although these results indicate that treatment performance remained high during the summer, more investigation is needed during the spring freshet where flows are anticipated to be higher because of the influx of melt waters and influent strength is expected to be stronger with the release of the frozen wastewaters that accumulated over the winter time. Treatment performance during the spring freshet is not well documented but is expected to be variable depending on the inherent characteristics of individual wetlands which vary in their capacity to accommodate increased volumes and influent strengths. The Environment Canada funded study looked at treatment performance at multiple locations within the wetland in an attempt to better understand the progression of treatment as effluents traversed the site. It was found that in most cases the wetlands had the capacity to accommodate either higher flows or loading rates. Both studies (IPY & EC) did indicated that a slow steady release of effluent into the wetland resulted in better treatment performance than if effluents were released quickly over a short period of time during a scheduled lagoon decant. This information suggests that wetland treatment is enhanced if the lagoon exfiltrates rather than being decanted quickly. A better management practice may be to decant smaller volumes more frequently rather than release a large volume all at once.

It was found that the SubWet 2.0 modelling program provided simulated results that were much closer to measured results than any of the other readily accessible predictive tools that currently exist. SubWet was successfully calibrated to all eleven wetlands and after

calibration the simulated (predicted) values were generally within 10% of the measured values. SubWet can provide wastewater managers the ability to predict the outcome of a variety of simulated operational scenarios and in doing so determine which management options are anticipated to provide the best treatment. This should be of great utility for those operators wanting to know how much of the lagoon's effluent can be released to the wetland and how frequently this can be done while still maintaining a desired level of treatment. SubWet can also be used to predict the capacity of existing wetlands to accommodate future population growth and associated increases in wastewater volume.

It is hoped that this document will be useful to wastewater managers, regulatory agencies and consulting engineers and planners. The work contained in this document has attempted to provide greater insight into the performance of natural wetlands for the treatment of domestic effluent and in doing so to remove some of the barriers that may have prevented a greater formal utilization of natural wetlands as part of an overall wastewater strategy for northern Canada.

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