HYDROLOGIC AND MICROBIAL RISK CHARACTERIZATION OF AN ARCTIC WETLAND TREATMENT AREA

by

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CONTENTS

LIST OF TABLES

LIST OF FIGURES

ABSTRACT

This study focused on the hydrologic characterization and microbial risk associated with a passive wastewater treatment wetland in Qamani'tuaq (Baker Lake), Nunavut. Site-specific field data was utilized in conjunction with hydrologic-contaminant modelling to simulate *E. coli* concentrations throughout the system. The results were applied to a quantitative microbial risk assessment (QMRA), along with information obtained through community consultation, to characterize the potential microbial health risk for multiple exposure pathways and hydrologic scenarios. Several water quality parameters exceeded treatment criteria specified in the Hamlet's water license, while short hydraulic retention times were the primary reason for ineffective treatment performance during spring freshet. Simulated *E. coli* concentrations throughout the system indicated that dilution from external watershed contributions was playing a large role in reducing bacteria concentrations, as opposed to biological degradation or treatment. Overall, the assessment predicted risk levels greater than the recommended health target (10^{-4}) for four common exposure pathways.

LIST OF ABBREVIATIONS AND SYMBOLS USED

°C	Degree Celsius
AGI	Acute Gastro-intestinal Illness
BOD5	Five Day Biological Oxygen Demand
CAMRA	Center for Advancing Microbial Risk Assessment
CCME	Canadian Council of Ministers of the Environment
CEQG	Canadian Environmental Quality Guidelines
DALYs	Disability-adjusted Life Years
DEM	Digital Elevation Model
DO	Dissolved Oxygen
DSM	Digital Surface Model
ECCC	Environment and Climate Change Canada
E. coli	Escherichia coli
GPS	Global Positioning System
HEC-HMS	Hydrologic Engineering Center's Hydrologic Modelling System
HRDEM	High-Resolution Digital Elevation Model
mm, cm, m, km	Millimetres, Centimetres, Metres, Kilometres
MDL	Minimum Detection Limit
NSE	Nash-Sutcliffe Coefficient of Efficiency
NPS	National Performance Standards
NWB	Nunavut Water Board
ORP	Oxidation Reduction Potential
QMRA	Quantitative Microbial Risk Assessment
RMSE	Root Mean Square Error
R_s	Relative Sensitivity
RWU	Residential Water Use
SCS	Soil Conservation Service
SMA	Soil Moisture Accounting

TAN	Total Ammonia Nitrogen
TIS	Tanks-in-series
TKN	Total Kjeldah Nitrogen
TP	Total Phosphorus
TSS	Total Suspended Solids
UTM	Universal Transverse Mercator
VSS	Volatile Suspended Solids
WHO	World Health Organization
WSP	Wastewater Stabilization Pond
WSER	Wastewater Systems Effluent Regulations
WTA	Wetland Treatment Area

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CHAPTER 1. INTRODUCTION

1.1 Background

Population growth and increased development in Arctic Canada have led to growing concerns with respect to wastewater treatment and associated impacts on human health and the environment (Chouinard et al., 2014). Similar concerns have been previously expressed and resulted in the development of the Canada-wide Strategy for the Management of Municipal Wastewater Effluent, which recommended regulations specific to wastewater systems located in Canada's northern territories (CCME, 2009). The extreme climate and remoteness of the Canadian Arctic restricts wastewater management options, and as a result most communities in Nunavut use passive wastewater systems which are generally limited to primary treatment with reduced pathogen removal (Hayward et al., 2014; Huang et al., 2017). As these passive treatment areas tend to be situated in close proximity to communities and within areas commonly used for recreation and food harvesting activities, human health effects are increasingly important (Nilsson et al., 2013; Daley et al., 2019). Exposure to pathogenic microorganisms present in partially treated wastewater, downstream from ineffective treatment systems, has been identified by Northern residents as a significant concern (Harper et al. 2015).

In order to address concerns regarding the effective treatment of wastewater in the community of Qamani'tuaq (Baker Lake), Nunavut, Dalhousie University is working with the Hamlet and Agnico-Eagle Mines Limited to propose upgrades for the passive wastewater treatment system. In early 2010 Agnico-Eagle began operations of Meadowbank Mine, a gold mine located approximately 80 km to the north of the community. In order to provide access from the mine to the community, Agnico-Eagle maintains a gravel road across the tundra. As part of the agreements to develop the mine in close proximity to the community, Agnico-Eagle is supporting community and environmental initiatives aimed to benefit the growing community.

1.2 Study Objectives

The overall purpose of this study is to assess the performance of wastewater treatment in Qamani'tuaq, and to increase the understanding of associated impacts on human health in the community. In addition, the results of the study will provide valuable information on the requirements for future wastewater treatment system upgrades for the community. Specifically, the study objectives are as follows:

1. Characterize the hydrology and contaminant loading of a wetland utilized for municipal wastewater treatment in Qamani'tuaq, Nunavut.
2. Construct, calibrate, and validate a hydrologic-contaminant transport model representative of the wastewater treatment wetland and surrounding watershed.
3. Evaluate the human health risks associated with the treatment area using a Quantitate Microbial Risk Assessment (QMRA) that is informed by the hydrologic-contaminant transport model.

CHAPTER 2. LITERATURE REVIEW

2.1 Wastewater Treatment in Nunavut

The Canadian territory of Nunavut is made up of 25 remote communities, with populations ranging from 150 to 8,000 people, resulting in a total of approximately 38,780 full-time residents (Nunavut Bureau of Statistics, 2019). Each community is spatially isolated, with no roads connecting the communities to each other or to Southern Canada. Due to the remote nature of the communities in Nunavut, each requires its own public works infrastructure such as wastewater treatment facilities. Municipal and industrial wastewater treatment in this territory, and similar cold climate regions, can be problematic due to the remote location of, and the limited access to communities, in addition to the physical constraints introduced by the harsh environment (Hayward et al., 2015). Additionally, communities throughout Nunavut face a variety of operational, financial, and technical limitations when dealing with wastewater and other municipal services (Yates et al., 2012). Mechanical wastewater treatment plants have been reported to experience many challenges in their application in these regions and are not always considered a feasible option for relatively small communities (Johnson et al., 2014). High capital and maintenance costs, and the requirement for technical supervision and treatment optimization, introduce many obstacles with respect to small communities with limited budgets. As a result, most of the communities in Nunavut utilize passive wastewater treatment systems.

2.1.1 Passive Wastewater Treatment in Nunavut

Passive wastewater treatment is a common treatment solution to manage wastewater in many communities throughout Nunavut, as passive systems utilize natural processes to reduce contaminant concentrations present within municipal and industrial wastewater. Commonly, wastewater is transported from holding tanks located at residential and municipal dwellings to a designated discharge and treatment area. While most hamlets in Nunavut use a trucked system for wastewater collection, three communities (Resolute Bay Rankin Inlet, and Iqaluit) utilize a piped system similar to those of larger municipalities (Daley et al., 2018). Once the wastewater is transported to the treatment area, it is most commonly discharged into a wastewater stabilization pond (WSP), or an un-engineered lake lagoon (Johnson et al., 2014). WSPs and lagoons are the first step in the treatment path, and use natural processes such as sedimentation, microbial decomposition, and filtration (Daley et al., 2015).

2.1.1.1 Wastewater Stabilization Ponds and Retention Lagoons

In Nunavut, WSPs operate as retention lagoons in which no discharge occurs during the winter months, which usually extend from mid-to-late September until June depending on the location. During this time, effluent within the lagoon is frozen or in liquid state close to freezing temperatures, therefore biological degradation is limited (Huang et al., 2018). The treatment season starts once the water bodies thaw, usually beginning in June. The treatment season is characterized by higher biological activity, resulting from warmer air temperatures and extended daylight (Huang et al., 2018). WSPs and lagoons with functioning decant structures operate with a scheduled decant, generally outlined in a water license issued by the Nunavut Water Board (NWB). These systems commonly store wastewater throughout the treatment season, before discharging either directly into the aquatic receiving environment or to a natural tundra wetland for further treatment in the late summer months (Huang et al., 2018). Lagoons lacking decant structures, continuously discharge effluent to the downstream environments, such as wetland treatment areas (WTAs) or marine aquatic environments, beginning in spring freshet and continuing until freeze-up (Hayward et al., 2018; Yates et al., 2012).

The ability to avoid chemical flocculants and mechanical equipment within WSPs and lagoons make these systems a feasible option despite a limited capital and operational budget (Huang et al., 2018). They also require minimal operation and maintenance compared to more technically complex mechanical or chemical systems, and therefore tend to be more practical for remote communities in which trained personnel may be limited or in which energy costs are high (Daley et al., 2015). However, as WSPs and lagoons rely on natural environmental and biological processes, they may experience treatment limitations due to the harsh arctic climate and may not be able to achieve treatment goals set out by regulatory bodies (Huang et al., 2018). Additionally, some hamlets do not utilize a WSP or lagoon, and discharge untreated effluent directly to un-engineered tundra wetlands, natural ponds, or marine receiving environments (Hayward et al., 2014; Yates et al., 2012; Ragush et al., 2015).

2.1.1.2 Wetland Treatment Areas

In the Arctic, tundra WTAs are often located in naturally occurring depressions on the tundra under wet conditions. These systems have variable physiographic features which influence plant communities and water retention, further impacting the overall treatment performance of the system (Yates et al., 2012). Multiple studies have suggested that the influx of organic matter, nutrients and hydrological inputs from the recurrent dispersion of effluent onto the tundra have formed these wetlands (Hayward et. al., 2014; Chouinard et al., 2014). Treatment wetlands are commonly referred to as polishing wetlands, as they provide additional treatment of effluent following primary treatment within WSPs or lagoons. In most communities, WTAs ultimately discharge effluent into marine receiving environments (Hayward et al., 2014).

Previous studies of both arctic and sub-arctic receiving wetlands have demonstrated the ability of these systems to improve municipal wastewater quality; however, there is limited information on the hydrodynamics and pollutant removal rates occurring in these wetlands (Hayward et al., 2014; Chouinard et al., 2014; Yates et al., 2012). The remote location of many Nunavut communities makes it challenging to collect the data necessary to parameterize quantitative treatment performance models, such as coupled hydrologic and water quality datasets.

Nevertheless, previous studies do suggest that the regulation of WTA inflows through the control of wastewater effluent allows for optimal hydraulic retention times and therefore better treatment performance throughout a system. With this is mind, many attributes of northern tundra WTAs are a product of the natural environment; and as such their physical, hydrological, and biogeochemical characteristics display significant intrasystem variability (Hayward et al., 2014). For instance, overloading of WTAs have been specifically noted during spring freshet when external hydrologic contributions, such as watershed runoff from snowmelt, is extremely high. During these high flow periods, treatment throughout the wetland is reduced due to short hydraulic retention times with contaminant reductions becoming a factor of dilution from the external hydrologic contributions (Hayward et al., 2014). These unanticipated and/or uncontrolled fluctuations in natural environmental processes, required to passively treat wastewater, may result in the release of partially treated wastewater to the surrounding aquatic and terrestrial environments (Daley et al., 2019). While properly engineered passive treatment systems have been identified to be more suited to small, remote Arctic communities when compared to other design alternatives, literature suggests that research gaps still exist with respect to passive wastewater treatment and associated human health effects.

2.1.2 Human Health Impacts

Exposure to pathogenic microorganisms present in partially treated wastewater, downstream from ineffective passive or mechanical treatment systems, has been identified by Northern residents as a public health concern. The release of inadequately treated domestic wastewater to the surrounding environment is of concern as the effluent is a natural vector for a variety of disease-causing microbes (Yapo et al., 2014; Bitton, 2005). Contact with harmful effluent via unintentional direct contact, cross contamination of drinking water sources, or bioaccumulation of contaminants in the food chain presents a risk of exposure to many pathogenic agents such as pathogenic Escherichia coli (*E. coli*), Salmonella spp., Campylobacter spp., rotavirus, Giardia spp., and Cryptosporidium spp. (Daley et al., 2019). As these pathogenic agents are transmissible via fecal-oral routes, commonly with a very low infectious dose, frequent interaction between community members and the receiving environment can lead to acute gastrointestinal illness (AGI) and other human diseases after exposure to low concentrations (Leclerc et al., 2002).

Although estimating the disease burden associated with exposure to wastewater in remote arctic communities can be challenging, literature has reported higher waterborne- and sanitation-related illness in Northern Canada compared to the southernmost parts of the country (Harper et al. 2015; Thomas et al. 2013; Parkinson et al., 2014). Human exposure to potentially harmful effluent is believed to be higher in these areas due to traditional interactions with the natural environment through food harvesting and recreational activities (Harper et al., 2011; Daley et al., 2018). A study conducted on self-reported cases of AGI in Inuit communities estimated 2.9 to 3.9 annual cases per person (Harper et al., 2015; Thomas et al., 2013). This is a higher incidence range than the 0.6 annual cases estimated nationally, and the 0.8 to 1.3 annual cases per person estimated from developing countries (Mathers et al., 2002; WHO, 2006a; Daley et al., 2019).

Additionally, while engineering assessments have demonstrated the ability of passive wastewater systems to reduce the levels of *E. coli* present within effluent, these levels still typically exceed those achieved with conventional wastewater disinfection in more temperate regions (Hayward et al. 2014; Krumhansl et al. 2015; Ragush et al. 2015; Yates et al. 2012). As *E. coli* is commonly used as a regulatory indicator of other pathogenic organisms, these levels present increased public health risks with respect to the contraction of AGI and other related diseases. Of these assessments, few have exclusively detailed the possibility of human exposure through interactions with the natural environment in arctic communities (Daley et al., 2018). While there is limited site-specific data available to evaluate the potential health risks associated with the passive systems in Nunavut communities, Daley et al. (2018) developed a conceptual model, supported by literature, to be used as a guide for the microbial risk assessment of these scenarios. From a public health perspective, further investigation is required to determine whether these systems meet regulatory guidelines set out for the protection of environmental and human health, and more specifically if the existing guidelines are adequate with respect to the associated microbial risk effluent presents to Nunavut communities, through human exposure to infectious pathogens.

2.1.3 Wastewater Treatment Regulations

Currently, wastewater effluent objectives in Nunavut are regulated by the NWB. The NWB is responsible for the use, management, and regulation of inland water throughout Nunavut (with the exception of National Parks), and its primary role is to license the use of water and deposits of waste (NWB, 2021); inland fresh water including, lakes, rivers, streams, wetlands, and groundwater, fall under the regulatory jurisdiction of the NWB. Regulatory authority varies for marine areas, generally with joint sanction between the NWB, Nunavut

Marine Council, and other areas of the public government (NWB, 2021). In order to regulate water-use and waste disposal in Nunavut, the NWB issues water licenses specific to the applicant, which can vary from hamlets to industrial or mining operations. While water licenses are issued by the NWB, compliance monitoring and enforcement is the responsibility of Indigenous and Northern Affairs Canada (NWB, 2021).

Canada-wide, wastewater systems are managed under the Wastewater Systems Effluent Regulations (WSER). These regulations were developed under the Fisheries Act to fulfill a commitment of the Canadian Council of Ministers of the Environment (CCME) strategy for the establishment of national municipal wastewater effluent standards (Johnson et al., 2014; CCME, 2009). The standardized effluent targets outlined in the CCME WSER include criteria of 25 mg/L for total suspended solids (TSS), 25 mg/L for carbonaceous biological oxygen demand (CBOD5), and 1.25 mg/L for un-ionized ammonia nitrogen (NH3-N) for municipal systems producing greater than 100 m^3/d (CCME, 2009). While these regulations set national standards for the quality of effluent, a grace period was allotted to Nunavut, and similar northern regions, prior to having to comply with the regulations (CCME, 2009). This grace period was initiated due to the harsh climate and limited treatment season in which northern wastewater lagoons and systems are employed (Johnson et al, 2014; Ragush et al., 2015). Additionally, information regarding potential environmental and human health risks associated with wastewater systems in use in the territory of Nunavut remains limited (Daley et al., 2018).

Lastly, the Canadian Environmental Quality Guidelines (CEQGs) for the protection of aquatic life are occasionally applied to wastewater treatment studies in Nunavut. While these guidelines are generally voluntary in nature, they can be used as a reference intended to protect all forms of freshwater and marine (including estuarine) aquatic life downstream of wastewater lagoons and wetlands (CCME, 2021). The guidelines include both chemical-specific fact sheets, categorized by media, and scientific criteria documents or supporting documents. These documents outline the guidelines for each substance and include the key scientific information and the rationale for the derivation of the guidelines (CCME, 2021). When comparing the CEQGs to municipal wastewater treatment systems utilized in

Nunavut, the target values are commonly applied to water quality results collected downstream of the treatment area as opposed to those samples collected at compliance locations outlined on a NWB water license.

2.2 Quantitative Microbial Risk Assessment

QMRA was introduced over 40 years ago as a tool to estimate human health risks associated with exposure to microbial hazards (Haas et al., 1999); it has been a widely recognised practice since at least 2004 for use in a variety of environmental settings (Haas et al., 2014; WHO, 2016). This risk assessment approach integrates the scientific understanding of pathogens, their fate and transport through natural and engineered systems, potential paths to human exposure, and ultimately the associated human health effects (WHO, 2016). QMRA generally follows a four-step process following problem formulation, that consists of hazard identification (1), exposure assessment (2), dose–response modelling (3) and risk characterization (4) (Haas et al., 2014; WHO, 2016; Owens et al., 2020). An additional key outcome from QMRA, is the identification of limited or missing data; this helps to identify research gaps and the need for additional data (CAMRA, 2021a).

QMRA is a particularly useful risk assessment method for evaluating risks associated with exposure to low levels of pathogens on an individual basis or across large populations (Haas et al., 2014). As this method uses mathematical models to estimate probability of infection based upon existing information related to human exposure, it can be utilized for scenarios with limited site-specific information (Daley et al., 2018). Depending on the availability of data, point or stochastic modelling can be used. Point models assess risk based upon one value for each parameter, while stochastic models utilize assumptions and probability functions to quantify uncertainty related to spatial and temporal information (Hass et al., 2014; Daley et al., 2018).

A limited number of studies have implemented QMRA to investigate health risks associated with hydrological events through a variety of environmental matrices such as drinking water (Sokolova et al., 2015; Taghipour et al., 2019), recreational water (McBride et al., 2013), and sewage (Kozak et al., 2020). As this approach has proved to be advantageous for water quality studies in settings with limited data and resources, it is believed to be an appropriate method when evaluating human health risks associated with passive wastewater treatment systems in the Arctic (Haas et al., 2014; WHO, 2016; Yapo et al. 2014; Daley et al., 2019).

2.2.1 Hazard Identification

As QMRA is a scenario driven field of study. Problem formulation or hazard identification may include a discussion or consideration of the situations, problems, and potentially the study locations to be addressed (CAMRA, 2021a). This step defines the purpose and scope of the investigation through the selection of relevant microbial agents of concern, the context in which they are found, and the associated range of health effects (WHO, 2016; Daley et al., 2018); the articulation of these steps is necessary when formulating a problem scenario in order to demonstrate the need for such an assessment.

The key outcome of the hazard identification step within the QMRA framework is the selection of hazards or pathogens of interest (CAMRA, 2021a). Pathogens of interest can be chosen through a process in which the exposure pathways, such as occupational or recreational activities, are considered according to the assessment objective and location. Additionally, when selecting hazards or pathogens of interest, health outcomes should be considered and may include adverse health effects ranging from respiratory issues to gastrointestinal illness (CAMRA, 2021a). Overall, acute, and chronic human health effects, severity, sensitive populations, and immunological response for specific pathogens must be defined (Rose et al., 2013).

When identifying hazards, it may not be possible to consider all related human pathogens specific to the QMRA, or dose-response information regarding specific pathogens of interest may be limited. Therefore, reference or surrogate pathogens should be considered as a representative option based upon their characteristics in relation to those pathogens of interest (WHO, 2016; CAMRA, 2021a). It is important to ensure that when the reference pathogen is controlled, from a human health perspective, all other pathogens of concern would be controlled. Therefore, when selecting reference pathogens local conditions, such as relevance to exposure pathways, environmental prevalence, and incidence and severity of illness or disease should be taken into consideration (WHO, 2016).

2.2.2 Exposure Assessment

The objective of the exposure assessment is to estimate the magnitude, duration, and timing of human exposure to the agent of interest through potential exposure pathways (CAMRA, 2021b: Daley et al., 2018). To do this, the pathways in which microorganisms could be transported from the source to a point of contact with humans must first be defined. Next, the amount of exposure that is possible between humans and the contaminants can be estimated (Rose et al., 2013). While identifying the exposure pathways can be straightforward, accurately measuring the true exposure requires a defined concentration of contaminant occurring simultaneously with the human receptor (Daley et al., 2018). As this can be challenging to measure, default assumptions are commonly made regarding contact with contaminated media and associated ingestion rates. To finalize the types and levels of exposure, the media and contact rates are then applied to activity pattern estimations or scenarios (Daley et al., 2018).

To simplify the exposure assessment, it can be broken down into 4 steps; (i) identifying the overall exposure pathways, (ii) defining the mechanisms of exposure through the aforementioned pathways, (iii) quantifying exposure through each mechanism and pathway, and (iv) characterizing the exposure through magnitude and frequency for the range of scenarios to be considered (WHO, 2016). When applying this assessment process to wastewater management, pathogen concentration in wastewater and lagoon water, number of people exposed, frequency of exposure and volume of water ingested during exposure must be defined (Yapo et al., 2014). Conducting a representative exposure assessment is necessary for accurate risk characterization and management (Rose et al., 2013).

2.2.3 Dose–response Assessment

The objective of the does-response assessment is to define a quantitative relationship between pathogen exposure and the probability that such exposure will lead to an adverse health response (Yapo et al., 2014; Daley et al., 2018). Depending on the purpose of the assessment, the human health outcomes may include infection, illness, and/or a measure of disease burden (WHO, 2016). A popular measure of disease burden is disability-adjusted life years (DALYs) (Daley et al., 2018). This metric is commonly used to define the overall community health burden, as it considers influences related to both the quantity and quality of life (WHO, 2016). DALYs has been adopted for use in the development of health-based treatment targets for both drinking water and wastewater guidelines (WHO, 2016).

Dose-response models utilize mathematical functions to link pathogen exposure and resultant health outcomes, such as infection or illness (Daley et al., 2019). Trusted dose-response curves for many microorganisms have already been developed and are commonly selected from published literature specific to the purpose of the assessment (WHO, 2016; Daley et al., 2018). Through clinical trials and data analyses, two dose-response models were developed, and have proven to be widely applicable for most microorganisms and exposure routes, the exponential and beta-Poisson models (Haas et al. 2014; Westrell et al., 2004). These dose-response models provide the assessor with probability of infection, based upon a single exposure to the pathogen of concern. To further determine the probability of illness (symptomatic cases) given that infection has occurred, morbidity ratios can be applied; these morbidity ratios are specific to the pathogens considered (Daley et al., 2019). Additionally, secondary transmission through person-to-person contact and immunity can be assessed within QMRA if desired, using dynamic risk models (WHO, 2016).

2.2.4 Risk characterization

The final step in the QMRA is risk characterization. This step integrates information from the previous three steps into a single mathematical model to quantify the measures of risk specific to the assessment (Daley et al., 2018; WHO, 2016). The risk or health outcome can be quantified using many different metrics such as probability of infection, probability of illness, expected number of illness cases and DALYs. Additionally, the time scale and populations of exposure may vary. These could range from a single exposure event of a single person to a series of exposure events of an entire population (WHO, 2016). Overall, the goal of the risk characterization is to define the level of health response resulting from a specific level of exposure to the agent of interest (Daley et al., 2018).

Additionally, risk characterization can range from a simple point-estimate, in which a dose is input into a dose-response function, to more complex models that consider uncertainty in model input parameters and variability across individuals and subpopulations (CAMRA, 2021c). The complex models are known as probabilistic risk assessments and are commonly performed using Monte Carlo statistical modelling (McBride et al., 2012). Monte Carlo analysis is utilized to calculate a full range of possible risks based upon the range of values for exposure, dose, and hazard, that are determined in the first three steps of the QMRA (WHO, 2016). This type of characterization is performed through a number of iterations in which, for each iteration, samples are drawn from the distribution of exposure, to build up a risk profile including statistics such as health outcome averages and worst-case scenarios (McBride et al., 2012; CAMRA, 2021c).

An important component of risk characterization is the identification and discussion of all assumptions, uncertainties, and variability (Daley et al., 2018). A sensitivity analysis can be conducted to identify which modelling inputs, such as pathogen concentrations, influence the variability and uncertainty in the risk output, or health outcomes (Haas et al., 2014; Daley et al., 2018). Including a sensitivity analysis in these types of risk assessment is specifically helpful to establish the most important sources of variability and uncertainty, which can be further used to identify research or information gaps and therefore focus control measures and additional data collection (WHO, 2016).

2.3 Coupled Hydrodynamic Modelling and QMRA

QMRA is a beneficial tool for use in screening-level assessments of human health risks associated with exposure to contaminated water sources; however, a common limitation of this approach is minimal or lacking input data (Sokolova et al., 2013). As pathogen concentrations in water sources are driven by upstream loading events, they tend to be highly variable, thus making it challenging for routine monitoring to detect potentially hazardous peaks in pathogen concentrations (Westrell et al., 2006; Sokolova et al., 2013;

Sokolova et al., 2015). Additionally, the minimum detection limit for water quality analytical methods can be greater than pathogen concentrations relevant to public health (Sokolova et al., 2013). As continuous monitoring of contaminant levels may not be economically or practically feasible (McCarthy et al., 2007), water quality data can be supplemented with hydrodynamic modelling (McBride et al., 2012; Taghipour et al., 2019). Hydrodynamic models can be used to simulate the effects of upstream loadings from contamination sources and therefore help to provide estimates regarding the transfer of contaminants to points of exposure (McBride et al., 2012; Sokolova et al., 2013).

Information and data collected through field studies, such as the fate and transport of fecal indicator bacteria, can be applied to hydrodynamic models to predict pathogen concentrations at exposure locations throughout various environmental conditions (Tolouei et al., 2019). When supplementing field monitoring with hydrodynamic modelling for use in QMRA, models must be developed and validated using appropriate data sets and must account for the complex physical and biological processes that impact the fate and transport of pathogens throughout the environment (McBride et al., 2012). Parameterized models, used to simulate fecal indicator bacteria and therefore pathogens of interest, can then be integrated with exposure and dose specific information to predict human health risk (Tolouei et al., 2019).

When considering remote passive wastewater treatment systems, such as those utilized in many Nunavut communities, detailed information regarding pathogen concentrations across various hydro-meteorological conditions can seldom be obtained from field sampling alone. Therefore, coupling hydrodynamic modelling with QMRA approaches provides an opportunity to overcome such research gaps and characterize the human health risk associated with passive wastewater treatment in a Nunavut community. While hydrodynamic modelling coupled with QMRA has proven to be useful when assessing the influence of various environmental conditions on microbial water quality, it remains an emerging discipline in which continued research is essential (Sokolova et al., 2013; McBride et al., 2012). Specific consideration should be applied to the pathogens of concern selected for assessment and the associated dose-response relationships, along with the processes that may influence contaminant fate and transport (McBride et al., 2012).

CHAPTER 3.METHODOLOGY

3.1 Research Strategy

In order to meet the study objectives, a modelling approach was taken to evaluate environmental and human health risks associated with the wastewater treatment area in Qamani'tuaq. Site-specific data were collected to calibrate and validate a hydrologic model that was developed to represent the WTA and surrounding watershed. An integrated contaminant transport model was then utilized, in conjunction with simulated data obtained from the hydrologic model, to estimate pathogen concentrations at several locations in the area (Figure 3.1.1). Locations of interest were identified based on information gained through onsite observation and through interviews with local community members and organizations. The potential for community exposure to the pathogens of concern at these locations was then defined and applied to a QMRA. Overall, this study aimed to assess the performance of wastewater treatment in Qamani'tuaq, and to increase the understanding of associated impacts on human health in the community. In addition, it is anticipated that the results of the study will provide valuable information on the requirements for future wastewater treatment system upgrades.

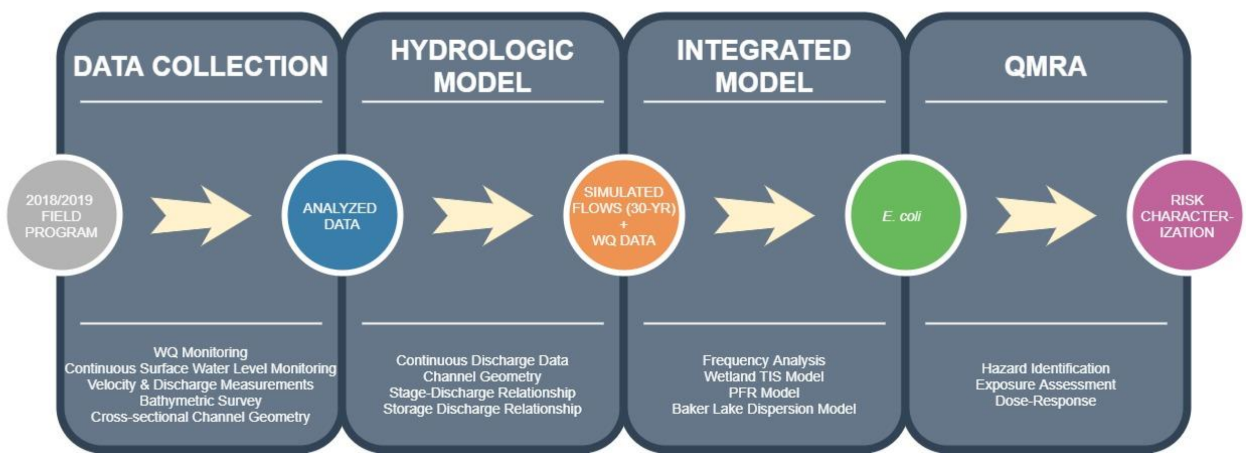

Figure 3.1.1 Flow diagram demonstrating research strategy

3.2 Site Description

The hamlet of Qamani'tuaq is located in the Kivalliq Region of central Nunavut at 64°19'05"N, 96°01'03"W; it is the only inland community in the territory. It is located to the west of the Hudson Bay on the northern shore of Baker Lake (Figure 3.2.1), which is a large freshwater lake with a surface area of approximately 1800 km^2 and a depth of approximately 60 m (Medeiros et al., 2012). The average annual precipitation for the area is 273 mm, with an average annual snowfall of 127 cm (Government of Canada, 2018). Annual average air temperatures for the area range from -31°C in January to 12°C in July (Government of Canada, 2018). The community of Qamani'tuaq is located in a zone of continuous permafrost and is underlain by till substrate with coarse gravels and sands with low ice contents (Throop et al., 2012). Catchment coverage in the area has been classified as Arctic tundra with moderate vegetation and some peat, including dwarf shrubs, grasses, mosses, fireweed, and cloud berry (Throop et al., 2012; Medeiros et al., 2012).

Figure 3.2.1 Location map for Qamani'tuaq study area

3.2.1 Wetland Treatment Area

The current wastewater treatment system consists of a 5.5 km^2 WTA situated within an 18 km^2 watershed, located approximately 1 km north of the hamlet of Qamani'tuaq (Figure 3.2.2). The WTA is the portion of watershed immediately impacted by raw wastewater, extending from the wastewater discharge location (holding cell) to the inlet of Finger Lake; the area is largely surrounding by fencing with some signage identifying the nature of the area. The inlet to Finger Lake is listed as the compliance point, or the end of the treatment area, according to the Nunavut Water Board (NWB) license for the Hamlet of Qamani'tuaq.

Figure 3.2.2 Wetland treatment area and surrounding watershed

As is common in many Nunavut communities, sewage is collected from commercial and residential dwellings by truck before being transported to the treatment area. The passive wastewater treatment system consists of an un-engineered holding cell that has a surface area of approximately 750 m^2, in which discharge is uncontrolled. Sewage exits the holding cell by exfiltration or by overtopping the berms, before flowing approximately 200 m downslope and entering Lagoon Lake; freshwater from upstream catchments enters the WTA at this location. From Lagoon Lake, diluted wastewater flows through a shallow wetland approximately 300 m east before discharging into Finger Lake. Runoff from the solid waste site has also been observed to enter the catchment at this location. Finger Lake then discharges into a channel approximately 1000 m in length before entering Airplane Lake. Airplane Lake receives freshwater from upstream catchments and ultimately drains into Baker Lake through Akkutuak Creek, 800 m to the south. The outlet of Akkutuak Creek, and the encompassing watershed, is approximately 2 km west of the drinking water intake location for the community. Figure 3.2.3 shows the treatment pathway.

Figure 3.2.3 Watershed treatment process and flow path

3.3 Field Monitoring Program

The hydrology and contaminant loading of the current treatment system was characterized during the 2018 and 2019 treatment seasons. Initially, it was anticipated that the 2018 treatment season would be considered as a preliminary sampling event, with key data collection to be conducted during the 2019 and 2020 treatment seasons. Unfortunately, due to travel restrictions implemented throughout Canada in March 2020 to address concerns related to Coronavirus disease (COVID-19), all site visits for the 2020 field season were cancelled. Sampling trips were therefore conducted on three occasions from June 2018 to September 2019. The dates of site visits, representative study periods, number of field days, and rounds of treatment performance samples collected are summarized in Table 3.3.1.

Table 3.3.1 Field monitoring program information

Arrival Date	Departure Date	Study Period	No. of Days with Flow	No. of WQ Sampling Events	No. of Flow Measurements
06/04/2018	06/12/2018	Spring freshet	2	1	1
06/10/2019	06/20/2019	Spring freshet	10	2	3 - 5
09/02/2019	09/10/2019	Late summer	8	1	2

The site was studied twice during spring freshet (June 2018 and 2019) to assess the performance of the treatment area during snowmelt, when uncontrolled treatment systems are commonly overloaded with thawed wastewater and watershed runoff entering the treatment area. A late summer trip was conducted in September 2019 to evaluate the system during the dry season, when flowrates through similar treatment areas are commonly minimized, and treatment performance is anticipated to be most favourable. Additionally, *in situ* monitoring devices were installed throughout the watershed for continuous data collection throughout the 2019 treatment season.

3.3.1 Water Quality Monitoring

In order to characterize the performance of the current wastewater treatment area, basic water quality parameters were monitored, and discrete water samples were collected. Water quality samples were analyzed for the following suite of parameters, commonly found in municipal wastewater effluent: total suspended solids (TSS), volatile suspended solids (VSS), total ammonia nitrogen (TAN), total Kjeldah nitrogen (TKN), total phosphorus (TP), pH, temperature, *E. coli*, and fecal coliforms. All sample analyses were completed by an accredited laboratory, with samples collected during 2018 submitted to Bureau Veritas (then Maxxam Analytics) in Montreal, Quebec and samples collected in 2019 submitted to H2Lab in Rouyn-Noranda, Quebec. Sample containers were received from the institution in which they were to be analyzed and were preloaded with preservatives specific to the requested suite of parameters. Samples were stored on ice at approximately 5^{o}C until the time of analysis and were submitted to the respective laboratories within 48 hours of collection.

Five primary water quality sample locations were selected to evaluate the treatment performance of the current system with respect to the parameters of interest (Figure 3.3.1). A sample was collected from an upstream channel and was utilized as a reference sample (REF), to represent freshwater in the surrounding area that had not been impacted by wastewater. A sample was collected from the holding cell/lagoon to represent raw water concentrations (LAG). A sample was collected at the discharge location of the WTA into the receiving environment (FL-I) and at three downstream locations prior to release into Baker Lake (FL-O, AL-I, GC).

In order to address concerns related to runoff from the adjacent solid waste site, select samples from the above sites were also analyzed for a suite of metals.

Four supplementary locations were sampled along the Baker Lake shoreline; three which were located between the outlet of the study watershed and the community's drinking water intake (BL-I, BL-MID, DW), and one from a stormwater culvert that discharged into Baker Lake in close proximity to the drinking water intake (SW). These samples were analyzed for bacteriological parameters, *E. coli* and fecal coliforms.

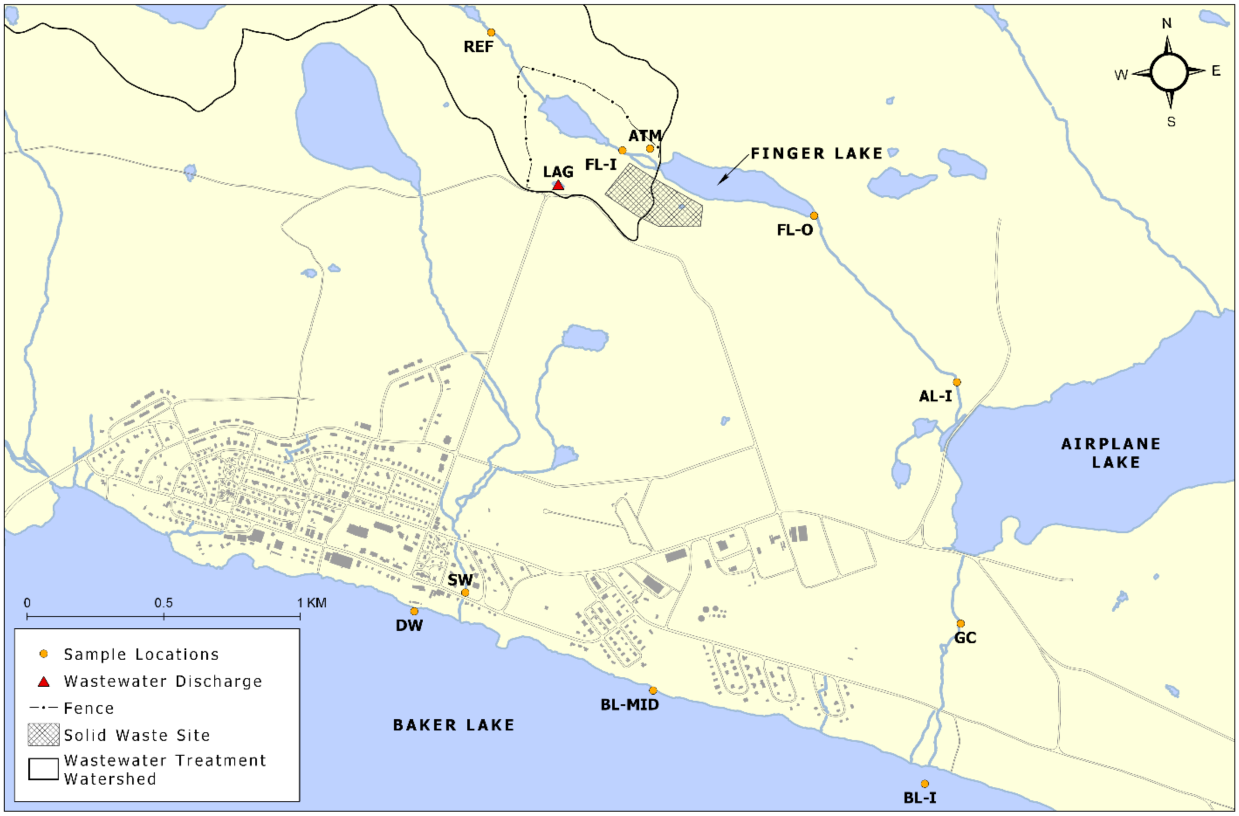

Figure 3.3.1 Sample location map

During sampling, *in-situ* water quality measurements were taken with a handheld YSI multiparameter water quality sonde 600 model (YSI Inc., Yellow Springs, Ohio, United States); these included pH, dissolved oxygen (DO), temperature, specific conductivity, and oxidation reduction potential (ORP). Daily measurements were taken during the 2019 spring freshet sampling trip, and at the time of each water quality sampling or stream gauging event during the other two site visits. All sondes were calibrated in Halifax according to the manufacturer's specifications (YSI Inc., 2020) prior to each site visit, while the DO probe was calibrated on-site prior to use.

3.3.2 Continuous Surface Water Level Monitoring

Water pressure and temperature were monitored continuously *in-situ* at the five main water quality sampling locations with HOBO Water Level Data Loggers (U20L-04, Onset® Computer Corporation, Bourne, Massachusetts, United States). An additional data logger was deployed in an open-air environment within the WTA to continuously monitor atmospheric pressure (ATM, Figure 3.3.1). The data loggers were programed to collect measurements at 30-minute intervals, and HOBOware Pro software was utilized to convert the pressure data to water level readings, while fully compensating for barometric pressure.

3.3.3 Surface Water Velocity and Discharge

Channel discharge was measured according to the velocity-area method outlined by Dingman (2002) using a USGS Model 625 Pygmy current meter (Gurley Precision Instruments, Troy, New York, USA), 1100 model indicator digital read-out, and 2 m wading rod. Channel and flood bank geometry was surveyed at three of the sample sites in September 2019 using a surveying level and rod (FL-O, AL-I, GC). The channel geometry was then used to parameterize Manning's equation to estimate bankfull discharge, which was used as high flow value when constructing stage-discharge relationships for each sampling location. A composite roughness coefficient was calculated for each site based upon field observations of the streambank and literature-based Manning's roughness coefficients (Sturm, 2001).

All discharge estimates were plotted against the corresponding stage and a regression analysis was conducted using the Microsoft Excel™ Trendline tool. The stage-discharge relationship was determined for each sampling site by setting the trendline to a Power function and ensuring that an adequate coefficient of determination (R^2) was achieved. The stage-discharge relationships were applied to the continuous water level data collected throughout the 2019 treatment season to develop hydrographs for each site.

3.3.4 Bathymetric Survey and Estimation of Storage Capacity

Bathymetric surveys were conducted in September 2019 for each of the two key waterbodies within the watershed, Finger Lake and Airplane Lake. Point-by-point bathymetry was measured over the water surface of each lake using a Garmin echoMAP 50dv combination fishfinder/chartplotter (Garmin International, Inc., Olathe, Kansas, USA). The global positioning system (GPS) points and associated depth measurements were then uploaded as point features in ArcGIS Pro 2.3.0 (2018 Esri Inc.), and bathymetric maps were generated. To create the bathymetric maps, additional point features with estimated water depths were added along the west portion of Finger Lake as data could not be collected onsite due to technical issues encountered with the instrumentation. Esri shapefiles were obtained from Government of Nunavut Community and Government Services Planning and Lands Division (2020) to represent the water body boundaries. The shapefiles contained waterbody polygons which were imported into ArcGIS and assigned a water surface elevation (z-unit) of zero. The Topo to Raster tool, available with the Spatial Analyst license, was then used to interpolate a hydrologically correct raster elevation dataset for each lake using the bathymetry point data and the waterbody polygon data as inputs. Next, the Contour function was used to create contour lines by joining the points with the same elevation from the raster elevation dataset. Finally, the Surface Volume tool, available with the 3D Analyst license, was used to calculate the surface area and volume of the region between the raster surface (lake depths) and the respective reference plane (reservoir boundaries).

The approximate storage volume and water surface area obtained from ArcGIS Pro were than used to develop a storage-discharge relationship for each waterbody. The crest height for each lake was estimated based upon the measured water level of the study site immediately downstream of the lake's outlet, FL-O for Finger Lake and GC for Airplane Lake. For this study, the crest height represented the minimum water volume in the lake before discharge occurred; the crest height would represent no flow conditions in the downstream channel. The storage-discharge relationship was then modelled in Microsoft Excel™ for each reservoir by increasing the volume of water stored in the reservoir and calculating the associated discharge through the outlet channel. For Finger Lake, Manning's equation was developed for the discharge relationships based upon surveyed cross-sectional channel geometry (Section 3.3.3). As water from Airplane Lake was observed to discharge through a culvert into Akkutuak Creek, flow through the culvert was calculated based upon Manning's equation and uniform partially full pipe flow equations.

The Manning's roughness coefficient was considered to vary as a function of the ratio of the flow depth to the culvert diameter (Camp, 1946).

3.4 Hydrologic Model: HEC-HMS

Hydrologic modelling was conducted to develop a representative model of the current wastewater treatment system and surrounding watershed. The Hydrologic Engineering Center's Hydrologic Modelling System (HEC-HMS) was selected as it is a generalized modeling system capable of simulating precipitation-runoff and routing processes for small, natural watershed systems (USACE, 2020). The goal of this portion of the study was to use HEC-HMS to develop a model that could accurately estimate peak flow values at specific locations throughout the watershed using both historical climate data, and future (projected) climate data, if desired.

3.4.1 Watershed Physical Description

A high-resolution digital elevation model (HRDEM) for the area was obtained from the Government of Canada – Open Government, CanSeries website (Government of Canada, 2019). Due to the low density of vegetation and infrastructure in northern Canada, only a digital surface model (DSM) dataset was available. The dataset was generated at a 5-metre resolution using the Polar Stereographic North coordinate system referenced to the UTM NAD83 (CSRS) coordinate system (Government of Canada, 2019). The watershed and subbasins were delineated using the Hydrology toolset available with the Spatial Analyst license in ArcGIS Pro. The shapefiles representing the delineated subbasins were then imported into HEC-HMS to provide the spatial context for the hydrologic elements within the basin model, such as subbasin boundaries and the locations of streams and reservoirs (Figure 3.4.1). Although the background map provided spatial context, it was not used in the computation process. Additional hydrologic elements, such as reach, reservoir, and junction elements were then added to the basin model and connected in a dendritic network to form the representative watershed system (Appendix A).

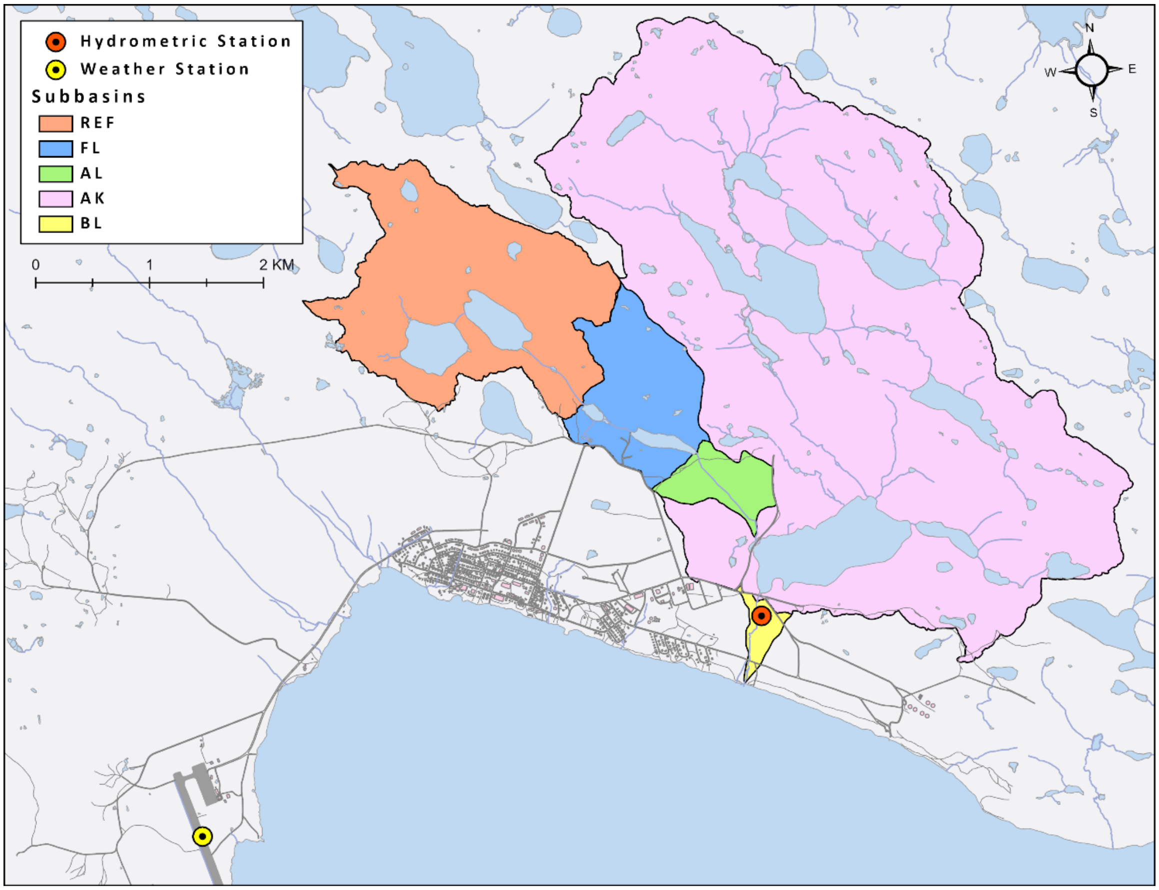

Figure 3.4.1 Subbasin map with weather and hydrometric stations.

3.4.2 Model Elements and Process Representations

The hydrologic model of the study site was comprised of five subbasin elements. All subbasin elements utilized the same mathematical models to represent each physical process occurring in the watershed; canopy interception, surface storage, infiltration, surface runoff, and baseflow. A conceptual representation of the hydrologic elements and processes is shown in Figure 3.4.2, while Sections 3.4.2.1 through 3.4.2.8 discuss the model methods in detail.

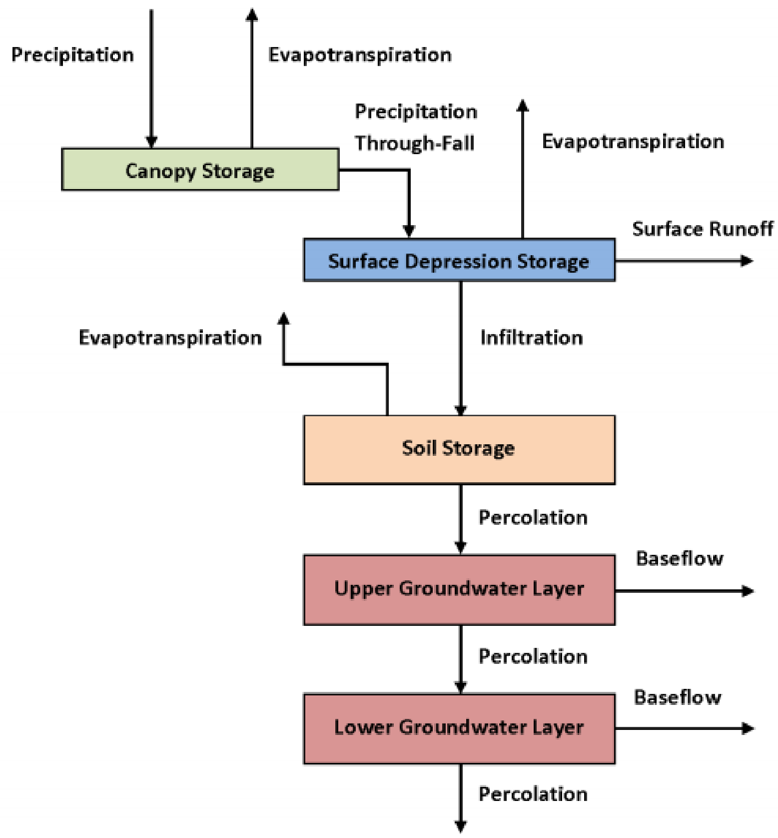

Figure 3.4.2 Hydrological elements used to characterize subbasins (Golder, 2013).

3.4.2.1 Canopy Method

The canopy method is an optional model within HEC-HMS, used to represent interception by vegetation. The simple canopy method was used whereby a maximum canopy storage capacity is specified; all precipitation was intercepted by the plant canopy until the storage capacity was reached. Excess precipitation would then fall to the surface and soil.

3.4.2.2 Surface Method

The surface method in HEC-HMS represents the depression storage of the ground or soil surface. Precipitation through-fall from the canopy layer arrives on the surface and is captured in storage before infiltrating into the soil. The simple surface model was selected, as in this model infiltration occurs at the rate in which the soil has the capacity to accept water, even when the storage capacity is not full. Surface runoff begins when the precipitation through-fall rate exceeds the infiltration rate, and the storage capacity is filled (USACE, 2020).

3.4.2.3 Loss Method

Twelve different loss (infiltration) methods are available in HEC-HMS. The soil moisture accounting method (SMA) was selected for this study as it was recommended for continuous simulations and was designed for use in combination with both a canopy and surface method (USACE, 2020). This method conceptualizes the landscape as three layers (soil storage, upper groundwater, and lower groundwater) to represent the dynamic movement of water throughout the vertical profile of the soil. Additionally, when used in combination with the linear reservoir baseflow method, this method allows for the lateral movement of water from upper and lower groundwater to baseflow (USACE, 2020).

Although the watershed is minimally developed, with no man-made impervious surfaces such as paved roads, the impervious parameter was assumed to be 95% for all subbasins. This value was used to better represent the frozen state of watershed soils during the early spring and summer months.

3.4.2.4 Transform Method

In HEC-HMS eight different transform methods are available, including a kinematic wave transform, a linear quasi-distributed method, and various unit hydrograph methods. The Soil Conservation Service (SCS) unit hydrograph method was selected for this study.

3.4.2.5 Baseflow Method

The linear reservoir option was selected for baseflow simulation, which uses a linear reservoir(s) to model the recession of baseflow after a storm event and is the only method that conserves mass within the subbasin. When this method is used in conjunction with the SMA loss method, the number of baseflow reservoirs correspond with the number of groundwater layers, and the lateral outflow from each groundwater layer is connected to the inflow for the respective baseflow reservoir (USACE, 2020).

3.4.2.6 Routing Elements

The reach element in HEC-HMS is used to represent a watershed feature that can have multiple inflows but only one outflow, and is commonly used to model rivers, streams, or channels. The reach element performs calculations using one of the nine available hydrologic routing methodologies, each of which differs in level of detail and applicability. The Muskingum-Cunge routing method was selected for this study as the routing parameters are recalculated every timestep based on channel properties and the flow depth (USACE, 2020). It is also recommended for reaches with a small slope, similar to those in the study watershed. This routing method allows for the use of data collected specific to the study site, such as Manning's roughness coefficient and reach length, slope, and cross-sectional geometry.

3.4.2.7 Reservoir Elements

The reservoir element in HEC-HMS is used to model reservoirs, lakes, and ponds, with one or more inflow and one computed outflow. The reservoir element provides 3 options for computing reservoir routing, all which assume the surface water in the reservoir to be level. The outflow curve method was selected to represent both reservoirs in the basin model, Finger Lake and Airplane Lake, as storage-discharge relationships had been developed as part of the bathymetric study conducted in September 2019 (Section 3.3.4). The computations in this method are completed with the Modified Puls algorithm with one routing step (USACE, 2020).

In the reservoir element there are three different options for specifying the storage relationship. The storage-discharge method, selected for this study, requires that a storage-discharge curve be selected from the Paired Data Manager. As preciously mentioned, the storage-discharge curves specific to each of the reservoirs, Finger Lake and Airplane Lake, were developed based on collected field data. HEC-HMS also offers three options for specifying the initial condition of the reservoir; the storage method was selected for simplicity.

3.4.2.8 Junction Elements

A junction element in HEC-HMS is used in the flow network to combine multiple inflows, often at a confluence, while assuming zero storage (USACE, 2020). For this model, a junction element was used to represent the watershed outlet into Baker Lake, where simulated and observed data would be compared.

3.4.3 Meteorology Description

In HEC-HMS the meteorological model contains the time series data responsible for preparing climate inputs. Climate data from Environment Canada was used to populate the meteorological model in HEC-HMS. The nearest Environment Canada Weather station to the study area was Baker Lake A, which was located at 64˚17’56” N, 96˚04’40” W approximately 4.5 km northeast of the watershed, at an elevation of 18.6 m (Figure 3.4.1). Due to the limited data available for the area, this weather station was assumed to be representative across all subbasins. RStudio Version 1.1.463 (RStudio Team, 2016) was utilized along with the R package ‘weathercan’ (LaZerte & Albers, 2018) to download historical climate data from Baker Lake A between January 1, 1982 and December 31, 1986. These dates were selected for the model’s calibration period as the weather station and a Water Survey of Canada hydrometric station (Section 3.4.4) both had 5 years of consecutive data during this time. Additional data were obtained from Baker Lake A for the model validation period extending from June 1, 2018 to September 30, 2019. Climate data available on a daily timestep were downloaded and imported directly into HEC-HMS, while data available on an hourly timestep were averaged or summarized into a daily timestep prior to use in the model. Environment Canada climate data required for the meteorological processes outlined below included total precipitation, average air temperature, dew point temperature, and average wind speed.

As solar radiation data for Qamani’tuaq was not available from Environment and Climate Change Canada for the required time periods (1982 to 1986, and 2018 to 2019), both shortwave and longwave radiation data were obtained from NASA Langley Research Center POWER Project funded through the NASA Earth Science/Applied Science Program. All solar radiation data were obtained on a daily timestep and converted from units of $MJ/m^2/day$ to W/m^2 for use in the HEC-HMS model. The data were assumed to be representative across all subbasins.

3.4.3.1 Evapotranspiration

Evapotranspiration is the process in which water is returned to the atmosphere through a combination of evaporation of free water from vegetation and land surfaces, and transpiration by vegetation. In HEC-HMS the meteorological model computes the potential evapotranspiration, which is the upper limit of available evapotranspiration based on atmospheric conditions; actual evapotranspiration is then calculated within each subbasin based upon soil water limitation (USACE, 2020).

The Penman Monteith method was selected for calculation of evapotranspiration for each subbasin in the model. For this method, which is completely dependent on atmospheric conditions, daily time-series of shortwave radiation, longwave radiation, windspeed, average daily temperature, and dewpoint temperature are required. A surface reference albedo was estimated based upon literature values for areas with similar ground coverage to the study watershed. The potential evapotranspiration was then calculated according to the method and procedure outlined in Allen et al. (1998).

3.4.3.2 Snowmelt

The only method available to calculate snow processes within HEC-HMS at the time of this study was a conceptual temperature index approach that used a degree-day approach to model snowpack dynamics. This method included a conceptual representation of cold energy storage within the snowpack, along with other factors, to compute the amount snowmelt for each degree above freezing (USACE, 2020). It utilized both a melt-rate coefficient and a cold-rate coefficient to track snowpack conditions. As the watershed is relatively low-relief, one elevation band was utilized to represent snowmelt across the entire watershed.

3.4.4 Hydrometric Data

Historical hydrometric data were obtained from Environment and Climate Change Canada (ECCC) for the study watershed from the hydrometric station at Akkutuak Creek near Baker Lake located at 64˚18’57” N, 95˚58’23” W (Figure 3.4.1). This station was located on Akkutuak Creek, approximately 200 m downstream of the final sampling location prior to discharge into Baker Lake. This station represented drainage for the entire study watershed with an approximate gross drainage area of 18 km^2 and was active from 1978 to 1990, including five consecutive years from 1982 to 1986 that coincided with the available climate data. The hydrometric data obtained from this station were utilized to optimize the HEC-HMS model during the calibration period.

3.4.5 Sensitivity Analysis

A local sensitivity analysis was completed to determine which model parameters would most significantly influence model performance. To evaluate the sensitivity of a parameter, a univariate analysis was completed in which the parameter of interest was varied by up to ±30% while all other model parameters were kept constant. The sensitivity of each parameter was tested with respect to both the simulated peak flow and the runoff volume at the watershed outlet. In order to compare the sensitivity of multiple model parameters, a relative sensitivity index (R_s) was determined for each parameter of interest according to Equation [3.4.1], (Haan, 2002):

$$R_s = \frac{\frac{\partial O}{O}}{\frac{\partial P}{P}} = \frac{\partial O}{\partial P} \cdot \frac{P}{O} \qquad [3.4.1]$$

Where:

O	=	The model output, or objective function
P	=	The parameter of interest, or input parameter
∂	=	Partial differential, or change in the variable

To assess the relative sensitivity of a parameter, the sensitivity index was classified according to the scheme outlined by Lenhart et al. (2002) in Table 3.4.1.

Table 3.4.1 Relative sensitivity classification scheme

R_s	Sensitivity Class
$0 \leq \| R_s \| < 0.05$	Small to negligible
$0.05 \leq \| R_s \| < 0.2$	Medium
$0.2 \leq \| R_s \| < 1.00$	High
$\| R_s \| \geq 1.00$	Very high

3.4.6 Model Calibration and Validation

Model calibration is a systematic approach of adjusting model parameters of interest until simulated results adequately represent observed data at a desired location within the watershed model. Calibration can be completed manually or using automated optimization algorithms. While all the mathematical models included in HEC-HMS are deterministic, the program offers two methods of automated optimization: deterministic and stochastic. For the study watershed, observed flow data obtained from the Akkutuak Creek hydrometric station was used to optimize model performance by automatically estimating the model parameters using the deterministic method. In this method, the initial parameter estimates are adjusted to minimize the difference between the simulated and observed streamflow data at the watershed outlet (USACE, 2020). Input parameters classified to have a high ($0.2 \leq |Rs| < 1.00$) or very high ($|Rs| \geq 1.00$) relative sensitivity were selected for use in the automated optimization for the study watershed. A variety of objective functions are provided in HEC-HMS to provide goodness-of-fit between the simulated and observed streamflow. Percent error in peak discharge was selected as the objective function for the automated calibration process; this method only considers the difference in the magnitude of peak flow and does not account for the timing of peak or the total flow volume. Following the automated optimization, temperature index parameters and the ATI-Meltrate function were manually adjusted until a satisfactory fit to observed hydrographs was achieved. The optimization trial extended from January 1, 1982 until December 1, 1986 while the objective function was minimized from January 1, 1984 until December 1, 1986 to allow for a 2-year warm-up period. The final values used for each method in the calibrated model, specific to each subbasin, are included in Appendix B.

Performance of the calibrated model was assessed using qualitative evaluation and a comparison of two summary statistics. The qualitative evaluation was the initial assessment and consisted of a visual comparison of the simulated hydrograph and the observed data hydrograph. The first evaluation statistic calculated was the Nash-Sutcliffe coefficient of efficiency (NSE), a relative goodness-of-fit indicator commonly used to evaluate the performance of hydrologic models (Equation [3.4.2]). The NSE is a dimensionless indicator that ranges from -∞ to 1, with higher values indicating better model agreement and values greater than 0.5, calculated on a monthly time step, generally reported to be acceptable (Moriasi et al., 2007). This evaluation statistic represents the ratio of the mean square error to the variance in observed data, subtracted from the unity, and therefore can be sensitive to extreme values (Legates & McCabe, 1999; Harmel et al., 2014).

$$NSE = 1.0 - \frac{\sum_{i=1}^{N}(O_i - P_i)^2}{\sum_{i=1}^{N}(O_i - \bar{O})^2} \qquad [3.4.2]$$

Where:

O_i = i^{th} observation for the constituent being evaluated

P_i = i^{th} simulated value for the constituent being evaluated

$\bar{O}$ = mean of observed data for the constituent being evaluated

N = total number of observations

The second evaluation statistic applied to the model was the RMSE-observations standard deviation ratio (RSR). The RSR was developed by Singh et al. (2004) and incorporates the benefits of error index statistics and additional information as suggested by Legates and McCabe (1999). It standardizes the root mean square error (RMSE) by calculating the ratio of the RMSE and standard deviation of measured data (Equation 3.4.3). While a RSR of 0 indicates a perfect model fit, with no residual variation, Moriasi et al. (2007) suggests that a value of less than 0.70 could be considered satisfactory.

$$RSR = \frac{\text{RMSE}}{\text{STDEV}_{obs}} = \frac{\sqrt{\sum_{i=1}^{N}(O_i - P_i)^2}}{\sqrt{\sum_{i=1}^{N}(O_i - \bar{O})^2}} \qquad [3.4.3]$$

To validate the model, a simulation was run from June 7, 2018 until September 30, 2019, with the warm-up period extending from June 2018 to June 2019. The simulation data were compared to discharge data collected during the field monitoring program (June 14, 2019 to September 3, 2019) using both goodness-of-fit indicators.

3.4.7 Frequency Analysis

A frequency analysis was completed to determine the annual maximum daily flow for return periods ranging from 2 to 100 years. To determine the peak flow scenarios, the hydrologic model was first run for a 30-year simulation period using historical climate data obtained from the Baker Lake A weather station between January 1, 1989 and December 31, 2019. The annual maximum daily flow for each subbasin was determined for each calendar year using the MAXIFS function in Microsoft Excel; this included the maximum flow at the watershed outlet. EasyFit 5.6 Professional statistical software was then used to fit the annual time series for each subbasin/outlet location to a lognormal probability density function, and the maximum flow was estimated for 2, 10, 25, 50 and 100-year return periods.

3.5 Contaminant Fate and Transport Models

An integrated model was developed to estimate *E. coli* concentrations at locations of interest throughout the study watershed, based upon simulated annual peak flow values and recorded water quality data. A modified tanks-in-series (TIS) model and a water mass balance (plug flow case) were used in sequence to assess the transport of *E. coli* within the WTA and downstream to the inlet of Baker Lake, respectively. A bacterial model with advection and dispersion was then used to predict *E. coli* concentrations along the shoreline of Baker Lake, between the outlet of Akkutuak Creek and the community's drinking water intake.

3.5.1 Modified Tanks-in-Series Model

A modified TIS model, recommended by Hayward and Jamieson (2015) for performance modeling of predominately surface flow wetlands, was used to represent the WTA up to the outlet of Finger Lake for each return period. The TIS model was based on a conventional TIS chemical reactor model, with modifications to account for external hydrologic contributions from the watershed that were cumulatively added along the length of the wetland (Hayward and Jamieson, 2015). The general mass balance was rearranged by Hayward and Jamieson (2015) for each tank in the modified TIS model to solve for the contaminant concentration leaving each wetland compartment, with the water balance components of evapotranspiration, precipitation, and infiltration assumed to be negligible (Equation [3.5.1]). To best represent the study watershed, the TIS model was developed to consist of two compartments, each with a series of 3 completely mixed tanks (*N*) possessing an equivalent hydraulic retention time (τ). The first compartment (1) was developed to represent the WTA, between the wastewater discharge location and the inlet of Finger Lake, while the second compartment (2) modelled contaminant transport throughout Finger Lake.

$$C_{out} = \frac{\left(\frac{Q_{in}}{Q_{out}}\right) C_{in} + \left(\frac{\frac{Q_{ws}}{N}}{Q_{out}}\right) C^* + \frac{k\tau C^*}{Nd}}{1 + \frac{k\tau}{Nd}} \qquad [3.5.1]$$

Where:

C_{in}	=	Concentration into tank 'N' [CFU/100 mL]
C^*	=	Background concentration [CFU/100 mL]
Q_{out}	=	Flow out of tank 'N' [m^3/d]
Q_{in}	=	Flow into tank 'N' [m^3/d]
Q_{ws}	=	External hydrologic contribution into the wetland segment [m^3/d]
N	=	Number of tanks

k	=	First order areal rate constant [m/d]
τ	=	Hydraulic retention time [d]
d	=	Average wetland depth [m]

To initiate the TIS model, the mass loading of *E. coli* was calculated from the daily projected water usage for the community (Q_{in1}) and the estimated *E. coli* concentration in the raw wastewater (C_{in1}). The annual projected water usage was determined using Equation [3.5.2], the standard design equation for a northern community with a population (*Pop.*) greater than 2,000 people (Heinke et al., 1991). The residential water use (RWU) was assumed to be 90 litres per person per day, as this was the reported engineering design standard used in Northern Canada for trucked water distribution systems (Smith & Emde, 1999). The annual projected water usage was assumed to be equal to the volume of wastewater produced. This volume was converted to a total daily discharge for the community based upon a 4-month long decant with the assumption that wastewater only enters the WTA from the lagoon between the beginning of June until the end of September. The initial *E. coli* concentration was presumed to be equal to the raw wastewater concentration, which was taken as the highest value recorded from the lagoon water quality data collected during the 2018 and 2019 treatment seasons.

$$RWU \times [-1.0 + (0.323 \times lnPop.)] \qquad [3.5.2]$$

The maximum flow predicted for each return period from the reference subbasin was then input into the TIS model as hydrologic contribution for component 1 (Q_{ws1}), while the maximum flow from the Finger Lake subbasin was input as the hydrologic contribution for component 2 (Q_{ws2}). An areal rate constant (*k*) of 70 m/year was used for the model simulation, as it was considered a conservative value when compared to studies completed for similar systems in Nunavut (Hayward and Jamieson, 2015).

3.5.2 Plug Flow Reactor Model

The transport of *E. coli* throughout the channel extending from the outlet of Finger Lake to the to the inlet of Baker Lake was then modelled as a simple plug flow case for a wetland with water gains. The equation presented by Kadlec and Knight (1996) was modified to account for negligible evapotranspiration and infiltration (Equation [3.5.3]). The inlet concentration (C_i) was taken from the *E. coli* outlet concentration from the third tank of component 2 (TIS model), while water gains from the surrounding watershed were considered as precipitation (P). The inlet flow velocity (q_i) was calculated as the ratio of the outlet flow from the third tank of component 2 (TIS model) to the surface area of the channel, and the areal rate constant was assumed to be equal to that used in the modified TIS model. The background concentration (C_a) was calculated based upon the reference *E. coli* concentration, the areal rate constant, and water gains.

$$C = C_a + (C_i - C_a) \times \left(\frac{q_i + Py}{q_i}\right)^{-\frac{k}{P}} \qquad [3.5.3]$$

Where:

C	=	Concentration in water, at location *y* [CFU/100 mL]
C_a	=	Background concentration [CFU/100 mL]
C_i	=	Inlet concentration [CFU/100 mL]
q_i	=	Inlet flow velocity [m/d]
P	=	Precipitation, or water gains [m/d]
y	=	Location of interest, ratio to total flow path length

Modelling this portion of the watershed as a plug flow case was believed to be representative of the system during spring freshet conditions, as meltwater was observed to flow as a channel over the frozen surface of Airplane Lake directly into Akkutuak Creek.

3.5.3 Advection-Dispersion Model

A near shore advection-dispersion model was used to simulate the transport of *E. coli* in the vicinity of the watershed outlet along the Baker Lake shoreline; between the outlet of Akkutuak Creek and the drinking water intake. The model followed the steady-state case in an infinite fluid, with advection along the shoreline (Equation [3.5.4]), as outlined by Chapra (1997). In order to represent the transport of *E. coli*, Baker Lake was assumed to be infinitely long and wide, and vertically well-mixed, with the waste source emanating from a line of infinitely small thickness at the shoreline. Horizontal diffusion and advection were assumed to be the sole transport mechanisms, with diffusion equal in all directions and a current (advection) ranging from 0.01 to 1 m/s along the shoreline, moving towards the drinking water intake. A conservative diffusion coefficient of 10^8 cm^2/s was used, as the suggested typical range for diffusion coefficients in natural waters in lakes varied from 10^2 to 10^8 cm^2/s (Chapra, 1997). The concentration of *E. coli* was then calculated for a mixing radius extending from 0 to 2 km, in 500 m steps, using a degradation rate of 0.24 d^{-1} at 20°C as suggested by Blaustein et al. (2013) for *E. coli* in a lake. The Arrenhius equation was then used to adjust the degradation rate to 2°C, the temperature applied throughout the integrated model to represent spring melt water temperatures. The model was run for each return period, with the waste load calculated using the respective flows and *E. coli* concentrations at the Akkutuak Creek outlet. A constant water depth of 6 m was utilized, to represent the depth in which the drinking water intake pipe is located along the shoreline, according to the NWB water license for the community.

$$c = \frac{W}{\pi H E} e^{\frac{U_x x}{2E}} K_0 \left[r \sqrt{\frac{k}{E} + \left(\frac{U_x x}{2E} \right)^2} \right] \qquad [3.5.4]$$

Where:

W	=	Waste load [$m^3/d \cdot CFU/100\ mL$]
H	=	Water depth [m]
E	=	Diffusion coefficient [m^2/d]
U_x	=	Current velocity along shoreline [m/d]
x	=	Location parallel to shoreline [m]
K_0	=	Modified Bessel function of the second kind
k	=	Bacteria decay rate [d^{-1}]
r	=	Mixing radius [m]

3.6 Quantitative Microbial Risk Assessment (QMRA)

The Microsoft Excel plug-in, Oracle Crystal Ball, was utilized to conduct a stochastic QMRA model aimed at characterizing the risks presented to Qamani'tuaq residents by the current wastewater treatment system. This risk assessment followed the QMRA framework outlined by the WHO (2016), including the identification of potential hazards, exposure assessment, dose-response assessment and ultimately risk characterization and was an adaptation of the model developed by Daley et al. (2019) to represent several other Nunavut communities. Further, Daley et al. (2018) and Daley et al. (2019) were utilized as primary references when developing the model as they provided valuable knowledge specific to microbial risk assessment associated with wastewater treatment systems in Arctic Canada.

Site-specific data such as water quality data and community knowledge, were utilized to parameterize the model in addition to data sourced from peer-reviewed literature. Overall, a total of 30 scenarios were modelled based upon this framework; a scenario was developed for each exposure pathway identified below (6) in combination with each flow scenario previously discussed (5).

3.6.1 Hazard Identification

The study site utilizes primary wastewater treatment with no effluent disinfection, in which similar systems have been identified to have low levels of pathogen removal (Huang et al., 2017). Therefore, the associated microbial hazard source identified was the release of partially treated wastewater into the receiving environment. While there are numerous pathogens of concern commonly found within wastewater, the scope of this project specifically focused on six pathogenic agents all transmissible by fecal-oral routes (Daley et al., 2019). The pathogenic agents in this assessment followed the framework presented by Daley et al. (2019), and included pathogenic *E. coli*, *Salmonella* spp., *Campylobacter* spp., rotavirus, *Giardia* spp., and *Cryptosporidium* spp. Each of these agents have been identified as either a prevailing or an emerging pathogen of concern to Arctic Canadian populations as ingestion is known to cause AGI and other harmful diseases, even at low doses (Goldfarb et al., 2013; Daley et al., 2018; Pardhan-Ali et al., 2012; Huang et al., 2018).

3.6.2 Exposure Assessment

Hydrodynamic modelling was used to simulate *E. coli* concentrations (a common fecal indicator organism) at a number of exposure locations, following the five modelled loading scenarios. As the indicator organism, the simulated *E. coli* concentrations throughout the WTA and receiving environment were used to infer the levels of the 5 other pathogens of concern using common ratios suggested in peer-reviewed literature (Table 3.6.1). While most ratios were sourced with reference to wastewater, surface water, or drinking water, an inference ratio of indicator *E. coli* to pathogenic Salmonella was not available. Therefore, the ratio between non-pathogenic and pathogenic strains of Salmonella was used in the model. (Daley et al. 2019).

Table 3.6.1 *E. coli*-to-pathogen inference ratios for use in QMRA as referenced in published literature.

Pathogen	Ratio (*E. coli: Path*)	References
Pathogenic *E. coli*	1: 0.08	Haas et al. (1999); Howard et al. (2006)
Salmonella spp.	1: 0.01	Fuhrimann et al. (2016); Hynds et al. (2014); Shere et al. (2002); Soller et al. (2010)
Campylobacter spp.	1: 10^{-5}	WHO (2006a & 2006b)
Rotavirus	1: 10^{-5}	Fuhrimann et al. (2017); Katukiza et al. (2014)
Giardia spp.	1: 10^{-5}	Machdar et al. (2013) (*general protozoa ratio)*
Cryptosporidium spp.	1: 10^{-6}	Fuhrimann et al. (2017)

Community-based information was utilized to delineate the exposure pathways in this assessment, and to therefore understand the human-environment interactions associated with the WTA. Localized knowledge was an important aspect of this assessment, and was gained through community consultation and interviews, including community forums, site-mapping exercises, and site inspections/assessments. Community consultation presented an opportunity for the public to provide feedback on and validation of the potential exposure scenarios. Community consultation specific to this study included community forums at the local Northern Store in June 2018 and September 2019, in which input was gained from approximately 75 to 100 community members. During the forum, a map of the hamlet and surrounding area was displayed to present the opportunity for individuals to identify locations of personal and/or traditional significance, and locations frequently used for activities such as trails, camps, fishing and harvesting. Additionally, key informant input was received from approximately 14 community members, through Hamlet Council and Senior Administrative Officer meetings, and discussions with Hamlet Staff responsible for wastewater collection and drinking water treatment. These interactions were completed between June 2018 and September 2019 and provided invaluable information regarding community concerns of the existing system and feedback with respect to anticipated system upgrades or relocation in the future.

Information obtained throughout this step of the assessment identified 6 major exposure scenarios. GPS points were collected at each exposure location to ensure an accurate spatial relationship was used when characterizing the potential risk. For each of the exposure scenarios, the transmission route for pathogens is through the accidental or intentional ingestion of contaminated water. Literature-based assumptions regarding potential ingestion volumes were applied to the assessment, following triangular distributions (minimum; most likely; maximum) as suggested in the framework outlined by Daley et al. (2018). It should be noted that frequency of exposure and population were not considered for this study, as the goal was to identify the overall risk the system could pose to a community member if exposed, without specifically calculating the risk based on exposure frequency or community populations. Each exposure scenario is discussed in the following sections and a summary of the related model inputs, including ingestion volume, is provided in Table 3.6.2.

3.6.2.1 Solid Waste Site

The first exposure pathway identified was human interaction with the community's solid waste site. The potential for exposure to the pathogens of concern exists through this pathway when depositing or sorting through materials located within the solid waste site that may have come in contact with partially treated wastewater, or when walking onto the berm adjacent to the waste site which separates the treatment area and the inlet to Finger Lake (sample location FL-I, Figure 3.3.1). During the 2018 and 2019 spring melt periods, wastewater was observed to overflow and seep through the surface of this berm, while mixed snowmelt and wastewater was observed to overtop materials present within the solid waste site. Accidental exposure through this pathway was assumed to have a high-contact exposure rate.

3.6.2.2 Wetland Travel

The second exposure pathway identified was through wetland travel, which included traversing the area by foot, all-terrain vehicle (ATV), or snowmobile. Although residents tend to avoid the lagoon and immediate area, the area downstream hosted no signage or fencing to indicate that the area may be a potential hazard. Additionally, areas of fencing present around the treatment area were in disrepair, making the treatment area easily accessible; it appeared that the fencing in place was likely to prevent caribou and other animals, rather than community members. ATV tracks were observed near the FL-O sampling location (Figure 3.3.1) during both field seasons, an area that had highly variable water levels; flooding was observed during Spring 2019 while minimal flow was observed during Fall 2019. Accidental water ingestion could occur following contact with soil, vegetation, clothing, or equipment that has been contaminated with effluent. Additionally, all-terrain vehicles and snowmobiles will, as they traverse the wetland, spray soil particles, and create droplets of water, which may be inadvertently ingested by the vehicle riders. Similar to the solid waste site, a high-contact exposure rate was applied to this pathway.

3.6.2.3 Mine Road near Airplane Lake

The third exposure pathway identified was the mine road located adjacent to the inlet to Airplane Lake (sample location AL-I). In June 2018 it was observed that the culvert at this location was overflowing; similar observations were not made during the 2019 spring freshet. A large amount of ATV and vehicle traffic was observed on this road, and therefore it was recognized as potential high-contact exposure pathway.

3.6.2.4 Shoreline Recreation

Shoreline recreation was identified as a potential exposure route during the 2019 field season. Snowmobiles, sleds and equipment were observed along the edge of Baker Lake near the outlet of Akkutuak Creek (sample location BL-I). During the spring 2019 site visit, an ATV was observed travelling across the frozen lake, and over the slushy shore area onto a land path about 100 m east of the creek outlet. Through community consultation, conservation officers noted that fishing occurred along the Baker Lake shoreline in this general area in both the spring and fall. As accidental ingestion may occur in these areas, a low-contact exposure rate was applied to this source.

3.6.2.5 Wharf

The community wharf, located approximately 1500 m west of the outlet of Akkutuak Creek into Baker Lake, was located as an additional exposure route (1500m from BL-I). Similar to the shoreline recreation exposure pathway in nature, this location hosted snowmobiles, sleds, and other equipment. Residents were also observed on foot in this location. Accidental exposure from this source was assumed to follow a low-contact exposure rate.

3.6.2.6 Drinking Water Intake

The last exposure scenario taken into consideration was through drinking water collected from the municipal drinking water treatment plant located along the shoreline of Baker Lake, approximately 2 km west of the outlet of Akkutuak Creek (DW-I). Although effective treatment should eliminate this pathway as a concern, it was included in the assessment in the case that the mechanical treatment plant experienced operational issues or was taken offline. Additionally, it was noted that multiple boil water advisories were in place for the community between November 2018 and September 2020 due to elevated turbidity, which would impede disinfection.

Table 3.6.2 Summary of QMRA exposure assessment scenarios and model inputs.

Exposure Pathway	Parameter	Distribution	Value[a]	Reference
Solid Waste Site	*E. coli* Concentration (CFU/100 mL)	Point-estimate	1.27×10^{5}	FL-O
	Ingestion Volume (mL)	Triangular	10; 35; 50	Fuhrimann et al. 2016 & 2017; WHO 2006b
Wetland Travel	*E. coli* Concentration (CFU/100 mL)	Point-estimate	5.79×10^{4}	FL-I
	Ingestion Volume (mL)	Triangular	10; 35; 50	Fuhrimann et al. 2016 & 2017; WHO 2006b
Mine Road near Airplane Lake Road	*E. coli* Concentration (CFU/100 mL)	Point-estimate	1.70×10^{4}	AL-I
	Ingestion Volume (mL)	Triangular	10; 35; 50	Fuhrimann et al. 2016 & 2017; WHO 2006b
Shoreline Recreation	*E. coli* Concentration (CFU/100 mL)	Point-estimate	1.01×10^{4}	BL-I
	Ingestion Volume (mL)	Triangular	3.8; 7.6; 22.8	Dorevitch et al. 2011; McBride et al. 2013
Wharf	*E. coli* Concentration (CFU/100 mL)	Point-estimate	8.86×10^{-3}	1500 m from BL-I
	Ingestion Volume (mL)	Triangular	3.8; 7.6; 22.8	Dorevitch et al. 2011; McBride et al. 2013
Drinking Water Intake	*E. coli* Concentration (CFU/100 mL)	Point-estimate	1.84×10^{-3}	DW-I, 2000 m from BL-I
	Ingestion Volume (mL)	Point-estimate	1000	

[a] Simulated with hydrodynamic model based on 30-year average flow scenario, *E. coli* concentrations from other return periods are included in Table 4.2.2.

[b] Triangular distribution (minimum; most likely; maximum).

3.6.3 Dose-Response Assessment

The dose-response assessment stage of a QMRA defines the relationship between exposure to the identified hazard and the probability of a health outcome; probability of infection was selected as the target health outcome for this assessment. The first step in the dose-response assessment is determining the dose of *E. coli* ingested by a person through the exposure scenario considered ($D_{E.\ coli}$). The dose is calculated by multiplying the concentration of indicator *E. coli* measured or modelled in the environmental media at the location of exposure (C_{dist}) by the volume of water ingested (V), accidentally or deliberately per event (Equation 3.6.1).

$$D_{E.coli} = C_{dist} \times V \qquad [3.6.1]$$

The inference ratios defining the relationship between indicator *E. coli* and each pathogen of interest (Table 3.6.1) were then applied to obtain the dose of each specific pathogen of interest (d). Finally, to predict the probability of infection associated with human exposure to pathogens of concern, a mathematical function is applied. In QMRA, two specific functions have been established as widely applicable to most organisms; the single-parameter exponential function, and the two-parameter beta-Poisson (Haas et al., 2014). The probability of infection, $P(d)$, based on a single dose of exposure, d, is shown for the exponential and beta-Poisson models in Equations 3.6.2 and 3.6.3, respectively (Haas et al., 2014).

$$P(d) = 1 - e^{-rd} \qquad [3.6.2]$$

$$P(d) = 1 - \left[1 + \left(\frac{d}{N_{50}}\right) \cdot \left(2^{1/\alpha} - 1\right)\right]^{-\alpha} \qquad [3.6.3]$$

In the exponential function, e represents the base of the natural logarithm, and r is the probability that one organism survives to initiate the health outcome; r is pathogen specific.

The pathogen-specific variable within the beta-Poisson model are the slope parameter, α, and the median infectious dose, N_{50}. The dose-response models utilized for the pathogens of interest in this risk assessment and the pathogen-specific variables are consistent with the QMRA framework outlined by Daley et al. (2019) for similar systems in Nunavut (Table 3.6.3).

Table 3.6.3 Dose-response models and input variables.

Pathogen	Model	Parameters	References
Pathogenic *E. coli* (EIEC[a])	Beta-Poisson	$\alpha = 0.16$ $N_{50} = 2.11 \times 10^{6}$	CAMRA (2015); Dupont et al. (1971)
Salmonella spp.	Beta-Poisson	$\alpha = 0.389$ $N_{50} = 1.68 \times 10^{4}$	CAMRA (2015); McCullough and Eisele (1951)
Campylobacter spp.	Beta-Poisson	$\alpha = 0.14$ $N_{50} = 890.38$	Black et al. (1988); CAMRA (2015)
Rotavirus	Beta-Poisson	$\alpha = 0.253$ $N_{50} = 6.17$	CAMRA (2015); Ward et al. (1986)
Giardia spp.	Exponential	$r = 0.020$	CAMRA (2015); Rendtorff (1954)
Cryptosporidium spp.	Exponential	$r = 0.057$	CAMRA (2015); Messner et al. (2001)

[a] Enteroinvasive *E. coli*

3.6.4 Risk Characterization

The Microsoft Excel plug-in, Oracle Crystal Ball was used for the risk characterization stage of the QMRA. Monte Carlo simulations were run for each contaminant loading scenario, specific to each exposure scenario for a total of 30 individual model simulations. For each simulation, the data distributions and point-estimate values specified within the exposure and dose-response assessments were used as model inputs. Samples were repeatedly drawn for 10,000 iterations to model the probability of the desired health outcome (Haas et al. 2014). The health outcome selected for this risk assessment was the probability of infection, based upon a single exposure to a pathogen of concern. A graphical comparison of risk of infection was completed for each scenario through a comparison of the minimum, median, maximum, 25th and 75th percentile statistics. The contaminant loading and exposure scenarios were compared, to assess the risk associated with various activities.

Each scenario was compared to the suggested health target of an annual risk of infection of less than 1 in 10,000 (10^{-4}) persons (Regli et al., 1991). This health target was developed based upon exposure to waterborne pathogens through potable water, with the assumption that a single day constitutes a single exposure, and that 2 L of water is consumed per day. This tolerable risk level also aligns with that proposed by WHO for water related infectious disease (WHO, 2006a). As there are currently no specific QMRA health-based targets applied to wastewater discharges in Canada (Daley et al., 2020), an annual risk of infection of less than 10^{-4} was deemed acceptable for this study. Although the exposure pathways in the wastewater-impacted environment in Arctic Canada differ from that common to potable water, the defined health target was believed to be acceptable for this study as every exposure pathway considered was based upon the accidental ingestion of contaminated water. Overall, the health target of less than 10^{-4} was assumed to be conservative in nature for the exposure pathways and ingestion volumes considered within the exposure assessment stage and ultimately for this QMRA.

CHAPTER 4. RESULTS AND DISCUSSION

4.1 Water Quality

This portion of the study investigated a range of physical, chemical, and biological parameters at 6 primary sample locations within the study site: 1 reference location and 5 throughout and downstream of the WTA. Sample results were compared to the NWB water license criteria outlined for wastewater treatment in Qamani'tuaq, specifically those collected from the compliance location (Table 4.1.1). Additional guidelines, such as the CCME guidelines for the protection of aquatic life (CEQG) (2021), the CCME National Performance Standards outlined within the WSER (2009), and the Guidelines for Canadian Recreational Water Quality (2012), were included for regulatory comparatives. While only the most conservative values were included within Table 4.1.1, further discussion is included in the following sections.

Treatment performance of the current system was then assessed by determining the contaminant reductions (improvements) from the lagoon to the compliance point, and from the compliance point to the outlet of Akkutuak Creek (sampling location GC). The assessment was conducted for 4 sampling dates across 2 treatment seasons in order to demonstrate the impacts of varying hydro-meteorological conditions. The results are presented in Table 4.1.2.

Supplementary samples and *in-situ* water quality measurements, such as metals, dissolved oxygen, and pH were completed; although these data were not utilized in the overall treatment performance assessment, they are discussed in the following paragraphs.

It should be noted that when including non-detect samples in statistical analyses throughout this study, a value half of the minimum detection limit (MDL) was applied.

Table 4.1.1 Summary of WTA water quality results from 2018 and 2019, compared to associated regulations/guidelines.

	Regulation/Guideline		REF		LAG		FL-I[a]		FL-O		AL-I		GC	
Parameter	**NWB[b]**	**Other**	**Mean**	**Max**	**Mean**	**Max**	**Mean**	**Max**	**Mean**	**Max**	**Mean**	**Max**	**Mean**	**Max**
E. coli (CFU/100mL)	$1x10^{4}$	400[c]	4	10	$1x10^{6}$	$8x10^{6}$	$3x10^{3}$	$8x10^{4}$	$1x10^{3}$	$3x10^{4}$	920	$3x10^{4}$	67	$2x10^{3}$
CBOD5 (mg/L)	80	25[d]	1	1	247	396	31	110	15	52	8	21	4	6
TSS (mg/L)	100	25[d]	1	1	90	131	14	46	10	17	8	14	9	12
VSS (mg/L)			1	1	133	310	40	150	30	110	12	40	5	7
TAN (mg/L)		18[e]	0.01	0.01	50.20	75.70	15.11	50.00	8.97	31.00	6.41	20.00	0.83	1.24
TKN (mg/L)			0.56	0.69	65.85	96.90	16.95	53.00	9.68	30.00	7.65	21.00	1.94	2.26
Un-ionized NH3 (mg/L)		0.019[e]	0.01	0.01	0.15	0.36	0.05	0.15	0.04	0.15	0.02	0.06	0.01	0.01
TP (mg/L)			0.01	0.03	8.14	10.60	2.41	8.40	1.35	4.30	0.98	2.90	0.16	0.24

[a] Compliance point

[b] Values required by the NWB water license with respect to max. concentrations of any grab sample at the compliance point, guidelines specific to FC and BOD5

[c] Guidelines for Canadian Recreational Water Quality (2012), single sample maximum concentration

[d] CCME National Performance Standard for municipal wastewater effluent (CCME, 2009)

[e] CEQG Guidelines suggested for the Protection of Aquatic Life (CCME, 2021)

Table 4.1.2 Treatment performance reductions throughout WTA based on 2018 and 2019 water quality data.

		FL-I[a]				GC[b]			
Parameter	**Date**	**6/10/2018**	**6/13/2019**	**6/19/2019**	**9/9/2019**	**6/10/2018**	**6/13/2019**	**6/19/2019**	**9/9/2019**
E. coli (CFU/100mL)	Influent	$8x10^4$	$6x10^6$	$6x10^5$	$8x10^6$	$8x10^4$	5400	$3.4x10^4$	5
	Effluent	$8x10^4$	5400	$3.4x10^4$	5	99	1800	110	1
	Log Reduction	0.0	3.0	1.2	6.2	2.9	0.5	2.5	0.7
CBOD5 (mg/L)	Influent	110	197	283	396	110	1	10	3
	Effluent	110	1	10	3	NA	3	2	6
	% Reduction	0	99	96	99	NA	-200	80	-100
TSS (mg/L)	Influent	28	131	94	108	46	2	7	2
	Effluent	46	2	7	2	NA	NA	5	12
	% Reduction	-64	98	93	98	NA	NA	29	-500
TAN (mg/L)	Influent	28	43.2	53.9	75.7	50	0.4	3.1	6.9
	Effluent	50	0.4	3.1	6.9	NA	1.22	1.24	0.04
	% Reduction	-79	99	94	91	NA	-190	60	99
TP (mg/L)	Influent	6.1	7.5	8.37	10.6	8.4	0.06	0.5	0.7
	Effluent	8.4	0.06	0.5	0.7	NA	0.2	0.24	0.04
	% Reduction	-38	99	94	94	NA	-233	55	94

[a] Treatment performance at FL-I represents contaminant reduction between the lagoon samples and FL-I samples (compliance point)

[b] Treatment performance at GC represents contaminant reduction between FL-I samples and GC samples (Akkutuak before the inlet to BL)

[c] Influent values for FL-I represent concentration measured within the lagoon, influent samples for GC are the concentrations

4.1.1 Lagoon Effluent

The water quality parameters measured within the lagoon appeared to be elevated when compared to wetland influent samples from similar studies (Yates et al., 2012; Hayward et al., 2015). *E. coli* and CBOD5 were noted to be of specific concern, as significant reductions would be required in order to meet the objective outlined by the NWB water license. While these samples were collected within the lagoon, they were compared to WTA influent samples from similar studies as the lagoon holding cell was not engineered and did not host a decant or control structure. Additionally, during the 2019 spring field visit, the berm appeared to be breached and wastewater within the holding cell was observed to enter the WTA shortly after being released from the collection trucks. Taken together, these observations indicate that the small holding cell is providing minimal treatment.

4.1.2 Escherichia coli

E. coli is a common indicator organism, used to indicate the presence of human pathogens within municipal effluents. As it is used to identify the presence of fecal contamination, it generally considered a useful water quality parameter for the evaluation of municipal wastewater treatment systems. The *E. coli* concentrations observed throughout the wetland and receiving area ranged from an average of 1 x 10^6 CFU/100 mL within the lagoon to an average of 67 CFU/100 mL at the most downstream sampling location (GC), near the outlet of Akkutuak Creek (Table 4.1.1). Samples collected midway through the wetland at the compliance location (FL-I) averaged at approximately 3 x 10^3 CFU/100 mL, with the highest concentration at this location reported as 8 x 10^4 CFU/100 mL. These concentrations demonstrate a reduction of *E. coli* throughout the treatment system, and while the average concentration measured at FL-I met the NWB water license criteria, the maximum concentration at the compliance point exceeded the criteria of 1.0 x 10^4 CFU/100 mL outlined for the system. Additionally, the maximum concentration of the most downstream sample (2 x 10^3 CFU/100 mL) exceeded the Guidelines for Canadian Recreational Water Quality (2012) of 400 CFU/100 mL.

Treatment performance throughout the wetland from the lagoon to the compliance point was variable, with log reductions ranging from of 0 to 3.0 during the spring freshet, to a maximum log reduction of 6.2 in the late summer of 2019. While these reductions are comparable to those of other systems (Yates et al., 2012), seasonal variability in the reduction of *E. coli* throughout the system could be attributed to reduced hydraulic retention times (HRTs) during the spring freshet, when increased volumes of water enter the system due to snowmelt and surface runoff. Additionally, *E. coli* reductions between the compliance point and the inlet to Baker Lake were variable ranging from a log reduction of 0.5 on June 13, 2019 to 2.5 on June 19, 2019. The area between these two points consists of a shallow channel passing through a large body of water, potentially indicating dilution or other external sources impacting the levels of *E. coli* measured.

4.1.3 Five-Day Carbonaceous Oxygen Demand

The concentrations of cBOD5 represents the amount of oxygen required to break down carbon-based organic material in water over the course of 5 days, with high concentrations indicating less available oxygen for other forms of aquatic life, such as fish. In conventional systems, primary treatment generally reduces cBOD5 concentrations by 25-40%, while secondary treatment reduces concentrations to 25 mg/L or less (CCME, 2009). Based on the water quality results available, the treatment wetland obtained a 96-99% reduction during both the spring freshet and late summer of 2019, with concentrations ranging from 1 to 10 mg/L at the compliance location. These levels would indicate that secondary treatment was achieved with respect to cBOD5, and that both the guidelines set out by the NWB water license (80 mg/L) and the CCME NPS (25 mg/l) were met. It should be noted that the sample collected in June 2018 showed a 0% reduction with a concentration of 110 mg/L at the compliance point; while this would indicate that ineffective treatment was being achieved, this sample was collected during spring freshet when the sewage began melting and flowing throughout the largely frozen watershed. Ultimately, due to the uncontrolled lagoon, effluent entered and short-circuited the wetland treatment system.

4.1.4 Total Suspended Solids

Total suspended solids represent the amount of solid particles, both organic and inorganic, present within a water body. Increased levels of TSS can result in reduced solar transmissivity to a water column, negatively impacting both water treatment and plant production. Conventional wastewater treatment systems have been noted to reduce TSS by 50-70%, while secondary treatment is defined by concentrations of 25 mg/L or less (CCME, 2009). Although TSS is a common water quality parameter used to evaluate the treatment performance of conventional systems, it has proved to be challenging when considering natural wetland environments as disturbances to vegetation and natural water inputs within a wetland can cause significant variability within TSS values (Balch et al., 2018). TSS levels varied significantly between the lagoon and compliance point and between the compliance point and the watershed outlet, with reductions ranging from -64 to 98% and from -500 to 25%, respectively (with negative values representing increases). The variability in these values demonstrate that TSS may not be the most representative water quality parameter for use in evaluating the passive treatment system in Qamani'tuaq.

4.1.5 Ammonia

The water quality parameter total ammonia nitrogen (TAN) is a combination of ammonium (NH_{4+}) and un-ionized ammonia (NH_3), with the partitioning between these two forms a function of both water pH and temperature. Water with high pH and high TAN can be toxic to the receiving environment, with acute toxicity for many fish species reported when TAN concentrations are greater than 18 mg/L, based upon a water pH of 7.5 and a temperature of 15°C (CCME, 2021). More specifically, the Canadian water quality guidelines for NH_3 in freshwater systems is 0.019 mg/L (CCME, 2021). Samples collected from the compliance point complied with the criteria for TAN, with an average of approximately 15 mg/L; however, the maximum concentration recorded at this location exceeded the guidelines at approximately 50 mg/L. The NH_3 concentrations exceeded those recommended for freshwater aquatic life, with an average of 0.05 mg/L and a maximum of 0.15 mg/L. These trends continued downstream of the compliance point, with maximum TAN concentrations greater than 18 mg/L and un-ionized NH_3 concentrations averaging between 0.04 and 0.02 mg/L at sampling locations within the receiving environment (FL-O and AL-I, Figure 3.3.1). Although the NWB water license does not outline criteria specific to TAN or un-ionized NH_3, the levels reported would present risks to the freshwater aquatic life within the receiving environment.

While ammonia concentrations within the receiving environment exceeded some of the suggested guidelines, treatment performance during the 2019 treatment season appeared to be effective when compared to that of similar systems (84 – 99%), with treatment efficiency ranging from 91 to 99% between the lagoon and compliance point (Yates et al., 2012). Treatment downstream of the compliance point appeared to be below average (60 – 90%), with ammonia increases noted during the spring freshet of 2018 and 2019.

4.1.6 Phosphorus

Phosphorus is a limiting nutrient in many freshwater systems, and when excess levels are present eutrophication, and therefore plant and algal growth and oxygen depletion, may be accelerated (Riemersma et al., 2006). Although specific guidelines related to phosphorus levels are not outlined within the NWB water license or the CCME NPS, baseline phosphorus levels within the watershed ranged from an average of 0.01 mg/L to a maximum of 0.03 mg/L, as suggested by the reference sample (REF, Figure 3.3.1). These levels would indicate a mesotrophic to meso-eutrophic environment (CCME, 2021). Wastewater loading into the wetland appeared to increase the phosphorus levels significantly, with TP levels ranging from an average of 8.1 mg/L within the lagoon, to 2.4 mg/L at the compliance point and 0.2 mg/L at the watershed outlet. These levels indicate hyper-eutrophic states extending from the lagoon to the watershed outlet (or the inlet to Baker Lake). These levels of TP in the receiving environment can lead to depleted dissolved oxygen levels, making it difficult for aquatic organisms to survive and may result in fish kills. Phosphorus management practices may be required to reduce the levels of phosphorus entering the receiving environment.

While TP concentrations within the receiving environment were elevated, treatment efficiency between the lagoon and compliance point during the 2019 treatment ranged from 94 to 99%, and was considered to be effective when compared to the removal (80 – 99%)

demonstrated by similar systems (Yates et al., 2012). Treatment downstream of the compliance point (55 – 94%) appeared to be below average, with reduced TP removal noted during the spring freshet of 2018 and 2019.

4.1.7 Other Water Quality Parameters

Heavy metals were analyzed for a portion of the samples collected throughout the field monitoring program, and a single water quality sample collected from the solid waste site leachate which entered the system within Finger Lake immediately downstream of the compliance point (FL-I, Figure 3.3.1). These samples were collected to evaluate the metals entering the system and receiving environment through both municipal wastewater and the solid waste site. Overall, the metals present within the lagoon wastewater samples were increased when compared to background levels (REF, Figure 3.3.1), but comparable to concentrations recorded for raw wastewater in other Nunavut communities (Yates et al., 2012). The heavy metals within the leachate sample were comparable or lower than those reported in the lagoon samples, except for cadmium, nickel, and zinc which were elevated. While cadmium concentrations were elevated within the lagoon and leachate samples, they decreased immediately downstream and were below suggested guidelines (CCME, 2021). Copper and nickel concentrations continued to exceed the guidelines for freshwater aquatic life further downstream within the receiving environment. Metal results are included in Appendix C.

Dissolved oxygen (DO), pH, temperature, and specific conductivity measurements were taken concurrently with each water quality sample collected. Overall, the pH of the samples ranged from 6.2 to 7.7 at the compliance point, which was within the pH range specified by the NWB water license and was comparable to the pH range observed within the reference wetland (6.9-8.2). The DO measurements in the reference site were comparable to those recorded in Akkutuak Creek, while specific conductivity was slightly elevated downstream. Spatial variation throughout the treatment area was observed, with low DO levels and high specific conductivity noted within the lagoon and FL-I samples, with DO increasing and conductivity deceasing further from the lagoon inlet. Lastly, temporal differences were observed within the wetland with DO levels ranging from 6.0 to 11.0

mg/L at FL-I and specific conductivity ranging from 64 to 187 μS/cm at the same location. Discrete water quality measurements are included in Appendix D.

4.2 Hydrologic and Wetland Modelling

A hydrological model was developed to simulate the flow regimes of the watershed-wetland system in response to variable meteorological situations. In order to parameterize the model, bathymetry data was collected for two waterbodies within the watershed, Finger Lake and Airplane Lake (Appendix E), and bankfull channel geometry was surveyed and discharge was estimated at 3 specific locations of interest, FL-O, AL-I, and GC (Appendix F). Using this information, storage discharge relationships were developed (Appendix G).

4.2.1 Sensitivity Analysis

To determine which parameters were most influential on the hydrological model, a relative sensitivity analysis was conducted to rank the model parameters. The parameters were ranked according to their sensitivity to peak flowrate and total volume simulated at the watershed outlet, based upon a pre-determined parameter input range (Table 4.2.1).

Table 4.2.1 Relative sensitivities for HEC-HMS input parameters, including final values.

Parameter	**Calibrated Value[1]**	**Low Input**	**High Input**	**Sensitivity Ranking**	
				Peak Flow	**Total Volume**
Canopy - Max Storage (mm)	1.00	0.56	1.04	Neg	Medium
Surface - Max Storage (mm)*	12.2-13.1	7	13	High	Neg
SMA - Max Infiltration (mm/hr)*	1.01-1.10	0.63	1.17	High	Neg
SMA - Impervious (%)	95	52.5	97.5	High	High
SMA - Soil Storage (mm)*	20.0-21.8	14	26	High	Neg
SMA - Tension Storage (mm)*	8.89-9.78	7	13	High	Neg
SMA - Soil Percolation (mm/hr)*	0.68-0.77	0.63	1.17	High	Neg
SMA - GW1 Storage (mm)	5	3.5	6.5	Neg	Medium
SMA - GW1 Percolation (mm/hr)	0.9	0.63	1.17	Neg	Medium
SMA - GW1 Coefficient (hr)	10	7	13	Neg	Medium
SCS UH - Lag Time (min)*	5511-8178	5600	10400	High	Neg

Neg = negligible

[1]Range of calibrated values across all subbasins

Parameters ranked as negligible for both peak flow and total volume are not shown, and only parameters ranking high or very high were selected for optimization within the model. The total water volume within the model was most sensitive to impervious surface coverage; this parameter was manually adjusted within HEC-HMS as it is a parameter that is site-specific and should be measured or estimated based upon a site inspection. Although there was no development on the study site, the final impervious value was set to 95% in an attempt to represent both permafrost and the frozen nature of the watershed surface for a significant portion of the year.

The magnitude of the peak flowrate was most sensitive to the infiltration, percolation, and storage parameters within the soil storage layer of the soil moisture accounting module, in addition to the maximum available surface storage parameter. The sensitivity of the model to these parameters is consistent with the observed hydrology of the tundra wetland, in which surface runoff appeared to be the dominant flow mechanism. Additionally, subsurface flow within the wetland was assumed to be negligible during the treatment season for modelling purposes, as no obvious indicators of subsurface flow (i.e. seepage areas) were observed during the site visits; this was done by setting the impervious value to 95%. Additionally, the presence of a shallow active layer throughout the watershed area would result in reduced groundwater influence. Lastly, the timing of the peak flowrate was most sensitive to the SCS unit hydrograph lag-time, which represents the time difference between the centroid of net precipitation and the peak discharge at the watershed outlet (USACE, 2020).

4.2.2 Model Calibration and Validation

The HEC-HMS model was developed based upon a daily-time step, with the calibration period extending from January 1, 1982 to December 1, 1986 (Figure 4.2.1) and the validation period extending from June 14 to September 9, 2019 (Figure 4.2.2). The model was calibrated to historical hydrometric data available for Akkutuak Creek (or sampling location GC) using the automated simplex optimization model available within HEC-HMS. An additional manual calibration was completed in order to further improve the model, and specifically focused on the temperature index snowmelt parameters. Final model parameter values for all methods are included in Appendix B.

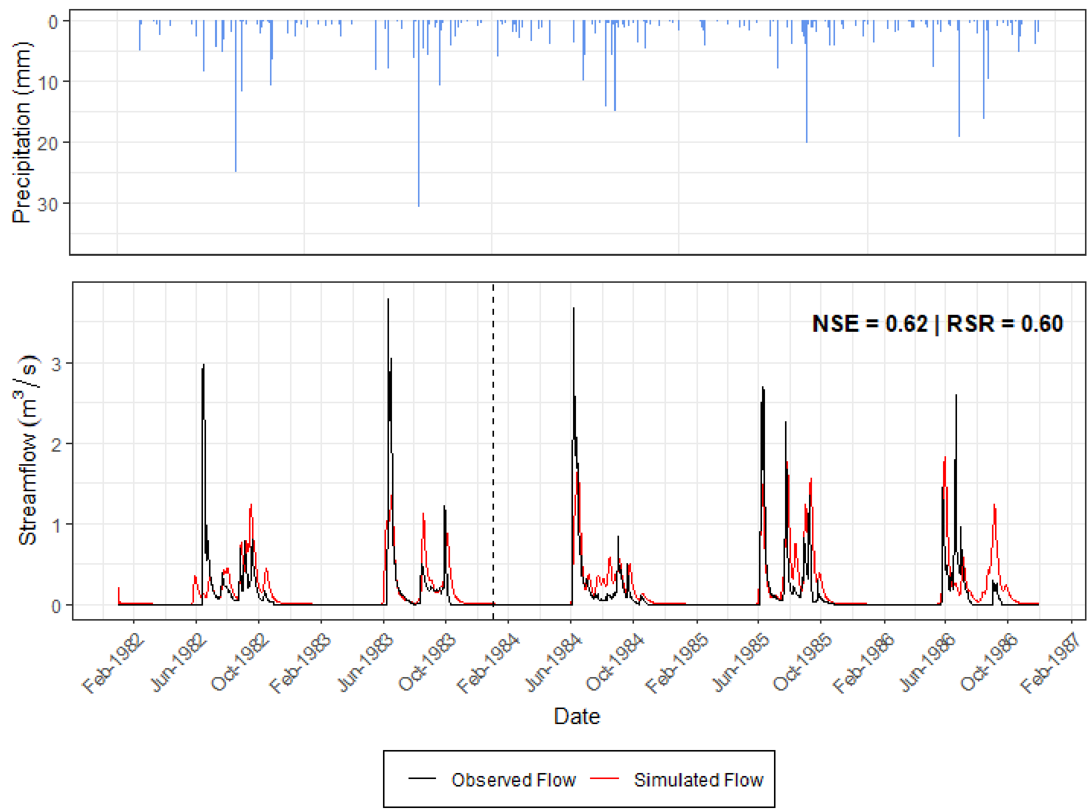

Figure 4.2.1 Hyeto-hydrograph at the outlet of Akkutuak Creek into Baker Lake for the calibration period (black dashed line represents the end of the warm-up period)

Overall, the calibrated model appeared to respond to precipitation events appropriately when compared to the observed hydrograph, with the key difference being the flow response to snowmelt during spring freshet (Fig 4.2.1), where discrepancies between observed and simulated were most pronounced. When considering the short validation period, the simulated peak flows appear to follow a similar response to precipitation events, with slightly higher peak flows observed (Figure 4.2.2). It should however be noted, that during the validation period the initial peak flow resulting from an influx of meltwater may not have been captured well during the field monitoring.

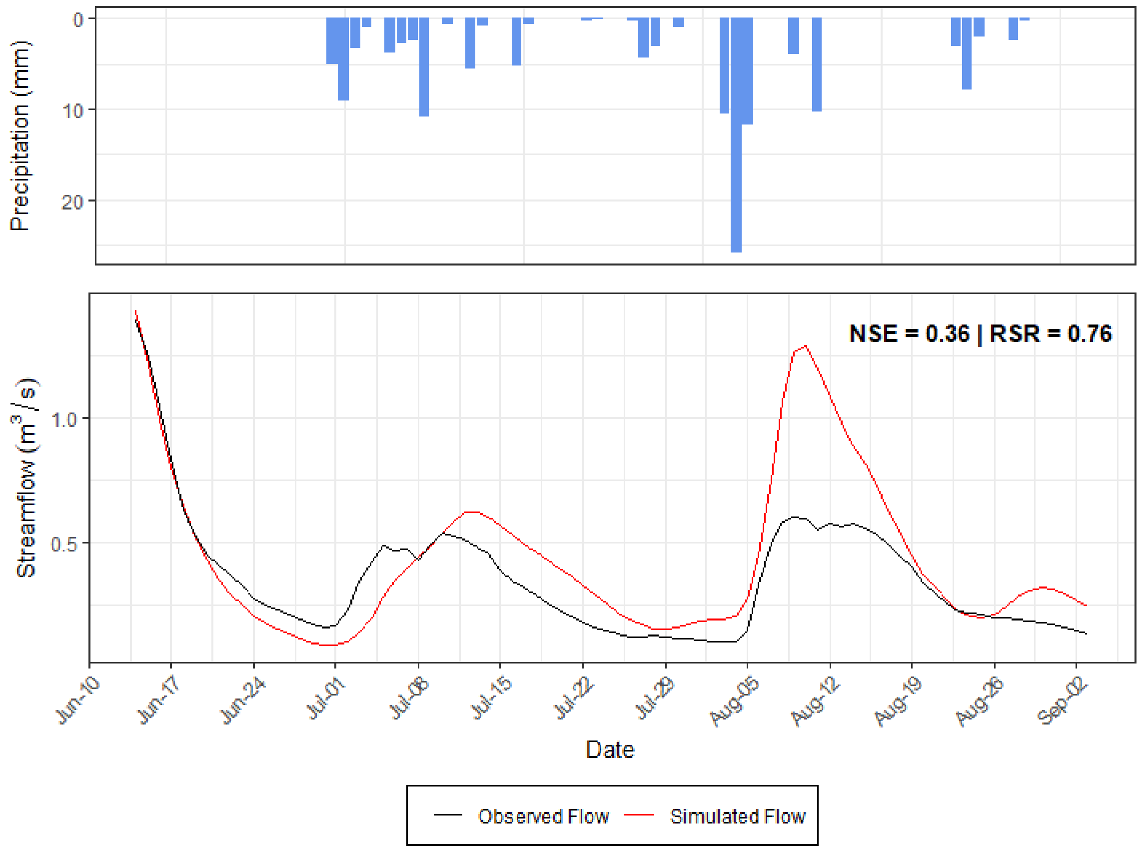

Figure 4.2.2 Hyeto-hydrograph at the outlet of Akkutuak Creek into Baker Lake for the 2019 treatment season and validation period (evaluation statistics calculated on a daily time-step)

In addition to a graphical comparison, calibration and validation goodness-of-fit metrics, NSE and RSR, were calculated to provide a quantitative comparison of the simulated and observed hydrographs. The model calibration was deemed satisfactory based on the evaluation statistics which were calculated on monthly average flowrates. The monthly NSE value of 0.62 exceeded the recommended value of 0.5 and the monthly RSR of 0.60 was below the recommended 0.70 (Moriasi et al., 2007). Due to the short extent of the validation period, evaluation statistics were calculated against daily values in which the model performed slightly poorer, with a mean daily NSE of 0.36 and a mean daily RSR of 0.76. Model performance was assumed to be acceptable as further data collection was not possible due to the COVID-19 pandemic travel restrictions; however, additional data collection would be recommended to better validate the model for use in future studies.

4.2.3 Hydrometric and Contaminant Loading Simulations

Once the model was deemed acceptable through the calibration and validation processes, a 30-year model simulation trial was run on a daily time step, extending from January 1, 1990 to December 31, 2019 (Figure 4.2.3).

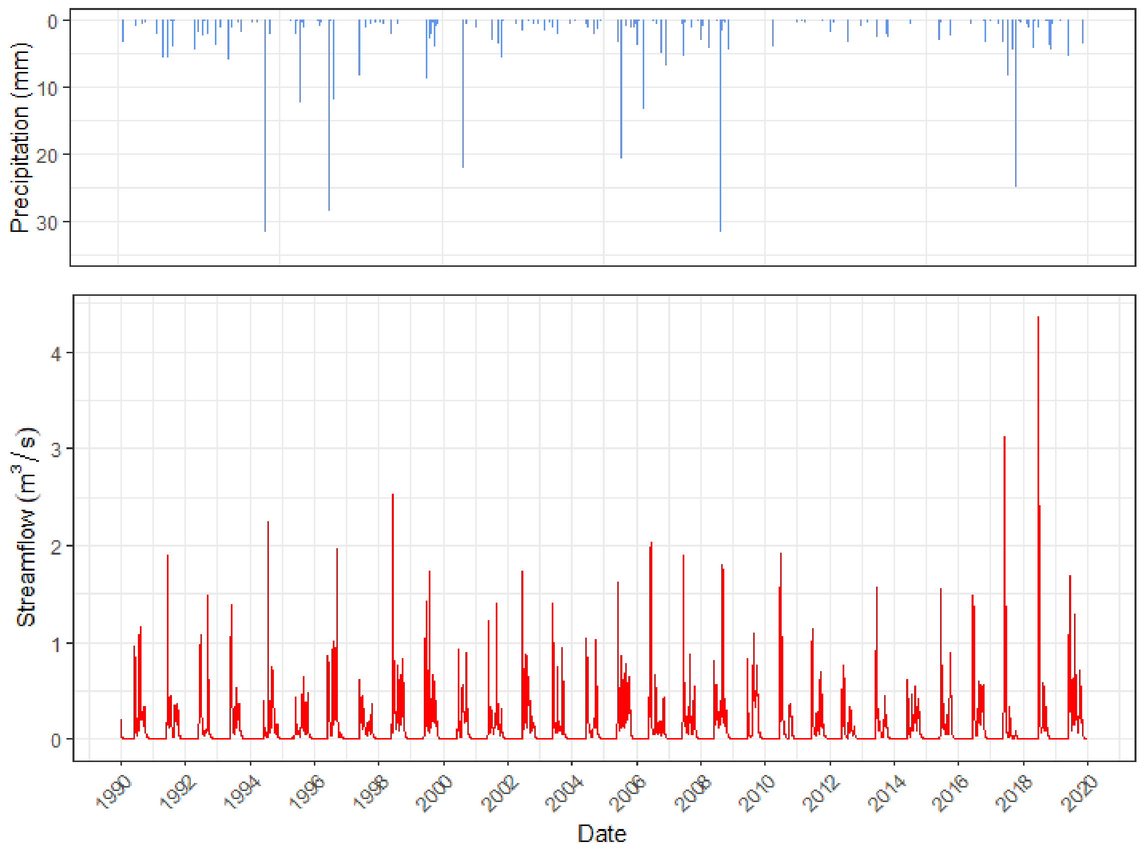

Figure 4.2.3 Simulated hyeto-hydrograph for outlet of Akkutuak Creek into Baker Lake

The watershed appeared to be snowmelt-driven, with annual peak flows consistently observed during spring freshet. The simulated hydrograph responded to increased precipitation, with a maximum flowrate of 4.36 m^3/s simulated on June 23, 2018. The annual cumulative precipitation for this year (2018) was recorded as 458.8 mm, a substantially higher precipitation depth when compared to other years between 1990 and 2019 in which annual cumulative precipitation ranged from 130.5 mm (2012) to 364.0 mm (2004).

The modelled flowrates throughout the simulation period were then extracted from HEC-HMS and applied to a contaminant fate and transport model for a number of loading scenarios. To develop the loading scenarios, a frequency analysis was completed in which the simulated annual maximum daily flow from each sampling location were fit to a lognormal probability distribution using EasyFit software. The probability distributions were then utilized to predict the maximum daily average flows for return periods ranging from 2 to 100 years. These maximum flow values were applied to an integrated TIS and plug flow reactor model, representing external hydrologic contributions, along with influent concentrations based upon the highest *E. coli* levels recorded within the lagoon and reference samples. Additionally, a 30-year mean flowrate (30-year average) was calculated for the treatment season across the simulation period for use as a more probable flow scenario. The integrated model was used to estimate *E. coli* concentrations throughout the treatment wetland and downstream watershed to the inlet of Baker Lake (Table 4.2.2).

Table 4.2.2 Modelled *E. coli* concentrations throughout WTA and recieving environment

	Modelled *E. coli* Concentration (CFU/100 mL)			
Return Period	**FL-I**	**FL-O**	**AL-I**	**BL-I**
30-year Average	$7.4\text{x}10^5$	$3.7\ \text{x}10^5$	1.2x105	$6.97\ \text{x}10^4$
2-year	$1.8\ \text{x}10^5$	$1.2\ \text{x}10^5$	$4.0\text{x}10^4$	$1.9\text{x}10^4$
10-year	$1.2\text{x}10^5$	$8.4\text{x}10^4$	$2.7\text{x}10^4$	$1.2\text{x}10^4$
25-year	$1.0\ \text{x}10^5$	$7.2\text{x}10^4$	$2.3\text{x}10^4$	$1.0\text{x}10^4$
50-year	$9.3\text{x}10^4$	$6.5\text{x}10^4$	$2.1\text{x}10^4$	$9.0\text{x}10^3$
100-year	$8.5\text{x}10^4$	$6.0\text{x}10^4$	$1.9\text{x}10^4$	$8.1\text{x}10^3$

The modelled *E. coli* concentrations specific to the locations of interest decreased with an increased influent flowrate, which can be seen by the maximum concentration of $7.4 \text{ x } 10^5$ CFU/100 mL at FL-I during the 30-year average scenario, compared to the $8.5 \text{ x } 10^4$ CFU/100 mL modelled during the 100-year return period for the same location. It should be noted that the model was utilized to simulate worst-case scenarios, as can be demonstrated by the maximum simulated *E. coli* concentration ($7.4 \text{ x } 10^5$ CFU/100 mL) at the FL-I sampling location compared to maximum observed *E. coli* concentration at the same location ($8.0 \text{ x } 10^4$ CFU/100 mL).

The 30-year average scenario, characterized by reduced watershed runoff when compared to the other scenarios modelled, resulted in a nominal retention time (HRT) was 1.4 days with external hydrologic contributions accounting for approximately 85% of the total flow through the system. Comparatively, the increased watershed runoff which characterized the 100-year return period resulted in a minimal retention time of 0.14 days or 3.5 hours. Although the reduced retention time would indicate less opportunity for treatment or degradation of *E. coli* within the WTA, the external hydrologic contributions account for 99% of the total flow into the system, indicating that dilution from external hydrologic contributions accounted for the majority of the contaminant reductions observed in this scenario. It should be noted that the nominal HRTs calculated within the model represent the theoretical maximum HRT; however, as wetlands are not 100% hydraulically efficient the actual HRTs would be shorter than the nominal values discussed.

Overall, the model indicated that the simulated *E. coli* reductions throughout the wetland during high flow scenarios, possibly comparable to those observed annually during spring freshet, could be attributed at least in part to dilution from external sources. Additionally, the nominal HRTs ranged from 1.4 to 0.14 days, which are significantly shorter than the minimal 14-day retention time specified by CSA standards (CSA, 2019). These short hydraulic retention times simulated by the model did not appear to provide adequate time for effective degradation of *E. coli* within the WTA prior to the compliance point. These factors may be contributing the exceedances observed within the water data collected throughout the 2018 and 2019 treatment seasons.

4.2.4 Advection-Dispersion Model

E. coli concentrations estimated at BL-I during the integrated modelling portion of this study were applied to a near-shore advection-dispersion model. This model was used to estimate the transport of *E. coli* along the Baker Lake shoreline between the outlet of Akkutuak Creek and the community's drinking water intake (Figure 4.2.4).

Disclaimer: *E. coli* concentrations away from shoreline may not be accurately represented in Figure 4.2.4.

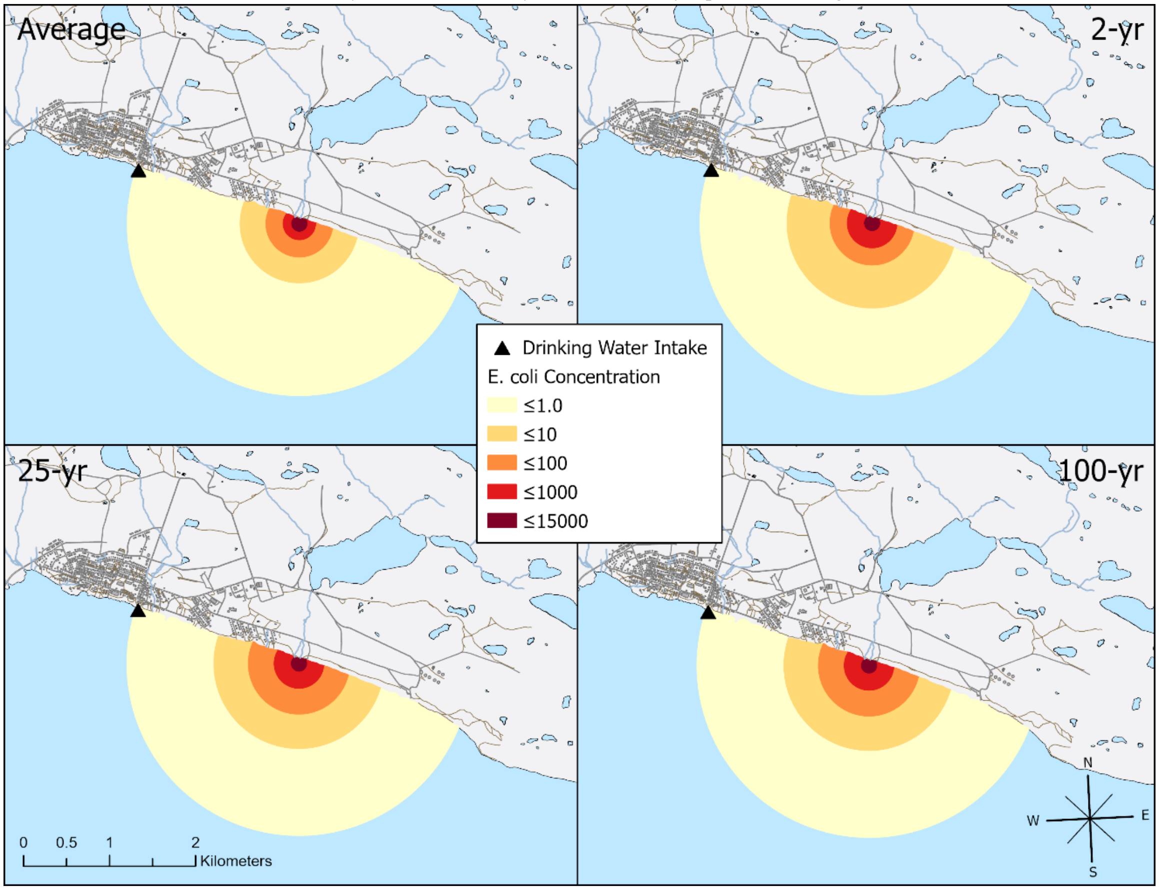

Figure 4.2.4 *E. coli* concentrations along Baker Lake shoreline between Akkutuak Creek and the drinking water intake based upon return period.

A small current moving parallel to the shore towards the drinking water intake, with a value of 0.01m/s, was applied to the advection-dispersion model to represent a conservative value. The low current selected for the model reduced diffusion along the shoreline, and therefore significantly increased the transport of *E. coli* towards the drinking water intake location. Even with the conservative assumption implemented, the projected *E. coli* concentrations reaching the drinking water intake from the wastewater treatment system were very low, with a modelled maximum concentration of less than 1 CFU/100 mL. Although the integrated model demonstrated the lowest concentration of *E. coli* entering Baker Lake during the 100-year return period simulation, the increased flowrate resulted in greater mass loading of *E. coli* overall and therefore higher levels further away from the point-source into the lake.

4.3 QMRA

Data collected for the completion of this risk assessment was provided by Qamani'tuaq residents on a voluntary basis through interviews, site-mapping, and public forums. These data were applied to the conventional risk assessment framework, along with site-specific physical, hydrological, and biogeochemical data. The local knowledge of activity patterns in wastewater-impacted environments was essential in the development of applicable exposure scenarios.

Of the 5 loading scenarios assessed, the 30-year average flow scenario presented the highest risk levels related to enteric pathogens through the exposure of wastewater associated with the WTA. Figure 4.3.1 utilizes a box-and-whisker plot to show the median, minimum, maximum, 25^{th} and 75^{th} percentile risk levels related to the 4 highest exposure scenarios and compared the probability of infection health target of 10^{-4} for this loading scenario. Box-and-whisker plots for the other loading scenarios are included in Appendix H. Of the six pathogens modeled, rotavirus and pathogenic *E. coli* were projected to pose the highest risk, followed by *Salmonella* spp., *Campylobacter* spp., *Giardia* spp., and *Cryptosporidium* spp. For two of the six exposure scenarios, the minimum probability of infection exceeded the proposed target of 10^{-4} as a maximum tolerable risk level, for all pathogens; these probabilities were seen for exposure at the solid waste site and through wetland travel. Both the wharf and drinking water intake exposure pathways demonstrated maximum risk levels that were less than 10^{-6} for all 6 pathogens of interest, therefore they were not included within the box-and-whisker plots.

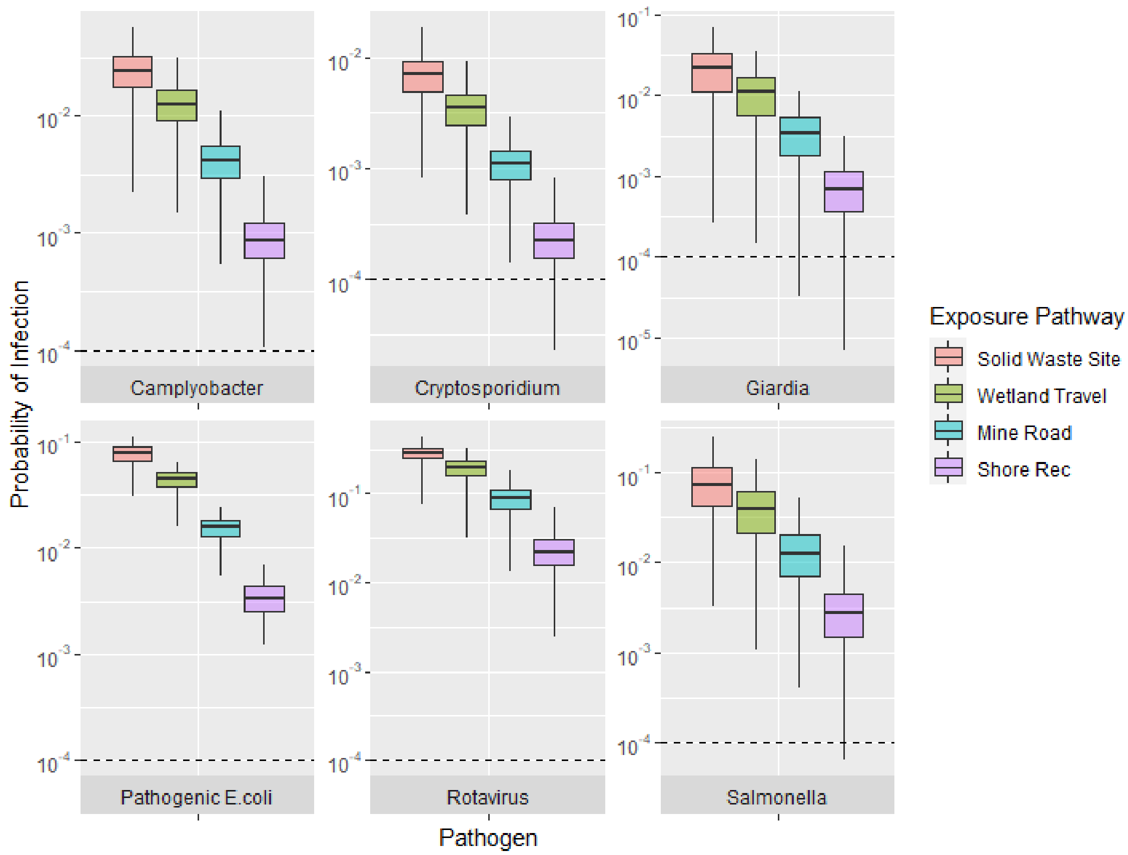

Figure 4.3.1 Box-and-whisker plot showing probability of infection caused by exposure to enteric pathogens through wastewater for a 30-year average flow scenario.

The results of this risk assessment identified 4 exposure scenarios common to Qamani'tuaq residents that may result in infection with a disease-causing pathogen at risk levels that are greater than those recommended (Regli et al., 1991; WHO, 2006a). The health target of an annual risk of infection of less than 10^{-4} was applied to the QMRA as more specific health-based targets were not available with respect to wastewater discharges in Canada (Daley et al., 2020).

These results demonstrated that exposure to pathogens through the solid waste site consistently posed the highest risk level across all scenarios modelled. Many residents were observed around this exposure location during each of the site visits completed, and although there is fencing surrounding the solid waste site, additional hazard mitigation may be warranted in this area. As well, the small roadway located at FL-I is fully accessible through the solid waste site or by tundra. Exposure through wetland travel posed the second highest risk levels, consistently. While exposure through this pathway was assessed specific to the FL-O sampling location, immediately downstream from the outlet of Finger Lake, additional risks may be present near the channel downstream of the WTA. There was no signage observed in this area indicating any potential hazards, or association with the treatment area, and site observations and conversation with residents identified the area as a commonly used path for travelling by both ATV and snowmobile. While residents tend to avoid the lagoon and fenced WTA, environmental interaction through these two pathways appeared to be frequent as there is minimal to no indication that they may be associated with or impacted by municipal wastewater.

Additionally, two of the exposure pathways located significantly further from the effluent source, the mine road near Airplane Lake and shoreline recreation along Baker Lake, continued to pose risks to the health of community members. These risk levels substantiate concerns made in other engineering assessments, such that increased adverse environmental and human health risks may be a result of undersized or un-engineered arctic wetland systems to increase (Hayward et al. 2018; Daley et al., 2019). Overall, the QMRA results show that the current wastewater management practices pose potential human health risks to the community.

4.4 Assumptions and Uncertainty

Due to the nature of this study, a number of assumptions were made in order to parameterize each of the models used and to overcome limitations introduced when representing the study watershed mathematically. Uncertainty is introduced with each assumption made, and through the collection of field data due to the discrete nature of sample collection. Assumptions and limitations specific to the hydrological model include:

- A stage-discharge relationship was determined for each sampling site using a line-of-best-fit and ensuring that an adequate coefficient of determination (R^2) was achieved. Limited flow data was available to develop these curves, which introduced uncertainty related to the continuous flowrates used for hydrologic model calibration.

- Within HEC-HMS, the impervious parameter was assumed to be 95% for all model subbasins in an attempt to represent the frozen state of watershed soils during the early spring and summer months. As a highly sensitive parameter, this may impact the model with ongoing climate change. Additionally, meteorological data applied to the model was collected from a weather station approximately 4.5km away and was assumed to be representative of the entire watershed. Specific data, such as snow accumulation could significantly impact external hydrologic contributions.
- Currently the only manner to simulate snowmelt hydrology within HEC-HMS is empirically, using a temperature index approach that includes a conceptual representation of the snowpack energy. Additionally, modelling of permafrost hydrology and snow interception is not available within HEC-HMS. Such hydrologic limitations increase model uncertainty, specifically as the study watershed receives over 50% of precipitation as snowfall, on average, and is located within a zone of continuous permafrost.
- Limited hydrometric data was available for model calibration and validation; calibration was based upon data from approximately 35 years ago, and validation was based only on 4-months of data collected in 2019.

Assumptions and limitations specific to the TIS and plug-flow reactor models include:

- The mass balance assumed evapotranspiration, precipitation, and infiltration to be negligible over the studied segments of the wetland, due to the short hydraulic retention time of surface flow in the wetland.
- Wastewater production was assumed to be equal to residential water use estimates.
- The rate constants and temperature correction coefficients were adopted from Hayward & Jamieson (2015) which introduces some uncertainty in the treatment performance estimations.

Assumptions and limitations specific to the advection-dispersion model include:

- The model utilized assumed that diffusion and advection were the sole transport mechanisms, and that Baker Lake was infinitely long and wide.

- A current moving parallel to the shore was estimated and remained consistent for up to 2 km away from the point source. This introduces uncertainty as model appeared to be highly sensitive to this parameter and field data could not be collected for parameterization of the model due travel restriction introduces as a result of the worldwide COVID-19 pandemic.

Assumptions and limitations specific to the QMRA include:

- The health risk assessment relied exclusively on *E. coli* as a fecal indicator organism and the use of pathogen inference ratios. Ideally, pathogen-specific data would be utilized to reduce uncertainty.
- Exposure and dose-response models were not specific to the presence of pathogens within Arctic environments.
- Ingestion volumes were estimated based upon values included within published literature.

While many uncertainties and limitations have been identified, this study was conducted to provide an increased understanding of the human health impacts associated with wastewater treatment in the community of Qamani'tuaq.

CHAPTER 5. CONCLUSIONS AND RECOMMENDATIONS

5.1 Conclusions

An integrated modelling and risk assessment approach was utilized to assess the performance of wastewater treatment in Qamani'tuaq, and to increase the understanding of associated impacts on human health in the community. The key objectives included (1) characterizing the hydrology and contaminant loading of the WTA, (2) developing a model to represent the WTA and surrounding watershed, and (3) utilize site-specific field and modelled data to conduct a QMRA aimed at evaluating the human health risks associated with the treatment area.

The modelling framework was comprised of a calibrated hydrological model (HEC-HMS) used to simulate daily flowrates and loading scenarios for a 30-year period across various hydro-meteorological conditions. The loading scenarios were coupled with a wetland model, using modified TIS and plug flow reactor methods, to predict *E. coli* concentrations throughout and downstream of the WTA. These site-specific concentrations were applied to a QMRA, along with community grounded knowledge to identify exposure risks. The microbial risk assessment was utilised to characterise the risk resulting from exposure to 6 pathogens of concern through pathways associated with the community's wastewater treatment area. Key findings of this study are outlined below.

Elevated water quality parameters, such as *E. coli* and cBOD5, were identified within the lagoon samples. This presents concerns to the treatment system due to the uncontrolled nature in which wastewater was observed to exit the lagoon holding cell. Site observations identified breaches within the berms during spring freshet, allowing raw wastewater to travel downslope and enter the WTA with no opportunity for settling or biodegradation, such that would occur in a typical lagoon or WSP. This may be one factor contributing to the exceedances identified at the compliance point.

The treatment performance assessment identified reductions in contaminants that were comparable to other systems for the majority of samples, while some concentration increases, and ineffective reductions were observed during spring freshet for both 2018 and 2019. Ineffective treatment performance during spring freshet is believed to be an indicator of system overloading during this time, potentially due to increased external hydrologic contributions resulting from snowmelt and surface runoff. Alternatively, ineffective treatment at this time could result from an influx of wastewater to the WTA, that had remained frozen throughout the winter and is rapidly melting and entering the system. These issues further demonstrate the need for either an engineered WSP or the addition of a decant structure to control the timing and duration of wastewater into the WTA.

The results of the hydrologic and integrated modelling approaches established *E. coli* concentrations throughout the watershed that suggested external hydrologic contributions had significant impact on the bacteria levels in the system. During increased loading scenarios, both the modelled HRTs and *E. coli* concentrations decreased throughout the system, indicating that dilution was playing a large part in the bacteria levels as opposed to biological degradation or treatment. Further, the modelled *E. coli* concentrations exceeded the water license criteria at the compliance point for all scenarios, and bacteria levels exceeding recreational water quality criteria were predicted in the receiving environment and Baker Lake, 1.8 km downstream from the WTA effluent.

Lastly, the results of the QMRA established risk levels greater than the recommended health target (10^{-4}) for 4 of the 6 exposure pathways. Rotavirus and pathogenic *E. coli* were projected to pose the highest risk to the community, followed by *Salmonella* spp., *Campylobacter* spp., *Giardia* spp., and *Cryptosporidium* spp. Risk characterization of the current wastewater treatment area identified unacceptable risk related to the probability of infection following exposure to the aforementioned pathogens.

Overall, both the water quality data collected throughout 2018 and 2019 treatment seasons, and the simulated data representing worst-case scenarios, provided evidence that the current wastewater treatment system does not consistently provide adequate treatment for the current municipal wastewater influent. In order to manage municipal wastewater for the community of Qamani'tuaq, mitigations may be required. Previous studies have demonstrated that engineered passive systems, incorporating controlled summer discharge schedules and risk communication messaging, are the most appropriate wastewater treatment options for Arctic communities (Daley et al., 2019).

5.2 Recommendations for Future Research

The limitations of this study have highlighted areas where future research is warranted; recommendations for future research include:

- Development of a more accurate integrated model through the collection of additional data. Due to travel restrictions, the data collection portion of this study was reduced and therefore model validation was limited. The collection of detailed bathymetry and near-shore current data within main water bodies, and additional streamflow data throughout the watershed would help in developing more accurate stage-discharge curves for use in model validation. If possible, model calibration to the compliance location would be ideal when analyzing results with respect to the NWB water license criteria.
- Further research should be conducted to understand what the most acceptable wastewater management options would be for the community and surrounding environment, regarding improvements to the current system. Mitigation options may include pre-treatment, system upgrades or relocation, or at a minimum signage to provide community knowledge of extent of environmental impacts from wastewater.
- The collection of additional site-specific frequency and exposure data could be collected and applied to the QMRA to further assess the risk related to the annual incidence of AGI or other related disease. Establishing probability of illness, annual incidence, or DALYS may prove to be more informative from a public health perspective.

REFERENCES

Allen, R. G., L.S. Pereira, D. Raes and M. Smith. (1998). Crop evapotranspiration – Guidelines for computing crop water requirements – *FAO Irrigation and drainage paper 56*. Published by FAO – Food and Agriculture Organization of the United Nations.

Balch, G., Hayward, J., Jamieson, R., Wootton, B., & Yates, C. N. (2018). Recommendations for the Use of Tundra Wetlands for Treatment of Municipal Wastewater in Canada's Far North. *Multifunctional Wetlands: Pollution Abatement and Other Ecological Services from Natural and Constructed Wetlands* (pp. 83–120). Springer International Publishing. https://doi.org/10.1007/978-3-319-67416-2_3

Bengtson, H. H. (n.d.). *Spreadsheet Use for Partially Full Pipe Flow Calculations.* CED Engineering. 36. Retrieved from: https://www.cedengineering.com/userfiles/Spreadsheet%20Use%20for%20Partially%20Full%20Pipe%20Flow%20Calculations.pdf [accessed 8 June 2020].

Bitton, G. (2005). *Wastewater Microbiology*. 3rd ed. Wiley, Hoboken, New Jersey, United States, p. 772.

Blaustein, R. A., Pachepsky, Y., Hill, R. L., Shelton, D. R., & Whelan, G. (2013). Escherichia coli survival in waters: Temperature dependence. *Water Research*, *47*(2), 569–578. https://doi.org/10.1016/j.watres.2012.10.027

Camp, T.R. (1946). Design of Sewers to Facilitate Flow. *Sewage Works Journal. 18*(3)

CAMRA [Center for Advancing Microbial Risk Assessment]. (2021a). *QMRA Framework – Hazard Identification.* http://qmrawiki.org/node/9 [accessed 27 April 2021].

CAMRA [Center for Advancing Microbial Risk Assessment]. (2021b). *QMRA Framework – Exposure Assessment.* http://qmrawiki.org/exposure-assessment [accessed 28 April 2021]

CAMRA [Center for Advancing Microbial Risk Assessment]. (2021c). *QMRA Framework – Risk Characterization, Uncertainty Propagation*. http://qmrawiki.org/content/uncertainty-propagation [accessed 30 April 2021].

CCME [Canadian Council of Ministers of the Environment]. (2021). *Canadian Environmental Quality Guidelines.* Available online from: https://ccme.ca/en/current-activities/canadian-environmental-quality-guidelines [accessed 27 April 2021].

CCME [Canadian Council of Ministers of the Environment]. (2009). *Canada-Wide Strategy for the Management of Municipal Wastewater Effluent*. Whitehorse, Yukon, Canada.

Chapra, S. C. (1997). Lecture 16: Lakes and Impoundments. In *Surface Water-Quality Modelling* (pp. 287-292). New York: McGraw-Hill.

Chouinard, A., Yates, C.N., Balch, G.C., Jørgensen, S.E., Wootton, B.C., Anderson, B.C. (2014). Management of tundra wastewater treatment wetlands within a lagoon/wetland hybridized treatment system using the subwet 2.0 wetland model. *Water 6*(3), 439–454. http://dx.doi.org/10.3390/w6030439

CSA (2019). Planning, design, operation, and maintenance of wastewater treatment in northern communities using lagoon and wetland systems. *CSA W203:19*. 72.

Daley, K., Jamieson, R., Rainham, D., Truelstrup Hansen, L., & Harper, S. L. (2019). Screening-level microbial risk assessment of acute gastrointestinal illness attributable to wastewater treatment systems in Nunavut, Canada. *Science of The Total Environment*, *657*, 1253–1264. https://doi.org/10.1016/j.scitotenv.2018.11.408

Daley, K., Jamieson, R., Rainham, D., & Truelstrup Hansen, L. (2018). Wastewater treatment and public health in Nunavut: a microbial risk assessment framework for the Canadian Arctic. Environmental *Science and Pollution Research, 25*(33), 32860–32872. https://doi.org/10.1007/s11356-017-8566-8

Daley, K., Castleden, H., Jamieson, R., Furgal, C., & Ell, L. (2015). Water systems, sanitation, and public health risks in remote communities: Inuit resident perspectives from the Canadian Arctic. Social *Science & Medicine, 135*, 124–132. https://doi.org/10.1016/j.socscimed.2015.04.017

Dingman, S. L. (2002). *Physical Hydrology 2nd Edition*. Waveland Press, Inc. Long Grove, Illinois, USA.

Dorevitch, S., Panthi, S., Huang, Y., Li, H., Michalek, A.M., Pratap, P., Wroblewski, M., Liu, L., Scheff, P.A., Li, A. (2011). Water ingestion during water recreation. *Water Research 45*(5), 2020-2028.

Fuhrimann, S., Nauta, M., Pham-Duc, P., Tram, N. T., Nguyen-Viet, H., Utzinger, J., Cissé, G., & Winkler, M. S. (2017). Disease burden due to gastrointestinal infections among people living along the major wastewater system in Hanoi, Vietnam. *Advances in Water Resources, 108*, 439–449. https://doi.org/10.1016/j.advwatres.2016.12.010

Fuhrimann, S., Winkler, M.S., Stalder, M., Niwagaba, C.B., Babu, M., Kabatereine, N., Halage, A.A., Utzinger, J., Cissé, G., Nauta, M. (2016). Disease burden due to gastrointestinal pathogens in a wastewater system in Kampala, Uganda. *Microbial Risk Analysis 4*, 16-28.

Golder [Golder Associates] (2013). *Hydrology Technical Support Document.* Hammond Reef Gold Project – Version 2. Retrieved from https://iaac-aeic.gc.ca/050/documents_staticpost/63174/97807/Hydrology_TSD_Version_2.pdf [accessed 10 June 2020].

Goldfarb, D.M., Dixon, B., Moldovan, I., Barrowman, N., Mattison, K., Zentner, C., Baikie, M., Bidawid, S., Chan, F., Slinger, R., 2013. Nanolitre real-time PCR detection of bacterial, parasitic, and viral agents from patients with diarrhoea in Nunavut, Canada. *Int. J. Circumpolar Health 72*, 19903. https://doi.org/10.3402/ijch.v72i0.19903.

Government of Canada (2019). *Open Government, High Resolution Digital Elevation Model (HRDEM) - CanElevation Series.* Retrieved from https://open.canada.ca/data/en/dataset/957782bf-847c-4644-a757-e383c0057995

Government of Canada (2018). *Canadian climate normal 1981-2010 station data*, *climate normal and averages.* Environment and natural resources. Retrieved from: http://climate.weather.gc.ca/climate_normals/ [accessed August 7, 2020].

Haan, C.T. (2002). *Statistical methods in hydrology*. Second Edition, Iowa State Press, 496 p.

Haas, C.N., Rose, J.B., Gerba, C.P., (2014). *Quantitative Microbial Risk Assessment. Second ed.* Wiley Publishing, Hoboken, New Jersey, USA.

Haas, C. R., Rose, J. B. & Gerba, C. P. (1999). *Quantitative Microbial Risk Assessment*. John Wiley and Sons, Inc., New York, USA.

Harmel, R. D., Smith, P. K., Migliaccio, K. W., Chaubey, I., Douglas-Mankin, K. R., Benham, B., Shukla, S., Muñoz-Carpena, R., & Robson, B. J. (2014). Evaluating, interpreting, and communicating performance of hydrologic/water quality models considering intended use: A review and recommendations. *Environmental Modelling & Software*, 57, 40–51. https://doi.org/10.1016/j.envsoft.2014.02.013

Harper, S. L., Edge, V. L., Ford, J., Thomas, M. K., Pearl, D. L., Shirley, J., & McEwen, S. A. (2015). Acute gastrointestinal illness in two Inuit communities: burden of illness in Rigolet and Iqaluit, Canada. *Epidemiology & Infection, 143*(14), 3048–3063. https://doi.org/10.1017/S0950268814003744

Harper, S. L., Edge, V. L., Schuster-Wallace, C. J., Berke, O., & McEwen, S. A. (2011). Weather, Water Quality and Infectious Gastrointestinal Illness in Two Inuit Communities in Nunatsiavut, Canada: Potential Implications for Climate Change. *EcoHealth, 8*(1), 93–108. https://doi.org/10.1007/s10393-011-0690-1

Hayward, J. L., Jackson, A. J., Yost, C. K., Truelstrup Hansen, L., & Jamieson, R. C. (2018). Fate of antibiotic resistance genes in two Arctic tundra wetlands impacted by municipal wastewater. *Science of The Total Environment, 642*, 1415–1428. https://doi.org/10.1016/j.scitotenv.2018.06.083

Hayward, J., Jamieson, R., Boutilier, L., Goulden, T., & Lam, B. (2014). Treatment performance assessment and hydrological characterization of an arctic tundra wetland receiving primary treated municipal wastewater. *Ecological Engineering*, *73,* 786–797. https://doi.org/10.1016/j.ecoleng.2014.09.107

Heinke, G. W., Smith, D. W., and Finch, G. R. (1991). Guidelines for the planning and design of wastewater lagoon systems in cold climates. *Canadian Journal of Civil Engineering, 18*(4): 556-567.

Howard, G., Pedley, S., Tibetemwa, S. (2006). Quantitative microbial risk assessment to estimate the health risks attributable to water supply: Can the technique be applied in developing countries with limited data? *Journal of Water and Health 4*(1), 49-65.

Huang, Y., Truelstrup Hansen, L., Ragush, C. M., & Jamieson, R. C. (2018). Disinfection and removal of human pathogenic bacteria in arctic waste stabilization ponds. *Environmental Science and Pollution Research, 25*(33), 32881–32893. https://doi.org/10.1007/s11356-017-8816-9

Hynds, P.D., Gill, L.W., Misstear, B.D. (2014). A quantitative risk assessment of Verotoxigenic E. coli (VTEC) in private groundwater sources in the Republic of Ireland. *Human and Ecological Risk Assessment 20*, 1446-1468.

Johnson, K., Prosko, G., & Lycon, D. (2014). *The challenge with mechanical wastewater systems in the Far North.* Conference proceeding paper at: Western Canada Water Conference and Exhibition. September 23-26, 2014. Regina, Saskatchewan.

Katukiza, A.Y., Ronteltap, M., Van der Steen, P., Foppen, J.W.A., Lens, P.N.L. (2014). Quantification of microbial risks to human health caused by waterborne viruses and bacteria in an urban slum. *Journal of Applied Microbiology 116*(2), 447-463.

Kozak, S., Petterson, S., McAlister, T., Jennison, I., Bagraith, S., & Roiko, A. (2020). Utility of QMRA to compare health risks associated with alternative urban sewer overflow management strategies. *Journal of Environmental Management, 262*, 110309. https://doi.org/10.1016/j.jenvman.2020.110309

Krumhansl, K. A., Krkosek, W. H., Greenwood, M., Ragush, C., Schmidt, J., Grant, J., Barrell, J., Lu, L., Lam, B., Gagnon, G. A., & Jamieson, R. C. (2015). Assessment of Arctic Community Wastewater Impacts on Marine Benthic Invertebrates. *Environmental Science & Technology, 49*(2), 760–766. https://doi.org/10.1021/es503330n

LaZerte, S., Albers, S. (2018). “weathercan: Download and format weather data from Environment and Climate Change Canada.” *The Journal of Open Source Software, 3*(22), 571. http://joss.theoj.org/papers/10.21105/joss.00571.

Leclerc, H., Schwartzbrod, L., & Dei-Cas, E. (2002). Microbial Agents Associated with Waterborne Diseases. *Critical Reviews in Microbiology, 28*(4), 371–409. https://doi.org/10.1080/1040-840291046768

Legates, D. R., and McCabe Jr., G.J. (1999) Evaluating the use of “goodness-of-fit” measure in hydrologic and hydroclimatic model validation. *Water Resources Research*, *35*(1), 233-241.

Lenhart, T., Eckhardt, K., Fohrer, N., & Frede, H.-G. (2002). Comparison of two different approaches of sensitivity analysis. *Physics and Chemistry of the Earth, Parts A/B/C, 27*(9–10), 645–654. https://doi.org/10.1016/S1474-7065(02)00049-9

Machdar, E., van der Steen, N.P., Raschid-Sally, L., Lens, P.N.L. (2013). Application of quantitative microbial risk assessment to analyze the public health risk from poor drinking water quality in a low income area in Accra, Ghana. *Science of the Total Environment 449*, 134-142.

Mathers, C. D., Stein, C., Fat, D. M., Rao, C., Inoue, M., Tomijima, N., Bernard, C., Lopez, A. D., & Murray, C. J. L. (2002). Global Burden of Disease 2000: Version 2 methods and results. *Global Programme on Evidence for Health Policy Discussion Paper No. 50*. Retrieved from https://www.who.int/healthinfo/paper50.pdf [accessed 22 April 2021].

McBride, G. B., Stott, R., Miller, W., Bambic, D., & Wuertz, S. (2013). Discharge-based QMRA for estimation of public health risks from exposure to stormwater-borne pathogens in recreational waters in the United States. *Water Research, 47*(14), 5282–5297. https://doi.org/10.1016/j.watres.2013.06.001

McBride, G. B., Stott, R., Papps, D., Palliser, C., Jenner, G., & Macdonald, G. (2012). Estimating health risks to water users: Marrying hydrodynamic models and risk models. *Water Practice and Technology, 7*(4). https://doi.org/10.2166/wpt.2012.078

McCarthy, D.T., Mitchell, V.G., Deletic, A., Diaper, C., (2007). Escherichia coli in urban stormwater: explaining their variability. *Water Science & Technology. 56*(11), 27–34.

Medeiros, A. S., Friel, C. E., Finkelstein, S. A., & Quinlan, R. (2012). A high resolution multi-proxy record of pronounced recent environmental change at Baker Lake, Nunavut. *Journal of Paleolimnology*, *47*(4), 661–676. https://doi.org/10.1007/s10933-012-9589-2

Moriasi, D. N., Arnold, J. G., Van Liew, M. W., Bingner, R. L., Harmel, R. D., & Veith, T. L. (2007). Model Evaluation Guidelines for Systematic Quantification of Accuracy in Watershed Simulations. *Transactions of the ASABE*, *50*(3), 885–900. https://doi.org/10.13031/2013.23153

Natural Resources Conservation Service (NRCS). (2007). *National Engineering Handbook: Chapter 16 Hydrographs*. Washington, DC.

NWB [Nunavut Water Board] (2021). *Sustaining the integrity of Nunavut's fresh water for the future.* Retrieved from http://www.nwb-oen.ca/ [accessed 22 April 2021]

Owens, C. E. L., Angles, M. L., Cox, P. T., Byleveld, P. M., Osborne, N. J., & Rahman, M. B. (2020). Implementation of quantitative microbial risk assessment (QMRA) for public drinking water supplies: Systematic review. *Water Research, 174,* 115614. https://doi.org/10.1016/j.watres.2020.115614

Pardhan-Ali, A., Wilson, J., Edge, V. L., Furgal, C., Reid-Smith, R., Santos, M., & McEwen, S. A. (2012). A descriptive analysis of notifiable gastrointestinal illness in the Northwest Territories, Canada, 1991–2008. *BMJ Open, 2*(4), e000732. https://doi.org/10.1136/bmjopen-2011-000732

Parkinson, A. J., Evengard, B., Semenza, J. C., Ogden, N., Børresen, M. L., Berner, J., Brubaker, M., Sjöstedt, A., Evander, M., Hondula, D. M., Menne, B., Pshenichnaya, N., Gounder, P., Larose, T., Revich, B., Hueffer, K., & Albihn, A. (2014). Climate change and infectious diseases in the Arctic: establishment of a circumpolar working group. *International Journal of Circumpolar Health, 73*(1), 25163. https://doi.org/10.3402/ijch.v73.25163

Ragush, C. M., Schmidt, J. J., Krkosek, W. H., Gagnon, G. A., Truelstrup-Hansen, L., & Jamieson, R. C. (2015). Performance of municipal waste stabilization ponds in the Canadian Arctic. *Ecological Engineering, 83,* 413–421. https://doi.org/10.1016/j.ecoleng.2015.07.008

Regli, S., Rose, J. B., Haas, C. N., & Gerba, C. P. (1991). Modeling the Risk From Giardia and Viruses in Drinking Water. *Journal (American Water Works Association), 83*(11), 76–84.

Riemersma, S., Little, J., Ontkean, G., and Moskal-Hébert, T. (2006). Phosphorus sources and sinks in watersheds: A review. 82 pp. *In Alberta Soil Phosphorus Limits Project. Volume 5: Background information and reviews.* Alberta Agriculture, Food and Rural Development, Lethbridge, Alberta, Canada.

Rose, J. B., Haas C. N., Guriam, P. L., Weir, M. H., Koopman, J., Eisenberg, J., Nicas, M., Shibata, T. (2013). *Theory and Practice of Quantitative Microbial Risk Assessment: An Introduction.* Center for Advancing Microbial Risk Assessment. Retrieved from http://qmrawiki.canr.msu.edu/images/6th_QMRA_Manual_2013.pdf [accessed 20 April 2021].

RStudio Team (2016). *RStudio: Integrated Development for R.* RStudio, Inc., Boston, MA URL http://www.rstudio.com/.

Shere, J.A., Kaspar, C.W., Bartlett, K.J., Linden, S.E., Norell, B., Francey, S., Schaefer, D.M. (2002). Shedding of Escherichia coli O157:H7 in dairy cattle housed in confined environment following waterborne inoculation. *Applied Environmental Microbiology 68*, 1947-1954.

Smith, D.W. and Emde, K.M.E. (1999). Effectiveness of wastewater lagoons in cold regions. *Biotechnological Applications of Cold-Adapted Organisms.* Springer, Berlin, Heidelberg. https://doi.org/10.1007/978-3-642-58607-1_16

Sokolova, E., Petterson, S. R., Dienus, O., Nyström, F., Lindgren, P.-E., & Pettersson, T. J. R. (2015). Microbial risk assessment of drinking water based on hydrodynamic modelling of pathogen concentrations in source water. *Science of The Total Environment, 526*, 177–186. https://doi.org/10.1016/j.scitotenv.2015.04.040

Soller, J., Embrey, M., Tuhela, L., Ichida, A., Rosen, J. (2010). Risk-based evaluation of Escherichia coli monitoring data from undisinfected drinking water. *Journal of Environmental Management 91*, 2329-2335.

Sturm, T. (2001). *Open channel hydraulics* (McGraw-Hill series in water resources and environmental engineering). Boston; Montreal: McGraw-Hill. Pg. 117

Taghipour, M., Shakibaeinia, A., Sylvestre, É., Tolouei, S., & Dorner, S. (2019). Microbial risk associated with CSOs upstream of drinking water sources in a transboundary river using hydrodynamic and water quality modeling. *Science of The Total Environment, 683*, 547–558. https://doi.org/10.1016/j.scitotenv.2019.05.130

Thomas, M. K., Murray, R., Flockhart, L., Pintar, K., Pollari, F., Fazil, A., Nesbitt, A., & Marshall, B. (2013). Estimates of the Burden of Foodborne Illness in Canada for 30 Specified Pathogens and Unspecified Agents, Circa 2006. *Foodborne Pathogens and Disease, 10*(7), 639–648. https://doi.org/10.1089/fpd.2012.1389

Tolouei, S., Dewey, R., Snodgrass, W. J., Edge, T. A., Andrews, R. C., Taghipour, M., Prévost, M., & Dorner, S. (2019). Assessing microbial risk through event-based pathogen loading and hydrodynamic modelling. *Science of The Total Environment, 693*, 133567. https://doi.org/10.1016/j.scitotenv.2019.07.373

United States Government, US Army Corps of Engineers (USACE) (2020). *Hydrologic Engineering Center Hydrologic Modeling System (HEC-HMS) User's Manual Version 4.5*. Retrieved from https://www.hec.usace.army.mil/software/hec-hms/documentation.aspx [accessed on 20 June 2020].

Westrell, T., Schonning, C., Stenstrom, T. A. & Ashbolt, N. J. (2004). QMRA (quantitative microbial risk assessment) and HACCP (hazard analysis and critical control points) for management of pathogens in wastewater and sewage sludge treatment and reuse. *Water Science & Technology 50*(2), 23–30. https://doi.org/10.2166/wst.2004.0079

WHO [World Health Organization]. (2016.) *Quantitative Microbial Risk Assessment: Application for Water Safety Management.* Geneva, Switzerland: WHO Press.

WHO [World Health Organization]. (2006a). *WHO guidelines for the safe use of wastewater, excreta and greywater. Volume 1.* Policy and regulatory aspects. Geneva, Switzerland: WHO Press.

WHO [World Health Organization]. (2006b). *Guidelines for the safe use of wastewater, excreta and greywater. Volume 2.* Wastewater use in agriculture. Geneva, Switzerland: WHO Press.

Yapo, R. I., Koné, B., Bonfoh, B., Cissé, G., Zinsstag, J., & Nguyen-Viet, H. (2014). Quantitative microbial risk assessment related to urban wastewater and lagoon water reuse in Abidjan, Côte d'Ivoire. *Journal of Water and Health, 12*(2), 301–309. https://doi.org/10.2166/wh.2013.051

Yates, C. N., Wootton, B. C., & Murphy, S. D. (2012). Performance assessment of arctic tundra municipal wastewater treatment wetlands through an arctic summer. *Ecological Engineering, 44,* 160–173. https://doi.org/10.1016/j.ecoleng.2012.04.011

YSI Incorporated. (2020) *6-Series Multiparameter Water Quality Sondes User Manual: Environmental Monitoring Systems Operations Manual Revision J*. Yellow Springs, Ohio, United States.

APPENDIX A: HEC-HMS DENDRITIC NETWORK

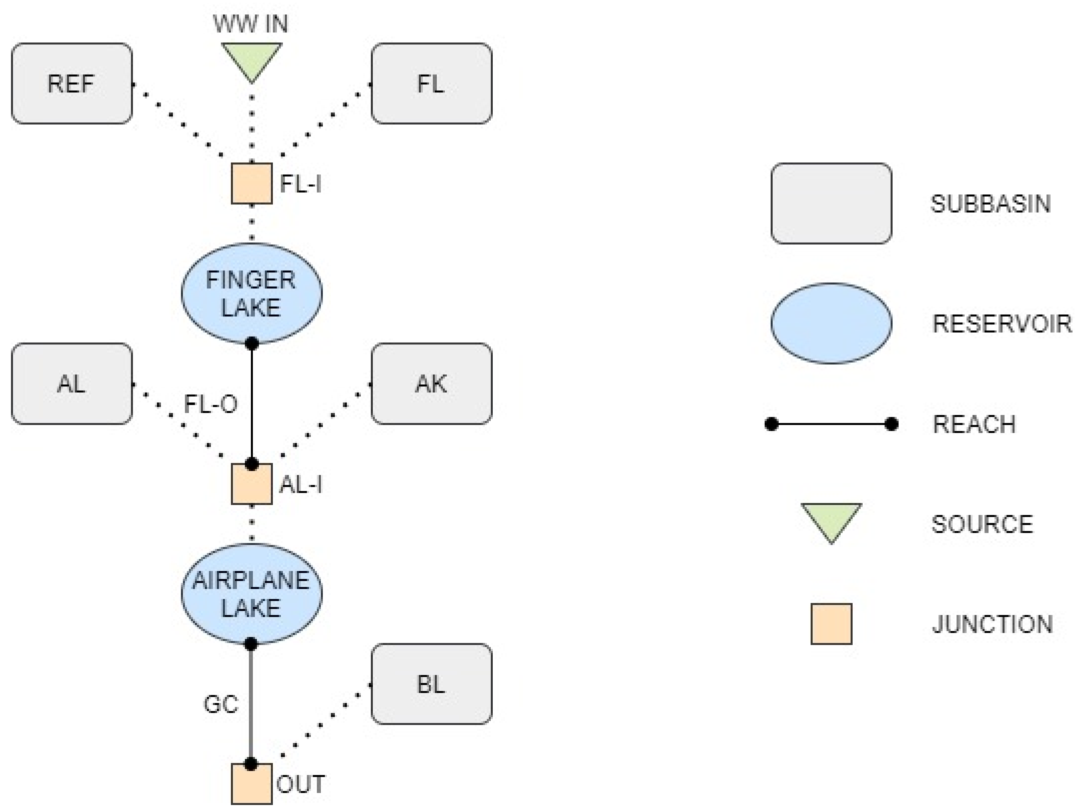

Figure A-1 Dendritic network representing Baker Lake watershed network.

APPENDIX B: HEC-HMS MODEL PARAMETERS

Table B-1 Canopy Storage – Input Parameters (Simple Canopy Method).

Subbasin ID	Initial Storage (%)	Max Storage (mm)
REF	0	1
FL-I	0	1
AL	0	1
FL-O	0	1
BL	0	1

Table B-2 Surface Storage – Input Parameters (Simple Surface Method)..

Subbasin ID	Initial Storage (%)	Max Storage (mm)
REF	0	13.1
FL-I	0	12.9
AL	0	12.4
FL-O	0	12.7
BL	0	12.2

Table B-3 Snowmelt – Input Parameters, ATI Meltrate Fucntion (Temperature Index Method).

ATI (^{o}C-day)	Meltrate (mm/^{o}C-day)
0.0	1.1
1.0	5.5
10.0	2.3
50.0	1.8
100.0	1.8
300.0	1.8

Table B-4 Infiltration and Percolation Losses – Input Parameters (Soil Moisture Accounting Loss Method).

			Soil Layer				Upper Groundwater Layer				Lower Groundwater Layer			
Subbasin ID	**Percent Impervious (%)**	**Maximum Infiltration Rate (mm/hr)**	**Initial Storage (%)**	**Total Storage (mm)**	**Tension Storage (mm)**	**Percolation Rate (mm/hr)**	**Initial Storage (%)**	**Total Storage (mm)**	**Coefficient**	**Percolation Rate (mm/hr)**	**Initial Storage (%)**	**Total Storage (mm)**	**Coefficient**	**Percolation Rate (mm/hr)**
REF	95	1.1	0	21.8	9.8	0.8	0	5	10	0.9	0	10	10	0.9
FL-I	95	1.1	0	21.3	9.6	0.7	0	5	10	0.9	0	10	10	0.9
AL	95	1.0	0	20.4	9.1	0.7	0	5	10	0.9	0	10	10	0.9
FL-O	95	1.1	0	20.9	9.3	0.7	0	5	10	0.9	0	10	10	0.9
BL	95	1.0	0	20.0	8.9	0.7	0	5	10	0.9	0	10	10	0.9

Table B-5 Surface Runoff – Input Parameters (SCS Unit Hydrograph Method).

Subbasin ID	**Lag Time (min)**
REF	6044
FL-I	5867
AL	5511
FL-O	5689
BL	8178

Table B-6 Baseflow – Input Parameters (Linear Reservoir Method).

Subbasin ID	Upper Groundwater Layer			Lower Groundwater Layer		
	Initial Baseflow (m^3/s)	Coefficient	Number of Reservoirs	Initial Baseflow (m^3/s)	Coefficient	Number of Reservoirs
REF	0.2	12	1	-	-	1
FL-I	0.2	12	1	-	-	1
AL	0.2	12	1	-	-	1
FL-O	0.2	12	1	-	-	1
BL	0.2	12	1	-	-	1

Table B-7 Reaches – Input (Muskingum-Cunge Routing Method).

Subbasin ID	Length (m)	Slope (m/m)	Manning's n	Index Flow (m^3/s)	Shape	Width (m)	Side Slope (xH:1V)
AL-I	245	0.006	0.04	1.07	Trapezoid	7.25	2.5
GC	875	0.015	0.07	0.93	Triangle	-	5.4

Table B-8 Snowmelt – Input Parameters, General (Temperature Index Method).

Parameter	Value
PX Temperature	0 ^{o}C
Base Temperature	-1 ^{o}C
ATI-Meltrate Coefficient	0.98
Wet Meltrate	3.3 mm/^{o}C-day
Rain Rate Limit	1 mm/day
Cold Limit	1.2 mm/day
ATI-Cold Rate Coefficient	0.99999
Water Capacity	3 %
Groundmelt	0 mm/day

APPENDIX C: METALS RESULTS

Table C-1 Metals results from water quality samples collected during the 2018 and 2019 treatment seasons, all values in mg/L.

	MDL[1]	CCME[2]	REF	LAG		FL-I		LEACH	FL-O	AL-I		GC
			2019 (n=2)	2018 (n=1)	2019 (n=2)	2018 (n=1)	2019 (n=1)	2019 (n=1)	2018 (n=1)	2018 (n=1)	2019 (n=1)	2019 (n=1)
Arsenic	**0.0005**	**5**	0	ND	1	ND	1	2	ND	ND	0	0
Cadmium	**0.00002**	**0.09**	0	ND	0.14	ND	0.01	0.49	ND	ND	0.01	0.03
Chromium	**0.0006**	**-**	1	ND	2	ND	2	1	ND	ND	1	1
Copper	**0.0005**	**2**	1	49	141	58	10	28	48	36	2	4
Nickel	**0.0005**	**25**	0	ND	4	ND	1	23	ND	ND	0	0
Lead	**0.0003**	**1**	0	ND	1	ND	0	0	ND	ND	0	0
Zinc	**0.001**	**30**	1	89	129	72	3	1290	45	31	1	4

[1] MDL = minimum detection limit

[2] CCME guidelines for the protection of freshwater aquatic life (2021).

APPENDIX D: DISCRETE WATER QUALITY RESULTS

Table D-1 Summary of discrete water quality measurements from 2019

Site	Parameter	Mean	Maximum	Minimum
REF	pH	7.35	8.16	6.90
	DO (mg/L)	13.21	13.99	12.17
	Conductivity (µS/cm)	52.80	68.00	38.00
	Temperature (^{o}C)	3.18	6.71	0.02
LAG	pH	6.93	7.28	6.51
	DO (mg/L)	3.17	4.50	2.38
	Conductivity (µS/cm)	847.33	1603.00	366.00
	Temperature (^{o}C)	5.69	8.85	1.40
FL-I	pH	6.97	7.70	6.20
	DO (mg/L)	10.95	13.55	6.40
	Conductivity (µS/cm)	98.10	187.00	64.00
	Temperature (^{o}C)	3.15	6.07	0.00
FL-O	pH	7.00	7.74	6.46
	DO (mg/L)	8.65	13.10	3.88
	Conductivity (µS/cm)	133.89	172.00	69.00
	Temperature (^{o}C)	3.35	5.32	0.00
AL-I	pH	6.96	7.30	6.50
	DO (mg/L)	10.22	13.20	7.77
	Conductivity (µS/cm)	137.63	172.00	69.00
	Temperature (^{o}C)	4.83	7.73	0.30
GC	pH	7.23	8.50	6.13
	DO (mg/L)	13.28	13.88	12.41
	Conductivity (µS/cm)	85.38	103.00	76.00
	Temperature (^{o}C)	3.72	5.98	0.50

APPENDIX E: BATHYMETRY MAPS

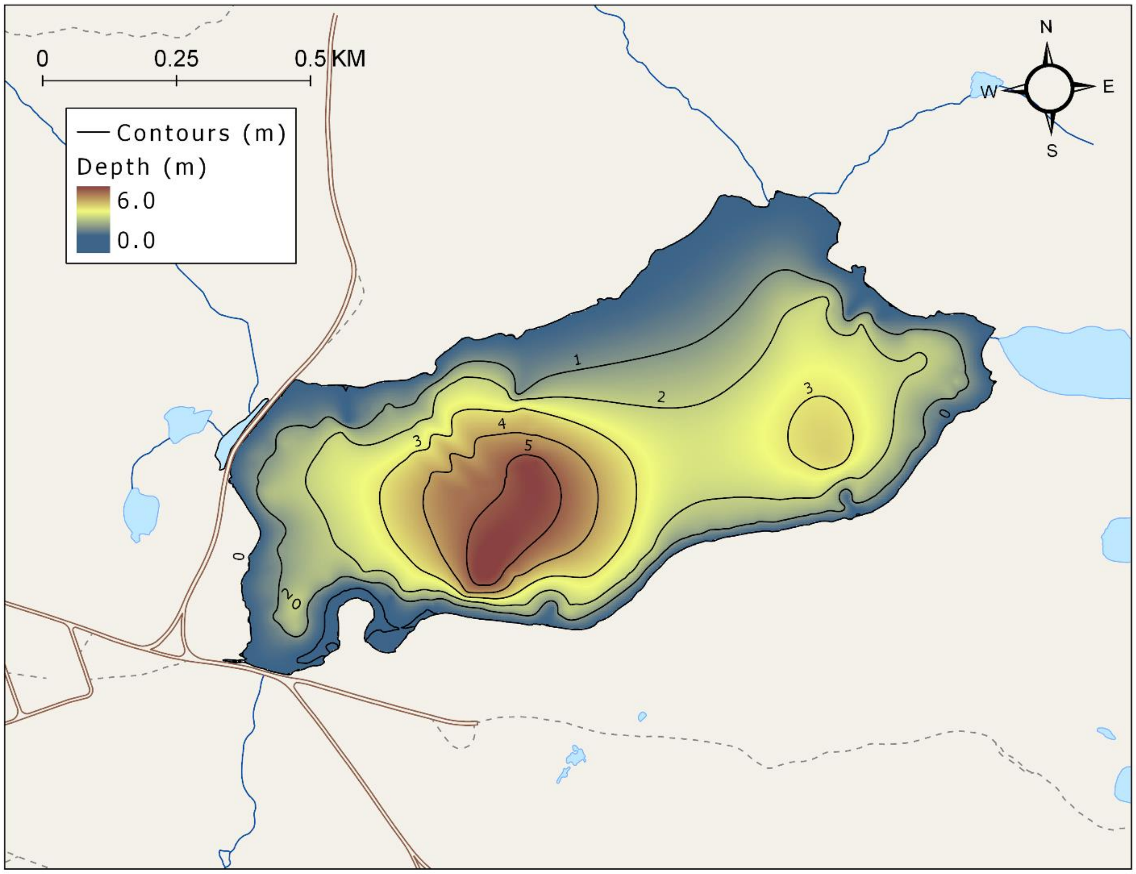

Figure E-1 Bathymetry map of Airplane Lake in Qamani'tuaq, NU.

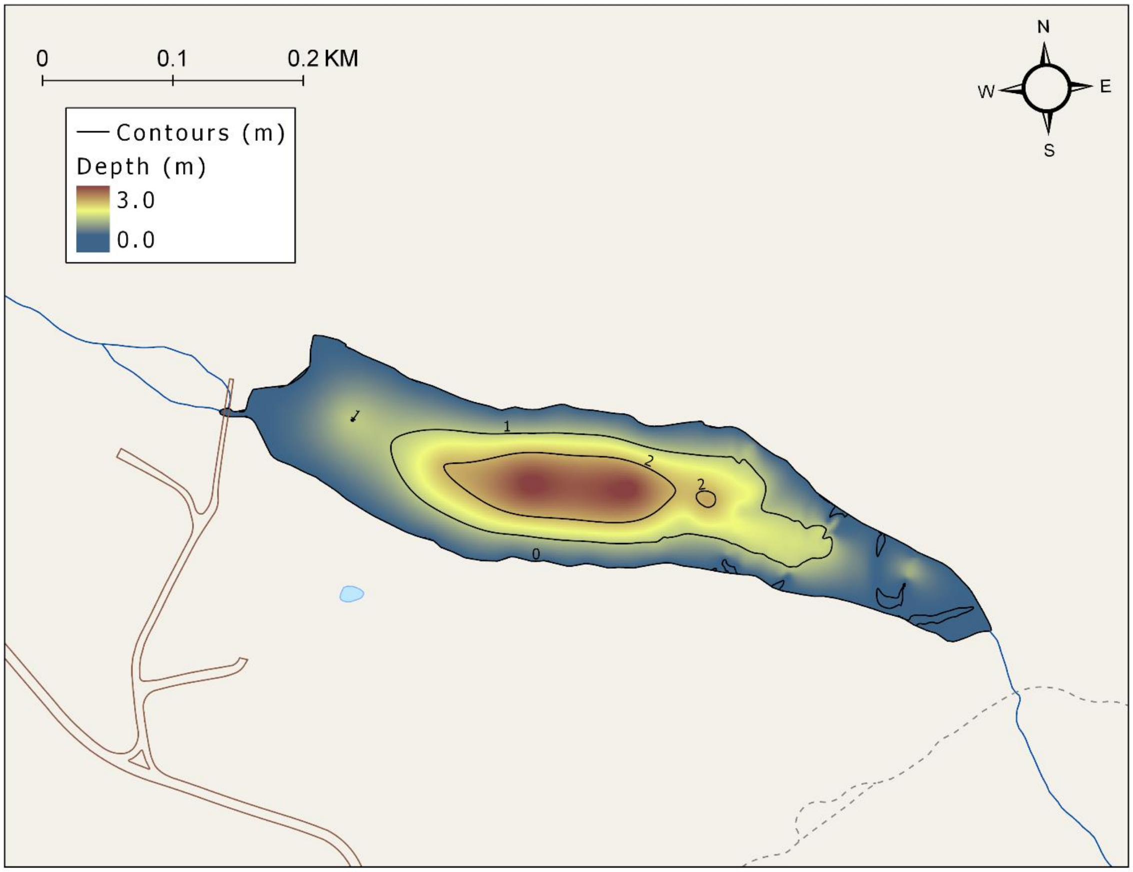

Figure E-2 Bathymetry map of Finger Lake in Qamani'tuaq, NU.

APPENDIX F: STORAGE-DISCHARGE RELATIONSHIPS

Table F-1 Finger Lake – Storage-Discharge Relationship

Storage (1000 m^3)	Discharge (m^3/s)
0	0.0
45	0.0
47	0.0
53	0.3
58	0.9
63	1.9
68	3.4
74	5.3
79	7.9
84	11.1
89	15.0
95	19.7
100	25.2
105	31.5
111	38.8
116	46.9
121	56.1
126	66.4

Table F-2 Airplane Lake – Storage-Discharge Relationship

Storage (1000 m^3)	Discharge (m^3/s)
0	0.0
1222	0.0
1281	0.0
1348	0.0
1415	0.2
1483	0.4
1550	0.6
1617	0.9
1685	1.2
1752	1.5
1820	1.9
1887	2.3
1954	2.8
2022	3.2
2089	3.7
2157	4.1
2224	4.6
2291	4.9
2359	5.2
2426	5.3

APPENDIX G: QMRA BOX-AND-WHISKER PLOTS

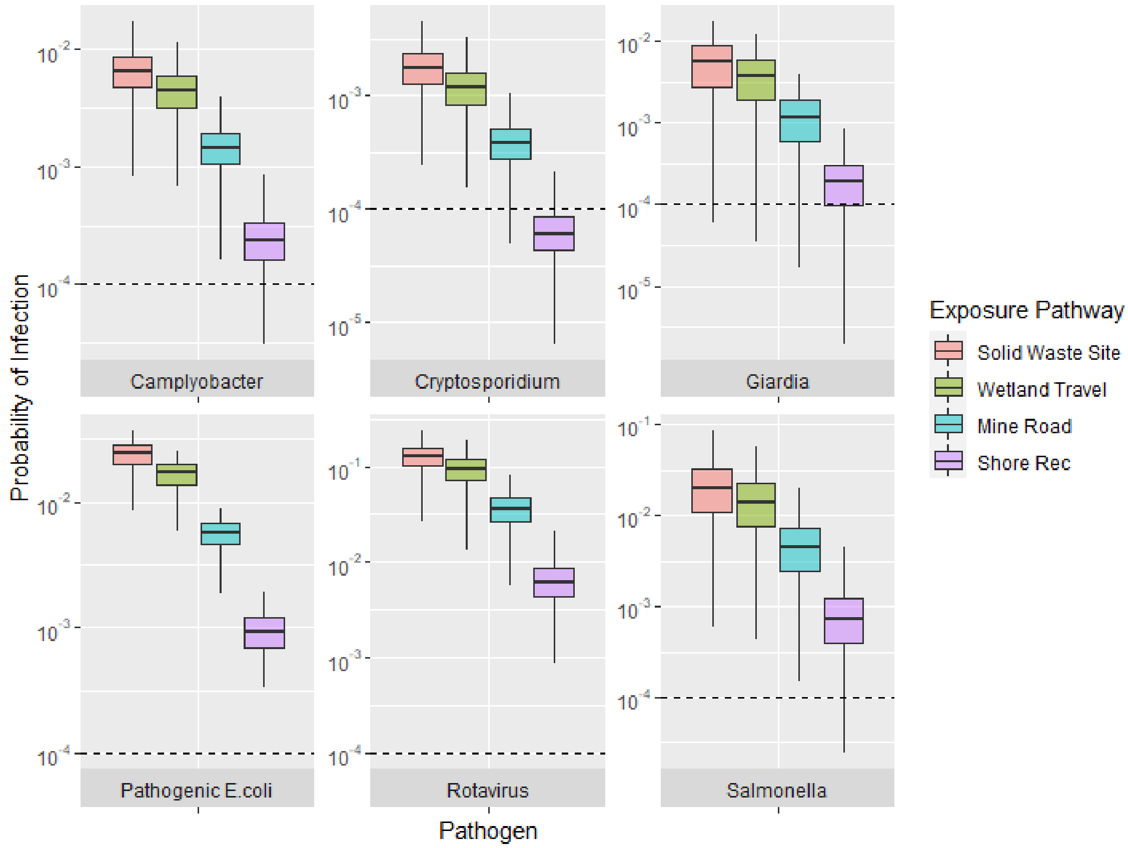

Figure G-1 Box-and-whisker plot showing probability of infection caused by exposure to enteric pathogens through wastewater for the 2-yr return period flow.

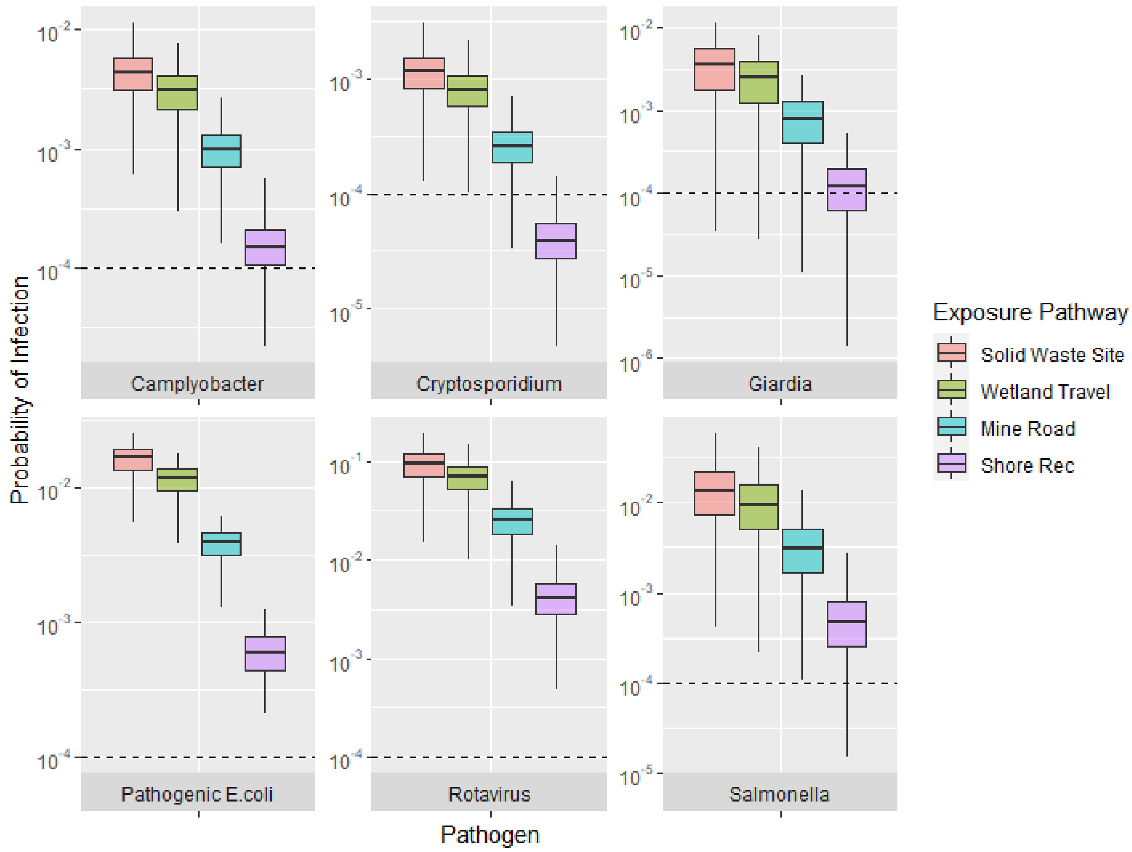

Figure G-2 Box-and-whisker plot showing probability of infection caused by exposure to enteric pathogens through wastewater for the 10-yr return period flow.

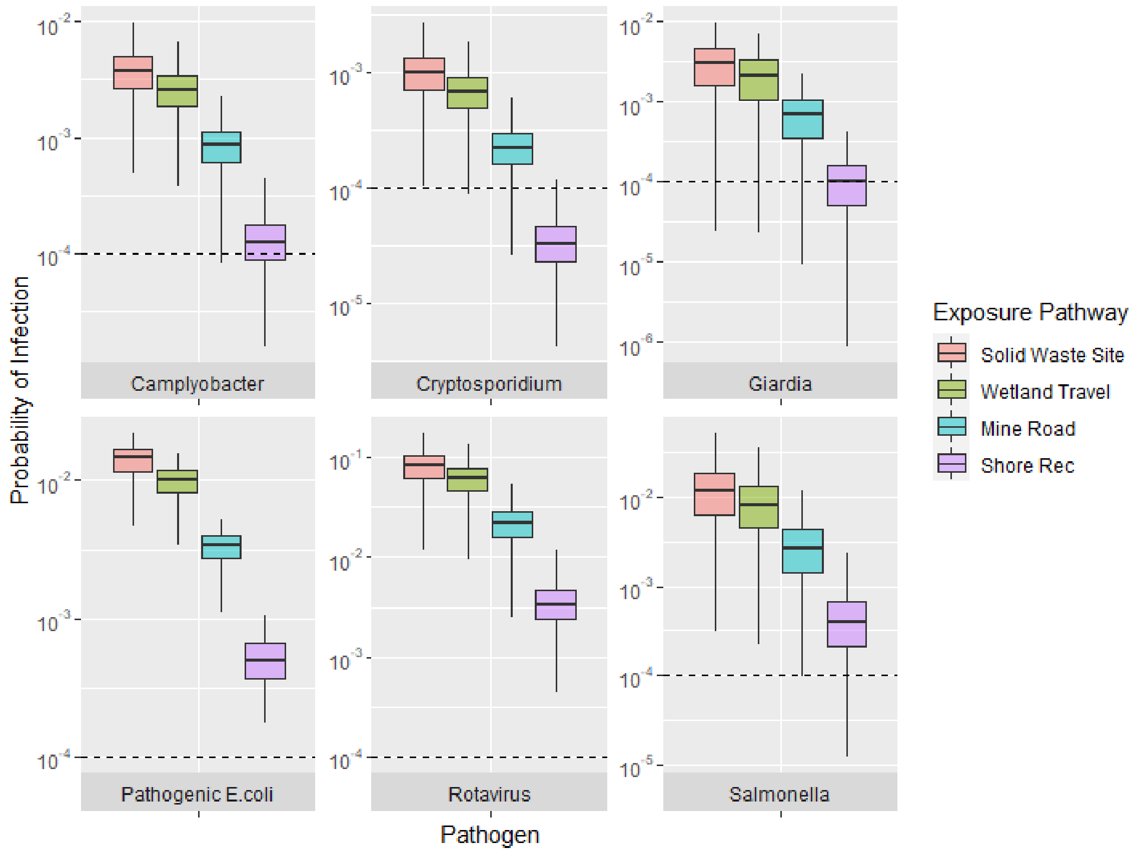

Figure G-3 Box-and-whisker plot showing probability of infection caused by exposure to enteric pathogens through wastewater for the 25-yr return period flow.

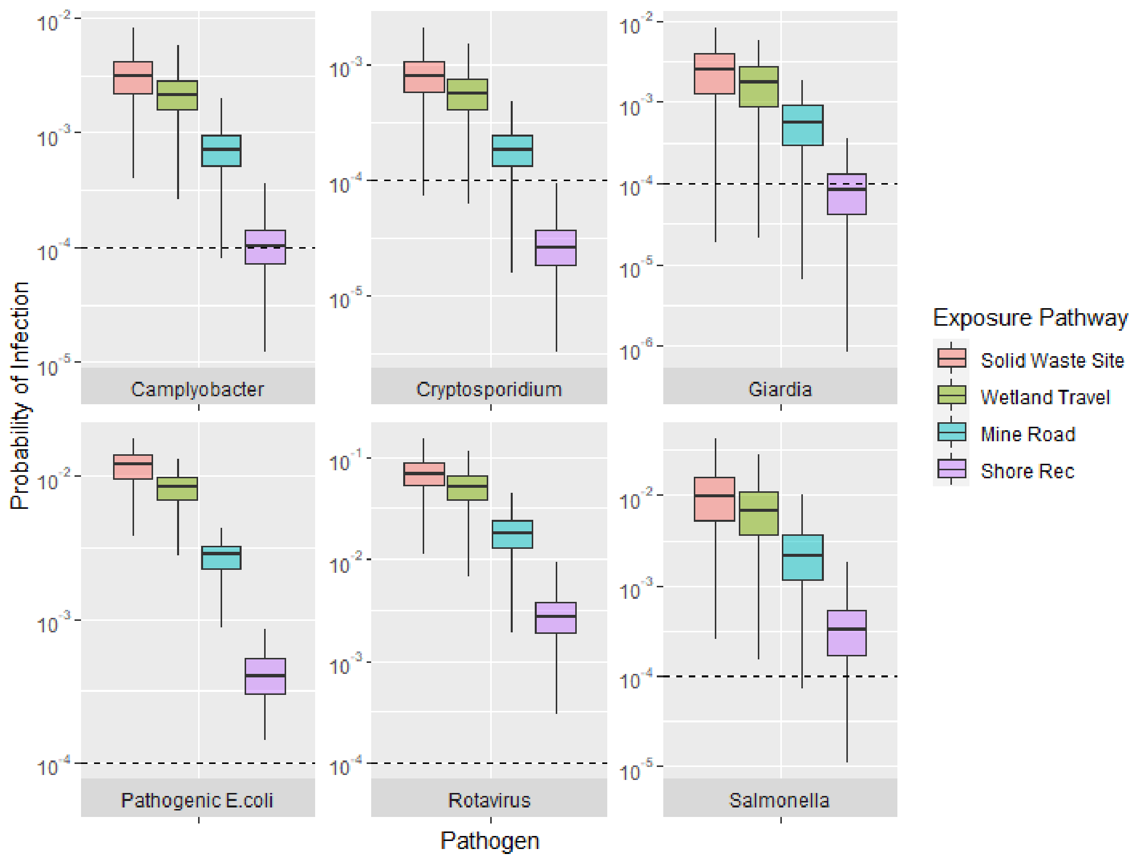

Figure G-4 Box-and-whisker plot showing probability of infection caused by exposure to enteric pathogens through wastewater for the 100-yr return period flow.