

Oxygen Dynamics and Carbon Removal in Municipal Waste  
Stabilization Ponds in Arctic Regions

by

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## **Dedication**

I would like to dedicate this work to the Inuit of Canada and the Canadian Arctic. I am eternally grateful for the experiences I have been privileged to experience the North, and my time there has enriched my life. The Inuit people willingly shared their joy, knowledge and pain with respect to their communities and environment. The Inuit are a proud, resilient and inviting people with a great deal of knowledge about their communities and the arctic environment, and will graciously share their knowledge with those willing to listen. The openness of the Inuit peoples to sharing their culture, humour, and struggles with me provided a powerful new perspective on Indigenous life and the impacts of Canadian policy.

The knowledge of the Inuit is kept within the fabric of their culture and by association their language. They are struggling to adapt and find a place within the economic and social fabric of western society. Life for those living in the Arctic is full of challenges and I bore witness to the painful ripples food insecurity, climate change, domestic violence, and substance abuse have left upon the people, their communities, and their culture. The creation of settlements in the 1960's continues to have impacts, both good and bad, however the people and their communities are bravely navigating the incorporation of their traditional knowledge into their new reality.

More support for the North's culture is needed. I believe Inuit culture is the source of strength and resilience that has allowed the Inuit to thrive in a beautiful, yet at times hostile and unforgiving, environment. Their culture and language is needed more than ever because of the rapid changes the Inuit are facing in their society and environment. We must recognize the knowledge contained in any culture is invaluable to all of humanity. Once a culture is gone it cannot be reconstituted, so the time to act is now, as traditional knowledge is quickly being lost with the passing of Inuit elders.

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## Abstract

Waste Stabilization Ponds (WSPs) are commonly used for municipal wastewater management in Canadian Arctic communities. However, there has been limited research on WSP performance in these regions, and how environmental and operational factors influence treatment processes in arctic WSPs. New wastewater discharge standards have been proposed for all Canadian municipalities and, with respect to arctic municipal wastewater treatment solutions, it is unclear if WSPs are capable of meeting these standards. The objectives of this research were to (i) characterize the performance of WSPs currently operating in arctic regions, (ii) identify environmental and operational variables that influence biological treatment processes, and (iii) develop recommendations for the design and upgrading of future and current arctic WSP systems. Four operational WSPs in the Canadian Arctic Territory of Nunavut were intensively monitored during a four year (2011-2014) period. The four WSPs were generally anaerobic, resulting in poor removal of carbonaceous biological oxygen demand (CBOD), with average effluent CBOD<sub>5</sub> concentrations exceeding 80 mg/l. A series of controlled mesocosm experiments were conducted to identify how environmental conditions (temperature and irradiance) and organic carbon loading variables influenced algae growth, oxygen dynamics and CBOD<sub>5</sub> removal rates. A process-based model was also developed to simulate oxygen dynamics and carbon removal and was successfully calibrated and validated using the experimental data. The experiments and process-based modeling demonstrated that the development of aerobic conditions, and increased rates of CBOD<sub>5</sub> removal was directly linked to the presence of an active algae population. The results also indicated that current organic loading rate guidelines adapted from non-arctic regions are not applicable, and that both initial carbon concentrations and daily areal carbon loading rates need to be decreased in order to facilitate aerobic environments in arctic WSPs. WSPs operating in cold climates are very sensitive to changes in operational and environmental conditions, and greater resiliency needs to be incorporated into design guidelines in order to meet more stringent discharge criteria.

## List of Abbreviations and Symbols Used

$\theta$	van't Hoff-Arrhenius temperature dependency coefficient
A	Phytoplankton concentration
AGS	Phytoplankton growth self-suppression
APHA	American Public Health Association
ANOVA	Analysis of Variance
B	Bacteria concentration
BOD	Biological oxygen demand
BOD <sub>5</sub>	5-day Biochemical oxygen demand
BGS	Bacterial growth self-suppression
°C	Degrees Celsius
C <sub>0</sub>	Initial concentration at time 0
C <sub>t</sub>	Concentration at time t
CBOD <sub>5</sub>	5-day Carbonaceous biochemical oxygen demand
CBOD <sub>5inf</sub>	Influent CBOD <sub>5</sub> concentration
CMFR	Completely mixed flow reactor
CCME	Canadian Council of Ministers of the Environment
ColumnZ	Depth of water column (total depth)
CO <sub>2</sub>	Carbon dioxide
COD	Chemical oxygen demand
C <sub>SO<sub>2</sub></sub>	Saturation concentration - oxygen
C <sub>SCO<sub>2</sub></sub>	Saturation concentration – carbon dioxide
DI	Daily volumetric loading
DO	Dissolved oxygen
ED	Equal distribution (factor)
F <sub>dis</sub>	Phytoplankton growth distribution function
GN	Government of Nunavut
H <sup>+</sup>	Hydrogen ion
HDPE	High-density polyethylene
HRAP	Hi-rate algae pond

$I_{av}$	Average irradiance across photic depth with phytoplankton
$I_{avclear}$	Average irradiance across photic depth with no phytoplankton present
OECD	Organisation for Economic Co-operation and Development
TAN	Total ammonia nitrogen
$I_o$	Surface incident irradiance
$K$	First order rate constant
$K_{ad}$	Phytoplankton death rate
$K_{as}$	Phytoplankton settling rate
$K_{bd}$	Bacteria death rate
$K_{bs}$	Bacteria settling rate
$K_c$	Half saturation of phytoplankton on carbon dioxide
$K_{O_2}$	Half saturation of bacteria on oxygen
$K_e$	Specific light attenuation coefficient of water/wastewater
$K_p$	Light abstraction by phytoplankton
$K_w$	Light attenuation coefficient of water and constituents
$Kl_{coeffCO_2}$	Coefficient of $CO_2$ transfer rate relative to oxygen
$Kl_{O_2}$	Oxygen transfer rate (piston velocity)
$K_s$	Half saturation of bacteria on substrate
LED	Light emitting diode
MDL	Method detection limit
$^{\circ}N$	Northern latitude
NIST	National Institute of Standards and Technology
NGSMI	National Guide to Sustainable Municipal Infrastructure
OSHA	Occupational Safety and Health Administration
$O_2$	Oxygen
ODE	Ordinary differential equation
$OH^-$	Hydroxide ion
OLR	Organic loading rate
$OUR_b$	Basal oxygen utilization rate of bacteria
$OUR_m$	Metabolic oxygen utilization rate of bacteria

OUR	Oxygen utilization rate (total/effective)
PVC	Polyvinyl chloride
PAR	Photosynthetic active radiation
S	Substrate (CBOD <sub>5</sub> ) concentration
SolCBOD <sub>5</sub>	Solubility Ratio of CBOD <sub>5</sub>
SNiP	Stroitelnye Normy i Pravila (Russian construction codes and regulations)
t	Time
TSS	Total suspended solids
U <sub>max<sub>a</sub></sub>	Max growth rate phytoplankton
U <sub>max<sub>b</sub></sub>	Max growth rate bacteria
USEPA	United States Environmental Protection Agency
V	Volume
v/v	Volume/volume (% by volume)
WSER	Wastewater system effluent regulations
WSP	Waste stabilization pond
Y <sub>ca</sub>	Yield factor of phytoplankton produce from CO <sub>2</sub> consumed
Y <sub>ca</sub> OY <sub>oa</sub>	Carbon dioxide used/ oxygen produced in phytoplankton production
Y <sub>cb</sub> O <sub>ob</sub>	Carbon dioxide produced/ oxygen used in bacteria production
Z	Photic zone depth

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## Chapter 1 Introduction

Wastewater treatment standards, in an effort to protect human health and the environment, are becoming more stringent in many jurisdictions around the world. In Canada, the Wastewater system effluent regulations (WSER), have recently been implemented by Environment Canada to provide nationally consistent regulations for the treatment of municipal wastewater. The WSER states municipal effluent discharged to receiving environments must have concentrations below 25 mg/l carbonaceous biochemical oxygen demand – Day 5 (CBOD<sub>5</sub>), 25 mg/l total suspended solids (TSS), 0.02 mg/l total residual chlorine, and 1.25 mg/l un-ionized ammonia. These regulations are to supersede provincial and territorial regulations, and are more stringent than existing regulations in many jurisdictions. However, at the time of this thesis publication, the Northwest Territories, Nunavut, and northern regions of Quebec and Newfoundland and Labrador are exempt from the WSER. It was recognized that the climate and socio-economic conditions in these regions made the application of the WSER particularly challenging, and that further research was required to define appropriate standards and technologies for these cold regions. Dalhousie University formed a research partnership with the Government of Nunavut in 2010 to assess the performance of wastewater systems in the territory and develop cost-effective treatment upgrade strategies. The Arctic is a challenging environment to successfully implement municipal infrastructure. The infrastructure solutions must be technically sound as well as work within constraints imposed by climatic, geographic, social, and economic challenges of the Arctic.

The average annual temperature in the arctic is sub-zero, and daily average temperatures exceed 0 °C for only two to three months a year (late June until early September). The Arctic has a short summer season that has a high number of daylight hours, and the most northern of latitudes experience continual 24 hour sunlight during the summer. Rarely do arctic communities experience greater than 300 degree days above 5 °C, and the most northern communities, such as Pond Inlet, typically experience less than 100 (Environment Canada, 2014a). The Arctic also covers a vast land area, resulting in great variation of climates amongst communities, and engineering solutions might not be broadly applicable to meet the requirements across the Arctic.

Communities in the Canadian Arctic Territory of Nunavut are remote and small (ranging from 140 to 7500 persons) with limited accessibility. There exists no inter-community road network, and while air transport is available continually throughout the year, communities are only ocean accessible in mid-summer until early fall. Communities receive one or two sea shipments of supplies and equipment during the short ice-free period. The limited accessibility presents a challenge for supporting the construction, operation, and maintenance of infrastructure as equipment and services are limited in these communities, and most specialized equipment and services must be sourced externally. Logistically this is complicated by the limited capacity and high cost of air freight, and the short duration of the more economical sea shipments.

Generally, the communities became inhabited by Inuit who used to live a migratory life in the local vicinity, however in some extreme cases, such as Grise Fiord, Inuit were moved from more southern locations to establish a permanent settlement (Bonesteel, 2006). These settlements were seen by the Canadian government as essential for maintaining sovereignty over the territory by maintaining a population presence as well as necessary for guaranteeing inhabitants' food supplies and providing essential services like health care and education, which were now seen as responsibilities of the Canadian Government (Bonesteel, 2006). The formation of permanent settlements has had a profound effect on their way of life. The culture is largely preserved but is still adapting to life in a stationary community. There is a high rate of unemployment, with over 20% of the employable workforce being unemployed. (Statistics Canada, 2006). This high unemployment is a result of limited economic opportunities in these remote communities. Consequentially there is limited wealth, and a ongoing struggle to maintain community infrastructure. The infrastructure is primarily supported by the federal government at this time.

Nunavut's education system is substandard in Canada with only 43% of Nunavut's total population and 34% of Nunavut's rural population (exclusion of Iqaluit from statistics) having high school diplomas (Statistics Canada, 2006). These statistics suggest that there is a probable void in local skilled labour for the design, construction, and operation of municipal infrastructure. The limited skilled labour pool further strains



the infrastructure costs in the North because external labour is often required, and comes at a higher cost.

Operational costs of infrastructure are also much higher because of an increased cost of energy and labour, and price of consumables. The average energy cost in Nunavut is 75 cents/kwh (CBC, 2011) compared to 11 cents/kwh in the rest of Canada (Canadian Electricity Association, 2009). The high power costs are due to a dependence on diesel generators. Operational staffing of complex infrastructure is also potentially expensive because these small communities are unlikely to have local people with the skill set required to operate and maintain it. If the process requires consumables the cost of operation increases greatly due to high shipping costs. Finally, costs in the case of failure of complex infrastructure are high as parts and/or expertise will most likely be brought in from the South.

It is also important to note that the Arctic is facing rapid climate change, and experiencing warmer average temperatures (NSIDC, 2015). Environment Canada's *Climate Trend and Variation Bulletin* (2014b) consistently depicts a warming trend in the Arctic. Infrastructure will need to be developed to meet the challenges posed by the changing climate.

The combined impact of climatic, social, and geographic factors are culpable for the failure of mechanical wastewater systems in northern communities, such as Iqaluit and Pangnirtung, to meet treatment performance and economic targets. With the poor history of conventional mechanical systems in the North, passive treatment systems, such as Waste Stabilization Ponds (WSPs), remain the most utilized and trusted wastewater management solution.

Northern water and wastewater systems differ from those commonly found in more southern latitudes because of the cold climate and year round permafrost. These conditions make buried infrastructure expensive or impractical, and as a result the wastewater infrastructure solutions common in southern latitudes, such as septic fields and piped water conveyance systems, are rarely used. A few communities, such as Iqaluit and Resolute, have wastewater and water piped conveyance systems. Resolute, Nunavut has a utilidor system consisting of insulated and heat traced pipes above ground and Iqaluit, Nunavut has a buried pipe system with strategically placed water reheat station.

Both piped conveyance systems are expensive and challenging to install, maintain and operate, and as a result most arctic communities use trucked water and wastewater systems. Households in communities operating with trucked water systems have holding tanks for their potable water and wastewater, and regular delivery and disposal of the respective products is operated by the hamlet. The inhabitants must ration their water until the next scheduled delivery, and the decreased access to potable water results in a decreased per capita water use and therefore a more concentrated wastewater (Smith, 1986).

In Nunavut, 22 of 25 communities use passive wastewater treatment technologies for their municipal wastewater solution. These are either WSPs or treatment wetlands, or a combination of the two. WSPs are used internationally as a treatment solution for municipal and industrial wastewater and have been demonstrated to be effective. However, their effectiveness depends on wastewater quality, environmental conditions, system design and operation. Wastewater stabilization ponds are shallow engineered water basins that are used to detain liquid waste from industries or municipalities and provide treatment prior to discharge to the environment. WSPs in Nunavut are generally designed as single cell ponds that are 2-3 m deep. The cold winters dictate that these systems are operated as intermittent, or controlled-discharge systems, as the surface is frozen for 9-10 months a year. The ponds are sized to have the capacity to retain 10 months of wastewater (a winters worth), and are decanted during the summer.

Arctic WSPs are atypical in terms of climate, operation and design when reviewing WSP literature; there is also limited information available on their performance and process dynamics available in peer reviewed journals. Very little research has been performed on any systems operating in the Canadian Arctic in the last 30 years. The objectives of this thesis were to:

- (i) characterize the treatment performance of current WSPs operating in arctic regions for the removal of WSER regulated contaminants,
- (ii) identify environmental and operational variables that influence biological treatment processes, and
- (iii) develop recommendations for the design and upgrading of future and current arctic WSP systems.

Chapter 3 provides results from a four year field study examining the treatment performance of four operational arctic single-cell WSPs in Nunavut for the removal of WSER regulated deleterious substances. Results are discussed in the context of system design and operation, and climate. This study also highlighted the impact of several design variables on treatment performance and elucidates potential WSP design features that are likely to improve treatment performance

Chapter 4 presents results from a series of mesocosm-scale experiments where model WSPs were operating under varying carbon loading rates, temperatures and irradiance conditions representative of the Arctic. Phytoplankton growth, oxygen state and CBOD<sub>5</sub> removal were closely monitored. It was observed that phytoplankton growth was sensitive within the range of environmental and operational conditions common in arctic WSPs, and the presence of an active phytoplankton population plays a critical role in the development of a facultative oxygen state and CBOD<sub>5</sub> treatment performance.

In Chapter 5, a process-based model that simulates oxygen dynamics and CBOD<sub>5</sub> removal in arctic WSPs is presented. In development of the model, it was discovered that common mathematical functions used for the modeling of depth integrated phytoplankton growth and populations in current ecosystem and WSP models performed poorly in representing phytoplankton in arctic systems because of the unique conditions (shallow, poorly mixed, and eutrophic). The development and the formulation of a depth integrated phytoplankton population model for shallow, poorly mixed eutrophic waters is presented. The model was calibrated using experimental results generated in Chapter 4. A sensitivity analysis was conducted to assess the primary factors that control CBOD<sub>5</sub> removal in arctic WSPs. The model highlighted that CBOD<sub>5</sub> removal performance is highly sensitive to temperature and phytoplankton activity. The model was used to assess the treatment capacity of single cell Arctic WSPs to provide direction on future design and operation.

In Chapter 6 major conclusions are presented, and recommendations are provided for refining arctic WSP design guidelines, and enhancing treatment performance. Recommendations for future avenues of research are also provided

## **Chapter 2 Literature Review**

### **2.1 Municipal Wastewater Characteristic and Impacts**

The creation of stationary human settlements has created a challenge with respect to the disposal of human sanitary waste. Sanitary waste, especially when concentrated in an area, poses a potential human and animal health hazard due to the potential presence of pathogens (Droste, 1997). To mitigate this hazard, water has been used a vehicle to remove human sanitary waste away from populated centers. Generally the wastewater following collection (and potentially treatment) is returned to a river, lake or ocean. Although the removal of sanitary waste from populated centers alleviates some of the human health risk associated with the presence of pathogens in the waste, its disposal has potential environmental impacts when discharged in high volumes without sufficient treatment.

Sanitary wastes contain high concentrations of nutrients, such as nitrogen and phosphorous, as well as elevated organic carbon concentrations, and when large quantities of nutrients are discharged to the environment they may have localized negative impacts on ecosystem health. In terms of receiving water environments, the discharge of wastewater with high nutrient concentrations may accelerate eutrophication (nutrient enrichment of an ecosystem) which is characterized by increased phytoplankton growth, and is recognized as disruptive and detrimental to the functionality of aquatic ecosystems (Smith et al., 1999). High concentrations of organic compounds being discharged to a receiving water may result in a localized depletion of oxygen (hypoxia) in the water body because of aerobic decomposition by bacteria (Servais et al., 1999). The most recognizable impact of high organic carbon discharge is fish kills as a result of hypoxic conditions, and if the loading is persistent “dead zones” develop (Hladyz et al., 2011). Additionally, wastewater may have compounds that are toxic to fish and wildlife, such as unionized ammonia (Thurston et al., 1981). To mitigate the impacts from wastewater discharges, a variety of technologies have been developed to treat municipal wastewater, ranging from passive systems that rely solely on natural processes to active systems that utilize mechanical devices and chemical additions to improve treatment

performance. One of the passive treatment technologies that is widely used is Waste stabilization ponds (WSPs).

## **2.2 Waste Stabilization Ponds**

Waste stabilization ponds are engineered basins that utilize natural processes for treatment. Within a WSP, biological processes involving phytoplankton and bacteria remove nutrients, reduce organic carbon concentrations and attenuate pathogens prior to environmental discharge (Kayombo et al., 2004). This technology represents one of the simplest, most reliable and cost effective municipal wastewater treatment solutions for rural communities (Kayombo et al., 2004; Mara et al., 1992). Cells, or ponds, may be linked in series or parallel to create a treatment system that dependably meets treatment performance requirements and has sufficient capacity, and redundancy (Kayombo et al., 2004; Shilton, 2005). The goal of a WSP design is not to remove all nutrients or all the pathogens from the wastewater, but to reduce them to an acceptable level for discharge to the environment (Ramalho, 1977).

Waste stabilization ponds are designed to have continuous or intermittent (controlled) discharge. Continuous discharge WSP effluent flows are determined by the volume of water (wastewater and precipitation) entering the system; the system operates with a constant volume and there are no provisions for storage (NGSMI, 2004). Intermittent discharge WSPs possess the capacity for storage, and provide flexibility by allowing the operator to control the timing and quantity of discharge. Controlling the timing allows for effluent quality management, and/or the timing of discharge to occur with favorable receiving water assimilation capacity conditions. When the climate varies greatly over the year, the WSP treatment performance and effluent quality also fluctuates, and the timing of discharge can be controlled to occur with the best effluent quality. This performance variability is particularly characteristic of temperate climates, where biological treatment is limited during the winters resulting in decreased treatment performance and poor water quality during and immediately following the winter (Gerardi, 2015). Intermittent discharge is also a potentially useful WSP attribute for reducing the impact on the receiving environment when the receiving water's

assimilation capacity fluctuates due to wet and dry seasons or biological activity (NGSMI, 2004; Libhaber and Orozco-Jaramillo, 2012).

To provide the desired treatment performance and process stability, WSPs are typically designed as a series of 2-5 cells. The creation of cells in series leads to better hydraulics (Marais, 1974; Persson, 2000), and a general improvement in water quality from the first cell to the last cell. As water proceeds through the cells, the water characteristics change (such as Total Suspended Solids (TSS)), and the treatment focus changes. To optimize treatment performance, cell design along the treatment train is modified to promote treatment processes consistent with the treatment objectives (Shilton, 2005). The first (primary) cell of a WSP system is typically anaerobic because the oxygen consumption by the bacteria, the main decomposers, is greater than the sum of atmospheric aeration and photosynthetic oxygen production, which typically is limited in the first cell (Alexiou and Mara, 2003). As wastewater travels through the series of cells, the oxygen concentrations typically increase as oxygen demanding substances are consumed by bacteria, and as larger phytoplankton populations increasingly contribute dissolved oxygen. Cells sequentially after the primary cell become facultative, or fully aerobic, once the oxygen demanding substances have been sufficiently oxidized.

### **2.2.1 Classification by Oxygen Concentration**

Cells are typically classified by their oxygen state (anaerobic, facultative or aerobic) as oxygen is a vital element in many biological and chemical reactions. The oxygen state of a WSP can be associated with general water quality properties and dominant wastewater treatment processes. However, the oxygen state of a WSP is dynamic, and annual variations in climate or operation may result in changes in the oxygen state, and as a result alter the classification of a WSP (Shilton, 2005).

#### **2.2.1.1 Anaerobic**

The first cell of most WSPs will typically be anaerobic, but the oxygen state is dependent upon the prevailing climate, cell design, and organic loading rates (Kayombo et al., 2004). Cells designed to operate as anaerobic are deeper than facultative or aerobic cells, and typically have operating depths of 3-5 m (Mara et al., 1992). The retention time

in the primary cell is generally 1 to 4 days, and rarely exceeds 5 days. This first cell is anaerobic because it receives the raw wastewater, and the organic carbon loading rates are high, typically greater than 100 g BOD/m<sup>3</sup>/day (Kayombo et al., 2004).

Phytoplankton typically does not thrive in these cells due to adverse water chemistry, and their lack of presence reinforces hypoxic conditions (Athayde, 2001). The main treatment objective for the primary cell is a reduction of TSS, and this is promoted by a deeper cell design resulting in a reduced resuspension of settled particles, and ample volume to create calmer water conditions, however they also contribute to the removal of organic carbon (Shilton, 2005). Despite the short retention times, the effluent of an anaerobic primary cell typically has concentrations of TSS and CBOD<sub>5</sub> that are 40-70% lower than the raw wastewater. The main removal mechanisms for organic carbon are methanogenesis, transformation to inorganic carbon, and sedimentation (Picot et al., 2002). Picot et al. (2002) identified methanogenesis to be the most important removal mechanism occurring in anaerobic ponds, accounting for 74% of the organic carbon removal.

#### 2.2.1.2 Facultative

Primary cells can be facultative under certain operational conditions, however facultative cells are more commonly located after the anaerobic cell of a WSP system (Kayombo et al., 2004). Facultative cells generally have a design operating depth of 1-2 m and retention times of 2-8 weeks depending on climate and operational factors (Marais, 1970; Mara et al., 1992). Facultative ponds are characteristically green in color due to the large phytoplankton population in the surface water (Shilton, 2005). Facultative ponds have an aerobic surface as a result of natural aeration along with oxygenation from photosynthetic activity of phytoplankton (USEPA, 2002). Despite the surface being aerobic, at depth the pond is generally anoxic due to the lack of phytoplankton activity, high (facultative and aerobic) bacterial activity, and the distance from the air/water surface. The primary treatment objective of facultative ponds is the oxidation of organic carbon (Shilton, 2005), and phytoplankton play a vital role by providing the oxygen necessary for the bacterial metabolism that oxidizes the carbon (Mara et al., 1992). Facultative cells are designed based on areal carbon loading rates (kg CBOD<sub>5</sub>/ha/day) to ensure there is sufficient oxygen available for the efficient oxidation of organic carbon

(Mara et al., 1992). Facultative cells are responsible for the vast majority of biological treatment, specifically CBOD<sub>5</sub> removal, occurring in WSPs (Shilton, 2005). Facultative cells have the most biological diversity, and treatment processes include: oxidation of carbon compounds by the bacteria in the oxygenated surface waters; methanogenesis at the bottom of the WSP; nutrient uptake by bacteria and phytoplankton; volatilization of gases; and the continued sedimentation of particulates (Mara et al., 1992).

#### 2.2.1.3 Aerobic

Aerobic cells are often referred to as maturation ponds or polishing ponds. As a result of treatment in the previous cells, aerobic cells receive water significantly lower in organic and nutrient concentrations. Aerobic cells are oxygenated throughout the depth and are typically shallow (less than 1.5 meter). The retention time is generally 5-15 days (Mara et al., 1992). These cells have a greater diversity of phytoplankton because of improved water quality (Shillinglaw and Piertese, 1977) The primary treatment objective of aerobic cells is the removal of pathogens (Von Sperling, 2005). The removal of pathogens is a result of a combination of UV disinfection and high pH (exceeding 9) (Curtis et al., 1992). Although aerobic cells are primarily tasked with the removal of pathogens, they promote phytoplankton growth, and as a result are effective at removing nutrients, specifically phosphorous (Silva et al., 1996) and nitrogen (Wriglet and Toerien, 1990; Somiya and Fujii, 1984) and further reducing dissolved organic carbon prior to discharge (Sah et al., 2012). Despite these benefits, aerobic cells have been cited to create challenges, specifically high TSS concentrations and elevated particulate organic carbon attributable to phytoplankton (Kim and Kim, 2000).

### **2.3 Biological Treatment Processes**

Biological treatment provides the foundation of organic carbon removal in most wastewater treatment systems, and is critical for the effective performance of WSPs. Effective biological treatment is dependent upon the system supporting a bacterial community that utilize (digest) complex carbon compounds to produce their energy and biomass. Additionally bacteria assimilate and transform nutrients, such as nitrogen and phosphorous, reducing nutrient concentrations in the effluent. The metabolism of bacteria



can be divided into two groups: aerobic and anaerobic, and is predicated on the availability of oxygen. In addition to bacteria, phytoplankton play a vital role in WSP treatment as they provide the oxygen for aerobic bacterial processes.

### **2.3.1 The Role of Bacteria**

Anaerobic digestion occurs in the absence of oxygen, and is characteristically found at depth in facultative WSPs or in anaerobic WSPs (Shilton, 2005). Under anaerobic conditions, organic carbon is digested by bacteria, through the process of hydrolysis, acidogenesis, and methanogenesis to produce methane, carbon dioxide, water and biomass (Chan et al., 2009). During the process anaerobic bacteria may utilize sulfate to oxidize organic carbon, resulting in the formation of the undesired by-product hydrogen sulfide (Oleszkiewicz and Sparling, 1987). Hydrogen sulfide has a putrid odour, and has been linked to adverse health effects such as headaches and fatigue (OSHA, 2005).

Aerobic digestion occurs in the presence of oxygen, and organic carbon compounds are oxidized with oxygen to create biomass and CO<sub>2</sub> (Chan et al., 2009). Aerobic digestion dominates in the presence of oxygen because it is more metabolically efficient (Metcalf and Eddy, 2003). Aerobic digestion benefits from being more rapid and having a shorter start-up time than anaerobic digestion (Leslie Grady et al., 2011). However, since aerobic digestion is metabolically more efficient, more biomass is generated and a greater amount of sludge is produced (Leslie Grady et al., 2011).

In general, WSP systems are designed to promote aerobic conditions in the majority of the system, because optimization of the organic carbon removal rate is a priority (Shilton, 2005). Additionally there is the general desire to avoid the production of noxious gases formed by anaerobic digestion, especially if located close to a community (Oleszkiewicz and Sparling, 1987).

### **2.3.2 The Role of Phytoplankton**

Phytoplankton is the broad classification of small free floating autotrophs that dominate the water column of most water bodies, including WSPs. Phytoplankton, in the presence of light, perform photosynthesis, a process where CO<sub>2</sub> is reduced and nutrients

are incorporated for the synthesis of carbohydrates. During photosynthesis oxygen is expelled from the cell as a waste product. In the absence of sufficient CO<sub>2</sub> some species are capable of incorporating other forms of carbon such as acetate or bicarbonate, however growth is much less efficient (Wiedeman and Bold, 1965; Azoz, 1982).

The production of O<sub>2</sub> by the phytoplankton is integral to the treatment performance of the WSP as it is required by the aerobic bacteria for the decomposition of detritus. The presence of healthy phytoplankton populations is symbiotic with bacteria as the phytoplankton provide O<sub>2</sub> to the bacteria and the bacteria respire and produce CO<sub>2</sub> which is utilized by the phytoplankton (Shilton, 2005).

Phytoplankton assimilate forms of dissolved nitrogen and phosphorous to construct new cells, removing these constituents from the water column. When phytoplankton die, these cells settle to the bottom of the WSP effectively removing the nutrients it has accumulated from the water column, sequestering them in the sediment, and reducing the amount of nutrients exported to receiving environments. (Schnoor, 1996). Phytoplankton preferentially uptake ammonium over nitrate as it is more energy efficient to incorporate (Dortch, 1990). This uptake of ammonium is an additional benefit because ammonia, the associated speciation, is highly toxic to most aquatic organisms (Thurston et al., 1981; Arthur et al., 1987), and as a result should be largely removed before discharge to the environment.

Phytoplankton exert a strong influence on the chemistry of WSPs by assimilating nutrients and carbon, either as CO<sub>2</sub>, bicarbonate or an organic carbon form, and expelling oxygen. Carbon dioxide assimilation has a large impact on the pH of a WSP. The decrease in CO<sub>2</sub> and bicarbonate concentrations due to photosynthesis modifies the carbonate buffering system, increases the pH by decreasing H<sup>+</sup> relative to OH<sup>-</sup> rendering the system more basic. The increase in pH influences other chemical constituents as their balances are pH dependent. The two major pH dependent processes of interest in WSPs are the ammonia- ammonium equilibrium and phosphorous precipitation through complexation with metal ions (Tchobanoglous and Schroeder, 1985).

The ammonia-ammonium ion equilibrium is dependent on pH with high pH (> 9) favouring the ammonia species, which exists in a gaseous state (Schnoor, 1996). The existence of a greater concentration of ammonia allows for a greater volatilization rate,

which is a mechanism for nitrogen removal from the water column. However, there are conflicting studies regarding the role of volatilization in nitrogen removal (Rockne and Brezonik, 2006). Some studies suggest that volatilization is responsible for over 90% of nitrogen removal (Pano and Middlebrooks, 1982), while some suggest volatilization is not a major contributor (Ferrara and Avci, 1982; Camargo and Mara, 2010).

In addition, shifts in pH that are attributable to phytoplankton growth has implications on the solubility, and biological availability, of phosphorous. Complexation reactions that result in the precipitation and sequestration of phosphorous to the bottom of a WSP cell are pH dependent. Phosphorous is more likely to form precipitates with aluminum and iron under acidic conditions, while under alkaline conditions phosphorous is likely to complex with magnesium and calcium (Reddy and D'Angelo, 1994).

Many WSP systems contain a great diversity of phytoplankton species, especially in the later cells (Amengual-Morro, 2011). The most common genera found in WSPs are from the phylum Chlorophyta; *Scenedesmus*, *Chlorella*, *Microactinium*, and *Chlamydomonas* and phylum Euglenophyta (Shillinglaw and Pietere, 1977; Picot et al., 1993; Pearson et al., 1987). In a turbid environment that is nutrient rich, such as a WSP, small motile phytoplankton with high growth rates tend to be most abundant because their small size and corresponding high surface area to volume ratio gives them a competitive advantage in nutrient assimilation. Their motility provides the ability to maintain an optimum position in the photic zone (Shilton, 2005). However, in an environment with stable irradiance and limited mixing non-motile phytoplankton may dominate. Pearson et al. (1987) found that the non-motile genera, *Chlorella*, physiologically, can outcompete many of the motile phytoplankton genera if it can maintain a suitable position in the photic zone and has been found in many maturation cells (Shilton, 2005).

In highly productive systems, where photosynthesis proceeds at a rapid rate, CO<sub>2</sub> can become depleted as it is utilized by algae. This has been found to occur in WSPs and as a result, species that can obtain carbon through other sources such as carbonates (Wiedeman and Bold, 1965), or species that have the ability to concentrate CO<sub>2</sub> in low CO<sub>2</sub> environments (Azoz, 1982), have a competitive advantage. They tend to dominate

WSP environments with lower carbon concentrations and high nutrient levels, such as aerobic cells.

Ammonia and sulphides have been found to have a negative effect on phytoplankton growth (Pearson et al., 1987; Konig et al., 1987). This has a strong influence on phytoplankton population dynamics in WSPs as there is often a high influent ammonia concentration and elevated levels of sulphide in the primary treatment cells due to prevailing anaerobic conditions (Pearson et al., 1987). It has been found that there are genera that are more tolerant of elevated ammonia and sulphide, such as *Chlorella* and *Chlamydomonas*, and they tend to dominate more heavily loaded WSPs (Konig et al., 1987).

### **2.3.3 Environmental and Operating Factors that Influence WSPs**

Temperature is the largest influencing factor on the biological treatment performance of a WSP, because chemical reactions, and by association biological rates, are exponentially dependent on temperature (Tchobanoglous and Schroeder, 1985). In general, a 10 °C increase in temperature results in an increase in reaction rate by 2-3 times in biological systems (Cossins and Bowler, 1987). As a result, bacterial and phytoplankton metabolism greatly increase with temperature, as does the associated removal rate of organic carbon and nutrients. Of particular importance to arctic WSPs, anaerobic digestion has been found to be very sensitive to temperature, and at temperatures below 10 °C has been demonstrated to proceed very slowly (Juanico et al., 2000). This would suggest that anaerobic digestion in arctic WSPs is limited.

In the case of WSPs, irradiance is also an influential factor on biological treatment because affects the photosynthetic rate of phytoplankton, with higher light oxygen production (Falkowski and Owens, 1978). The increased oxygen production promotes aerobic conditions, and organic carbon removal rates are expected to increase. As an additional benefit, irradiance provides heat to the WSP increasing the temperature and biological reaction rates (Klemetson, 1983).

Organic carbon loading and concentrations in the WSP have also been shown to have an impact on performance. As has been discussed, aerobic digestion in facultative WSPs is required for the efficient removal of organic carbon. If organic carbon areal

loading rates are excessive there is a prevailing oxygen deficit for the bacteria and sub-optimal organic carbon removal rates are realized (Kayombo et al., 2004). Additionally, high organic carbon concentrations, which is a potential result of high areal loading rate and/or poor treatment performance, has been observed to impede phytoplankton growth (Athayde, 2001). As mentioned earlier facultative systems are dependent on phytoplankton in order to maintain aerobic conditions in the surface waters (Shilton, 2005).

### **2.3.4 Performance of WSPs**

Waste stabilization ponds have been proven to provide cost effective wastewater treatment in rural areas where land costs are low (Tsagarakis et al., 2001). The impact of discharging effluent from WSPs, like any other wastewater treatment solution, is dependent on the receiving environment and the quantity and quality of effluent. WSPs can provide a level of treatment that is equivalent or exceeds conventional secondary treatment mechanical systems (Finney and Middlebrooks, 1980; USEPA, 2002). However, caution has to be exercised as overloaded or improperly designed systems may have effluent that consistently or periodically fails to meet wastewater treatment regulations, and can have a negative impact on the environment and human health (Middlebrooks, 1987; Okoronkwo and Odeyemi, 1985).

Properly designed and operated WSPs have been shown to have effluent quality which under most conditions meet regulatory requirements for TSS and CBOD<sub>5</sub> (Finney and Middlebrooks, 1980; USEPA, 2002). WSPs can also efficiently remove nutrients with nitrogen removal rates of 80% (Ferrara and Avci, 1982) and phosphorous removal rates of greater than 60% (Mara, 1992) documented in previous studies. However, there are many systems in operation that occasionally or routinely exceed guidelines because of incomplete design (Finney and Middlebrooks, 1980).

#### **2.3.4.1 Performance of arctic WSPs**

There have been very few published studies on the performance of WSP systems in arctic, or sub-arctic regions. Miyamoto and Heinke (1979) investigated the performance of a municipal wastewater WSP operating in the arctic community of Inuvik

in 1971. The system is a single cell WSP with a surface area of 25 ha and an operation depth between 0.3 and 2.5 m depending on season. The Inuvik WSP in 1971 had been in operation for 15 years, and was treating wastewater from a population of 3500 serviced by a utilidor system. Miyamoto and Heinke (1979) reported that the WSP reduced the BOD concentrations by 80% and 71% in the summer and winter respectively, and consistently reduced the TSS concentrations by 85%. TSS concentrations were typically less than 30 mg/l and BOD<sub>5</sub> concentrations were less than 40 and 51 mg/l in the summer and winter respectively. Prince et al. (1995) studied an intermittent discharge WSP system (4 anaerobic cells, one facultative cell and one maturation cell) operating in northern Alberta, Canada. They reported that effluent quality from the system was superior to that of most conventional systems, with the exception of TSS levels which was attributed to algae populations. In a treatment performance assessment of a WSP consisting of 4 cells in series with supplemental aeration located in Fort Nelson, BC, Canada, Prosko et al. (2007) found it achieved a high quality effluent with concentrations of BOD<sub>5</sub> and TSS below 24 and 22 mg/l, respectively. Although the above studies provide insight into the potential performance of WSPs in cold climates, the performance results are not directly transferable to systems operating in Nunavut and the Northern Arctic. Results lack transferability because the three systems described have greatly different operating conditions than single cell WSPs operating in Nunavut, and the climate in the Northern Arctic is still significantly cooler in the summer than the climates of communities in the referenced studies.

## **2.4 Arctic WSP Design Guidelines**

Dawson and Grainge (1969) were the first to propose WSP design criteria for Arctic and Sub-arctic regions. Smith (1986) and Heinke et al. (1991) continued on from Dawson and Grainge's (1969) work, and produced design guidelines that have been used for the past 30 years.

Dawson and Grainge (1969), in their report, recommended summer operation depths (1.2 to 1.5m) and winter operation depths of (1.8 to 2.4 m) and recommended 8 to 12 month detention times. Dawson and Grainge (1969), from their experience in Yellowknife and Inuvik, recommended a BOD loading of 22 kg/ha/day, which is

equivalent to a hectare of surface area for every 250 person, for long detention WSPs; this corresponds with recommendations by the USEPA (1983) for areal loading rates of 11-22 kg/day/hectare for systems operating under conditions with winter temperatures below 0 °C. The USEPA (1983) also suggests detention times greater than 180 days, especially for winter effluent. Dawson and Grainge (1969) state that, for a loading rate of 22 kg/ha/day it is reasonable to expect an 80% BOD<sub>5</sub> removal and a 4 log coliform reduction for single cell WSPs operating in Sub-arctic regions. Smith (1986) and Heinke et al. (1991) also recommended that organic loading rates not exceed 22 kg/day/hectare to achieve a high level of treatment (80% removal of TSS and BOD<sub>5</sub>). They also recommend that single cell WSP systems be designed for a 365 day storage time.

The use of multi-cell systems is widely accepted to result in improved treatment performance and WSP effluent quality (Heinke et al., 1991; Dawson and Grainge, 1969). Barjenbruch (2005) in his assessment of the performance of a small WSP in Germany suggested that optimizing the primary cell would be one of the most effective ways of improving effluent quality. The USEPA (1983) and SNIIP(1996), a Russian regulatory body, also recommend that a WSP system consist of at least 3 cells in series.

## **2.5 Models of WSP Performance**

Performance and design models for WSPs fall into three general categories; (i) empirical, (ii) reactor (kinetic) based or (iii) process based. Empirical models are generated by fitting linear or non-linear regression equations to predict treatment performance as a function of environmental or operational factors. Kinetic models incorporate the physical hydraulic conditions (well-mixed, plug flow, or plug flow with dispersion) and utilize determined reaction rates related to the prevailing environmental and operational conditions to predict treatment. Process based models attempt to utilize knowledge of the biological, physical and chemical processes that are deemed important to describe the system mathematically, generally as a system of differential equations. All three forms of models can be valid and useful for design and performance prediction. However, they differ in data requirements, ability to be generalized, and ease of application.

### **2.5.1 Empirical**

Empirical equations for the treatment performance of WSPs for BOD and nutrient removal have been developed by numerous authors to aid in the design of WSPs (e.g. McGarry and Pescod, 1970; Larsen, 1974; Gloyna, 1976). These equations attempt to relate loading, hydraulic and environmental factors (e.g. temperature) to the observed treatment performance, and they range in complexity. McGarry and Pescod (1970) is the most simplistic as it only uses the BOD<sub>5</sub> areal loading rate, while Larsen (1974) is on the other side of the spectrum and is based on dimensionless design. Larsen (1974) developed a complex equation consisting of definable (but complex) design variables that might be used to determine the necessary surface area to meet treatment performance for provided conditions.

### **2.5.2 Reactor Based**

Reactor based models functionally attempt to advance purely empirical relationships by incorporating the hydraulic conditions, assessing final conditions through rate controlled processes, and contain a time component. The choice of hydraulic conditions, kinetic formula, and kinetic rate constants are rooted in physical processes but in practice kinetic equations still may contain a large amount of empiricism. The choice of hydraulic conditions (completely mixed, partially mixed or plug flow) potentially has a large impact on model results, and can be difficult to quantify. Many models describe the hydraulics as either completely mixed or plug flow because it greatly simplifies the problem. However, in reality, most hydraulic regimes fall somewhere between the two conditions (Thirumurthi, 1974).

Predicting the kinetics of a reaction requires the specification of a reaction rate constant. However, these “constants”, such as kinetic rate of CBOD<sub>5</sub> removal, encompass a set of complex processes, some which might not be fully understood and/or are difficult to measure and determine. In practice, rate constants used in kinetic models are experimentally derived and are only representative of the experimental conditions, such as pH and temperature, under which they were derived. Reported values in the literature can vary greatly depending on the context of the experiment (Marais, 1974; Finney and Middlebrooks, 1980). Natural systems have a plethora of water quality



parameters that can affect the reaction rates making it challenging to determine and apply representative rate constants (Thirumurthi, 1974).

### **2.5.3 Process Based**

The advancement and access to personal computers has provided the possibility of developing comprehensive mathematical process-based models of complex systems. Sophisticated models have been developed for treatment wetlands (Langergraber et al., 2009) and activated sludge systems (Henze et al., 1995). Process-based models attempt to decouple and represent the important processes occurring in a system that are entangled and lumped in both empirical and statistical models. The largest advantage of process based models is that they explicitly state assumptions about the system and the processes, unlike purely statistical or empirical models (Cuddington et al., 2013). Process-based models are a tool that can be used to gain insight into treatment mechanisms, identify system limitations, aid in design optimization, and ultimately predict performance. The utilization of process-based models is cost effective, however the development is difficult and data intensive (Cuddington et al., 2013) There are many existing process-based models of WSPs, however they tend to be very focused on specific processes such as predicting the oxygen balance during WSP thaw in cold climates (Banks, 2003), nitrogen dynamics (Ferrara and Avci, 1982), stratification dynamics (Gu and Stefan, 1995) or BOD dynamics (Giraldo and Garçon, 2002). The specificity of model design limits the wide applicability of these models and they are generally most suitable for gaining further insight into the processes occurring in the WSPs. There are a few models that attempt to model general WSP dynamics, incorporating as many biological, physical, and chemical processes as possible (Fritz, 1979; Beran and Kargai, 2005; Moreno-Grau et al., 1988; Sah et al., 2011). However data for a wide range of systems are not readily available making it difficult to perform robust model validations, which are necessary for wide acceptance of a model. WSPs still lack a comprehensive mathematical process-based model that is widely accepted by the engineering design community (Sah et al., 2012).

#### **2.5.4 Model Performance**

Finney and Middlebrooks (1980) reviewed models for the prediction of BOD removal from WSPs, which included empirical models proposed by McGarry and Pescod (1970) and Larsen (1974), and reactor based models proposed by Marais (1974) and Thirumurthi (1974). They found that none of the models adequately predicted the treatment performance of four WSPs that ranged in latitudinal location in the interior United States. Ellis (1995b) in his attempt to develop an empirical regression model for BOD and fecal coliform treatment in the Cayman Islands examined existing models, and determined that none were appropriate. Through examination of the work of Finney and Middlebrooks (1980), Middlebrooks (1987), and Ellis (1995a; 1995b) it is apparent that caution needs to be exercised when using empirical or reactor based design equations for the prediction of WSP treatment effectiveness because the models are heavily influenced by the climatic and operational conditions used to derive them. Middlebrooks (1987) concluded the best current design equation is an empirical loading model which has been developed with data from systems with representative local climatic conditions, which relates influent BOD loading to effluent BOD concentrations.

#### **2.5.5 Models of Arctic WSPs**

Limited development of design equations for arctic WSPs has occurred, and Smith et al. (1985) noted that the extreme climate of the Arctic renders readily used design equations unusable because of the much different climate the relationships were derived from. The USEPA (1983) guidelines reflect this; for cold climates they simply recommend adhering to a organic loading rate less than 22 kg/ha/day, and this loading rate has consistently been supported by others (Heinke et al., 1991; Smith, 1986). In a review of the literature, no design equations developed with consideration of arctic climates were found. Banks et al. (2003) created a model specific to arctic WSPs to examine the environmental and operational conditions that led to facultative/aerobic conditions at the onset of summer in an arctic WSP, however it has not been utilized to assess the treatment performance of arctic WSPs or tested more broadly on additional arctic WSPs.

## **2.6 Research Gaps**

Upon completion of a review of the literature, four broad research gaps with respect to the design and operation of single cell WSPs in the Arctic were identified:

1. No peer-reviewed treatment performance assessments of single-cell arctic WSPs, with the characteristics of those operating in the Nunavut, were identified.
2. The guidelines for arctic WSP design are very limited and do not reflect the diverse climate of the Arctic.
3. Design equations for WSPs lack broad application, and design equations for Arctic WSPs do not exist.
4. No validated comprehensive process-based model of WSPs exists, and no models have been developed with the consideration of assessing treatment performance of WSPs operating in the Arctic.

## **Chapter 3 Performance of Municipal Waste Stabilization Ponds in the Canadian Arctic**

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### **3.1 Introduction**

Nunavut is an arctic territory consisting of all the land mass of northeastern Canada above 60 °N and the regional islands. Nunavut has a small but widely distributed population with 32,000 inhabitants spread across 25 communities and over 2 million square kilometers; communities are extremely remote with no connecting roads making transportation of goods and labour expensive (Government of Canada, 2013). The population of hamlets in Nunavut ranges from 130 to 2800 with median and average populations of 1000 and 1200 people, respectively (Government of Nunavut, 2014). The arctic climate is characterized by long cold winters and short solar intense summers. Lakes and ponds remain frozen until late June or early July and begin to freeze in mid-September. All of Nunavut experiences continuous permafrost. Communities in Nunavut have average air temperatures below zero for all months except for the summer (June, July, August and, depending on location, September) when temperatures rise into the single digits (Environment Canada, 2014).

Simple infrastructure solutions tend to be preferred in the Canadian Arctic due to the environmental, financial and logistical constraints. In 21 of 25 hamlets in Nunavut the entire community is on trucked water and wastewater service, with residences and buildings receiving drinking water and sewage pick-ups every 2 to 3 days depending on the service schedule and water use (Nunavut Water Board, 2014). It should be noted that raw sewage tends to be more concentrated in communities on trucked water systems as

the decreased accessibility to water significantly lowers water use (90 litres/person/day) when compared to the Canadian national average of 274 litres/person/day (Smith, 1986).

Wastewater treatment systems in the territory are generally required to meet territorial effluent quality criteria of 180 mg/l Total Suspended Solids (TSS) and 120 mg/l biochemical oxygen demand (BOD<sub>5</sub>) (Nunavut Water Board, 2014). New national standards for municipal wastewater systems have recently been implemented for southern regions of Canada, with all municipal systems producing greater than 100 m<sup>3</sup>/d having to meet effluent quality criteria of 25 mg/l TSS, 25 mg/l carbonaceous BOD<sub>5</sub>, and 1.25 mg/l un-ionized ammonia (Government of Canada, 2012). These regulations have yet to be applied to northern Canada due to the limited knowledge of the performance of existing systems.

Currently, wastewater systems in northern Canada are designed according to specifications outlined in the *Cold Climate Utilities Manual* (Smith, 1986) with additional guidelines for sizing and constructing WSPs in the Arctic (Heinke et al., 1991). In most Nunavut communities, one year detention WSPs have been designed as controlled discharge storage ponds, sized to hold 365 days of wastewater generation, as discharge during the winter is challenging and effluent quality is expected to be poor due to limited biological activity. These systems were mostly designed to act as facultative pond systems, with operating water depths between 1.5 – 2.5 m. However, some WSPs are deeper (3 – 6 m), as the system design may have been constrained by topography and geology. The effluent from arctic WSPs is typically discharged in late summer or early fall just prior to freeze-up, allowing for approximately 60 days of wastewater treatment during the ice-free summer season. The available OLR recommendation for facultative WSPs operating in cold climates is 11 – 22 kg BOD<sub>5</sub>/ha/d (USEPA, 1983) in order to promote aerobic treatment environments. Aerobic biological processes are generally desirable in municipal wastewater treatment because they produce a better effluent quality, are less temperature sensitive, and have a shorter start-up time in comparison to anaerobic processes (Chan, 2009).

The effectiveness of WSPs for the treatment of municipal wastewater has been demonstrated in temperate climates (Finney, 1980; Shilton, 2005; Barjenbruch and Erler, 2005) and there are a number of guidance manuals for their design and operation

(e.g., USEPA 1983). The use of WSP technology for wastewater treatment in arctic regions, however, is poorly understood, and it is unclear if design practices used in southern jurisdictions are applicable. The climate of arctic regions likely poses the greatest constraint on treatment performance, but the influent wastewater quality, and operational regime (e.g., single cell treatment, controlled annual decant) will also influence effluent quality.

There is a paucity of published research concerning the performance of one year detention, single cell WSPs in the Arctic. There is also limited understanding of how these systems function during the summer treatment season with respect to the level of oxygenation and treatment kinetics. The overall goal of this study was to assess the performance of four municipal WSPs in the Canadian Arctic. Specific objectives of the research were to: (i) characterize the biogeochemical (e.g., pH, dissolved oxygen, temperature), environmental (e.g., degree days above 0°C, basin dimensions) and operational (e.g., daily loading rates) factors of arctic WSP systems and the potential for aerobic treatment, and (ii) determine the treatment performance and range of effluent quality from a one year detention arctic WSP. This paper focuses on the removal of CBOD<sub>5</sub>, TSS, and ammonia-nitrogen, as these are the parameters that will be regulated under the Canadian municipal wastewater regulations (Government of Canada, 2012).

## **3.2 Methods**

### **3.2.1 Study Sites**

All four communities in this study have trucked drinking water and wastewater systems, and represent a broad geographical distribution of communities as shown in Figure 3.1. The studied systems are also representative of WSP designs across Nunavut. Table 3.1 provides an overview of the location, population and climate (degree days) for each of the study communities while Table 3.2 provides general information on WSP design and operation conditions at each site. Degree days above 0 and 5 °C (as annual averages [1981-2010] for the ambient air temperature) are provided in Table 3.1 as they aid in describing the arctic climate's imposed limitations for biological treatment as it pertains to temperature and climatic variability across the arctic region. Included in Table

3.2 is organic loading rates (OLRs) of the systems, as determined by the influent organic carbon content per a unit area per a time (CBOD<sub>5</sub>/ha/d). The operation of each WSP is briefly discussed in the following section.

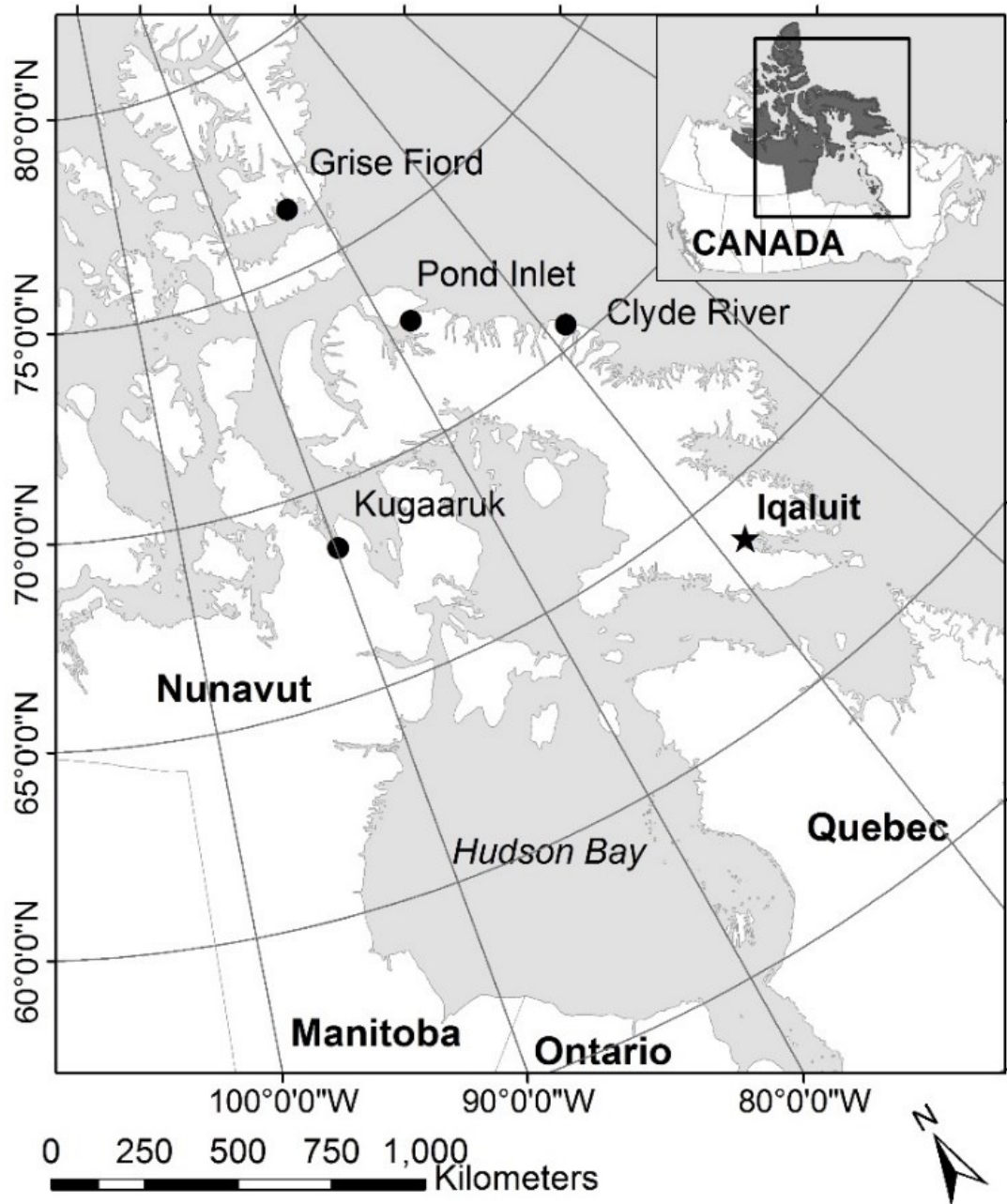


Figure 3.1 Map of geographical locations of study communities.

Table 3.1 Study communities' size, location and climate information.

Community	Location	Population <sup>1</sup>	Degree days above °0 C <sup>2</sup>	Degree days above °5 C <sup>2</sup>
Clyde River	70° 28' 26" N, 68° 35' 10" W	1004	382	64
Kugaaruk	68° 31' 59" N, 89° 49' 36" W	878	660	243
Grise Fiord	76° 25' 3" N, 82° 53' 38" W	157	402	79
Pond Inlet	72° 41' 57" N, 77° 57' 33" W	1612	473	99

<sup>1</sup> Government of Nunavut population estimates (Government of Nunavut, 2014)

<sup>2</sup> Average annual degree days above 0 and 5 °C from 1981-2010 (Environment Canada, 2014a)

Table 3.2 Study WSP design characteristics.

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Community	Construction year		Surface area (ha)		Average operating depth (m)		Average summer cell volume (m <sup>3</sup> )		Daily volumetric load (m <sup>3</sup> /d) <sup>3</sup>	Areal organic loading (kg/m <sup>2</sup> /d)
	P <sup>1</sup>	S <sup>2</sup>	P	S	P	S	P	S		
Clyde River	1976	2011	6	0.5	0.1	2.3	6600	5000	93	P <sup>4</sup> : 57 T <sup>5</sup> : 16
Kugaaruk	2008		1.0		5.4		54000		76	28
Grise Fiord	1997		0.4		1.5		6000		16	25
Pond Inlet	2006		4.0		1.9		80000		110	15

<sup>1</sup>Primary cell

<sup>2</sup>Secondary cell

<sup>3</sup>Annual water use reports (Nunavut Water Board)

<sup>4</sup>Based on primary surface area

<sup>5</sup>Based on total surface area (primary and secondary)



#### 3.2.1.1 Pond Inlet

Pond Inlet's WSP is a single cell system constructed in 2005. The WSP was designed to be a facultative cell with an average depth of 1.9 m during the summer, but there are areas of the cell that are deeper (e.g., near the middle a depth of 3.2 m was measured). The surface area during the summer is approximately 4 ha and the estimated volume of the cell is 80,000 m<sup>3</sup>. The WSP is decanted annually between early September and early October over a period of 3 weeks. Wastewater is pumped over the berm of the WSP and then flows in a channel (500 m) before ending in the marine environment (Eclipse Sound).

In the fall of 2010 the WSP was emptied for an inspection of the HDPE liner and was offline until the spring of 2011. Wastewater was sent to an alternative disposal site from the fall 2010 to late spring 2011 at which time the WSP began to receive wastewater again.

#### 3.2.1.2 Kugaaruk

Kugaaruk has a single cell WSP, with an additional small decant cell located between the main cell and a tundra treatment wetland. Kugaaruk's WSP is a deeper system with an operating depth of 5.4 m. From mid to late summer, wastewater effluent is pumped from the main cell into the decant cell. The water from the decant cell seeps through the permeable berm and is dispersed across the tundra treatment wetland before flowing into the marine environment (Pelly Bay).

#### 3.2.1.3 Clyde River

Clyde River has two cells in their WSP system; the original smaller cell, referred to as the primary cell, was constructed in 1976 and rehabilitated in 2011 when the secondary cell was added to increase capacity. The primary cell covers an area of 0.6 ha and the secondary cell has a surface area of 1.5 ha. The cells are operated in a semi-parallel fashion with both cells receiving wastewater directly from sewage trucks. Due to the limited capacity of the primary cell and physical location (being separated from the discharge location by the secondary cell), water is transferred from the primary cell into the secondary cell periodically when the primary cell is full. The only decant location is on the side of the secondary cell facing Patricia Bay, the marine receiving water. The

intent of the primary cell is to provide preliminary treatment of the wastewater prior to entering the secondary cell, and to increase system capacity. The decanted wastewater flows through a vegetated filter strip (2.34 ha) prior to entering the marine receiving water environment of Patricia Bay. The annual decant occurs in the fall, just prior to freeze-up.

#### 3.2.1.4 Grise Fiord

Grise Fiord has a single cell WSP system which is 0.4 ha in size and has a depth of 1.5 to 2 m. The effluent in the WSP is typically pumped out over a period of 3-4 days in early September into a small 0.6 ha natural treatment wetland before draining into the marine environment (Jones Sound).

### 3.2.2 Data Collection Timeline

Water quality sampling of the WSPs occurred during the summers of 2011-2014 and the sampling periods for each community are provided in Appendix A. The remoteness of the communities and the associated high travel costs, especially for Grise Fiord, limited the frequency of visits to the study sites and as a result, not all sites were visited the same number of times. Efforts were made to sample three of the WSPs (Pond Inlet, Clyde River, Kugaaruk) at the start and end of the summer treatment season. The WSPs in Pond Inlet and Clyde River were also sampled around the middle of the treatment season in 2011-14 and 2013, respectively. Start, middle and end of the treatment season were defined as late June/early July, late July/early August and early/mid-September, respectively.

### 3.2.3 Continuous Water Quality Data Collection

The temperature, pH, and dissolved oxygen (DO) in the WSPs were measured electrochemically with multi-parameter sondes. Handheld versions were used for spot measurements while YSI 6-Series and EXO *in situ* multiparameter sondes (YSI Inc., Yellow Spring, OH) were deployed in the WSPs during the first visit of a field season, to record parameters hourly in the WSPs until the end of the treatment season. In addition, HOB0 temperature/light pendants, temperature/water level loggers and ROX DO probes

(Onset Computer Corporation, Cape Cod, MA) were used to capture data at other depths and to validate the continuous measurements captured by the *in situ* sondes.

### **3.2.4 Discontinuous Sampling of Wastewater in the Treatment System**

#### 3.2.4.1 Sampling strategy and apparatus

Wastewater samples were collected from the wastewater trucks as they discharged into the WSP and at multiple locations in each WSP; these sampling locations were repeated during each sampling event. All grab samples were collected in clean, Milli-Q rinsed, polypropylene bottles. The raw wastewater samples were collected from the trucks using a sampling pole with polypropylene bottles attached.

When possible, sampling of the WSP was conducted from an inflatable boat. When this was not possible due to logistical or weather constraints, samples were taken with a fully extended sub-surface pole sampler (Environmental Remediation Equipment, Inc., Montreal, QC) from the shore of the WSP. When sampling from the boat, the sub-surface pole sampler or an acrylic bacon bomb sampler (Koehler Instrument Company, Inc., Bohemia, NY) allowed for retrieval of samples at variable depths. Occasionally, surface samples were retrieved by hand because of logistical challenges and physical limitations. Samples were taken from 3 to 5 sites in each WSP cell to examine spatial variability.

#### 3.2.4.2 Sample analysis

Samples that were collected in Clyde River and Pond Inlet were chilled (4 °C), and flown to Iqaluit to be analyzed in the Northern Water Quality Laboratory at the Nunavut Research Institute within the holding times denoted in the American Public Health Association (APHA) Standard Methods (American Public Health Association, 1999) or the manufacturer's instructions. Samples collected in Kugaaruk were chilled and flown to Yellowknife (Northwest Territories, Canada) and processed by Taiga Environmental Laboratory, an accredited commercial lab. Due to Grise Fiord's remote location, a mobile lab space was erected to process samples. The methodologies used on-site and in the Iqaluit and Grise Fiord laboratories are briefly described here.

Analysis of CBOD<sub>5</sub> was performed in standard 300 ml Wheaton™ BOD bottles in duplicate following the APHA standard method 5210 B (American Public Health Association, 1999). A nitrification Inhibitor (Hach®, Loveland, Colorado) was added to the samples as directed. A Thermo Scientific™ DO Probe Orion™ 083005MD (Fisher Scientific, Ottawa, ON, Canada) was used with their Orion Star™ series meters (Fisher Scientific) to measure the dissolved oxygen.

Total ammonia was measured with the Thermo Scientific™ Orion™ High-Performance Ammonia Electrode attached to the Orion Star™ series meters (Fisher Scientific). Ionic Strength Adjuster (Fisher Scientific) was added to all samples as directed in the manufacturer's instructions to ensure consistency.

Total Suspended Solids analysis was performed following the APHA standard methods 2540 D (American Public Health Association, 1999) with Whatman™ 934-AH 47 mm glass fiber filters (Fisher Scientific).

### **3.2.5 Calculation of Degree Days**

Degree days are calculated by finding the average daily temperature for each day over the year and subtracting the reference temperature. For example, if the average temperature on a particular day was 10 °C and the reference temperature was 5°C then that day produced 5 degree days above 5°C (10 °C – 5 °C). To obtain the degree days for the year, the calculations for each day are then summed for the year. Degree days are commonly applied in agriculture for the assessment of plant and insect development and analysis of the viability of plants or crops in a specific geographical location or climate. Degree days were calculated from collected weather data from Clyde River, Kugaaruk, and Pond Inlet for 2012-2014 and compared against historical degree days from 1981-2010 available from the Climate Normal database maintained by Environment Canada (2014).

Degree days are used in this publication to provide a metric of comparison for the potential for phytoplankton growth and biological wastewater treatment between the study sites and other published studies of WSP performance. Since it was observed that the surface water temperatures in the WSPs rose well above that of the air temperature, the degree days were also calculated for the surface water in the WSPs. The degree days

of the surface water is of interest because its possible impact on the activity level of the microbes and phytoplankton.

### **3.2.6 Kinetics of CBOD<sub>5</sub> Removal in the WSPs**

First order rate constants of the removal of CBOD<sub>5</sub> were calculated for Pond Inlet and Clyde River for the years of 2012, 2013, and 2014. The calculated rate constants are considered estimated, and have been calculated from a limited data set. The rate constants have been provided to the reader as a metric of comparison to other wastewater treatment systems, and as a value that can be used to calculate approximate detention times required to meet CBOD<sub>5</sub> removal objectives with single cell WSPs in the Arctic. To our knowledge, at the time of publication, there are no first order rate constants that have been calculated from arctic single cell WSPs published in literature. They were calculated assuming a completely mixed batch reactor; this assumption is corroborated by sampling results showing the WSPs to be relatively homogeneous. For simplicity, the continuous addition of raw wastewater to the WSP was omitted. The omission of the additional raw wastewater into the calculation generates a conservative rate constant because it under represents the amount of CBOD<sub>5</sub> treated, and the actual rate constant is expected to be marginally higher. The impact of this omission is expected to be minor because the additional wastewater added during the summer treatment season only represents 1/7 – 1/6<sup>th</sup> of the total volume of wastewater. The first order rate constants for CBOD<sub>5</sub> removal in the WSPs were calculated as follows:

$$k = -\ln(C_t/C_o)/t$$

Where: *k* is the first order rate constant (day<sup>-1</sup>), *C<sub>o</sub>* is the CBOD<sub>5</sub> concentration in the WSP during the first visit of treatment season (mg/l); *C<sub>t</sub>* is the CBOD<sub>5</sub> concentration in the WSP at the end of treatment season (mg/l); and *t* is the time interval between the two sampling events (days).

### **3.2.7 Statistics**

Student's T-tests were used to compare the CBOD<sub>5</sub>, TSS, and ammonia concentrations in the raw wastewater, the WSP at the start of summer, and in the WSP at the end of the treatment season from each study site using a 5% significance level. Statistical analysis revealed that the concentrations of CBOD<sub>5</sub>, TSS and ammonia in

wastewater samples from each WSP and sampling time (start, middle or end of treatment season) did not differ significantly ( $p>0.05$ ) across the sampling years, which allowed for the cross-year pooling of samples. Also, the samples obtained from different locations in each WSP were pooled for each sampling event because there were no statistical differences between samples ( $p>0.05$ ). Sampling results from Pond Inlet in 2011 were not included in the statistical analysis because of the difference in operation during this year.

### **3.3 Results and Discussion**

#### **3.3.1 Characterization of Biogeochemical Environment**

The Arctic's climate is a strong factor in treatment performance as the average daily air temperatures for most communities remain below 10°C during the short summer. As a result the WSP water temperatures, averaged over the depth, are usually below 10°C. Chemical and biological reaction rates are strongly influenced by temperature, with rates responding to a change in temperature in an exponentially manner according to the van't Hoff-Arrhenius relationship (Tchobanoglous and Burton, 1991). These average air temperatures would suggest limited capacity for biological activity as it has been shown that activity is greatly reduced below 10°C (Bartsch and Randall, 1971; Lettinga et al., 2001). However, the surface waters of the WSPs were shown to be substantially warmer than the average over the depth, reaching temperatures of 15-20°C for limited periods of time (Table 3.3). The elevated temperatures occurred during clear sky periods when the incident solar radiation was high, resulting in warming of the WSPs. The effect of temperature on WSP performance is well documented and illustrated well in a sensitivity analysis of WSP design equations conducted by Kehl et al. (2009). A closer look at the surface temperature profiles in the WSPs suggests shallow cells can achieve temperatures that would be permissive for biological growth and treatment, particularly within surface layers where the accumulated degree days above 5°C were found to range from 243 – 374 in the Pond Inlet, Kugaaruk, and Clyde River WSPs (Table 2.3); which were significantly ( $p<0.05$ ) higher than the average air temperature degree days for these locations (Table 3.1).

Table 3.3 Biogeochemical characteristics in the Pond Inlet, Clyde River and Kugaaruk WSPs. Average values of surface water temperature, pH and dissolved oxygen, with the range (min – max).

Location	Year	Temperature (°C) <sup>2</sup>	degree days above 5 (°C) <sup>2</sup>	pH	Dissolved oxygen (mg/l)
Pond Inlet	2011	8.7 (1.9 – 12.7) <sup>5</sup>	- <sup>4</sup>	8.0 (7.2 – 9.0)	5.7 (< MDL <sup>3</sup> – 29.0)
	July 5 – Sept 5 2012	11.0 (5.2 – 19.0)	351	7.6 (7.3 – 7.8)	< MDL
	June 18 – Sept 6 2013	7.9 (0.8 – 18.0)	313	7.5 (7.1 – 7.7)	0.3 (<MDL – 1.3)
	July 4 – Sept 11 2014	10.5 (4.0 – 21.5)	386	7.8 (7.6 – 8.0)	0.1 (<MDL – 0.9)
	2012	-	-	Spot avg <sup>1</sup> : 7.2	Spot avg <sup>1</sup> : 1.6
Clyde River Primary	July 5 – Sept 7 2013	7.9 (0.6 – 17.5)	243	7.5 (7.2 – 7.7)	< MDQL
	2014	7.2 (0.0 – 16.2)	245	7.3 (6.9 – 7.6)	0.0 (<MDL – 0.9)
	June 26 – Sept 8				
Clyde River Secondary	2012	8.6 (3.9 – 13.3)	324	Spot avg <sup>1</sup> : 7.4	Spot avg <sup>1</sup> : 3.5
	July 5 – Sept 7 2013	8.0 (2.7 – 13.9)	246	7.4 (7.2 – 7.6)	< MDL (<MDL – 1.1)
	June 26 – Sept 5 2014	7.9 (0.7 – 12.7)	300	7.4 (7.3 – 8.0)	< MDL
Kugaaruk	2012	11.2 (7.7 – 17.2)	374	7.6 (7.4 – 7.8)	0.2 (<MDL – 0.4)
	2013	9.6 (0.4 – 20.9)	356	7.2 (6.8 – 7.4)	0.8 (<MDL – 2.8)

<sup>1</sup> Spot averages taken during field visits and not from continuous data

<sup>2</sup> Pond surface water temperature

<sup>3</sup> Method detection limit (0.2 mg/l)

<sup>4</sup> Not available

<sup>5</sup> Range – minimum and maximum

The pH of the WSPs was neutral to slightly basic (Table 3.3) and tended to rise marginally during the summer, peaking at the height of the summer in late July. The increase of pH is attributable to biological activity as the carbonate – bicarbonate equilibrium is impacted by phytoplankton using carbon dioxide (Talling, 1976). The WSPs in Pond Inlet and Grise Fiord in 2011, where high phytoplankton productivity was observed, had elevated average pHs of 9 and 10.8 during the summer treatment period, respectively. These peaks in pH were associated with supersaturated oxygen conditions in the WSPs and water samples being visibly green indicating a major algae bloom.

However, during the sampling seasons of 2012-2014 the WSPs at the three major study sites; Pond Inlet, Clyde River, and Kugaaruk, would be classified as anaerobic due to the lack of detectable dissolved oxygen levels (Table 3.3). The characteristic foul odours associated with anaerobic system was also detected throughout the treatment season. Odours associated with anaerobic activity are reportedly from the volatile fatty acids, mercaptans and gaseous hydrogen sulfide from the anaerobic degradation (Oleszkiewicz and Sparling, 1987). In contrast, in 2011, the WSPs in Pond Inlet and Grise Fiord both were found to be supersaturated with DO and have elevated pH levels suggesting that WSPs in the Arctic can be facultative, and even aerobic, for parts of the treatment season if properly designed, and favorable weather conditions occur. The Grise Fiord WSP was only monitored using grab sampling techniques for a one week period in 2011 prior to decant, and the continuous monitoring equipment in Pond Inlet was not installed until several weeks after the spring thaw in 2011. This unfortunately limits our ability to fully understand the conditions that facilitated the elevated phytoplankton production in 2011. However, comparison of the degree days above 5°C in Pond Inlet revealed that 2011 was significantly warmer with 205 degree days above 5°C (Environment Canada, 2014) relative to the historical average of 99 (Environment Canada, 2014). It may be that more clear sky days accompanied the increase in temperature, and the combination of these factors resulted in prolific growth of phytoplankton. It is also likely that operational differences, such as lower areal organic carbon loading and reduced operational depth, significantly contributed to the facultative nature of Pond Inlet's WSP during the 2011 treatment season.



### 3.3.2 Treatment Performance

#### 3.3.2.1 Carbonaceous biochemical oxygen demand (CBOD<sub>5</sub>)

The improvement in CBOD<sub>5</sub> between the raw wastewater and the water in the WSPs at the start of the summer (Figure 3.2) can be attributed to settling processes, as little to no biological activity would be expected in the WSPs when temperatures approach 0°C (Tchobanoglous and Burton, 1991; Reed et al., 1995). The WSPs in Kugaaruk and Clyde River's secondary cell contained the lowest concentrations of CBOD<sub>5</sub> at the onset of summer (Figure 3.2). The greater depth of the Kugaaruk WSP promotes organic matter being sequestered at the bottom of the WSP and reduces the organic matter that can be released back into the water column after settling. The recent construction of Clyde River's secondary cell means there would be minimal sludge accumulation. Both characteristics, deep and young operational age, appear to limit the release of organics back into the water column. Kugaaruk and Clyde River also had lower CBOD<sub>5</sub> concentrations in the raw wastewater when compared to Grise Fiord and Pond Inlet, and the better raw water quality also may contribute to better WSP water quality. Clyde River's poor primary cell water quality at the start of the summer is attributed to 40 years of continual operation. High rates of sludge accumulation have been documented in arctic WSPs (Miyamoto and Heinke, 1979), which further supports settling as an important removal mechanism of CBOD<sub>5</sub> in these arctic systems.

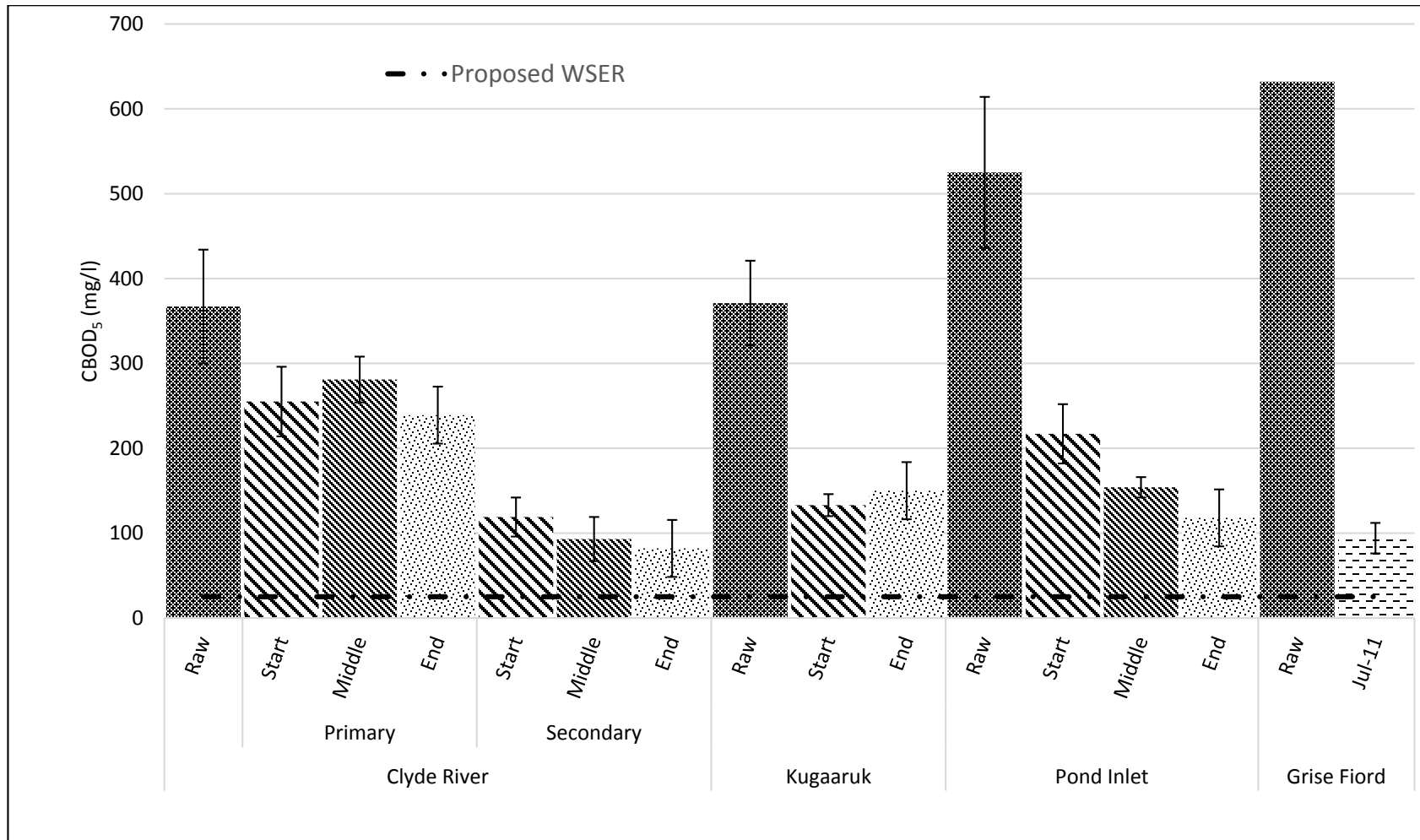


Figure 3.2 Mean CBOD<sub>5</sub> concentrations in the raw wastewater and in the WSP during the treatment season (error bars represent 95% confidence interval).

There were observable trends that suggest there was biological treatment of CBOD<sub>5</sub> in the shallower systems (Pond Inlet, and Clyde River) during the summer, as lower concentrations occurred at the end of the season. However, the trend was only statistically significant ( $p < 0.05$ ) for Pond Inlet. Clyde River's secondary cell had particularly large variability in the CBOD<sub>5</sub> values and was attributed to inconsistent operation of the system. Clyde River's secondary cell receives both wastewater from the primary cell and directly from sewage trucks. Kugaaruk, the deepest WSP, exhibited no observable CBOD<sub>5</sub> removal during the summer treatment season.

Raw wastewater quality in arctic communities is generally stronger with respect to CBOD<sub>5</sub>, TSS, and total ammonia concentrations when compared to wastewater quality entering wastewater treatment systems in the South (Smith, 1986). Generally, this is attributed to the decreased per capita residential water use. However, our results illustrate that even between arctic communities there can be significant differences ( $p < 0.05$ ) in raw wastewater quality (Figures 3.4). There are challenges with respect to treating more concentrated raw wastewater as it can result in WSPs being overloaded with organic material, which appears to be the case for the systems included in this study. Previous recommendations for OLRs suggest a maximum of 22 kg BOD<sub>5</sub>/ha/day for northern systems meant to be operated as facultative, but researchers have advocated for even lower loading rates in the Arctic (Dawson and Grainge, 1969; Heinke et al., 1991; Smith, 1986). The Pond Inlet and Clyde River WSPs (based on a total system surface area) met the recommendation of  $OLR \leq 22$  kg BOD<sub>5</sub>/ha/day, while Grise Fiord and Kugaaruk were above (Table 3.2). Deteriorating treatment performance is expected above the recommended loading limit (Heinke et al., 1991). However, our study shows that even for the systems that met the recommended organic loading rate (Pond Inlet and Clyde River), a facultative environment during the summer treatment season was not achieved, suggesting that maximum recommended loading rates of 22 kg/ha/day are too high.

#### 3.3.2.2 Total suspended solids

Solids removal in arctic WSP systems is quite effective with most systems producing an effluent quality approaching 25 mg/l TSS. Clyde River's primary WSP cell was observed to have the highest TSS concentration (58 mg/l) while the Kugaaruk WSP

had the lowest (25 mg/l) (Figure 3.3). Again, these different results may be attributed to the depth and age of the systems, with systems that are younger and/or deeper performing the best due to limited resuspension of solids and limited algae growth. TSS can exceed the Nunavut Water Board standard of 180 mg/l during phytoplankton blooms, as was the case in Grise Fiord in 2011, where the TSS concentration in the WSP reached 438 mg/l. Phytoplankton growth has previously been reported to create challenges for some WSP systems in regards to meeting effluent TSS requirements (Shilton, 2005). This is important to consider as system designs that increase phytoplankton growth likely will have high effluent TSS levels, if additional measures for phytoplankton removal are not considered.

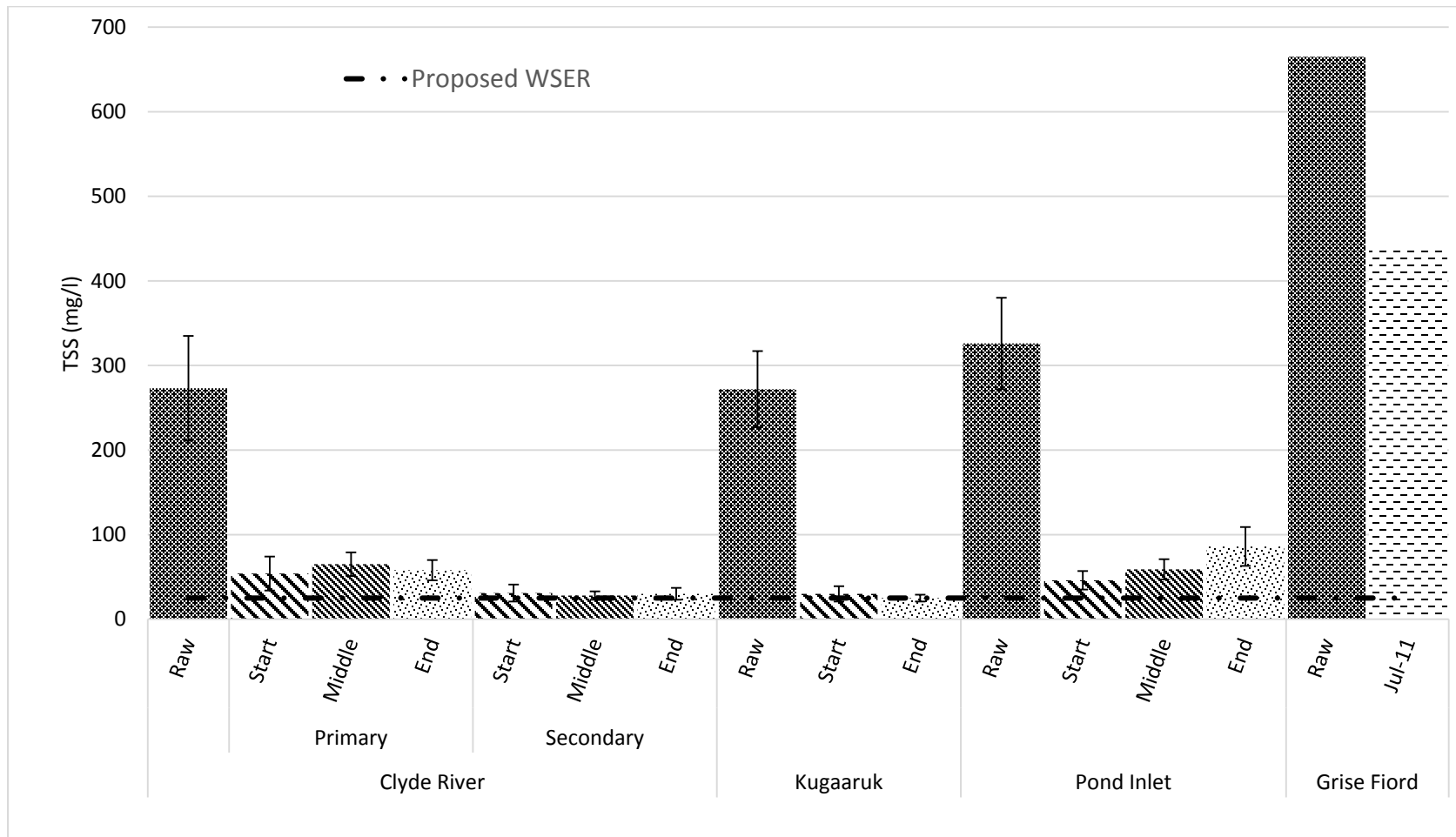


Figure 3.3 Mean TSS concentrations in the raw wastewater, and in the WSP during the treatment season (error bars represent 95% confidence interval).

### 3.3.2.3 Un-ionized and total ammonia

Figure 3.4 illustrates that the systems that were the least biologically active, as defined by minimal CBOD<sub>5</sub> removal over the summer (Clyde River (Primary cell) and Kugaaruk), achieved limited reductions in Total Ammonia Nitrogen (TAN) between the raw wastewater and the WSP water quality, with no significant ( $p > 0.05$ ) removal observed over the summer. This lack of removal is in contrast to the WSPs of Grise Fiord and Pond Inlet, where reductions during the treatment season are observed. This observation is in accordance with CBOD<sub>5</sub> results and would suggest that biological activity has a role in the reduction of TAN concentrations (Figure 3.4). The increase in TAN levels at the end of the treatment season in Clyde River's secondary WSP cell was not anticipated, but is likely explained by the large transfers of wastewater from the primary cell to the secondary cell in the fall resulting in increases in TAN concentrations in the secondary cell. Lastly, in the case of the Grise Fiord WSP in 2011, the decrease in TAN is likely attributed to phytoplankton uptake and volatilization of un-ionized ammonia due to high pH ( $> 10$ ). Efficient removal of TAN in cold climate WSPs has been observed in other studies and has been associated with volatilization of un-ionized ammonia when pH is above 8 (Rockne and Brezonik, 2006), however, a growing body of evidence point to the increased uptake by phytoplankton and not volatilization as the main pathway of ammonia removal at elevated pH (Valero and Mara, 2010). Within all of the systems, except the Grise Fiord WSP, the TAN was primarily in the ionized form. This is a result of the neutral pH within most of the systems. Grise Fiord had an elevated pH, presumably due to phytoplankton metabolizing carbon dioxide which shifts the ammonium-ammonia equilibrium towards higher ammonia concentrations thereby increasing the availability for the phytoplankton and the potential for volatilization.

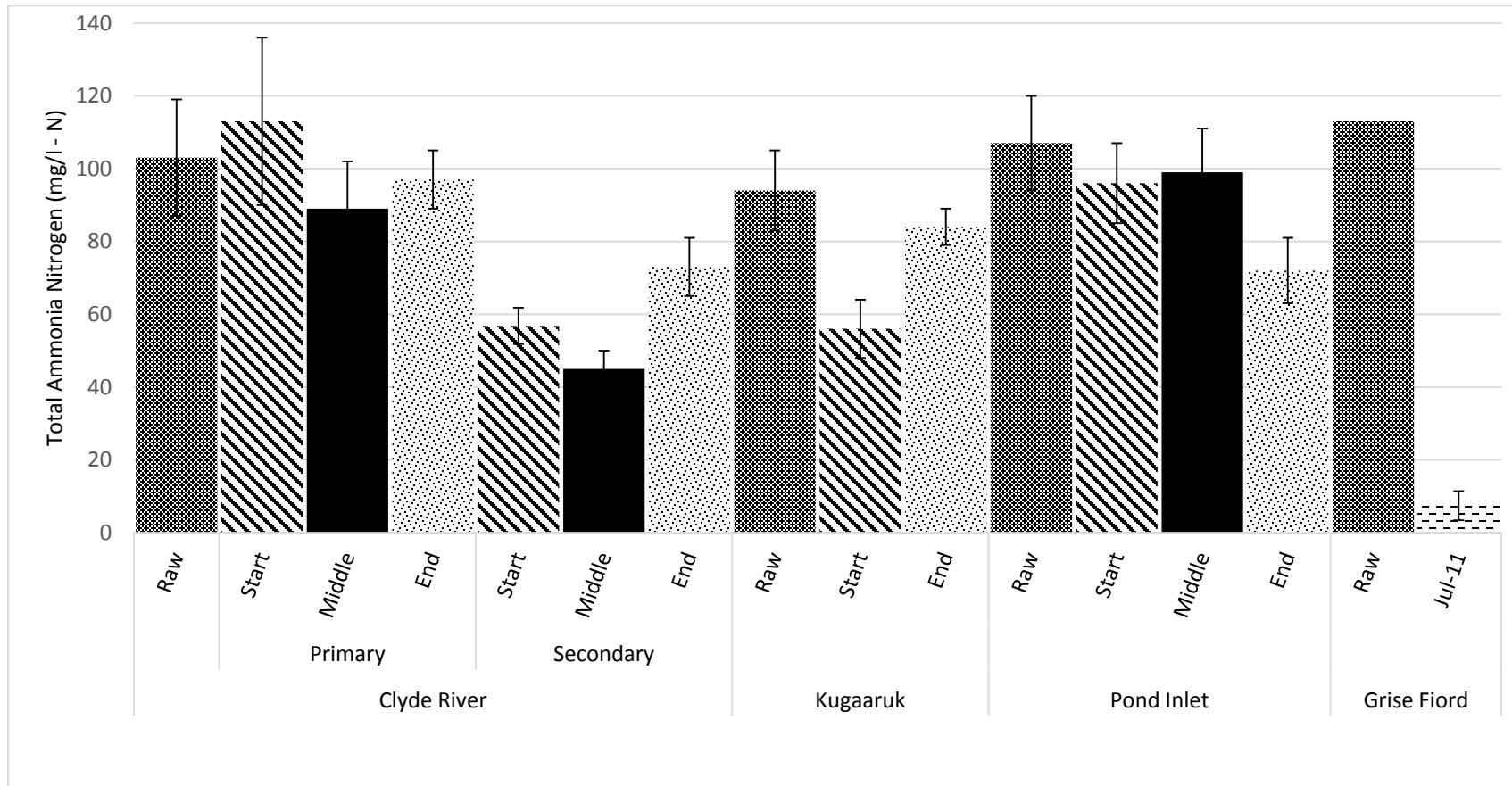


Figure 3.4 Mean ammonia nitrogen (TAN) in the raw wastewater, and in the WSP during the treatment season (error bars represent 95% confidence interval).

### **3.3.3 Comparisons to Other Northern WSP Systems**

To the best of our knowledge, our study is the first to perform an extensive multi-year study of single cell Arctic WSPs that are operated with an annual decant event. Other studies have examined the performance of WSPs in Sub-arctic or Arctic regions, but the systems have been functionally different, and the climate significantly warmer, than the locations in this study. Prince et al. (1995) examined a WSP system in Northern Alberta (58 °N) which consisted of four anaerobic cells, one treatment cell (facultative and <1.5 m deep) and one long retention storage cell in series. At the time of fall decant the effluent BOD<sub>5</sub> and TSS concentrations were well below 25 mg/L. Miyamoto and Heinke (1979) assessed the performance of a single-cell continuous flow WSP in Inuvik, NWT, Canada (68 °N) that covered an area of 25 ha and documented that the system could produce effluent with concentrations of 40 mg/l BOD<sub>5</sub> and 20 mg/l TSS. Prosko et al. (2007) investigated an aerated WSP containing 4 cells in series in Fort Nelson, BC, Canada (59 °N), and found it achieved effluent concentrations of BOD<sub>5</sub> and TSS below 24 and 22 mg/l, respectively. The performance of these other systems in Northern Canada attests to the potential effectiveness of WSPs in Arctic or near-Arctic regions. However, the WSP designs in these studies were generally more sophisticated, including several cells in series or mechanical aeration, or much larger (i.e., Inuvik). The climates of their study sites were also significantly warmer than the climate in our study sites (Table 2.1), with Inuvik, Fort Nelson and Northern Alberta having historical averages of 750, 1330, and 1230 degree days above 5°C, respectively (Environment Canada, 2014a). It is apparent from these studies, though, that WSP systems can achieve secondary treatment standards in cold climates if they are large enough and/or are supplemented with mechanical aeration.

### **3.3.4 Implications for the Design of Arctic WSPs**

Results of previous studies in the Arctic/sub-Arctic suggest that WSP designs will need to be greatly enhanced from a single cell system to a multi-cell system if they are to produce an effluent quality that would meet secondary treatment standards. The implementation of multi-cell WSPs would increase retention times, improve system



hydraulics, and isolate sludge deposits in the first cell(s) of the system. Configuring cells to achieve increases in length to width ratios would also improve system hydraulics and generate plug flow behaviour (Persson, 2000).

First order rate constants ( $k$ ) for the removal of  $\text{CBOD}_5$  over the summer treatment seasons were determined for individual years in Pond Inlet and Clyde River based on the assumption that the WSPs could be modeled as batch reactors and ignoring the continual influent raw wastewater, which makes the calculated rate constants modestly conservative (Table 3.4). The  $k$  values ranged from  $0 - 1.3 \times 10^{-2} \text{ d}^{-1}$ . The calculated rate constants are much lower than those recommended for use in established design equations for non-Arctic systems, which range from  $6 \times 10^{-2}$  to  $1.9 \times 10^{-1} \text{ d}^{-1}$  when assuming  $10^\circ\text{C}$ , which was approximately the average temperature of the WSPs (USEPA, 1983; Thirumurthi 1974). This large discrepancy urges caution when applying WSP design equations developed in the South to single cell retention WSPs in the Arctic, as they do not appear to appropriately account for the extreme climate and the unique design and operation. Rate constants for WSPs in this study, when biological treatment was observed to be occurring, had a range where the highest was double that of the lowest recorded rate, suggesting that treatment performance on an annual basis can be highly variable. There is a general trend that suggests that warmer years, as measured by surface temperature and degree days above  $5^\circ\text{C}$  (Table 3.3), resulted in higher rate constants (Table 3.4). However, due to different lengths and starting times of the monitoring seasons, it is difficult to draw conclusions from the limited number of observed years. As a measure of temperature and length of frost free season, the variation of degree days of the monitored communities (Table 3.1) suggests large potential differences in the biological treatment rates between arctic communities. Using the average  $k$  value obtained from Pond Inlet of  $9.7 \times 10^{-3} \text{ d}^{-1}$ , it can be calculated that it would take 230 summer treatment days (defined as frost free days) to achieve an effluent with a  $\text{CBOD}_5$  of 25 mg/l. Considering that the short summer provides only about 60 treatment days per year, four years of retention in the current WSP would be required to meet the southern wastewater effluent regulations. It should be noted that this estimate is likely conservative and does not fully account for the continual addition of wastewater.

Table 3.4 Calculated first order rate constants (k) for CBOD<sub>5</sub> removal

Location	Year	k <sup>a</sup> (days <sup>-1</sup> )	
Pond Inlet	2012	1.3 x 10 <sup>-2</sup>	
	2013	6.3 x 10 <sup>-3</sup>	
	2014	9.3 x 10 <sup>-3</sup>	
	Average	9.7 x 10 <sup>-3</sup>	
Clyde River	2012	-	
	Primary	2013	5.9 x 10 <sup>-3</sup>
		2014	4.3 x 10 <sup>-3</sup>
	Secondary	2012	2.9 x 10 <sup>-3</sup>
		2013	5.6 x 10 <sup>-3</sup>
		Average	5.7 x 10 <sup>-3</sup>

<sup>a</sup> k =  $-\ln(C_t/C_o)/t$ , where C<sub>o</sub> = Concentration start of season;  
C<sub>t</sub> = Concentration end of season; t = Season duration

Operational water depths appear to strongly impact treatment performance in terms of CBOD<sub>5</sub>, TSS, and un-ionized ammonia removal. The physical design is directly related to areal loading rates and impacts water temperatures, gas exchange and incident solar radiation; all of which are factors that strongly influence the oxygen state and biogeochemical processes of a WSP. While shallow cells (< 1.5 m) promote biological activity by increasing the average water temperature and the gas air-water transfer boundary, which generally improves carbon and nutrient removal, they are not without their challenges as they promote phytoplankton growth, which potentially results in increased un-ionized ammonia and TSS concentrations, as was seen in Grise Fiord 2011. The Kugaaruk WSP, with its greater depth, demonstrated the benefit of a deeper WSP with respect to sequestration of contaminants in the sludge as it had the best water quality at the start of summer when compared to the other systems. Kugaaruk, conversely, demonstrates that deep systems lack the characteristics required to promote biological treatment over the summer with no measureable change in CBOD<sub>5</sub> over the summer. TAN concentrations were also significantly (P<0.05) higher at the end of the treatment season. Anaerobic wastewater treatment processes have a greater dependence on temperature and slower microbial growth rates and start-up times than aerobic processes

(Chan, 2009). Deeper cells have other benefits though, as they provide greater storage capacity (based on a volume to surface area ratio) through the winter. The utility of deeper anaerobic cells has been demonstrated as the systems in Fort Nelson (Prosko et al, 2007) and Northern Alberta (Prince et al., 1995) both utilized deep primary cells as their first cell in their multi-cell WSP systems.

From a design perspective it is apparent that both shallow cells (< 1.5 m) and deeper cells have benefits, however, the data from the present study suggests that a single cell system cannot achieve secondary treatment standards. The optimal design for WSP systems in the Arctic would therefore likely include more than a one cell arrangement where a deep anaerobic cell(s) is used for primary treatment and is followed, in series, by a shallow cell(s) loaded at a reduced organic loading rate to provide biological treatment.

### **3.4 Conclusions**

This study characterized the level of oxygenation and treatment performance within WSPs used in the Canadian Arctic. WSP systems used in this extreme climate function as controlled discharge storage ponds, where biological treatment processes are constrained to relatively short time periods (approximately 60 d) where air temperatures average between 7-10°C. However, extended photoperiods at these high latitudes can contribute to warming of surface water layers, and surface water temperatures in the range of 15 – 20°C can be achieved. The facultative WSPs in this study were primarily anaerobic throughout the majority of the treatment seasons monitored. These systems are effective for the removal of suspended solids. The WSPs studied were challenged to meet secondary wastewater treatment objectives for the removal of biological oxygen demanding material, as reduced biological treatment rates associated with low temperatures and near-anaerobic conditions were observed. Accordingly, current design guidelines for Arctic WSPs should be reviewed to ensure that WSPs will operate in the intended oxygen state and meet the treatment goals. A possible strategy for improving treatment is the use of multi-cell pond arrangements employing a combination of deep anaerobic cells, proceeded by shallow facultative cells receiving lower organic loading rates.

## **Chapter 4 Understanding the Influence of Light, Temperature and Organic Loading Rates on Oxygen Dynamics and Carbon Removal in Waste Stabilization Ponds Operating in Arctic Environments**

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### **4.1 Introduction**

Waste stabilization ponds (WSPs) are passive treatment systems that are strongly influenced by the ambient climate (Heinke et al., 1991). WSPs are a commonly used technology for treating agricultural, industrial and municipal wastewater, providing effective treatment of oxygen demanding substances, nutrients, suspended solids and pathogens (Shilton, 2005). WSPs are a popular technology for small remote communities as they are simple in design and operation, requiring minimal operator expertise, and are inexpensive in both capital and operational costs when compared to conventional mechanical treatment systems (Heinke et al., 1991; Mara et al., 1992).

Most communities in the Canadian northern territory of Nunavut utilize WSPs for municipal wastewater treatment. WSPs in Nunavut are typically designed as single cell storage WSPs with the capacity to store the volume of municipal wastewater generated over 11-12 months. These systems typically remain frozen for approximately 9-10 months of the year, with a short 2-3 month ice-free period during the arctic summer. Once a year, typically in late summer/early fall, WSPs are decanted over a 1-3 week period (depending on the size of the system) to provide capacity for the wastewater generated during the following year. Decant occurs in late summer because that is when the best effluent quality is assumed to have been achieved. Elevated temperatures and solar irradiance experienced during the summer presumably facilitate biological treatment, and results in improved effluent quality by the time of fall decant.

Ragush et al. (2015) monitored four municipal WSPs in Nunavut during the summer treatment seasons of 2011-2014 to characterize WSP conditions (chemical, biological and physical properties) and assess their performance in the context of the new Canadian Wastewater Systems Effluent Regulations. Ragush et al. (2015) identified that new design guidelines and strategies would need to be developed for arctic WSPs to meet the more stringent effluent quality criteria. It was found that systems that were designed to operate as facultative WSPs were found to be generally anaerobic with limited phytoplankton populations, and it was speculated that the areal organic loading rates (OLRs) (kg CBOD<sub>5</sub>/ha/day) exceeded the OLRs that would be required for the WSPs to be facultative or aerobic in an arctic climate. It is likely that anaerobic conditions in the WSPs negatively impacted the WSPs' CBOD<sub>5</sub> treatment performance (Mara et al., 1992; Shilton, 2005). The limited data available in the literature makes it difficult to predict the environmental and operational conditions that will lead to facultative conditions in arctic WSPs, but observations and literature suggest that water temperature (Lettinga et al., 2001), irradiance (Shilton, 2005), and OLR (USEPA, 1983) strongly influence WSP treatment performance and oxygen status.

Literature further highlights the impact of phytoplankton populations on WSP treatment processes. Mara et al. (1992) state “(algal growth) is the whole basis of WSP treatment,” and in an overview of WSP design guidelines it is typical that at least one cell in the system is incorporated to promote prolific algal growth (Shilton, 2005). Algae provide a nutrient sink through nutrient assimilation (Middlebrooks et al., 1999), and an oxygen source for heterotrophic/aerobic bacteria (Shilton, 2005). Heterotrophic bacteria are more efficient at removing oxygen demanding material under aerobic conditions when compared to anaerobic conditions (Chan et al., 1999), and as a result, treatment efficiency in terms of CBOD<sub>5</sub> removal is greatly enhanced by the presence of phytoplankton (Mara et al., 1992).

To better understand how environmental and operational factors influence phytoplankton dynamics and arctic WSP performance, a bench scale study of single cell WSPs operating under simulated arctic conditions was conducted. In the bench scale study, biological, chemical, and physical parameters were measured during 34-40 day simulations of arctic WSPs. The bench scale analysis of the arctic WSPs was used to

assess the impact of irradiance, temperature, and organic loading conditions on: (i) oxygen status, and (ii) removal of carbonaceous oxygen demanding material.

## **4.2 Methods**

### **4.2.1 Experimental Design**

Model WSPs were constructed out of transparent polyvinyl chloride (PVC) pipes (15.25 cm in diameter and 1.25 m in length) that were capped at one side, arranged vertically, and filled with synthetic wastewater (Figure 4.1). The experiment was performed in a temperature-controlled chamber. A synthetic wastewater recipe (Appendix C) was developed with comparable chemical, biological, and physical characteristics to wastewater contained in Pond Inlet, Nunavut's WSP (Ragush et al., 2015) at the start of the summer treatment season (late June/early July). The experiment was designed as a factorial design with 4 factors and 2 levels (Table 4.1) creating 16 unique conditions. Each set of unique conditions was tested in duplicate, creating a total of 32 experimental columns. A control column for phytoplankton growth, filled with Modified Bold 3N (University of Texas, 2015), and irradiated at 225  $\mu\text{e}/\text{m}^2/\text{s}$ , was included within each trial. The low irradiance condition was used for the control column because the low light attenuation characteristics of the Modified Bold 3N was likely to result in the phytoplankton experiencing conditions of high irradiance and inhibited growth (photoinhibition) (Dauta et al., 1990). Two initial carbon concentrations were simulated, because it was postulated that the initial carbon concentration of the WSP at the beginning of the biologically active summer period may impact treatment performance. Also, synthetic wastewater that was chemically comparable to raw untreated wastewater was added daily at two different rates to simulate an OLR comparable to that received by arctic WSPs (15 kg/ha/d) as well as a reduced rate to simulate how treatment performance may change under reduced loading conditions. Water lost due to evaporation and from sampling was replaced with distilled water daily to maintain a constant fluid volume. Trials ran for 34 to 40 days, and samples were drawn every 5-7 days from the top sampling port, located approximately 20 cm below the water surface.

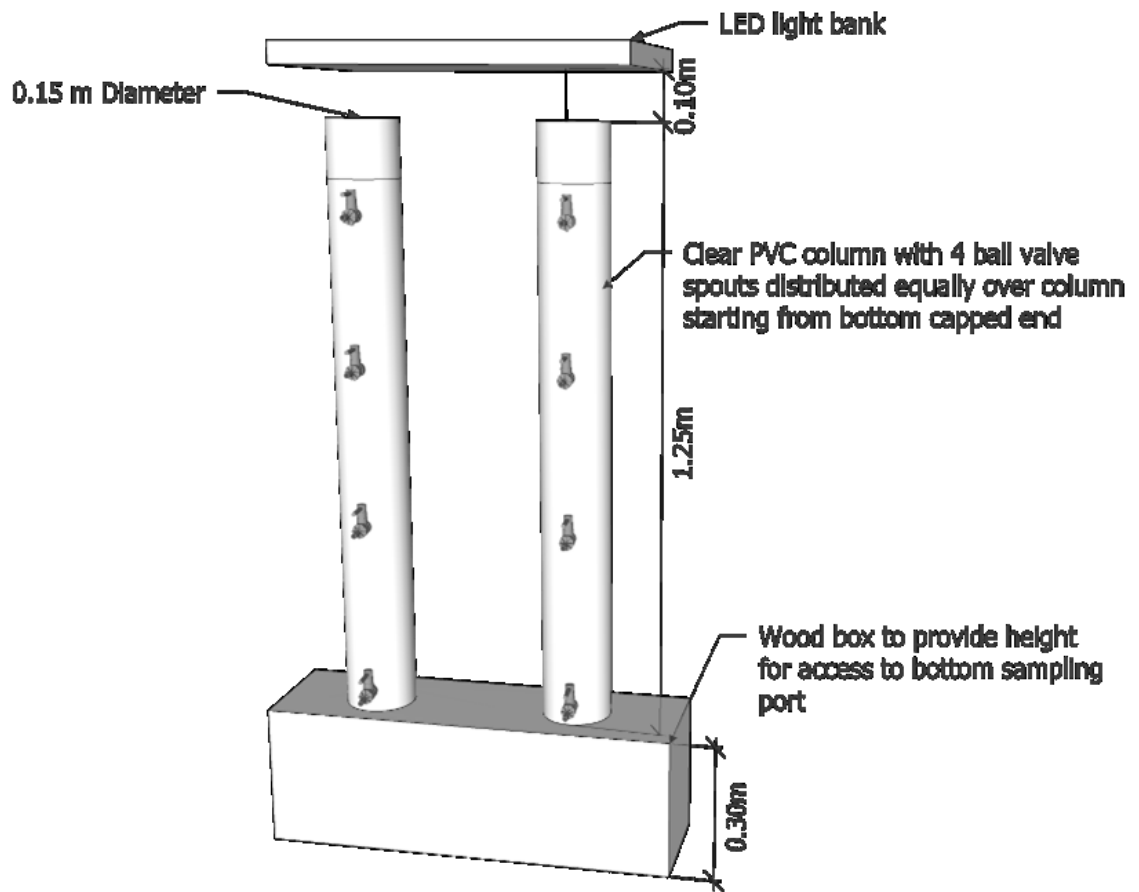


Figure 4.1 Diagram of the experimental apparatus.

Table 4.1 Factorial design – experimental factors and levels.

<b>Factors</b>	<b>Levels</b>	
Temperature	5 °C	15 °C
Irradiance	225 PAR (ue/m <sup>2</sup> /s)	1050 PAR (ue/m <sup>2</sup> /s)
Organic Loading Rate	3.8 kg CBOD <sub>5</sub> /ha/d	15 kg CBOD <sub>5</sub> /ha/d
Initial CBOD <sub>5</sub> Concentration	80 mg/l	240 mg/l

Two levels for the initial CBOD<sub>5</sub> concentration were used. A value of 240 mg/L represented the typical concentration of CBOD<sub>5</sub> in the Pond Inlet WSP at the start of the summer treatment season (Ragush et al., 2015a). A lower level of 80 mg/L was included to assess how WSP systems would perform under a scenario in which a pre-treatment step (i.e. anaerobic cell) was included to reduce CBOD<sub>5</sub> before the facultative cell. Temperatures of 5 °C and 15 °C were used as this range was representative of the observed surface water temperatures over the summer in several monitored arctic WSPs (Ragush et al., 2015). LED light banks were used to irradiate the columns and provide a light spectrum comparable to the solar spectrum. Two irradiance conditions were evaluated, with the high irradiance condition (1050 ue/m<sup>2</sup>/s) being representative of a clear sky irradiance around solar noon. 225 ue/m<sup>2</sup>/s, roughly one quarter of the maximum incident irradiance, was the lower irradiance condition used as it was perceived worst-case scenario with respect to observed average irradiance in Pond Inlet over the summers of 2012-2014, which ranged from 292 to 355 ue/m<sup>2</sup>/s.

## 4.2.2 Materials

### 4.2.2.1 Lights

Four Atlantik V1's LED light banks, designed to simulate the solar spectrum, from Orphek (Sao Paulo, Brazil) were installed 10 cm above the columns to provide irradiation. Each light had two columns situated under it (Figure 4.1) that were strategically placed to provide equal amounts of irradiance, as measured by Photosynthetic Active Radiation (PAR) using a MQ-200 Quantum Meter from Apogee (Logan, Utah, USA), to each column. The irradiance from the light systems was not



uniform over the surface area of each column as the light systems are composed of 68 LEDs. It was found that at a distance of 10 cm from the source the irradiance varied by +/- 10% across the surface of the columns.

#### 4.2.2.2 Synthetic wastewater

A synthetic wastewater was developed based upon the wastewater characteristics measured in raw wastewater and the wastewater in the WSP at the start of the treatment season in Pond Inlet, Nunavut (Ragush et al., 2015a). In development of a representative synthetic wastewater, literature was consulted with respect to both the composition of wastewater (Huang et al., 2010; Sophonsiri and Morgenroth, 2003; Heukelekain and Balmat, 1959; Raunkjaer et al., 1994), and the composition of synthetic municipal wastewater used in previous wastewater treatment studies (Canizares et al., 2010; OECD, 2001; Nopens et al., 2001; Tokuz, 1991). The reviewed studies demonstrated that municipal wastewater characteristics vary greatly with local diet. Special consideration was made for Raunkjaer et al. (1994) because they analyzed wastewater in Denmark. It was theorized that the diet of people in Denmark was the most analogous to the diet of people in the Canadian Arctic. A representative mixture of inorganics, fatty acids, proteins, and carbohydrates was formulated, and the developed synthetic wastewater recipes are provided in Appendix C.

Humic acids were added to the synthetic wastewater to provide color, while diatomaceous earth was added to represent inorganic total suspended solids. These two substances were primarily used to replicate the light attenuation characteristics of the WSP in Pond Inlet, Nunavut, which was measured to have a light attenuation coefficient of  $14 \text{ m}^{-1}$ . Diatomaceous earth was added in accordance with the previously measured concentration of non-volatile suspended solids in the Pond Inlet WSP.

#### 4.2.2.3 Vessel

Clear 15.25 cm diameter PVC pipes were cut in 1.25 m lengths. Four barbed PVC ball valves were installed on the PVC column, at the bottom of the column and approximately every 30 cm vertically, as sampling ports (Figure 1). The columns were wrapped with a black plastic sheet from the bottom up to 10 cm below the water surface to eliminate irradiation from other sources or diffracted light entering the side of the

column. Vessels were constructed of clear PVC to allow for visual observations of the column.

#### 4.2.2.4 Bacterial culture

Preserved bacterial cultures from the Pond Inlet WSP were used to provide a microbial seed. Wastewater samples from the WSP in Pond Inlet were collected in the summers of 2013 and 2014. Aliquots from these samples were centrifuged (3,200 x g) for 5 minutes to pellet the WSP's microbiome. The supernatant was decanted off leaving the pellet and a small amount of liquid. Glycerol was added to the tube to make a 40% (v/v) glycerol solution. The cultures were stored in a -20 °C freezer. Preserved bacterial cultures were reinvigorated by warming the tube at room temperature and then pouring the contents of the tube into sterilized synthetic wastewater media representative of the initial WSP conditions for the next trial. The cultures were incubated at 20 °C for 3-5 days prior to the start of the trial. At the beginning of the trial, 20 ml of the cultured bacteria was added to each column, with the exception of the control column for algae growth.

#### 4.2.2.5 Phytoplankton culture

Two species of phytoplankton were maintained in Modified Bold 3N medium (University of Texas, 2015): *Chlorella vulgaris* and *Chlamydomonas reinhardtii*. These species were identified to be present in three arctic WSPs during the summer of 2011 (Pond Inlet, NU, Grise Fiord, NU, and Coral Harbor, NU). Cultures were obtained from the National Research Council of Canada's Institute for Marine Biosciences (NRC-IMB, Halifax, Nova Scotia). The cultures were grown in 250 ml Erlenmeyer flasks under standard florescent lights for 10-14 days at room temperature (20-22 °C). At the start of the trial, 20 ml of each algal culture was added to every column in the trial.

### 4.2.3 Sampling

On a daily basis, electrochemical readings for dissolved oxygen concentration (DO), temperature, and pH were taken in the surface water (top 5 cm). Columns were sampled every 5-7 days from the top sampling port located 20 cm below the surface

water. The sampling port was first flushed (about 50 ml), and then a sample was collected in sterile 50 ml falcon tubes.

#### **4.2.4 Sample Analysis**

Analysis of CBOD<sub>5</sub> followed the APHA standard method 5210B (American Public Health Association, 1999) and was performed in standard 300 ml Wheaton™ BOD bottles. To eliminate nitrification, a nitrification inhibitor (Hach®, Loveland, Colorado) was added to the BOD bottles as directed by the manufacturer's instructions.

Phytoplankton was enumerated with a Motic BA310MET-T light microscope (Richmond, BC, Canada) using a Hausser Scientific improved Neubauer Bright-line hemacytometer (Fisher Scientific, Ottawa, ON, Canada). Two aliquots of each sample were loaded into the slide (one on each available cell) and five 1.00 mm x 1.00 mm squares were counted, four corners and the middle. The counts from the two aliquots were averaged.

Heterotrophic bacteria were enumerated by spread plating appropriate serially diluted samples taken from the top sampling port on R2A Agar (BD Difco, Fisher Scientific). The agar plates were incubated 5 days at 20 °C prior to enumeration.

pH and DO were measured using electrochemical probes in the Thermo™ Orion™ (Fisher Scientific) series. The probes were connected to an Orion Star™ series portable multiparameter meter (Fisher Scientific).

#### **4.2.5 Data Analysis**

To test for the effects of the different factors on the response variables several approaches were employed. For oxygen status and CBOD<sub>5</sub> removal both a logistic regression analysis and a four-way analysis of variance ANOVA were conducted on the 32 trials (16 unique conditions in duplicate) to assess the relationship between the response variables and the four factors (light, temperature, initial CBOD<sub>5</sub> concentration, OLR). For oxygen status, an aerobic condition was categorized as attaining a DO concentration greater than 2 mg/L at any time during the trial. Therefore, for the logistic regression, a positive result was recorded if the DO concentration exceeded 2 mg/L anytime during the trial, whereas in the ANOVA the number of days that it took until the

DO concentration first exceeded 2 mg/L was used as the response variable. The same methodology was employed for assessing CBOD<sub>5</sub> treatment, but with a measurement of CBOD<sub>5</sub> below 30 mg/L as the response variable and a positive response for the logistic regression analysis assigned to trials where CBOD<sub>5</sub> fell below 30 mg/L. 30 mg/l was chosen as a critical value because it was found that biologically active columns, columns containing high concentrations of phytoplankton and bacteria, had base concentrations of 15-25 mg/l CBOD<sub>5</sub> attributable to the oxygen demand of the organisms (Figure 4.3). The ANOVA method was compromised by the presence of censored data, as some trials always were anaerobic or never achieved a CBOD<sub>5</sub> concentration below 30 mg/L. In the case of censored data, a value of 40, the last day of measurement, was used as the response variable value for the trial. An ANOVA analysis was also used to assess the influence of the experimental factors on maximum phytoplankton populations. The datasets used for ANOVA were tested for the assumptions of normality and equal variance by plotting, and were not observed to violate the assumptions. All statistical analysis was conducted in Minitab, with a significance level of  $p < 0.05$  chosen *a priori*.

#### 4.2.6 First Order Rate Constants for CBOD<sub>5</sub> Removal

First order rate constants ( $k$ ) for CBOD<sub>5</sub> removal were also calculated according to equation 4.1. The rate constants were only computed using data from the logarithmic declining component of the removal curves, omitting the final stages when a plateau in CBOD<sub>5</sub> was observed. The addition of wastewater was omitted from the calculation providing a conservative first order rate constant.

$$k = -\frac{\ln\left(\frac{c}{c_0}\right)}{t} \quad (4.1)$$

Where:  $k$  = first order rate constant ( $\text{day}^{-1}$ ),  $c$  = concentration at time  $t$  (mg/l),  $c_0$  = Initial concentration at time 0 (mg/l), and  $t$  = time (days)

#### 4.3 Results

The four experimental factors (temperature, irradiance, initial carbon concentration, and OLR) were all found to have a significant effect ( $p < 0.05$ ) on at least two of the response variables (Table 4.2). Temperature and initial carbon concentration were found to be significant factors ( $p < 0.05$ ) for all CBOD<sub>5</sub> and dissolved oxygen

responses. Temperature and irradiance were observed to be the only two factors which significantly influenced maximum phytoplankton populations.

Table 4.2 Significance of factors on characteristics related to dissolved oxygen concentration, phytoplankton populations and CBOD<sub>5</sub> removal. Shaded cells denote factors identified as significant based on  $p < 0.05$ .

Factors	Phytoplankton	CBOD <sub>5</sub> removal		Dissolved oxygen	
	max	(>30 mg/l)		(>2.0 mg/l)	
	population	ANOVA	Logistic	ANOVA	Logistic
Temperature 5 & 15 °C	< 0.01	< 0.01	< 0.01	< 0.01	< 0.01
Irradiance 225 & 1050 ue/m <sup>2</sup> /s	0.02	< 0.01	0.10	< 0.01	0.03
Initial CBOD <sub>5</sub> 80 & 250 mg/l	0.60	< 0.01	< 0.01	< 0.01	0.03
OLR 3.8 & 15 kg/ha/d	0.91	0.01	0.10	0.23	0.03

#### 4.3.1 Dissolved Oxygen

Dissolved oxygen concentrations were impacted by all four experimental factors, with the logistic regression showing that all factors were significant ( $p < 0.05$ ) (Table 2 and Figure 2). Higher temperature and irradiance increased DO concentrations above the threshold of 2.0 mg/l earlier in the trial, while higher OLR and initial carbon concentration delayed the development of DO concentration greater than 2.0 mg/l. In all trials conducted at 15 °C, a measurable level of oxygen was observed before the completion of the trial period, with only the trials conducted with high initial carbon concentrations and low irradiance not achieving oxygen saturation. At a temperature of 5 °C, only the columns with a low initial carbon concentration and high irradiance reached a state of oxygen saturation before the end of the trial. Columns at 5 °C with high initial carbon concentration which also had low light and/or high OLR, were classified as anaerobic for the duration of the trial. Results highlighted that higher temperature and lower initial carbon concentrations were the most significant factors in promoting higher DO concentration.

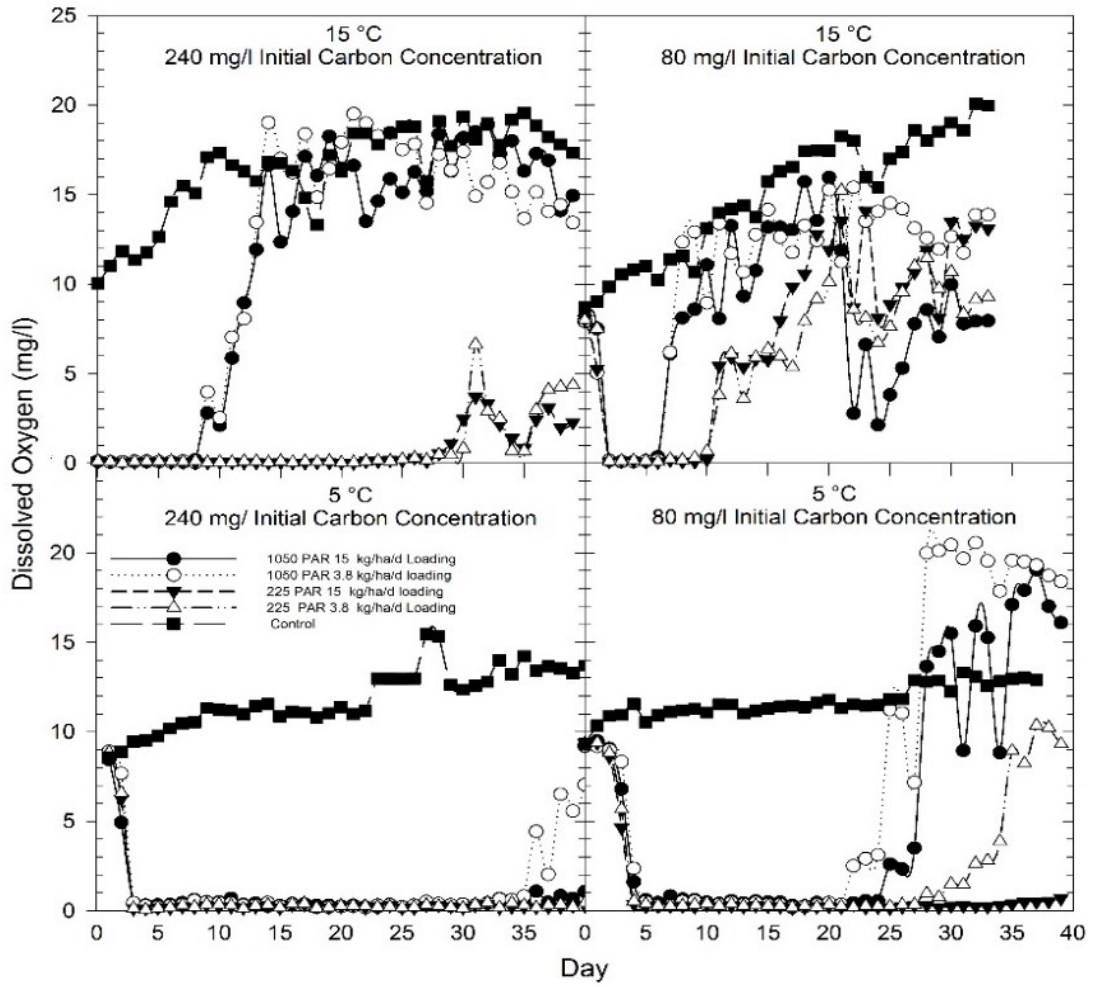


Figure 4.2 Measured dissolved oxygen concentrations at the column surface for the duration of the each of the 16 unique trials.

#### 4.3.2 CBOD<sub>5</sub>

CBOD<sub>5</sub> removal to < 30 mg/l by the end of the trials was shown to be significantly negatively influenced ( $p < 0.05$ ) by higher initial carbon concentration and positively influenced by temperature using logistic regression analysis, but all four factors were significant using the ANOVA (Table 2). First order rate constants (Table 4.3), in general, increased with higher temperature and irradiance and decreased with higher OLRs and initial carbon concentrations. There was minimal change in the first order rate constants between columns operation at 15 °C with 240 mg/l and 80 mg/l initial carbon concentration, and suggests that the treatment rate is not impacted by carbon concentration between these levels at this temperature. At 15 °C the impact of the other factors (irradiance, OLR, initial carbon concentration), based on the relative change of CBOD<sub>5</sub> removal rates and final concentrations, are much less pronounced than at 5 °C.

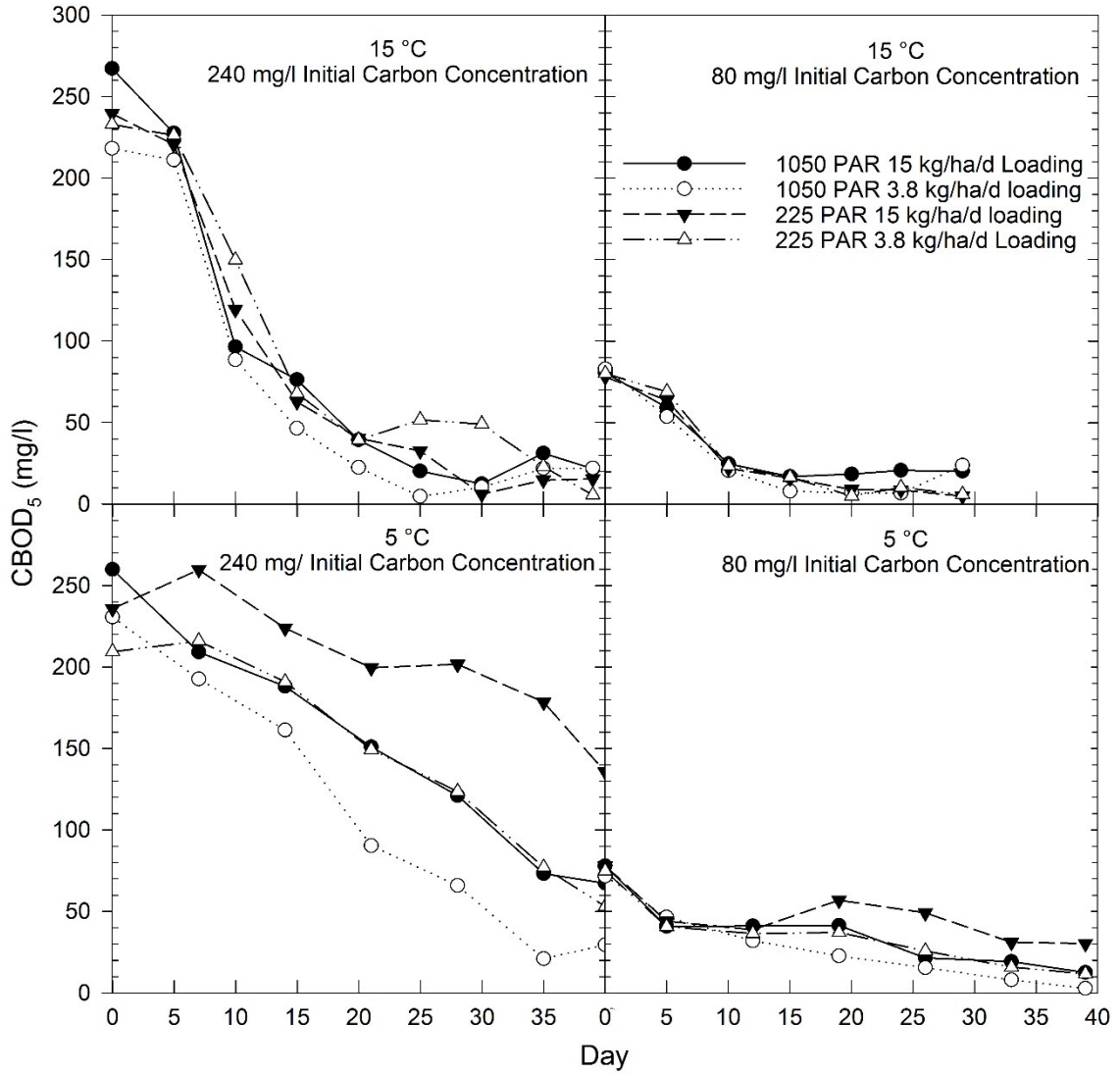


Figure 4.3 Measured CBOD<sub>5</sub> concentrations of water taken from the top sampling port (20 cm below surface) for the duration of each of the 16 unique trials.



Table 4.3 Calculated first order rate constants (k) for CBOD<sub>5</sub> removal of each unique trial.

Temperature (°C)	Light ( $\mu\text{e}/\text{m}^2/\text{s}$ )	Initial CBOD <sub>5</sub> (mg/L)	Organic Loading Rate (kg/ha/d)	First order rate constant (d <sup>-1</sup> )
15	1050	240	15	0.10
			3.8	0.15
		80	15	0.10
			3.8	0.16
	225	240	15	0.09
			3.8	0.09
		80	15	0.11
			3.8	0.14
5	1050	240	15	0.03
			3.8	0.08
		80	15	0.05
			3.8	0.08
	225	240	15	0.02
			3.8	0.04
		80	15	0.03
			3.8	0.05

Columns that were operated at 15 °C all had similar CBOD<sub>5</sub> concentrations at the end of trial, and were all measured to have end of trial CBOD<sub>5</sub> concentrations < 25 mg/l). Notably, columns receiving the high irradiance possessed slightly higher final CBOD<sub>5</sub> concentrations (Figure 4.3). Most of the columns reached concentrations less than 25 mg/l after 25 days of treatment, with the exception of the columns receiving low irradiance and high initial carbon concentrations and OLRs. CBOD<sub>5</sub> concentrations in columns operated at 15 °C decreased more rapidly than at the 5 °C temperature (Figure 4.3) therefore resulting in higher first order rate constants for CBOD<sub>5</sub> removal (Table 4.3).

At 5 °C, columns with an initial carbon concentration of 80 mg/l CBOD<sub>5</sub>, high irradiance and low OLR attained final concentrations below 25 mg/l. In contrast, all columns with 240 mg/l initial CBOD<sub>5</sub> exceeded 25 mg/l after 40 days, and, in the cases of high OLRs, columns contained elevated levels of CBOD<sub>5</sub> (60-130 mg/l) at the end of the trial. Only the columns with low OLR and high irradiance were approaching 25 mg/l CBOD<sub>5</sub> (29 mg/l average) after 40 days. Figure 4.3 illustrates that both the final CBOD<sub>5</sub> concentrations and removal rates in columns operating at 5 °C were more strongly affected by irradiance, initial carbon concentration and OLR than those operated at 15 °C.

### **4.3.3 Phytoplankton and Bacteria**

Phytoplankton populations increased with higher temperature and irradiance and decreased with higher initial carbon concentration, and OLR (Table 4.2,  $p < 0.05$ ). From Figure 4.4, it is apparent that temperature had the greatest influence on phytoplankton growth and population in the columns, where phytoplankton populations rose to more than  $10^6$  cells/ml within the first 10 days at 15 °C while populations remained at least one order of magnitude lower at 5 °C for a longer duration of the trial, and sometimes for the entire 40 day trial. The statistical analysis (Table 4.2) showed evidence that irradiance has an important influence on phytoplankton population. Figure 4.4 provides evidence that phytoplankton are more sensitive to initial carbon concentration, irradiance and OLR changes at low temperature, with greater differences in populations and growth observed at a temperature of 5 °C when compared to 15 °C. In contrast, maximum bacteria

populations and population growth rates did not appear to be impacted by any of the factors (Figure 4.5).

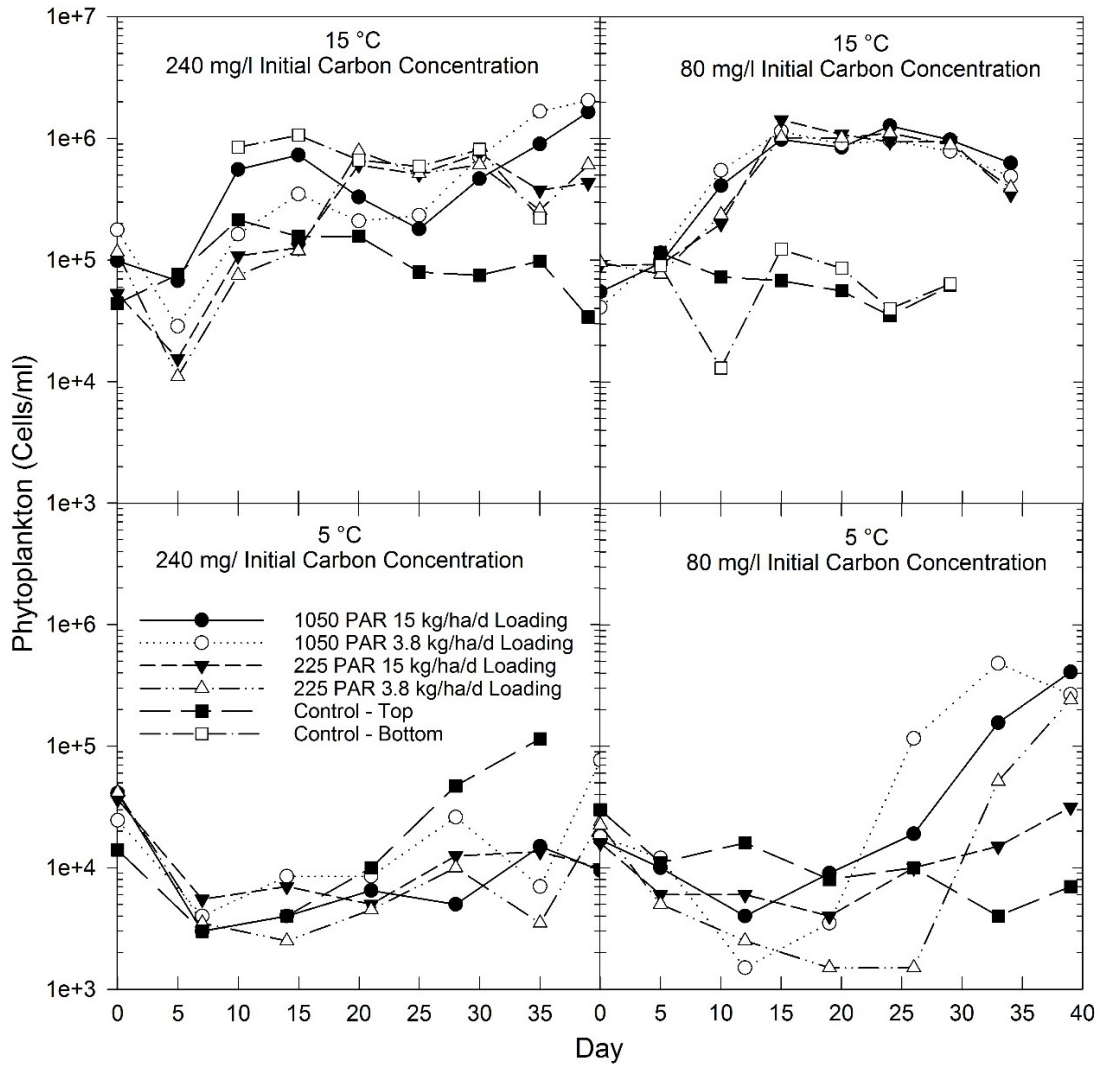


Figure 4.4 Measurements of phytoplankton populations in samples taken from the top sampling port (20 cm below surface), with exception of control – bottom, for the duration of each of the 16 unique trials.

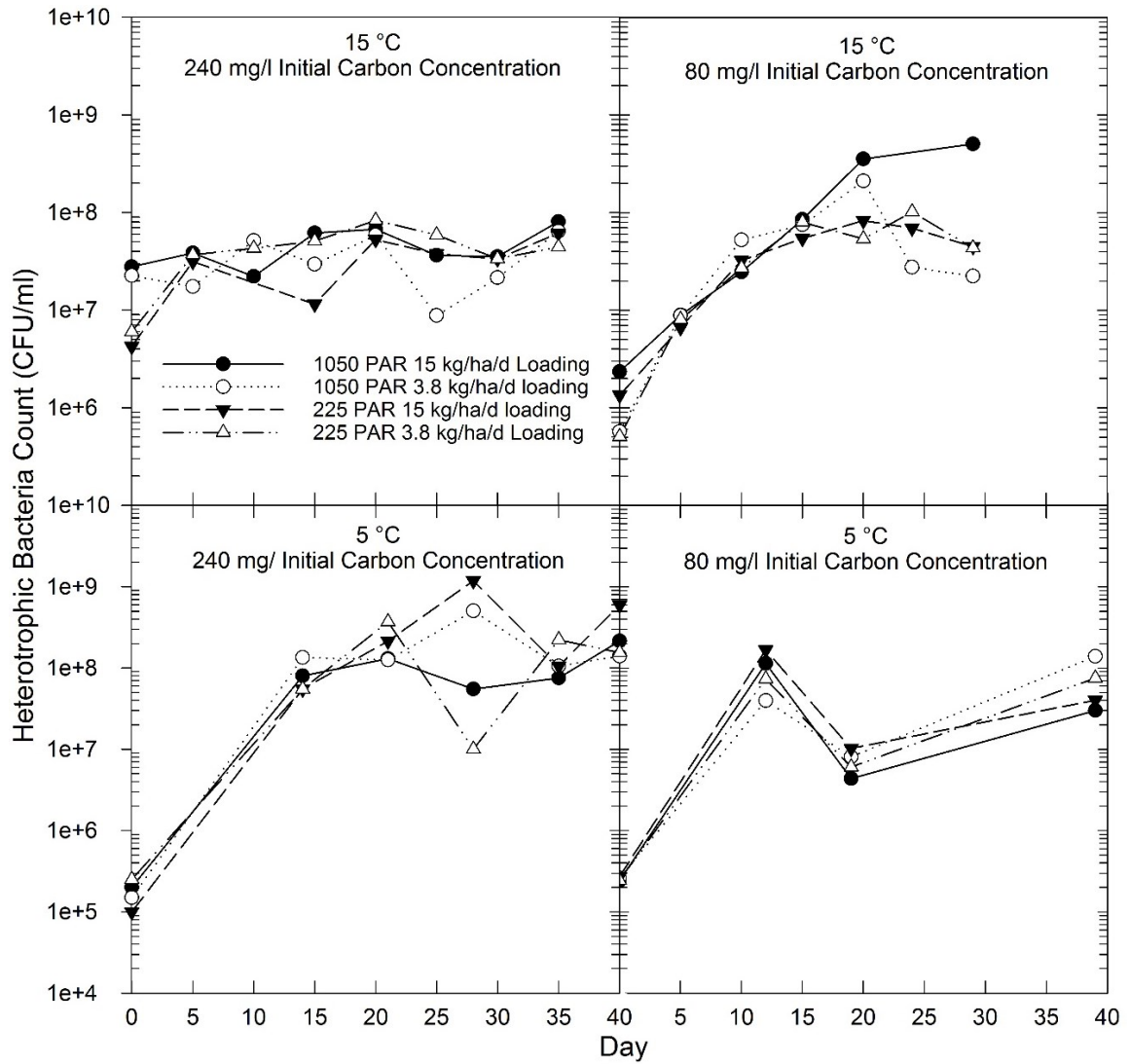


Figure 4.5 Measurements of heterotrophic bacteria populations in samples taken from the top sampling port (20 cm below surface) for the duration of each of the 16 unique trials.

#### 4.4 Discussion

All four experimental factors were found to impact the dissolved oxygen status and CBOD<sub>5</sub> removal rates in the model WSPs. In order of importance the factors were; temperature, initial carbon concentration, irradiance and OLR. Temperature was observed to be the most important factor, and the tested responses were more sensitive to irradiance, OLR and initial carbon concentrations in the trials conducted at 5 °C compared to 15 °C. All four factors influenced phytoplankton populations, which resulted in differences in the availability of dissolved oxygen through photosynthesis, and CBOD<sub>5</sub> removal rates. CBOD<sub>5</sub> removal rates were negatively impacted by the lack of oxygen when the model WSPs were anaerobic.

Microbial and phytoplankton growth and metabolism have been shown to increase exponentially with temperature within a suitable temperature range (generally 5-35 °C) (Ratkowsky et al., 1982; Eppley, 1972). Although maximum phytoplankton populations achieved over the trial duration in the test columns were higher at 15 °C than at 5 °C, as tested by ANOVA, it is unclear if the maximum attainable populations are truly higher at 15 °C as phytoplankton populations were still rising in columns operating at 5 °C at the end of the trial (Figure 4). It is also of interest that populations of phytoplankton in the trials conducted at 15°C increased throughout the trial, whereas in trials conducted at 5 °C decreases in the suspended phytoplankton population in the water column were observed prior to a noticeable growth phase. The reason for the initial decrease in phytoplankton population is not known, but it is possible that an acclimation period was required by the phytoplankton at lower temperature. Phytoplankton in the control columns increased quite slowly at 5 °C, in contrast to the phytoplankton populations in the simulated WSPs with low initial carbon concentrations, which increased much faster. This is an unexpected result, and it was not apparent why phytoplankton populations in the control would fail to grow as well at 5 °C as those in the simulated WSPs with low carbon concentrations. Dauta et al. (1990) also found that phytoplankton growth was more sensitive to light at cold temperatures 5 °C, and the apparent sensitivity to shifts in WSP conditions at low temperature reiterates the sensitivity of WSPs operating in cold climates. The strong relationship between phytoplankton community size and oxygen confirms that the phytoplankton community

is a strong determinant of oxygen state in Arctic WSPs. The increased phytoplankton population size and growth rate at higher temperature coincided with increased measurable oxygen concentrations earlier in the trial and higher CBOD<sub>5</sub> removal rates.

The initial carbon concentration was found to significantly ( $p > 0.05$ ) affect DO and CBOD<sub>5</sub> concentration. The bench scale experiment provides evidence that systems overloaded with organic carbon result in WSPs with lower phytoplankton growth rates and populations, anaerobic conditions and decreased CBOD<sub>5</sub> treatment performance. This is consistent with the findings of König (1984) and Athayde (2001) that showed facultative WSPs and Wastewater Storage and Treatment Reservoirs (WSTR) (batch reactor where the cell is filled over a short duration) with elevated levels of organic carbon loading/carbon concentrations had lower phytoplankton biomass, and potentially result in anoxic condition. In our experiment, the phytoplankton growth rates were initially impeded by higher initial carbon concentration, however the maximum measured phytoplankton concentrations ( $>10^6$  cells/ml) in columns that appeared to reach a climax community, were un-impacted and are comparable to concentrations found in WSPs in Northeast Brazil by Athayde (2001). It appears that high initial carbon concentrations are detrimental to phytoplankton growth, however, the initial carbon concentration had no observed impact on heterotrophic bacteria population or growth rates. The ability of the bacterial community to flourish in the columns with high carbon concentration, while the phytoplankton population struggles, results in an extended duration of anaerobic conditions and reduced CBOD<sub>5</sub> treatment performance. Variations between phytoplankton populations in the simulated WSPs were largely responsible for observed differences in oxygen status, and CBOD<sub>5</sub> treatment performance, as phytoplankton populations provide the oxygen required for aerobic respiration of the heterotrophic prokaryotic microbial population. It appears likely that reducing the organic carbon concentration in cell(s) at the onset of the summer treatment season will increase phytoplankton growth rates, promote facultative conditions in WSPs, and result in greater CBOD<sub>5</sub> removal rates.

Increased irradiance promoted increased phytoplankton growth, DO concentrations, and CBOD<sub>5</sub> removal rates. Again, the increased DO concentrations and CBOD<sub>5</sub> removal rates are attributed to increased phytoplankton activity; increased

phytoplankton oxygen production increased DO concentrations and stimulated oxygen demanding substance removal through aerobic respiration by the heterotrophic prokaryotic microbial community. Due to the high light attenuation coefficient of  $14 \text{ m}^{-1}$ , which was measured in the Pond Inlet WSP, and simulated in the columns, only 1% of the irradiance penetrates to a depth of 30 cm. This reduced light transmittance through the WSP limits the productive depth of the phytoplankton, or the photic zone. Light attenuation appears to be more limiting in Arctic WSPs, as compared to other studied WSPs, and is attributed to the highly concentrated wastewater that these WSPs are receiving (Smith, 1986). Shilton (2005), in his review of the literature, noted that typically phytoplankton have been observed to inhabit the top 40 cm of primary cells, but were capable of expanding their range to a depth of 50-60 cm in following cells.

The OLR was shown to have the least impact of the four experimental factors on the tested responses. The minimal impact is attributed to the daily addition of wastewater (and by association carbon) being minimal relative to the existing carbon load that has accumulated over the winter. At  $15 \text{ }^\circ\text{C}$ , there was minimal difference between final  $\text{CBOD}_5$  concentrations in columns operating with the two OLRs. This suggests that single-cell WSPs operating with an average temperature of  $15 \text{ }^\circ\text{C}$  during the summer may have an OLR of at least  $15 \text{ kg CBOD}_5/\text{ha}/\text{day}$  with minimal impacts on treatment performance. Columns at  $5 \text{ }^\circ\text{C}$  had reduced phytoplankton populations (Figure 4) and elevated  $\text{CBOD}_5$  concentrations (Figure 3) for the duration of the trial when initial carbon concentration was high (particularly at low irradiance). To prevent the development of conditions with reduced phytoplankton population and elevated  $\text{CBOD}_5$  in WSPs, OLRs may need to be lower than  $15 \text{ kg CBOD}_5/\text{ha}/\text{day}$  as the duration when arctic WSPs reach  $15 \text{ }^\circ\text{C}$  is potentially short. When columns were operated at  $5 \text{ }^\circ\text{C}$ , with high initial carbon concentrations, and low irradiance, the final  $\text{CBOD}_5$  concentration at the end of the trials with high OLR was above  $130 \text{ mg/l}$ . In contrast, the  $5 \text{ }^\circ\text{C}$  columns with low OLR but the same initial carbon concentrations possessed  $\text{CBOD}_5$  concentrations of  $60 \text{ mg/l}$  by the end of trial. This demonstrates that for WSPs mainly operating at low temperatures ( $5 \text{ }^\circ\text{C}$ ), as is potentially the case of arctic WSPs for a large portion of the duration of the summer treatment period, the OLR during the summer treatment period can become an important operational design parameter that will impact  $\text{CBOD}_5$  treatment. A careful

assessment of the operational temperature of arctic WSPs is critical for the determination of OLRs for arctic WSPs and prediction of treatment performance, a finding that is in agreement with a previous study on the treatment performance of four arctic WSP systems in Nunavut (Ragush et al., 2015a).

#### **4.5 Conclusion**

This bench scale experiment demonstrated that adequately sized and designed WSPs can meet commonly applied secondary treatment standards for CBOD<sub>5</sub> (<25 mg/l), even under adverse arctic conditions when temperatures are as low as 5 °C during the summer treatment period. The CBOD<sub>5</sub> first order treatment rate coefficients of the simulated WSPs in this study ranged from 0.03-0.08 d<sup>-1</sup> at 5°C and 0.10-0.16 d<sup>-1</sup> at 15°C. These values are within the range of the rate constants used in design equations provided by US EPA (1983) (0.04-0.10 and 0.09-0.23). Results from this experiment suggest that selection of a lower (more conservative) rate constant (i.e.  $\leq 0.05$  d<sup>-1</sup>) is appropriate for Arctic WSPs.

Temperature, initial carbon concentration, irradiance, and OLR were all found to significantly influence oxygen state and CBOD<sub>5</sub> removal. Higher temperature and irradiance increased CBOD<sub>5</sub> removal rates while higher OLR and initial carbon concentrations resulted in lower CBOD<sub>5</sub> removal rates. This study provides evidence that CBOD<sub>5</sub> treatment performances and oxygen states of WSPs operating in the temperature range of 5 to 15°C, the temperature range of Arctic WSPs, are highly sensitive to environmental and operational changes. This finding is attributed to the sensitivity of phytoplankton community dynamics in this low temperature range, as phytoplankton populations were found to be closely linked to the oxygen state and CBOD<sub>5</sub> treatment performance of these systems.

This study highlights that: (i) maximizing the average temperature in the WSP, and (ii) decreasing the initial carbon concentrations at the onset of the summer treatment season are design strategies that are likely to improve CBOD<sub>5</sub> treatment rates and effluent quality in Arctic WSPs. CBOD<sub>5</sub> removal rates are also likely to be improved by supplementing the dissolved oxygen through aeration to promote the reduction of CBOD<sub>5</sub> in cells with high carbon concentrations (as persistent anaerobic conditions were found to



decrease treatment rates). Finally, increasing the average irradiance or increasing the transmittance of irradiance to greater depths will likely increase the CBOD<sub>5</sub> removal rate by promoting aeration by phytoplankton. In terms of design, the use of multi-cell WSP systems, where deep, anaerobic storage cells are used for winter storage, and then decanted into shallow (< 1 m) facultative treatment cells during the summer season, would be a logical upgrade strategy. Decreasing operational depth in facultative cells can be expected to improve operational temperature and increase average irradiance while the use of deep winter storage cells for pre-treatment will help to reduce the initial carbon concentration and is likely to improve the transmittance of irradiance in following cells.

## **Chapter 5 Representing Phytoplankton Growth in Eutrophic Systems – Insights from Modeling Oxygen Dynamics and Carbon Removal in Arctic Waste Stabilization Ponds**

### **5.1 Introduction**

Waste Stabilization Ponds (WSPs), in essence shallow highly eutrophic water bodies, are widely used for municipal wastewater treatment serving to reduce the CBOD<sub>5</sub> concentration prior to discharge. However, the design and operation of arctic WSPs are typically different than those used in warmer climates due to the prevailing cold temperatures, and short ice-free time periods. Arctic WSPs are operated as controlled discharge storage ponds; raw wastewater is continuously received into the WSP year round, but effluent is only discharged once per year, typically in late summer/early fall during a period of 2-3 weeks. The surfaces of the arctic WSPs stay frozen for 9-10 months and influent wastewater temperatures will quickly approach 0 °C, limiting the capacity for biological treatment. As a result, WSPs at the start of the summer treatment season, or ice free period, contain high concentrations of oxygen demanding substances (five day carbonaceous biochemical oxygen demand (CBOD<sub>5</sub>) >200 mg/l). The reduction in CBOD<sub>5</sub> concentrations during the summer treatment season is highly variable (Ragush et al. 2015a), and the limitations and best operational practices of single cell WSPs operating in arctic environments have not been deeply investigated.

The current best practices were developed from the performance of systems operating in northern climates and expert experience (Dawson & Grainge, 1969; Heinke et al., 1991). However, these design guidelines and operational best practices presently in use in the Arctic were meant to meet less stringent effluent regulations (Nunavut Water Board, 2015) than are currently being implemented across Canada (Government of Canada, 2012). Also, the systems used to develop the guidelines were generally; i) located in cold temperate (such as northern interior United States or Canada) or sub-arctic climates and/or are ii) continuous flow systems (US EPA, 1983). The applicability of such guidelines for the design of single cell arctic WSPs under the progressing regulatory framework warrants further scrutiny.

To better understand the climatic and operational factors influencing the performance of single cell WSPs in cold climates Ragush et al. (2015b) used a bench

scale model system to examine the effect of temperature, irradiance, organic loading rate and initial carbon conditions. In this experiment, mesocosms were constructed to represent WSPs operating in the Arctic operating for 40 days, representing the length of the summer treatment season in many Nunavut (Canada) communities. A factorial design was utilized to assess how arctic temperatures, irradiance, organic loading rate, and organic concentration at the beginning of the summer treatment season would influence the development of aerobic conditions and CBOD<sub>5</sub> treatment performance. Using a statistical analysis, Ragush et al. (2015b) found that all four factors impacted the oxygen state and CBOD<sub>5</sub> removal rates. To help direct future research and examine the operational limitations of arctic WSPs, it was concluded that a mechanistic model would deepen the understanding of the biological treatment processes occurring in the system. The model was derived with the intent that it may be used to assess knowledge gaps and may further be developed into a tool to assess arctic WSP design and optimization. During the development of the model, it was found that models examined from the literature poorly represented phytoplankton growth in our eutrophic environment with high light attenuation. The poor phytoplankton growth representation lead to a holistic approach to derive a mathematical representation capable of capturing the dynamics witnessed in bench-scale experimentation through consideration of the phytoplankton's response to the light and growth stressors. This chapter presents the development of a novel mathematical representation of phytoplankton growth in environments that are eutrophic and have high light attenuation, and is likely to have merit for others ecosystems where phytoplankton is light limited. Incorporating the novel depth integrated phytoplankton growth representation, we present a process-based model to predict dissolved oxygen and CBOD<sub>5</sub> concentrations in WSPs and provide an assessment of the local sensitivity of parameters of such a model through a one-factor-at-a-time (OFAT) sensitivity analysis. A brief discussion of simulation results of the sensitivity analysis is provided in the context of WSP design, but may also potentially provide insight into management of other eutrophic systems.

## 5.2 Model Development

The use of process-based models to design and evaluate wastewater treatment processes, specifically activated sludge systems, is well established (Orhon & Artan, 1994), and principles from these systems have also been coupled with ecosystem models and applied to WSPs (Beran & Kargi, 2005; Fritz et al., 1979; Buhr & Miller, 1983; Banks et al., 2003; Moreno-Grau et al., 1996). The models presented in the aforementioned literature display a large range in complexity and formulations, depending upon the studies objectives and design characteristics of the system. These models were reviewed and assessed for their applicability to our system and focus; predicting dissolved oxygen levels and CBOD<sub>5</sub> removal in WSPs operating in arctic environments. Banks et al. (2003), an adaption of Buhr & Miller (1983.), presented a box model of the photic zone (vertical depth zone where phytoplankton can grow photosynthetically), and forms the cornerstone of the model presented in this paper. However when Banks et al.'s formulation was applied to the system presented by Ragush et al. (2015b) it was found that it was unable to adequately predict the oxygen state and CBOD<sub>5</sub> concentration. The poor agreement is believed to be due to inherent assumptions present in the formulation of Buhr & Miller (1983) whose system was a high-rate algal pond, which is shallow and has a paddle system engineered to create a continually well-mixed conditions. The continually well-mixed nature of the system being examined by Buhr & Miller is inconsistent with single cell WSPs operating in the Arctic, and it was determined that the model poorly represented phytoplankton in our eutrophic system. A new modeling approach, making several adaptations to Buhr & Miller (1983), was developed to better represent the features of an arctic WSP such as greater depth and limited mixing. The focus of our study was the mechanisms of CBOD<sub>5</sub> removal and dissolved oxygen concentration dynamics, and in particular, the model needed to adequately represent the length of time required for algae populations to create an aerobic (> 2 mg/L dissolved oxygen) environment under arctic temperature and light conditions. One of the ultimate objectives of this work is to identify organic loading regimes for arctic WSPs that facilitate the formation of aerobic environments within the relatively short (approximately 40-60 days) summer treatment season.

Figure 5.1 provides an overview of the model along with references to the key equations listed in Table 5.1 that were used to represent the major processes. The model was developed as a box model of the photic zone, and state variables and parameters were vertically-integrated over the depth of the photic zone. External inputs into the photic zone were additional wastewater, and irradiance. Exports from the photic zone were bacteria and phytoplankton through sinking. Gas exchange of oxygen and carbon dioxide between the atmosphere and photic zone was included as a transboundary interaction. Within the photic zone the growth and dynamics of bacteria and phytoplankton populations and their metabolites of oxygen, carbon dioxide, and carbon (in the form of CBOD<sub>5</sub>) were modeled. Nutrients other than carbon, such as nitrogen and phosphorous, were excluded because their concentrations in both field scale and experimental WSPs are high, and it was assumed that they would not impact biological processes by being limiting (Ragush et al., 2015a; Ragush et al., 2015b). Table 5.2 provides a list of model parameters and their description.

Matlab, version R2015b, by MathWorks (Massachusetts, USA) was used to implement a numerical solution to evaluate the system of equations. The simulations were initialized with phytoplankton and bacteria concentrations that were reported by Ragush et al. (2015 b) at the beginning of the experiment. The model formulation is discussed in the following section and the discussion will refer back to the model equations as numbered in Table 5.1. Equations used to explain the development, but not included in Table 5.1 will include, the prefix 5, the chapter number.

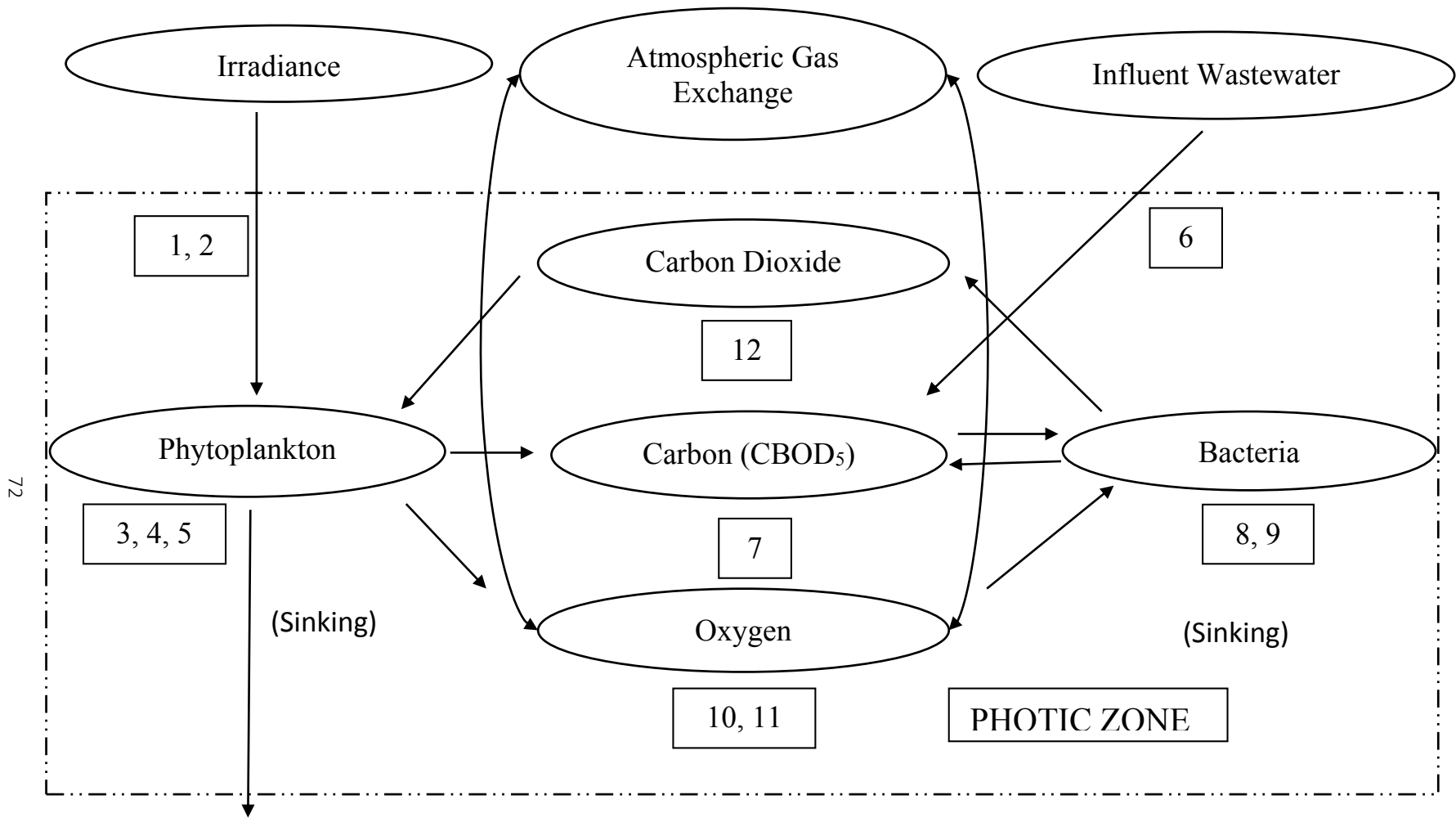


Figure 5.1 Diagram of modeled processes with listed applicable equations next to process arrows.

Table 5.1 List of model equations.

#	Equation	Description
1	$I_{av} = I_0 \frac{1 - e^{-(K_w + K_p * A) * Z}}{(K_w + K_p * A) * Z}$	Average Irradiance across depth (considering shading by phytoplankton)
2	$z_{1\%} = \frac{\log(0.01)}{-K_e}$	Depth of 1% light transmittance (negating phytoplankton)
3	$Fdis = \left( (1 - ED_{factor}) * \left( 1 - e^{-4e^{-(AGS)^A}} \right) + ED_{factor} \right)$	Growth inhibition of phytoplankton as caused by crowding
4	$U_a = U_{max} Fdis \frac{CO_2}{K_{CO_2} + CO_2} * \frac{I_{av}}{I_{av} + I_{halfsat}}$	Growth rate of phytoplankton
5	$\frac{dA}{dT} = (U_a - K_{ad} - K_{as}) * A$	Change in phytoplankton = (Growth – death – settling) * phytoplankton
6	$CBOD_{5inf} = Raw * SolCBOD_5$	Addition of CBOD5 into photic zone = CBOD5 concentration * solubility
7	$\frac{dS}{dT} = -OUR * B + DL * \frac{CBOD_{5inf}}{V} * \frac{ColumnZ}{Z} + K_{ad} * A * 0.5 + K_{db} * 0.7 * B$	Change in CBOD5 = utilization bacteria + daily loading + inputs of phytoplankton death + inputs bacteria
	Note S >= 0	
8	$U_b = U_{max_b} * \frac{S}{K_s + S} * \frac{O_2}{K_{O_2} + O_2} * (1 - BGS * B)$	Bacteria growth rate = carbon substrate limitation * Oxygen limitation * self-limitation
9	$\frac{dB}{dt} = (U_B - K_{br} - K_{bs})B$	Change in bacteria = (growth – death – settling) * bacteria
10	$\text{If } IO_2 > OUR_b * B$ $OUR = OUR_m * \frac{O_2}{O_2 * K_{O_2}} + OUR_b$ Else $OUR = \frac{O_2}{B}$	Oxygen utilization rate – depends on the available oxygen and the bacterial population
11	$\frac{dO_2}{dt} = Y_{oa} * U_a * A - OUR * B + Kl_{O_2} * \frac{Area}{V} * (CS_{O_2} - O_2)$	Change in oxygen = Production by phytoplankton – consumption by bacteria + aeration

12	$\frac{dCO_2}{dt} = \frac{Y_{cb}}{Y_{ob}} * OUR * B - Y_{ca} * U_a * A + Kl_{CO_2} * \frac{Area}{V} * (CS_{CO_2} - CO_2)$	Change in carbon dioxide = production by bacteria – consumption by phytoplankton + aeration
13	$CBOD_5(t) = S(t) + 0.5*(A(t))$	CBOD5 = Carbon pool CBOD <sub>5</sub> + CBOD <sub>5</sub> of phytoplankton

Table 5.2 List of model state variables and constants.

Symbol	Definition	Value & Units
<i>State Variables</i>		
A	Phytoplankton concentration (algae)	mg/l Wet Mass
B	Bacteria concentration	mg/l Wet Mass
S	Substrate concentration (carbon)	mg/l (CBOD <sub>5</sub> )
O <sub>2</sub>	Oxygen concentration	mg/l
CO <sub>2</sub>	Carbon dioxide concentration	mg/l
Fdis	Reduction in phytoplankton growth due to preferred distribution reducing irradiance	Unitless
I <sub>av</sub>	Average irradiance across photic depth (Z) with phytoplankton	$\frac{ue}{m^2} s^{-1}$
<i>Constants</i>		
CBOD <sub>5inf</sub>	Influent CBOD5 concentration	550 mg/l
ColumnZ	Depth of water column (total depth)	1.25 m
C <sub>sO<sub>2</sub></sub>	Saturation concentration oxygen	11.3 mg/l (5 °C) 8.9 mg/l (15 °C) NIST (2015)
C <sub>sCO<sub>2</sub></sub>	Saturation concentration carbon dioxide	1.01 mg/l (5 °C) 0.75 mg/l (15 °C) Benson & Krause (1984)
DI	Daily volumetric loading	0.0125 or 0.05 l/d
I <sub>avclear</sub>	Average irradiance across photic depth (Z) with no phytoplankton	(Eqn 1) $\frac{ue}{m^2} s^{-1}$
I <sub>o</sub>	Surface incident light	225 & 1050 $\frac{ue}{m^2} s^{-1}$
K <sub>w</sub>	Attenuation coefficient of the wastewater	14 m <sup>-1</sup>
SolCBOD <sub>5</sub>	Solubility Ratio of CBOD <sub>5</sub>	0.5
V	Volume	0.0228 m <sup>3</sup>
Z <sub>1%</sub>	Photic zone depth (1% measured irradiance)	(Eqn 2) m



### 5.2.1 Temperature

The temperature dependence of chemical, physical and biological processes was modelled based on the van't Hoff-Arrhenius relationship (equation 5.1) from Metcalf & Eddy (2003) who provide a range of 1.024 – 1.08 for  $\theta$  for biological processes, and for aeration rates, Elmore and West (1961) suggests a value of 1.024 (equivalent to a  $Q_{10}$  of 1.3). For simplicity, in the model all temperature dependent processes were modelled with a  $\theta$  of 1.024. The van't Hoff-Arrhenius relationship was not applied to phytoplankton maximum growth rate because literature supported a larger temperature dependence (Dauta (1990) would suggest a  $Q_{10}$  of 2-2.5), and calibrated values of 0.32 and 0.75 day<sup>-1</sup>) were used at temperatures of 5 °C and 15 °C respectively.

$$Rate_{Temperature} = Rate_{20C} * \theta^{(T-20)} \quad (5.1)$$

### 5.2.2 Phytoplankton

Modeling of phytoplankton populations and growth must account for the vertical distribution of the population and the vertical and local variation in irradiance, nutrients, and metabolites. The modeling of phytoplankton growth in a WSP requires unique mathematical consideration because the nature of the environment is atypical as it has a high light attenuation coefficient, poor mixing, and is nutrient rich. In this instance, it is the depth of light penetration, and not a defined mixed depth, that defines the zone where phytoplankton can grow. Additionally the phytoplankton biomass is limited at a carrying capacity, despite there being an apparent ample concentration of nutrient. This carrying capacity is most likely attributed to stressors from limitations of cellular mass transfer rates and localized nutrient and metabolite concentrations in high phytoplankton density spaces. The development of the depth integrated phytoplankton response is discussed below, and the mathematical representation of the response requires careful consideration of three factors: i) migration, ii) self-shading and iii) localized resource competition. The coupling of these forces to form the phytoplankton growth equation is discussed below.

#### 5.2.2.1 Production irradiance curves

The growth rate of phytoplankton increases with increasing irradiance until an optimal irradiance results in a maximum growth rate, after which a decline in growth rate

is observed with increasing irradiance due to photoinhibition (Dauta, 1990). The observation of photoinhibition has been documented in small batch experiment where phytoplankton is confined and subjected to the high irradiance. However, provided the individual phytoplankton has motility that can overcome the local physical transport forces (such as in a stagnant water body), it can be rationalized that the phytoplankton population will migrate deeper in the water column, removing the stressor, and minimizing the negative impact of photoinhibition. Such a response would lead to a threshold irradiance, where if exceeded, vertically integrated phytoplankton population growth rates will not increase or decrease with increasing irradiation, and it can be rationalized that the inclusion of photoinhibition in this model unnecessary. Prior to model development, the depth integrated growth rate of phytoplankton populations was assessed using piece-wise integration of the production-irradiance relationships of an equally distributed population over the depth range. The results supported the use of a rectangular hyperbola such as the Michaelis-Menten function ( $X/(X+X_{\text{halfsat}})$ ), used in this paper, for the production-irradiance curve.  $I_{\text{halfsat}}$ , the phytoplankton's half saturation growth rate for irradiance, in this model is constant. However, the parameterization of  $I_{\text{halfsat}}$  has been shown to be dependent on phytoplankton species, environmental conditions (specifically temperature) and the structure of the mathematical model (Beran & Kargi, 2005; Moreno-Grau et al. 1996; Dauta et al. 1990). As a result, literature values of  $I_{\text{halfsat}}$  range greatly and must be carefully selected when being used in a model of phytoplankton growth because model dynamics are potentially highly sensitive to  $I_{\text{halfsat}}$  values, as is the case in this model (Table 5.5).

Examining the expected physiological response of phytoplankton populations under low and high irradiance conditions helps to provide clarity to the utility of Michaelis-Menten for modeling of depth integrated phytoplankton growth. Under low light conditions, when irradiance is essentially limiting growth, a small increase in irradiance will rapidly increase the phytoplankton population growth rate as the irradiance is limiting. The use of the Michaelis-Menten function for the depth integrated phytoplankton population at high irradiance levels, where photoinhibition of individuals will occur, requires clarification. Assuming already high levels of surface irradiance (well beyond where photoinhibition occurs in individuals), an increase in surface

irradiance will not result in a change in the depth integrated phytoplankton growth rate under the conditions where phytoplankton have control on their vertical position because there is no change in the available irradiance to the phytoplankton population. Initially this is counter intuitive as one would expect higher irradiance to increase phytoplankton growth, but if the phytoplankton population can be expected to optimize its growth, phytoplankton at a highly irradiated shallow depth will use their motility to migrate to a deeper depth with optimal irradiance. From an energy perspective, there is no net change in input light energy to the phytoplankton (between the two scenarios of phytoplankton at a shallow depth with less surface irradiance and phytoplankton at a deeper depth with higher surface irradiance) because in the high irradiance scenario water and particles have absorbed/deflected the additional energy prior to reaching the phytoplankton cells, and consequentially no net change in growth rates can be expected. From a mathematical perspective, the phytoplankton-irradiance relationship under stagnant water conditions is likely saturating, and may share the similar phytoplankton-nutrient relationships. This type of response is well suited to being represented by hyperbolic functions such as the Michaelis-Menten function.

#### 5.2.2.2 Phytoplankton available irradiance

Solar radiation provides the energy for photosynthesis, and the total (vertically integrated) phytoplankton production will be proportional to the amount of energy absorbed. Not all of the irradiance that reaches the surface of the water column can be utilized by the phytoplankton because light energy is also absorbed or reflected by particles. Phytoplankton also acts as a particle and absorbs irradiance, reducing the available irradiance to other individuals at greater depth (i.e. effectively increasing light attenuation), and as the population increases the available irradiance per individual must decrease, known as self-shading. There is also localized competition for resources, in the form of nutrients and metabolites (oxygen and carbon dioxide), between phytoplankton and this competition is expected to limit the maximum density of the phytoplankton population (Mellard et al., 2011). A density-dependent, or logistic growth, imposes a constraint on the maximum population production, because the phytoplankton population cannot utilize all the irradiance as the phytoplankton population grows because of other limiting factors, such as the exchange of metabolites.

As a result of our assumption that individual phytoplankton optimize their growth conditions, the phytoplankton are distributed around the optimal irradiance depth. Furthering this concept, the depth over which phytoplankton are distributed can be expected to broaden with increasing population as individuals seek out areas with the best growth conditions; a behaviour which was qualitatively observed in the experimental columns as a noticeable dark green band would vertically broaden in the column during initial stages of the trial.

The transmittance of light as it travels a distance through a media is commonly described by Beer-Lambert's law:

$$I_z = I_0 e^{-kz} \quad (5.2)$$

Where:  $I_0$  = irradiance at surface (depth 0 m),  $I_z$  = irradiance at depth  $z$  ( $\mu\text{e}/\text{m}^2/\text{s}$ ),  $k$  = attenuation coefficient ( $\text{m}^{-1}$ ), and  $z$  = depth (m)

The attenuation coefficient is a property of the water and contents of the water. The attenuation coefficient can be split into different contributors such as water, color, particulates and phytoplankton (Lorenzen 1972). When modeling a system with phytoplankton, if modeling integrated growth, it is required to define a depth, or photic zone. The most common way to do this is:

$$-\frac{\text{Ln}(0.01)}{k} = z_{1\%} \quad (5.3)$$

$z_{1\%}$  may be interpreted as the photosynthetic active depth, or the depth where 1% of the surface light may be measured in a water column with attenuation properties of  $k$ . The definition of the photic zone as the depth to 1% of measured irradiance is suggested as appropriate because it is unlikely that much phytoplankton growth will occur at depths greater, especially in systems with high light attenuation properties. Nonetheless, it must be recognized as an arbitrary definition that potentially impacts model parameterization.  $z_{1\%}$  is not impacted by the intensity of irradiance at the surface, and initially the depth of the photic zone being expressed based upon this definition may seem erroneous. However, in the development of a model that vertically integrates the photic zone, if the photic zone is defined by an irradiance minimum the photic depth changes with a change in surface irradiance or water characteristic, and creates new challenges with respect to model implementation and interpretation.

Phytoplankton concentrations change with depth and time, and therefore  $k$  was split into two contributors;  $k_w$  (considered a property of the water and its constituents), and  $k_p$  (a property of the phytoplankton concentration).

$$k = k_w + k_p(A) \quad (5.4)$$

Where:  $k_w$  = attenuation coefficient of water and constituents ( $m^{-1}$ ),  $k_p$  = attenuation coefficient of phytoplankton ( $m^{-1}/[mg/l]$ ), and  $A$  = phytoplankton concentration ( $mg/l$ )

$k_w$  is a property of the water and all its constituents, and was considered homogenous and not to change over the duration of the simulation.  $K_p$  is a property of the phytoplankton and accounts for the absorption of light by phytoplankton. Although neither the phytoplankton concentration nor the irradiance is constant with depth, an appropriately parameterized box model is not compromised by using the averages in the depth. Behrenfeld & Falkowski (1997) in their work assessing integration of phytoplankton-irradiance relationships demonstrated that using averages at higher levels of abstraction can be a valid approach, provided that parameterization is consistent with the level of abstraction.

The average light in the photic zone can be approximated by incorporating attenuation into Beer-Lambert's law and integrating over the photic zone and averaging over the depth:

$$I_{av} = \frac{1}{Z_{1\%}} \int_0^{Z_{1\%}} I_0 e^{-kz} dz = \frac{1 - e^{-(k_e + k_p * A)(Z_{1\%})}}{(k_e + k_p * A)Z_{1\%}} \quad (1)$$

The typical (average) irradiance in the photic zone was used in the Michaelis-Menten equation for the irradiance-growth response of phytoplankton (Equation 4). Additionally, this method of formulation is beneficial because it incorporates light absorption by phytoplankton as  $I_{av}$  is reduced with increasing phytoplankton.

### 5.2.2.3 Phytoplankton distribution

If light was the only controlling factor of phytoplankton growth, it would be optimal for phytoplankton to grow in large concentrations over a narrow depth where irradiance was optimal. However, although populations tend to reside in greatest concentrations near the depth of optimal irradiance, there is a limitation to the phytoplankton density, and the population expands out from the area of optimal

irradiance as it grows. It is theorized by the authors that the limitations imposed by extracellular mass transfer rates (diffusion) of metabolites, likely carbon dioxide and oxygen, is the strongest force that structures the vertical distribution. The constraint of a maximum phytoplankton density is a critical element in modeling phytoplankton dynamics in nutrient rich systems with minimal currents. Conceptually, phytoplankton can only obtain its physiological maximum growth rate at low populations, when individuals are not experiencing some limiting force, and the growth rate of the population must decrease with increasing population (competition) for resources. Populations that experience self-limitation are commonly described with logistic growth models. Logistic growth has been used extensively to describe the growth of bacteria populations, and the theory has been expanded upon to create more complete expressions such as the Gompertz model (Contois, 1959; Zwietering et al., 1990), however, it has not as commonly been explicitly applied to phytoplankton growth.

In our model, phytoplankton growth was formulated on the premise that the phytoplankton population, having motility, will distribute itself within the water column to optimize its growth rate, and the growth rate of the phytoplankton population will approach its maximum with respect to light. However, as the population grows, individuals will inhabit areas with less optimal irradiance, and the net result is that the integrated population growth rate decreases. The decrease in population growth rate due to competition for metabolites due to crowding and the associated increased vertical distribution of phytoplankton population is mathematically represented by Gompertz model, a generalized logistic model, and is denoted as  $F_{Dis}$  (equation 3). The maximum of  $F_{Dis}$ , is the maximum growth rate of a phytoplankton population (when the growth rate is not limited by light, nutrients or density), and the minimum value of  $F_{Dis}$  effectively the value of ED (equal distribution factor), can be interpreted as the integrated growth rate at the carrying capacity in the photic zone population. The value of ED was theoretically approximated by integrating the growth rate of a large equally distributed population over the photic zone. AGS (Phytoplankton growth self-suppression) defines the relationship of the growth rate to population density. The parameters of  $F_{Dis}$ , AGS and ED can be estimated through review of literature for maximum population growth rates of phytoplankton, however AGS and ED were largely used in calibration of the model

because more experimentation of the relationship between phytoplankton density and phytoplankton growth is needed.

### **5.2.3 Bacteria**

A logistic growth model (Equation 8), with a death term was used to describe bacteria growth. Bacterial growth suppression (BGS) is approximately (an approximation due to the inclusion of bacteria death terms) the inverse of the carrying capacity, as BGS multiplied by the maximum bacteria population will provide a value of 1, and result in bacteria growth of zero. The aerobic metabolism was based on an oxygen utilization rate (OUR) (units of  $\text{mg O}_2/\text{mg bacteria day}^{-1}$ ) that consisted of the basal oxygen utilization rate ( $\text{OUR}_b$ ) required to sustain the existing population, and an additional oxygen utilization rate ( $\text{OUR}_m$ ) required for the population to grow (Equation 10). It was reasoned that in the case there was less oxygen available than desired by the bacteria, the bacteria would use all the oxygen (resulting in an OUR equal to the available oxygen concentration divided amongst the bacteria concentration). The model does not consider the potential anaerobic growth of the heterotrophic bacteria. The removal of  $\text{CBOD}_5$  was equivalent to the amount of oxygen used, and the production of  $\text{CO}_2$  was computed based on the stoichiometry of bacteria growth.

### **5.2.4 Carbon Cycling**

Phytoplankton and bacteria return carbon back into the organic pool upon death (Equation 7). From literature it was estimated that 1 mg of dry mass phytoplankton has a chemical oxygen demand (COD) of 1 mg (Boyd 1973) and 1 mg of bacteria results in a COD of 1.4 mg (Gaudy Jr. et al. 1964). The general relationship of 1 mg/l  $\text{CBOD}_5$ : 2 mg/l COD was used resulting in 0.5 mg  $\text{CBOD}_5/\text{mg}$  phytoplankton and 0.7 mg  $\text{CBOD}_5/\text{mg}$  bacteria. Additionally, the  $\text{CBOD}_5$  of the phytoplankton was also accounted for in equation 13 by adding the  $\text{CBOD}_5$  of the phytoplankton to the  $\text{CBOD}_5$  of the organic pool.

## 5.3 Results and Discussion

### 5.3.1 Model Calibration and Performance

Experimental results from Ragush et al. (2015b) were used to calibrate the model. The model calibration was performed by fitting the model to the experimental results of CBOD<sub>5</sub> and dissolved concentrations obtained at 5 °C, and then validating against experimental results generated at 15 °C. Maximum phytoplankton growth rates ( $U_{max_a}$ ) were set to the values provided by Dauta (1990).  $U_{max_a}$  was then calibrated at each temperature condition, however, the calibration values (0.32 and 0.75 days<sup>-1</sup> at 5 and 15 °C, respectively) represented a minor adjustment from growth rates provided by Dauta (0.3 and 0.7 days<sup>-1</sup>). During the calibration of maximum phytoplankton growth rate, the model was calibrated at each temperature with a 240 mg/l initial carbon concentration, and validated at 80 mg/l initial carbon concentrations. The values of the calibrated parameters are provided in Table 3. The performance was qualitatively compared to the experimental CBOD<sub>5</sub> treatment and measured dissolved oxygen concentrations of Ragush et al. (2015b) (Figures 5.2 and 5.3 respectively). Visual inspection suggests the model captures the general trends and effectively differentiates system dynamics for the various conditions. Statistical assessment of the goodness of fit of the model was not performed because the focus of this paper is the modeling approach for an arctic WSP and the examination of such a model through a sensitivity analysis. The absolute fit to the data was not the paramount focus of this work. It was of the authors opinions that much broader testing of the model would be required before the statistical assessment of the goodness of fit would be meaningful.



Table 5.3 Manually calibrated model parameters.

Parameter	Definition	Units	Value
Ihalfsat	Irradiance half saturation of phytoplankton	$\frac{\mu\text{e}}{\text{m}^2} \text{s}^{-1}$	30
K <sub>ad</sub>	Phytoplankton death rate	day <sup>-1</sup>	0.05
K <sub>as</sub>	Phytoplankton settling	day <sup>-1</sup>	0.05
K <sub>bd</sub>	Bacteria death rate	day <sup>-1</sup>	0.025
K <sub>bs</sub>	Bacteria settling	day <sup>-1</sup>	0
K <sub>c</sub>	Half saturation of phytoplankton on carbon dioxide	$\frac{\text{mg CO}_2}{\text{l}}$	0.044
K <sub>O<sub>2</sub></sub>	Half saturation of bacteria on oxygen	$\frac{\text{mg O}_2}{\text{l}}$	0.256
K <sub>p</sub>	Light abstraction by phytoplankton	$\frac{\text{m}^{-1}}{\text{mg/l}}$	0.013
K <sub>l<sub>coeffCO<sub>2</sub></sub></sub>	Oxygen transfer rate (piston velocity)	$\frac{\text{m}}{\text{day}}$	0.256
K <sub>l<sub>O<sub>2</sub></sub></sub>	Carbon Dioxide transfer rate (piston velocity)	$\frac{\text{m}}{\text{day}}$	0.17 (@ 20 °C)
K <sub>s</sub>	Half saturation of bacteria on substrate	$\frac{\text{mg CBOD}_5}{\text{l}}$	80
Our <sub>b</sub>	Basal oxygen utilization rate of bacteria	$\frac{\text{mg O}_2}{\text{mg bac}} \text{day}^{-1}$	0.10 (@ 20 °C)
Our <sub>m</sub>	Metabolic oxygen utilization rate of bacteria	$\frac{\text{mg O}_2}{\text{mg bac}} \text{day}^{-1}$	0.55 (@ 20 °C)
U <sub>max<sub>a</sub></sub>	Max growth rate phytoplankton	day <sup>-1</sup>	0.75 (@ 15 °C) 0.32 (@ 5 °C)
U <sub>max<sub>b</sub></sub>	Max growth rate bacteria	day <sup>-1</sup>	5
Y <sub>ca</sub>	Yield factor of phytoplankton produced for CO <sub>2</sub> consumed	$\frac{\text{mg CO}_2}{\text{mg Phytoplankton}}$	2.18
Y <sub>caOY<sub>oa</sub></sub>	Carbon dioxide/ oxygen produced	mg CO <sub>2</sub> / mg O <sub>2</sub>	1.30
BGS	Bacterial Growth Self Suppression	l/mg	0.01
AGS	Phytoplankton growth Self suppression	Unitless	0.1
ED	Equal Distribution Factor	Unitless	0.45

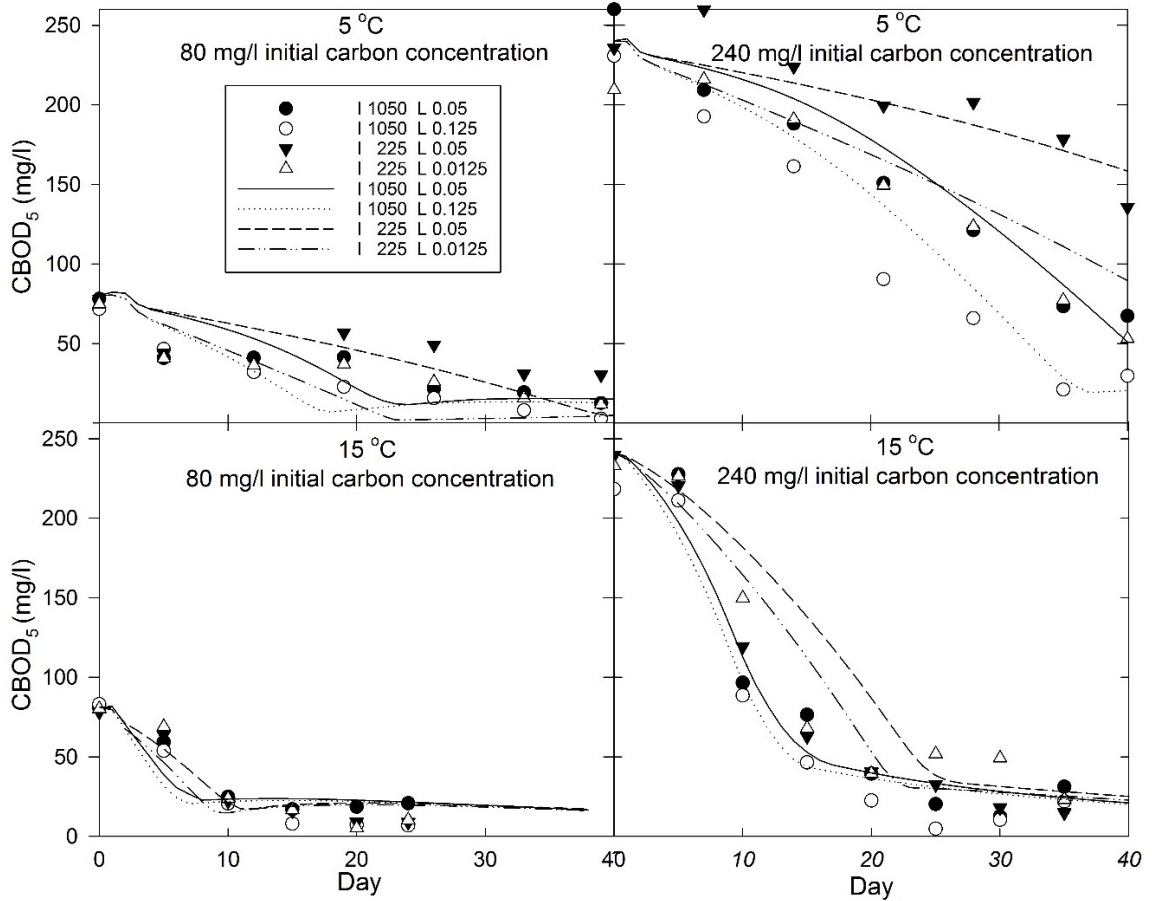


Figure 5.2. Model performance for  $CBOD_5$  concentration prediction (lines) compared to experimental results from Ragush et al. (2015b) (symbols).  $I$  is the modeled and experimental irradiance ( $\mu\text{e}/\text{m}^2/\text{s}$ ) and  $L$  is the volumetric loading rate ( $\text{m}^3/\text{day}$ ).

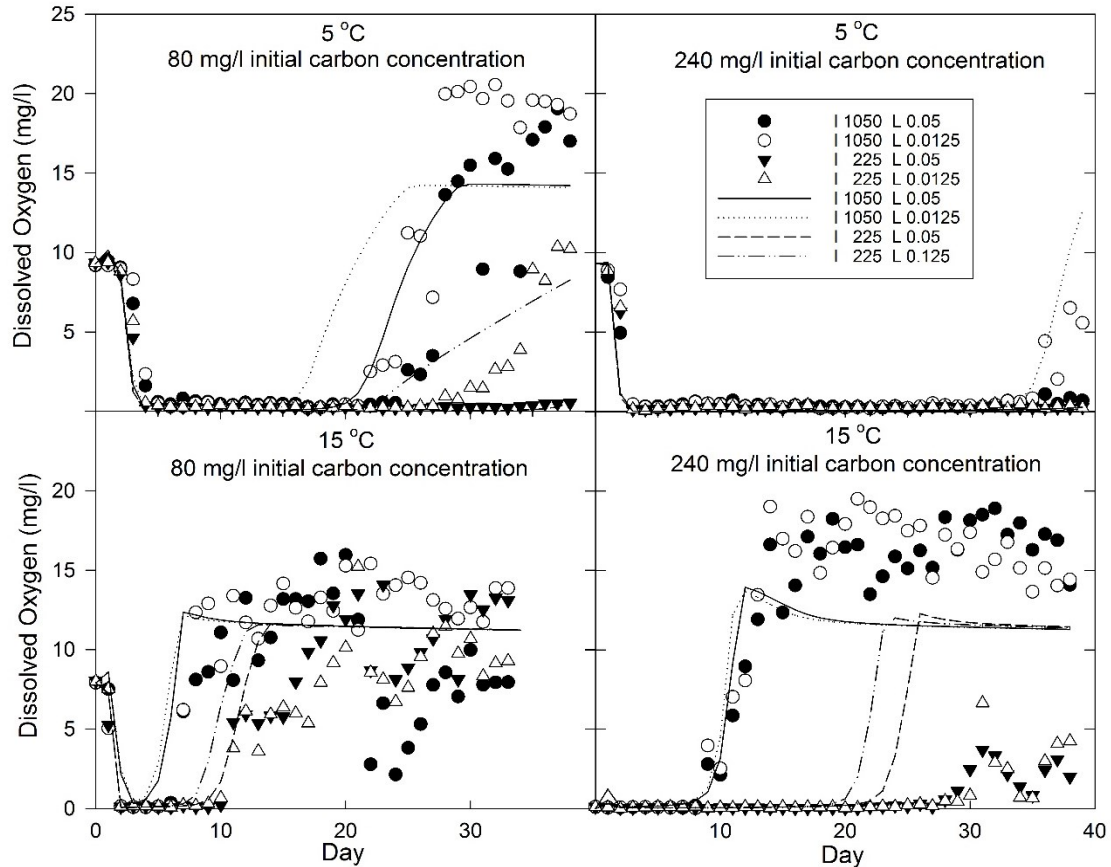


Figure 5.3. Model performance for dissolved oxygen concentration prediction (lines) compared to experimental results from Ragush et al. (2015b) (symbols). I is the modeled and experimental irradiance ( $\mu\text{e}/\text{m}^2/\text{s}$ ) and L is the volumetric loading rate ( $\text{m}^3/\text{day}$ ).

Of particular note is that the model was able to capture the influence of organic loading rates, and initial carbon concentrations on dissolved oxygen and  $\text{CBOD}_5$  concentrations. These are two key parameters that WSP designers are able to control. These theoretical findings suggest arctic WSPs can obtain an effluent concentration for  $\text{CBOD}_5$  that meets secondary standards (25 mg/l) with lowered areal loading rates, and more importantly lowered carbon concentrations at the onset of the summer treatment season.

Good agreement between model and experimental results was qualitatively observed, however, several inconsistencies provide insight into areas that are not well represented by the model and require further research. Generally an under prediction of the maximum dissolved oxygen (as measured at the surface) occurred, and the prediction of when a measurable oxygen concentration occurred earlier than experimentally found. The under prediction of the maximum oxygen may be attributed to the expected gradient

of oxygen over the photic zone. The model's predicted values are average concentrations across the photic zone whereas the measured values were taken at a location which was likely the oxygen maximum; further instrumentation throughout the photic zone would be necessary to better assess the model. The early prediction of oxygen concentration, especially under low light conditions, may be attributed to the model not considering diffusion of oxygen from the photic zone to deeper into the water column, and suggests that model performance could likely be improved by expanding it to a two box model, where a deep (non-photoc) zone is included.

The model only considers aerobic metabolism of bacteria for the removal of CBOD<sub>5</sub>, and due to the good agreement with experimental results this appears to be a reasonable simplification. However, when anaerobic conditions prevail, especially under low light conditions with minimal oxygen production by photosynthesis, the model under predicts the treatment performance. The incorporation of anaerobic processes is likely to improve the robustness and prediction performance under low light condition and cold conditions.

### **5.3.2 Sensitivity Analysis**

A one-factor-at-a-time (OFAT) method, a local sensitivity analysis method, was performed post calibration. An OFAT does not assess interaction, and results of the OFAT may be impacted by the values of other parameters set during the calibration. The sensitivity analysis was carried out on the 20 parameters in Table 5.1. The parameter range tested was chosen based upon values reported in the literature, listed in Table 5.4. In the OFAT, parameters were set to the calibrated value (Table 5.3) and one parameter at a time was varied over 5 equally-spaced levels, that ranged between the high and low values reported in the literature when available (Table 5.4) or else a range of (+/- 25%).

Table 5.4 Parameter values from literature

Parameter	Definition	Units	Reported Values	Sources
I <sub>halfsat</sub>	Irradiance half saturation of phytoplankton		60	Dauta et al. (1990)
			34	Moreno-Grau et al. (1996)
			220	Beran & Kargi (2005)
K <sub>ad</sub>	Phytoplankton death rate		0.05	Lawrence & McCarty (1970)
			0.001	Moreno-Grau et al. (1996)
			0.05-0.25	Schnoor (1996)
K <sub>as</sub>	Phytoplankton settling/respiration		0.2 m/d	Schnoor (1996)
			0.05	Moreno-Grau et al. (1996)
K <sub>bd</sub>	Bacteria death rate		0.035	Moreno-Grau et al. (1996)
			0.1	Buhr & Miller (1983)
			0.06	Beran (2005)
			0.06-0.015	Metcalf & Eddy (2003)
K <sub>bs</sub>	Bacteria respiration/settling rate		0.085 (+/- 25%)	Moreno-grau (1996)
K <sub>CO2</sub>	Half saturation of phytoplankton on carbon dioxide		0.044 (+/- 25%)	Buhr & Miller (1983)
K <sub>O2</sub>	Half saturation of bacteria on oxygen		0.256	Buhr & Miller 1983
			0.128	Banks et al. (2003)
			1	Metcalf & Eddy (2003)
K <sub>p</sub>	Light abstraction by phytoplankton		0.138 – 0.0249	Lorenzen (1972) Li (2009)

$K_s$	Half saturation of bacteria on substrate		25-100 (60) 150	Metcalf & Eddy (2003) Lawrence & McCarty (1970)
$Kl_{\text{coeffco}_2}$	Carbon Dioxide transfer rate (piston velocity)		0.893 1	Boogerd et al. (1989) Schnoor (1996)
$Kl_{\text{O}_2}$	Oxygen transfer rate (piston velocity)		0.15 0.189 0.24	Schnoor Chu & Jirka (2003) Deacon (1975)
$OUR_b$	Basal oxygen utilization rate of bacteria		0.15 (+/- 25%)	Henze (1978)
$OUR_m$	Metabolic oxygen utilization rate of bacteria		0.85 (+/- 25%)	Henze (1978)
$U_{\text{max}_a}$	Max growth rate phytoplankton		0.3 (5 °C) 0.7 (15 °C) 0.5 0.48 (5 °C) 0.78 (15 °C) 1.13 (@20 °C) 1.5 (@20 °C)	Dauta et al. (1990) Moreno-Grau et al. (1996) Buhr & Miller (1983) Banks (2003) Schoor (1996)
$U_{\text{max}_b}$	Max growth rate bacteria		4.95 5.0 2-10	Banks (2003) Moreno-Grau et al. (1996) Metcalf & Eddy (2003)
$Y_{ca}$	Yield factor of phytoplankton produced for CO <sub>2</sub> consumed		2.18 1.83 1.82	Fogg (1953) Cramer & Myers (1948) McKinney (2004)
$Y_{CaO}Y_{oa}$	Carbon dioxide used/ oxygen produced/ phytoplankton production		1.25 – 1.37	Fogg (1953) Cramer & Myers (1948) McKinney (2004)
BGS	Bacterial Growth Self Suppression	mg/l <sup>-1</sup>	0.002– 0.05	Estimated

AGS	Phytoplankton growth Self suppression	Unitless	0.02 – 0.5	Estimated
ED	Equal Distribution	Unitless	0.25 – 0.6	Estimated

Sensitivity coefficients (SC) were developed for six responses. The six responses included in the sensitivity analysis were chosen to be consistent with the problem being probed (predicting the timing when dissolved oxygen first exceeds 2 mg/l, and the timing of CBOD<sub>5</sub> concentrations being reduced to 30 mg/l) and four responses related to biological state variables (the max bacteria population, timing of max bacteria population, max phytoplankton population, and timing of max phytoplankton population). The sensitivity coefficient provides a non-dimensional measure of relative influence of a parameter to the relative change in the response (Downing et al. 1985). The sensitivity coefficients were calculated according to Equation 5.5, and evaluated over a range of five parameter values (the origin and two higher and lower) to determine an average SC over the parameter range (equation 5.6). The SC was taken to be the average to smooth out non-linearities within the relationship. Sensitivity analysis was performed at both lighting and temperature conditions at an initial carbon concentration of 240 mg/l and 0.0125 l/d loading rate to examine if the sensitivity of the parameters varied with environmental conditions. Displayed in Tables 5.5 and 5.6 are the sensitivity coefficients for the two responses (of the six) of greatest interest; timing of dissolved oxygen concentration exceeding 2 mg/l and timing of CBOD<sub>5</sub> concentration below 30 mg/l. Insights from Table 5.5 and 5.6 will be discussed further in this section.

$$SC(P)_i = \left| \frac{dR}{dP_i} \right|_{P_0} = \frac{\frac{R_i - R_0}{R_0}}{\frac{P_i - P_0}{P_0}} \quad (5.5)$$

Where: R = response vector, P = parameter vector, SC(P) = Sensitivity Coefficient of parameter p, and O = origin of parameter value (middle value of range tested)

$$C(P) = \frac{\sum_i^n SC(P)_i}{n} \quad (5.6)$$

Table 5.5. Parameter sensitivity coefficient for timing of dissolved oxygen concentration exceeding 2 mg/l

<b>Temperature (°C)</b>	<b>5</b>	<b>5</b>	<b>5</b>	<b>15</b>
<b>Light (ue/m<sup>2</sup>/s)</b>	<b>250</b>	<b>250</b>	<b>1000</b>	<b>1000</b>
Ihalfsat	0.41	0.47	0.49	0.67
Kad	0.54	0.38	0.77	0.45
Kas	0.11	0.14	0.16	0.13
Kbs	0.03	0.01	0.02	0.01
Kbd	0.70	0.17	0.31	0.03
Ks	0.02	0.10	0.06	0.17
Ko2	0.00	0.04	0.02	0.09
Kc	0.00	0.01	0.00	0.01
Kp	0.03	0.10	0.02	0.04
Klcoeffco2	0.00	0.00	0.00	0.00
Klo2	0.73	0.26	0.47	0.14
OUR <sub>b</sub>	0.01	0.08	0.04	0.15
OUR <sub>m</sub>	0.03	0.05	0.01	0.19
Umaxa	0.98	1.46	1.32	1.74
Umaxb	0.03	0.01	0.02	0.01
Yca	0.55	0.66	0.65	0.61
YcbObo	0.00	0.00	0.00	0.00
YcaOYoa	0.56	0.68	0.66	0.68
BGS	0.82	1.50	1.45	2.60
AGS	0.34	0.33	0.40	0.21
ED	0.15	0.57	0.37	0.74



Table 5.6. Parameter sensitivity coefficients for timing of CBOD<sub>5</sub> concentration below 25 mg/l

	Temperature (°C)		Light (ue/m <sup>2</sup> /s)	
	5	15	5	15
	250	250	1000	1000
Ihalfsat	0.29	0.86	0.46	0.47
Kad	0.48	0.31	0.54	0.25
Kas	0.10	0.09	0.11	0.04
Kbs	0.02	0.00	0.01	0.01
Kbd	0.73	0.26	0.36	0.20
Ks	0.00	0.03	0.01	0.15
Ko2	0.01	0.02	0.01	0.03
Kc	0.00	0.01	0.00	0.01
Kp	0.03	0.06	0.02	0.01
Klcoeffco2	0.00	0.00	0.00	0.00
Klo2	0.71	0.26	0.47	0.13
ourb	0.13	0.03	0.05	0.10
ourm	0.04	0.02	0.03	0.11
Umaxa	0.88	1.10	1.18	0.85
Umaxb	0.03	0.00	0.01	0.02
Yca	0.50	0.57	0.60	0.41
YcbObo	0.00	0.00	0.00	0.00
YcaOYoa	0.51	0.58	0.59	0.43
BGS	0.23	0.68	0.03	0.91
AGS	0.29	0.28	0.41	0.34
ED	0.12	0.39	0.27	0.32

Utilizing the sensitivity results from all 6 responses a cumulative report was constructed to provide a qualitative assessment of parameter sensitivity across the range of temperature and irradiance conditions, and provides a relative sensitivity of the parameters. Table 5.7 displays the number of sensitivity coefficients of the 6 responses exceeding 0.1 (a value that was arbitrarily assigned as being an indicator of a sensitive parameter) for a parameter under noted temperature and irradiance conditions. Table 5.7 also provides a parameters cumulative exceedances of sensitivity coefficients > 0.1 over the range of temperature and irradiance (right column total) and exceedances under each temperature/irradiance pairing (row at the bottom of the table). Finally, Table 5.7 provides an importance ranking that blends the SC rankings of the 6 responses into one to provide a qualitative assessment of parameter sensitivity.

Table 5.7 Cumulative sensitivity index organized by parameter and temperature/irradiance conditions. Value denotes number of SI indices greater than 0.1 for 6 tested categories.

	Temperature (°C)		Irradiance (ue/m <sup>2</sup> /s)		Total	Importance Ranking
	5	15	5	15		
Ihalfsat	4	6	5	5	20	2
Kad	4	5	6	5	20	5
Kas	5	3	5	2	15	12
Kbs	2	0	0	0	2	15
Kbd	5	5	5	3	2	9
Ks	0	1	0	3	4	14
KO2	0	0	0	0	0	18
Kc	0	0	0	0	0	20
Kp	0	1	0	0	1	16
KlcoeffCO2	0	0	0	1	1	19
KIO2	4	5	5	4	18	10
ourb	3	3	4	3	13	11
our <sub>m</sub>	1	0	0	4	5	13
Umaxa	6	5	5	5	21	1
Umaxb	0	0	0	0	0	17
Yca	6	5	5	5	21	4
YcbObo	0	0	0	0	0	21
YcaOYoa	5	5	5	5	20	6
BGS	5	5	5	6	21	3
AGS	6	5	5	4	20	8
EDFactor	4	5	5	5	19	7
<b>Total</b>	60	59	60	60		

The cumulative sensitivity report suggests that parameter sensitivity was not strongly influenced by the selection of irradiance and temperature conditions, as the model was consistently sensitive to the same parameters under all tested environmental conditions. However, the sensitivity coefficients for certain parameters can vary greatly with changing environmental conditions, as is the case for oxygen aeration rate (Kl<sub>O2</sub>) and bacterial growth self-suppression (BGS) (Tables 5.5 and 5.6). The results of the sensitivity analysis suggest that model outputs are more sensitive at lower temperature.

The sensitivity analysis highlights the model's sensitivity to phytoplankton growth parameters, as six of the top seven cumulatively ranked parameters are directly linked to phytoplankton growth rate or metabolism. BGS was the only parameter not related to phytoplankton ranked in the top seven. The model was also found to be more sensitive to  $K_{L_{O_2}}$ , the aeration rate, at low temperature, while being more sensitive to BGS, the bacteria carrying capacity, at high temperature.

Critical assessment of the sensitivity analysis provides insight into model dynamics and limiting processes under different conditions. The increased sensitivity to parameters at lower temperature suggests that arctic WSPs are more unstable and sensitive to operational and environmental changes than temperate climate WSPs. The sensitivity of the model to phytoplankton growth parameters agrees with research that has shown the importance of phytoplankton in the removal of  $CBOD_5$  and establishing facultative conditions in WSPs (Shilton, 2005). The increase in sensitivity coefficients of  $K_{L_{O_2}}$  at low temperature, suggests that atmospheric aeration processes are more important at lower temperatures, because phytoplankton populations grow more slowly and provide less oxygen to the system. The increased sensitivity to BGS at the higher temperature suggests that bacteria carrying capacity, for which the mechanism(s) are not well understood, is a major controller of dynamics in WSPs when the bacteria are not limited by oxygen or nutrients.

The sensitivity analysis highlights the importance of parameters related to biological processes, such as phytoplankton and bacterial growth, death and metabolism as the model was found to be highly sensitive to many of the related parameters. Calibration and comparison of outputs to additional bench and field scale datasets would reduce uncertainty in the selection of these parameters. Phytoplankton growth in the model is strongly dependent on the integrated light function, and improved characterization of the mathematical representation of vertically integrated growth would improve the confidence of predictions.

## 5.4 Simulation of Reduced Carbon Loading

A scenario where organic carbon loading and initial carbon concentration were reduced to two thirds, 160 mg/l and 10 kg/ha/d respectively was simulated to further assess treatment capacity of an arctic WSP. In this simulation the low temperature (5 °C) and low light condition (225 ue/m<sup>2</sup>/s) were assumed, as this was considered the worst case scenario. The model was initialized with concentrations of 50 mg/l bacteria and 1 mg/l phytoplankton. The simulation results suggest that at these reduced loading conditions, an Arctic WSP can be expected to achieve a CBOD<sub>5</sub> concentration of 25 mg/l at the end of the treatment season. However, the WSP is expected to be primarily anaerobic. Figure 5.4 illustrates that CBOD<sub>5</sub> treatment rates increase as the simulation progresses, and this increase is attributed to the increased oxygen availability as a result of increasing phytoplankton population. The simulation results highlight again that slow phytoplankton growth rates at 5 °C inhibits the development of facultative conditions and CBOD<sub>5</sub> treatment performance.

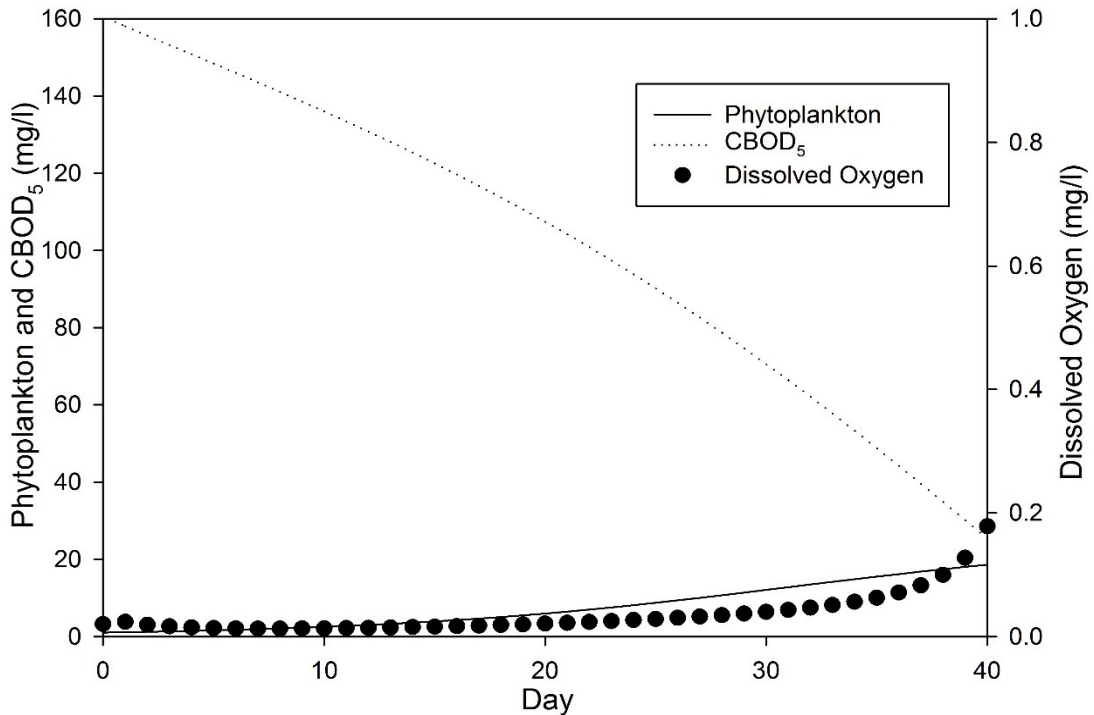


Figure 5.4 Simulation of arctic WSP dynamics at low temperature (5°C) and irradiance (225 ue/m<sup>2</sup>/s), 160 mg/l initial CBOD<sub>5</sub> concentration, and 10 kg CBOD<sub>5</sub>/ha/d.

## 5.5 Conclusion

The box model of the photic zone of WSPs operating under arctic conditions showed good agreement with experimental results with respect to predicting concentrations of dissolved oxygen and CBOD<sub>5</sub>. The model suggests that the phytoplankton growth rates, and consequentially the population, is severely limited at 5 °C and that this is largely responsible for observed differences in WSP treatment performance in the temperature range of 5 – 15°C. CBOD<sub>5</sub> removal and bacteria growth were observed to be controlled by dissolved oxygen concentrations, and the model was found to be highly sensitive to parameters that increased the input of oxygen to the photic zone, particularly phytoplankton growth rates and aeration. When dissolved oxygen concentrations increased and no longer controlled the dynamics, the bacterial carrying capacity was found to be a sensitive parameter, and suggests that bacterial metabolism becomes the dominant influence. The sensitivity analysis revealed that the model is highly sensitive to representation of phytoplankton growth and population in the photic zone, and highlights the need for more research into: i) the modeling and parameterization of phytoplankton growth under different limiting conditions (nutrient, light, density), and ii) depth-integration representations of phytoplankton growth and populations.

## Chapter 6 Conclusions

### 6.1 Summary and Conclusions

Monitoring of four municipal arctic WSPs operating in the Territory of Nunavut demonstrated that current design guidelines for single cell arctic WSPs do not lead to effluent quality characteristic of secondary wastewater treatment. The field study confirmed that the arctic climate poses a challenge for CBOD<sub>5</sub> treatment, and the bench-scale experiments and process-based modeling indicate that it is likely a result of slow growth rates and metabolism of bacteria and phytoplankton. CBOD<sub>5</sub> treatment was also hindered by the near-anaerobic conditions found to exist in the studied WSPs. Further, monitoring of systems highlighted that effluent quality from a WSP decreases with the age of WSPs. This was particularly evident in the case of Clyde River's primary cell which had poor performance, and the poor performance is attributed to the advanced operational age without the removal of the settled sludge from the bottom.

Results from the mesocosm experiments demonstrated the importance of temperature to WSP treatment performance. The establishment of a phytoplankton population appears highly sensitive between 5 °C and 15 °C, as at the lower temperature phytoplankton struggled to become established in the water column. This is of particular importance to arctic WSPs as this is their general operating temperature range. Additionally, it was observed that elevated initial carbon concentrations also resulted in decreased CBOD<sub>5</sub> removal rates, and suggests that heavily loaded WSPs create a toxicity towards the biological entities, phytoplankton or bacteria, and impedes treatment. It is theorized that the apparent toxicity is toward phytoplankton, and this decreased phytoplankton growth furthers the problem of persistent near-anaerobic conditions. Results suggest that the combination of low operational temperatures and high areal and concentration loadings in these arctic systems are problematic for the development of facultative oxygen conditions and treatment performance.

A process based model was developed and had good qualitative agreement with the experimental mesocosm results. The greatest novelty within the modeling work was the presentation and development of a function for depth integrated phytoplankton growth and populations in shallow, poorly mixed eutrophic waters. The approach was

explored because common ecological and WSP phytoplankton growth functions were performing poorly in modeling phytoplankton in arctic WSP systems. The methodology used to develop the function is firmly grounded on principles of ecology and population behaviour, and may find broader application than just WSP systems. Despite the good qualitative agreement of the model and experimentation, the model must undergo further scrutiny and testing on other applicable systems before confidence in the model would allow it to be used as an engineering design tool. This by no means detracts from the insight it has provided, or may provide in other applications.

The mesocosm WSP experiments, combined with process-based modeling, elucidated the relationships between operational (initial organic carbon concentration and OLR) and environmental (temperature and irradiance) variables and CBOD<sub>5</sub> removal and oxygen state of arctic WSPs. Results from the bench-scale experimentation highlight that water temperature, solar irradiance, organic loading rate, and initial carbon concentration all significantly impacted the CBOD<sub>5</sub> treatment performance and oxygen state. Higher temperature and solar irradiance increases treatment performance, while higher organic loading rates and initial carbon concentrations decrease treatment performance.

Modeling results highlighted that CBOD<sub>5</sub> removal in these arctic WSPs is likely oxygen limited. Consequentially, the CBOD<sub>5</sub> treatment performance of a WSP is highly sensitive to environmental and design changes in the arctic context because of the sensitivity of phytoplankton growth, which is heavily relied upon to provide the oxygen for the development of facultative conditions. The modeling results suggest that, if the goal of reaching secondary treatment is to be achieved consistently, design and operational practices need to robustly account for the potential of variable annual environmental conditions. Robust design and operation is required to ensure that sensitive biological processes occurring in WSPs continue to support biological treatment even during adverse climatic conditions.

Although monitoring, experimental, and modeling results are in agreement that arctic WSPs, as they are currently operated in Nunavut, will not meet secondary treatment standards, the results are encouraging that appropriately designed and organically loaded arctic WSPs are capable of meeting WSER standards. Model simulations and bench-scale experiments showed that WSPs with lower OLRs, and more

importantly lower initial carbon concentration, resulted in the development of aerobic conditions and CBOD<sub>5</sub> concentrations at the end of the treatment season less than 25 mg/l in the WSP.

## **6.2 Application of Research Findings to Arctic WSP Design**

The findings from this research indicate that design improvements should focus on: (i) maximizing the average temperature in the WSP, and (ii) decreasing the initial carbon concentrations at the onset of the summer treatment season to reduce CBOD<sub>5</sub> in the WSP effluent. Consideration should be made for supplementing dissolved oxygen levels through mechanical aeration to promote the reduction of CBOD<sub>5</sub> in cells with high carbon concentrations. Finally, design strategies should also focus on improving the transmittance of irradiance to greater depths in the WSP.

The water temperature of the WSPs was identified as the most important factor influencing the development of a facultative oxygen state and CBOD<sub>5</sub> removal. The significant improvement in CBOD<sub>5</sub> treatment and oxygen state in experimental WSPs operated at 15 °C compared to 5 °C (typical temperature range for arctic WSPs in the summer) demonstrates that efforts to maximize water temperature are expected to improve treatment results; although it must be recognized that temperature is an environmental constraint. An evaluation of the WSP's heat balance could be included within WSP design procedures to determine expected operational temperatures, as this will aid in selecting the appropriate OLR and predicting treatment performance of the WSP. Although temperature is an environmental constraint, a WSP designer can exert influence over the summer operating temperature. The most practical means of controlling WSP water temperature is through modifying the cell depth. Reducing the depth will increase the solar irradiance/volume ratio and generally can be expected to increase the average temperature of the WSP. Other potential methods that could be explored to increase the operating temperature of arctic WSPs include: (i) optimizing solar heating through orientation and siting, (ii) implementing solar collecting baffles or tubing, (iii) providing insulation to prevent conductive ground heat loss, and (iv) reducing heat loss by wind convection through siting and construction of windbreaks.



The results from the mesocosm WSPs also provide evidence that the continual loading of a WSP over the biologically dormant winter period presents a significant problem for wastewater treatment as arctic WSPs will contain high CBOD<sub>5</sub> concentrations at the start of the summer treatment season. Treatment performance could be improved if CBOD<sub>5</sub> concentrations within facultative/aerobic treatment cells are reduced at the onset of summer. In order to decrease the initial carbon concentration, a pre-treatment process will need to be added to the existing single cell systems. For example, a system comprised of two cells (or more) in series, with a deep primary cell and additional shallow secondary cell(s), is expected to provide better treatment compared to existing single cell systems. The deep primary cell would provide winter storage and pre-treatment. The primary cell is expected to reduce carbon concentrations of the wastewater before a secondary cell, conservatively by 50% (Shilton, 2005). The subsequent cells would act as the summer treatment cells and their performance is expected to be improved with the lower initial carbon concentration of the influent. Mesocosm experiments and modeling suggest that WSPs are likely to achieve wastewater effluent comparable to secondary treatment if the initial carbon concentration is reduced to 160 mg/l CBOD<sub>5</sub> and are modestly loaded over the summer treatment period. Once the carbon concentration is low enough in a cell (the level will depend on WSP operation and environmental conditions but our experiments would suggest < 80 mg/l CBOD<sub>5</sub> would be appropriate), phytoplankton growth is likely to improve leading to facultative conditions and will promote higher CBOD<sub>5</sub> removal rates.

Since CBOD<sub>5</sub> treatment is likely oxygen limited in arctic WSPs, mechanical aeration of the cells with high carbon concentration (deep primary cells or a cell after the deep primary cell) may be an effective tool to reduce the CBOD<sub>5</sub>. Not only will aeration likely reduce CBOD<sub>5</sub> in aerated cells, it will promote lower CBOD<sub>5</sub> concentrations in latter cells facilitating phytoplankton populations that can maintain facultative conditions. Alternatively, methods of pre-treatment prior to the primary cell of the WSPs, such as mechanical presses/filters and small sedimentation basins may also be economical solutions to reducing the initial CBOD<sub>5</sub>. However, a mechanical solution would require careful consideration of socioeconomic factors as mechanical solutions may have inherent operational challenges due to the remoteness and extreme climate of the Arctic.

The experimental results, along with findings from the process-based modeling, identified that the high light attenuation properties of the wastewater generated in arctic communities creates a design challenge because there is an apparent limited depth range, or photic zone, where phytoplankton were seen to thrive. This limited depth range was visually apparent in the mesoscale experiments, and theoretical modeling indicated that phytoplankton were likely to become density limited as growth was constrained to a small depth range. If arctic WSPs are intended to operate as facultative, the depth of cells utilized for facultative treatment may need to be decreased to provide sufficient irradiance per a unit volume, promote phytoplankton growth, and provide sufficient biological aeration throughout the depth. This design strategy aligns with decreasing the depth of the WSP to improve the average temperature. Pre-treatment steps to reduce light attenuating substances such as particulates and color would also improve light penetration.

On a cautionary note, despite the fact that higher irradiance conditions resulted in greater phytoplankton growth and higher CBOD<sub>5</sub> first order removal rates in mesocosm experiments, lower irradiance conditions were found to produce lower final CBOD<sub>5</sub>, TSS, and un-ionized ammonia concentrations. Higher irradiance is associated with higher final CBOD<sub>5</sub> concentrations because a larger phytoplankton community is supported, which demands more oxygen itself when it degrades. Phytoplankton itself is a suspended particulate, and will contribute to the measurement of TSS. Finally, high concentration of phytoplankton may increase un-ionized ammonia because phytoplankton removes CO<sub>2</sub> in the water, and causes an increase of the pH. In order to meet stringent effluent quality criteria for CBOD<sub>5</sub>, TSS, and un-ionized ammonia from highly productive WSPs a final polishing step such as a rock or sand filter may be require. The rock or sand filter would remove the phytoplankton, reducing the associated CBOD<sub>5</sub> and TSS concentration, as well as provide mechanical aeration decreasing the pH, and also stripping some un-ionized ammonia in the process.

### **6.3 Contribution to the Advancement of Knowledge**

This study includes results of a field monitoring program that documented the CBOD<sub>5</sub>, TSS and un-ionized ammonia treatment performance of four single cell WSPs operating in arctic climates. The monitoring program occurred over four years and covered the diversity of design and size of systems across Nunavut. This study produced the most comprehensive treatment performance dataset for arctic WSPs currently available.

This study provided insight into the process dynamics of CBOD<sub>5</sub> treatment in single cell arctic WSPs and lays the foundation for potential design and operational improvements. Although the results from this study are most relevant to arctic WSPs, novel approaches in the experimental and modeling methods, as well as some of the larger revelations with respect to CBOD<sub>5</sub> treatment in WSPs, have broad applicability for the WSP community.

Bench-scale mesocosms of single-cell WSPs were created for the assessment of how temperature, irradiance, organic loading and initial organic concentration impact dissolved oxygen concentration and CBOD<sub>5</sub> treatment. To the best of the author's knowledge, there are no comparable bench-scale studies of single-cell systems. The experiment provides a methodology for future bench-scale research involving WSPs, specifically the utilization of synthetic wastewater and LED light systems. The process of seeding the mesocosms with bacterial and phytoplankton communities from arctic WSPs is also novel. The experimental results provided strong evidence that high initial organic carbon concentrations reduces treatment performance. Additionally, although the impact of irradiance and temperature on treatment performance of WSPs in temperate regions is well documented, this is the first study to present the magnitude of their impact in the temperature range of 5 °C and 15 °C. The experiment identified the challenges and limitations of WSP operation in arctic climates, specifically the slow phytoplankton growth rates. The experiments further identified both the importance of the environmental conditions (temperature, and irradiance) as well as operational conditions (specifically initial carbon concentration). The experiments provided evidence, supporting field monitoring results, that systems currently receive areal carbon loading

rates that are too high, but highlights that arctic WSPs can be an effective CBOD<sub>5</sub> treatment solution if designed and operated under certain constraints.

The process based model that was developed within this research also contained several novel contributions. Of particular note is the new approach for simulating the growth dynamics of a depth integrated phytoplankton population in a poorly mixed environment. The sensitivity analysis that was performed provided new insight into the sensitivity of WSP biological treatment processes to temperature. The model was used to identify, for the first time, organic loading rates for arctic WSP systems that are likely to facilitate facultative treatment environments.

Finally, the largest impact of this work was a framework that coupled process-based modeling, bench scale experimentation, and field monitoring to examine the optimization of WSPs treatment performance. For example, a process-model can be utilized to assess the effectiveness system improvements such as utilizing multi-cell system or implementing aeration. Effective improvement strategies identified by the process-model can then be admitted to a bench-scale experimentation program for further testing. Finally, strategies found to improve treatment in the bench-scale experiments then can be presented the opportunity for implementation in the field or in pilot studies. This approach of utilizing all three components to inform WSP design and optimization is expected to be cost efficient. Construction, and failure, of WSPs is expensive particularly in remote locations), and utilizing all opportunities and methods to fully assess design and operation of WSPs prior to system implementation will help ensure they are successful at achieving desired wastewater treatment standards.

## 6.4 Avenues of Future Research

This work did not directly address the assessment of design improvements of arctic WSPs, and there is a great potential for this research. Some areas requiring assessment are:

- Impact of operational depth
- Multi-cell systems
- Pre-treatment technologies
- Polishing technologies

The bench scale experiments assessed only two irradiance conditions (225 and 1050  $\mu\text{e}/\text{m}^2/\text{s}$ ) and two temperatures (5 °C and 15 °C), did not possess a temperature gradient with depth, and samples were only collected in the photic zone. Additionally, the temperatures and irradiance in the experiment were constant. Benefits would be derived from research that:

- Fills in the gaps in the irradiance and temperature range to create a more complete picture of the relationship between environmental variables and the dissolved oxygen state and CBOD<sub>5</sub> treatment performance.
- Assesses the impact of stratification dynamics on treatment performance.
- Includes dynamic light and temperature regimes, representative of the Arctic, to validate the utilization of mesocosms for assessing design alternatives.
- Examine the water quality variability over depth by sampling at multiple depths.
- Determining if the CO<sub>2</sub> limitation of phytoplankton growth is expected to negatively impact CBOD<sub>5</sub> removal.

The process-based model had good agreement with the results of the experiments. However, through the development and analysis of the model research, avenues were identified that would lead to advancements in the understanding of WSP process and dynamics as well as modeling practices. These avenues include:

- Research into the parameterization of phytoplankton and bacteria metabolisms
- Further assessment of the developed depth integrated phytoplankton growth function
- A study on how increased photosensitivity of phytoplankton at lower temperature can be mathematically represented; specifically within a depth integrated phytoplankton growth function.

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## Appendix A Water Quality in the WSPs during the Summer Treatment Season

Location	Sampling period	Sampling year	# of samples <sup>1</sup>	CBOD <sub>5</sub> (mg/l)	TSS (mg/l)	Total Ammonia Nitrogen (mg/l-N)	Un-ionized ammonia (mg/l-N)
Pond inlet	Start	2012-2014	22	217 ± 35	46 ± 11	96 ± 11	0.40 ± 0.09
	Middle	2011 <sup>2</sup>	5		253	70.6	7.4
		2012-2014	24	156 ± 12	59 ± 12	94 ± 12	0.77 ± 0.16
	End	2011	4	169	179	32	0.07
2012-2014		21	109 ± 7	71 ± 16	72 ± 9	0.40 ± 0.08	
Clyde River (Primary)	Start	2012-2014	8	255 ± 41	54 ± 20	113 ± 23	0.15 ± 0.05
	Middle	2013	3	281 ± 27	65 ± 14	89 ± 13	0.14 ± 0.03
	End	2012-2014	10	239 ± 24	58 ± 12	97 ± 8	0.25 ± 0.11
Clyde River (Secondary)	Start	2012-2014	8	119 ± 23	31 ± 10	57 ± 5	0.14 ± 0.03
	Middle	2013	4	93 ± 26	28 ± 5	45 ± 5	0.20 ± 0.11
	End	2012-2014	10	82 ± 24	30 ± 7	73 ± 8	0.21 ± 0.09
Kugaaruk	Start	2012-2013	8	133 ± 13	30 ± 9	56 ± 8	0.12 ± 0.03
	End	2012-2013	12	150 ± 8	25 ± 4	84 ± 5	0.44 ± 0.09
Grise Fiord	July	2011	4	94 ± 18	438 ± 226	5.5 ± 4.3	5.0 ± 4.1

95% confidence intervals provided

<sup>1</sup>Based on parameter with least amount of samples

<sup>2</sup>Provided for context – confidence intervals not provided because of large sampling variability and unique nature of 2011

## Appendix B Raw Wastewater Quality

<b>Location</b>	<b># of samples</b>	<b>CBOD<sub>5</sub> (mg/l)</b>	<b>TSS (mg/l)</b>	<b>Total Ammonia Nitrogen (mg/l - N)</b>	<b>Un-ionized ammonia (mg/l - N)</b>
Grise Fiord <sub>1</sub>	3	632	665	113	6.6
Pond Inlet	23	525 ± 89	326 ± 54	107 ± 13	1.56 ± 0.30
Clyde River	15	367 ± 67	273 ± 62	103 ± 16	0.64 ± 0.15
Kugaaruk	8	371 ± 50	272 ± 45	94 ± 11	2.26 ± 0.56

95% confidence intervals provided

<sup>1</sup>Insufficient number of samples to calculate a representative confidence interval

## Appendix C Chemical Composition of the Synthetic Wastewater.

Chemical	Chemical Formula	Raw water (mg/l)	Initialization Concentration (mg/l)	Initialization Concentration (1/3 organic strength) (mg/l)
Peptone		220	145	36
Casein		220	55	14
Starch	(C <sub>6</sub> H <sub>10</sub> O <sub>5</sub> ) <sub>n</sub>	160	35	9
Glucose	C <sub>6</sub> H <sub>12</sub> O <sub>6</sub>	170	105	26
Sodium Acetate	C <sub>2</sub> H <sub>3</sub> O <sub>2</sub> Na	270	180	45
Oleic acid		290	145	36
Humic Acid		20	20	20
Soluble Starch		60	15	4
Sodium Hydroxide (1 N)	NaOH	1 ml/l	1 ml/l	0.5 ml/l
Monopotassium phosphate	KH <sub>2</sub> PO <sub>4</sub>	80		30
Dipotassium phosphate	K <sub>2</sub> HPO <sub>4</sub>	20		30
Magnesium chloride	MgCl <sub>2</sub>	50		45
Ammonia chloride	NH <sub>4</sub> Cl <sub>4</sub>	400		300
Cupric sulphate	CuSO <sub>4</sub>	0.5		0.2
Ferrous sulphate	FeSO <sub>4</sub> · 7 H <sub>2</sub> O	6		3
Calcium chloride dihydrate	Ca Cl <sub>2</sub> · 2 H <sub>2</sub> O	55		38
Manganous chloride tetrahydrate	MnCl <sub>2</sub> · 4 H <sub>2</sub> O	0.4		0.3
Aluminum chloride hexahydrate	AlCl <sub>3</sub> · 6 H <sub>2</sub> O	22		1.7
Zinc sulphate	ZnSO <sub>4</sub>	0.8		0.2
Sodium Carbonate	Na <sub>2</sub> CO <sub>3</sub>	25		25
Cobalt chloride	CoCl <sub>2</sub> · 6 H <sub>2</sub> O	4.0 x 10 <sup>-3</sup>		1.6 x 10 <sup>-3</sup>
Sodium Molybdate Dihydrate	Na <sub>2</sub> MnO <sub>4</sub> 2 H <sub>2</sub> O	5.0 x 10 <sup>-3</sup>		3.6 x 10 <sup>-3</sup>
Diatomaceous Earth		63		5

## Appendix D Matlab Code of Process-Based Model

```
%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%% Parameters and Constants%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%
Ihalfsat = 30; % ue/m2/s
Kad = 0.05; % Algae death rate 1/day
Kas = 0.05; % Algae settling rate 1/days
Kbs = 0.0; % bacteria settling rate 1/d
Kbd = 0.025; % Respiration rate 1/days
Ks = 80; % mg BOD/l
Ko2 = 0.256; % mg O2/l
Kc = 0.044; % mg CO2/l
Kp = 1.3*10^-1; % light abstracted per concentration algae m^-1/mg/l
Klcoeffco2 = 1; %piston velocity ratio CO2/O2
KlO2 = 0.17; %piston velocity oxygen m2/s
OURb = 0.1; % basal oxygen consumption (mg O2 / mg bac day)
OURm = 0.55; % max oxygen consumption attributable to metabolism(mg O2 / mg bac day)
Umaxa = 0.32; % Max algal growth rate 0.75 day-1 15C and 0.32 day-1 5 c
Umaxb20 = 5; % max growth rate bacteria (days-1)
Yca = 2.18; % (mg CO2 consumed/ mg algae) growth related
YcaOYoa= 1.30; %mg co2/mg o2 consumed by algae
YcbOob = 1.375; % (mg CO2 produced/ mg bacteria)
BGS = 0.010; %Bacterial growth self suppression
AGS = 0.1; %Phytoplankton growth self suppression
EDFactor = 0.45; %multiplicative factor of Umaxa when evenly distributed
Ke = 14; % light attenuation coefficient(m-1)

%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%CHANGEABLE PARAMETERS%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%

Simlength = 40; %Days
Io =225; %Surface incident light (Ue/m2/s)
T =5; % Temperature (Celsius)
Tk=T+273.15;
Ai = 1; % Initial Alg0ae pop (mg/l)
Bi = 1; % Initial bacteria pop (mg/l)
Si =80; % Initial substrate (mg cbod5 / l)
ColumnZ = 1.25; % depth column (m)
Dl = 0.0125; % Daily Loading (l)
SA = 182; % Surface area (cm2)
CBOD5inf=550; % CBOD5 of Raw (mg/l)
SolCBOD5 = 0.5;
Raw= CBOD5inf*SolCBOD5;
%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%CONSTANTS%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%
Area = SA*1/10000; % area m2
V = ColumnZ*Area; % volume (m3)
Simstep = 1/10000; %Days
steps= Simlength/Simstep+1;

%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%PREALLOCATION%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%
A=zeros(steps,1);
BT=zeros(steps,1);
CO2T = zeros(steps,1);
O2T = zeros(steps,1);
```

```

UbT=zeros(steps,1);
dBT=zeros(steps,1);
dA=zeros(steps,1);
ST=zeros(steps,1);
dST=zeros(steps,1);
Ua=zeros(steps,1);
dO2TP=zeros(steps,1);
dO2TC=zeros(steps,1);
dO2TT=zeros(steps,1);
CBOD5=zeros(steps,1);

%((((((((((((((((((((((INITIALIZATION))))))))))))))))))))))))))))))))))%
A(1,:) = Ai; % Initial Algae pop (mg/l)
BT(1,:) = Bi; % Initial bacteria pop (mg/l)
BB(1,:) = Bi;
ST(1,:) = Si; % Initial substrate (mg cbod5 / l)
CBOD5(1,:)=ST(1,:)+Ai.*0.5+Bi.*0;
SB(1,:) = Si;
CO2T(1,:) = 380*10^-6*0.034 * exp(2400*(1/(T+273.15)-1/298.15))*44000;
%Initial carbon dioxide (mg/l)

%((((((((((((((((((Temperature dependent
variables))))))))))))))))))))))))))))))%

Umaxb = Umaxb20*1.024^(T-20);
OURm= OURm*1.024^(T-20); %metabolic rate adjusted for temperature
OURb = OURb*1.024^(T-20); %basal rate adjusted for temperature
Klo2 = Klo2*1.024^(T-20);
Klco2 = Klcoeffco2*Klo2;

%--- initial dissolved oxygen (henry's law temperature dependent)-----
O2T(1,:) = exp(-139.34411 + 1.575701*10^5/Tk - 6.642308*10^7/Tk^2 +
1.2438*10^10/Tk^3 -8.621949*10^11/Tk^4); % 9.7 @ 15C and 12.1 @ 5C
%----- gas sat of water -----
Cso2 = exp(-139.34411 + 1.575701*10^5/Tk - 6.642308*10^7/Tk^2 +
1.2438*10^10/Tk^3 -8.621949*10^11/Tk^4);
% saturation of O2 mg/l
Csco2 = 380*10^-6*0.034 * exp(2400*(1/(T+273.15)-1/298.15))*44000;
% saturation of CO2 mg/l

% depth 99% light attenuated
Z=log(0.01)/-(Ke);
Iavclear = Io*(1-exp(-Ke*Z))/(Ke*Z); %average of light over 99% light
attenuation depth with no phytoplankton

for t=1:(steps-1)

    Iav= Io * (1-exp(-(Ke+Kp.*A(t,:))*Z))./((Ke+Kp.*A(t,:))*Z); %Average
irradiance considering algae

```

```

    if O2T(t,:) > OURb.*BT(t, :)*Simstep

OUR=OURm.*(ST(t, :)/(Ks+ST(t, :))).*O2T(t, :)/(O2T(t, :)+Ko2)+OURb;
%Oxygen utilization rate metabolism MM based
    else
        OUR = O2T(t, :)/BT(t, :); % oxygen utilization rate under
anaerobic conditions
    end

%~~~~~ bacteria growth~~~~~

    UbT(t, :) =
Umaxb.*(ST(t, :)/(Ks+ST(t, :))).*O2T(t, :)/(Ko2+O2T(t, :)).*(1-
BGS.*(BT(t, :)));
    dBT(t, :) = (UbT(t, :).*BT(t, :)-Kbs.*BT(t, :)-Kbd.*BT(t, :))*Simstep;
    BT(t+1, :)=BT(t, :)+dBT(t, :);

%~~~~~Algae Growth~~~~~
    Fdis=(Umaxa-Umaxa.*EDFactor).*(1-exp(-4*exp(AGS.*
A(t, :))))+Umaxa.*EDFactor;
    Ua(t, :)=Fdis.*Iav./(Iav+Ihalfsat).*(CO2T(t, :)/(Kc+CO2T(t, :)));
    dA(t, :)=(Ua(t, :).*A(t, :)-Kad.*A(t, :)-Kas.*A(t, :))*Simstep; % growth
- settling - respiration
    A(t+1, :)=A(t, :)+dA(t, :);

%~~~~~ Substrate growth~~~~~
    dST(t, :) = (-
OUR.*BT(t, :)+Dl*Raw/(V*1000)*ColumnZ/Z+Kad.*A(t, :)*0.5+Kbd.*BT(t, :)*0.7)
*Simstep; %uptake by bacteria + input + cell death algae + death
bacteria
    ST(t+1, :)=ST(t, :)+dST(t, :);

    for i=1:5
        if ST(t+1,i)<0 %doesn't allow for substrate to go negative
            ST(t+1,i)=0;
        end
        CBOD5(t+1, :)=ST(t+1, :)+A(t+1, :).*0.5+BT(t+1, :).*0;
    end

%~~~~~Oxygen change~~~~~

    dO2TP(t, :) = Yca./YcaOYoa.*Ua(t, :).*A(t, :)* Simstep; % Production
by phytoplankton
    dO2TC(t, :) = -OUR.*BT(t, :)*Simstep; %consumption by bacteria
    dO2TT(t, :) = KIo2*Area/(Area*Z).*(Cso2-O2T(t, :))*Simstep; %Gas
transfer
    O2T(t+1, :)=O2T(t, :)+dO2TP(t, :)+dO2TC(t, :)+ dO2TT(t, :);

%~~~~~carbon dioxide change~~~~~

```

```
    dCO2PC = (YcbOob.*OUR.*BT(t,:)- Yca.*Ua(t,:).*A(t,:))*Simstep;  
%production by bacteria - consumption by phytoplankton  
    dCO2T = Klco2.*Area/(Area*Z).*(CSCO2-CO2T(t,:))*Simstep; %gas  
transfer  
    CO2T(t+1,:) = CO2T(t,:) + dCO2PC + dCO2T;
```

```
end
```

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