

**ASSESSMENT OF WATER QUALITY IMPACTS IN MARINE
ENVIRONMENTS RECEIVING MUNICIPAL WASTEWATER
EFFLUENT DISCHARGES IN NUNAVUT**

by

Mark Greenwood

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ABSTRACT

Implementing newly developed federal wastewater discharge criteria represents a challenge to Canada's northern jurisdictions. Prior to implementation, an understanding of the impacts associated with current discharge scenarios is desired in order to identify the risks associated with current wastewater management practices. The objectives of the research were to better understand mixing processes, define extents of Initial Mixing Zones, and identify human and environmental risks through the completion of tracer studies and water quality monitoring at three study sites throughout Nunavut. Differences in effluent transport and Initial Mixing Zone extents between studies highlighted the significance of ambient conditions, timing and rates of discharge, and effluent quality on environmental and potential human health risk. These findings provide insight into what factors to examine when assessing wastewater discharge scenarios in northern communities, and can help identify which systems may be at highest risk for impacts as a result of their current discharge regime.

LIST OF ABBREVIATIONS AND SYMBOLS USED

%	Percent
°C	Degrees Celsius
µg/L	Micrograms per Litre
µS/cm	Micro Siemens per Centimetre
AANDC	Aboriginal Affairs & Northern Development Canada
ASTM	American Society for Testing and Materials
CBOD ₅	Five-Day Carbonaceous Biochemical Oxygen Demand
CCME	Canadian Council of Ministers of the Environment
CFU/100 mL	Colony Forming Units per 100 mL
cm	Centimetre
CEQG	Canadian Environmental Quality Guideline
CSRS	Canadian Spatial Reference System
CWRS	Center for Water Resources Studies
DFO	Department of Fisheries and Oceans
DO	Dissolved Oxygen
E	Eastern Hemisphere
<i>E. coli</i>	<i>Escherichia coli</i>
<i>e.g.</i>	<i>Exempli gratia</i>
EC	Environment Canada
EDO	Effluent Discharge Objective
EQO	Environmental Quality Objective
ERA	Environmental Risk Assessment
<i>et al.</i>	<i>Et alii</i>
<i>etc.</i>	<i>Et cetera</i>
GN	Government of Nunavut
GPS	Global Positioning System

hr	Hour
HT	High Tide
<i>i.e.</i>	<i>Id est</i>
IMZ	Initial Mixing Zone
IT	Incoming Tide
kg	Kilogram
km	Kilometre
L	Litre
LT	Low Tide
m	Metre
m ²	Metres Squared
m ³	Metres Cubed
mg/L	Milligrams per Litre
mL	Millilitre
mm	Millimetre
MPN/100 mL	Most Probable Number of Colony Forming Units per 100 mL
MWWE	Municipal Wastewater Effluent
N	Northern Hemisphere
<i>n</i>	Number of Samples
NAD83	North American Datum 1983
NH ₃	Un-ionized Ammonia
NH ₃ -N	Un-ionized Ammonia Nitrogen
NH ₄	Ammonium
NH ₄ -N	Ammonium Nitrogen
NPS	National Performance Standards
NRI	Nunavut Research Institute
NU	Nunavut

NWB	Nunavut Water Board
NWT	Northwest Territories
OT	Outgoing Tide
pH	$-\log_{10}[H^+]$
Q	Discharge
RTK	Real-Time Kinematic
RWT	Rhodamine WT
SAO	Senior Administrative Officer
TAN	Total Ammonia Nitrogen
TC	Total Coliforms
TN	Total Nitrogen
TP	Total Phosphorus
TRC	Total Residual Chlorine
TSS	Total Suspended Solids
U.S. EPA	United States Environmental Protection Agency
USA	United States of America
UTM	Universal Transverse Mercator
W	Western Hemisphere
WSER	Wastewater Systems Effluent Regulations
WSP	Wastewater Stabilization Pond
WTA	Wetland Treatment Area
WWTP	Wastewater Treatment Plant

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CHAPTER 1. INTRODUCTION

In 2003, the Canadian Council of Ministers of the Environment (CCME) began efforts to develop a nationwide approach for the management of Municipal Wastewater Effluent (MWWE). This led to a framework for the development of the national strategy, the *Canada-Wide Strategy for the Management of Municipal Wastewater Effluent* (the *CCME Strategy*). The *CCME Strategy* focused on four main objectives: 1) Providing consistency between the regulatory frameworks of municipal, provincial/territorial, and federal jurisdictions, 2) Adoption of the use of environmental risk assessment in MWWE decision making, 3) The coordination of the collection, organization, interpretation, and dissemination of applicable scientific and engineering research and 4) Understanding the economical implications related to new criteria (Hydromantis Inc. et al., 2005; Strategic Alternatives et al., 2006). The *CCME Strategy* was completed in 2009.

In 2012, National Performance Standards (NPS) were introduced by Environment Canada (EC) to harmonize the nation-wide treatment requirements for municipal wastewater (Government of Canada, 2015; CCME 2009). The EC Wastewater Systems Effluent Regulations (WSER) stipulate that all wastewater treatment facilities which collect greater than 100 m³/d must comply with discharge quality objectives of 25 mg/L for Five-day Carbonaceous Oxygen Demand (CBOD₅) and Total Suspended Solids (TSS), and 1.25 mg/L for Un-ionized Ammonia Nitrogen (NH₃-N). In recognition of the unique challenges associated with wastewater treatment in Canada's Northern provinces and territories, a grace period was granted to the Northwest Territories, Nunavut, and above the 54th parallel in Quebec and Newfoundland and Labrador to facilitate research on northern treatment facilities. The resulting research is meant to inform the development of regulations specifically for the Northern provinces and territories.

In cooperation with the Government of Nunavut (GN), Dalhousie University is conducting research to assess and optimize MWWE discharge systems to minimize environmental impacts in arctic communities. As part of this project, an understanding of the existing effects of MWWE discharges on marine receiving water environments is

required. This specific component of the research program focuses on characterizing the mixing processes and assimilative capacity within the arctic surface water systems receiving these MWWWE discharges. This research was undertaken in three Nunavut communities that possess varying discharge scenarios.

The work presented herein is ultimately meant to support the development of a framework for environmental risk assessment of municipal wastewater systems in Nunavut. There are unique conditions in the Far North that should be considered within the standardized framework used to assess overall human and environmental risks associated with MWWWE discharges and to specify regulatory discharge criteria.

1.1 WASTEWATER TREATMENT IN NUNAVUT BACKGROUND

The significant differences in climates between the territories that occupy the Northern regions of Canada and those in the South (e.g., South of 60° latitude) bring significant changes to how MWWWE is managed. The approaches followed for the collection, transportation, treatment, and disposal all can differ greatly when compared against their southern counterparts.

Due to the aforementioned climate differences, the treatment mechanisms that are widely implemented in southern communities that rely on more moderate temperatures (e.g., biological treatment) do not have the same treatment efficiency and reliability in northern communities (Hydromantis Inc., 2005). In addition, northern communities tend to be smaller in populations, have different MWWWE characteristics due to differing water usages, and are in much more remote locations. Conventional wastewater treatment plants (WWTPs) have repeatedly been cited as an inappropriate option for many remote and relatively small communities. The prohibitively high capital and maintenance costs paired with intensive requirements for technical supervision and optimization renders mechanical treatment plants a less favorable choice for most communities in Nunavut (Yates et al., 2012; Krkosek et al., 2012; Hayward et al., 2014; Chouinard et al., 2014). As a result, passive methods of municipal wastewater treatment tend to be the most applied approach in Nunavut due to the low operation and maintenance requirements. In most communities, passive treatment of wastewater in Nunavut occurs during a three to

four month period spanning from the spring freshet in June to the freeze-up in September. This period is termed the treatment season.

The majority of communities in Nunavut are small in size with populations ranging from 130 to 2800, with median and average populations of 1000 and 1200 people, respectively (Government of Nunavut, 2014). Water uses in Nunavut's communities range from approximately 13 – 140 m³/day, in line with reported estimates related to wastewater production for trucked water delivery and wastewater collection (i.e., 90 L/person/day; Heinke, 1991). Water consumption values are on average three times less than the national average (e.g., 110 L/capita/day in Nunavut vs. 329 L/capita/day overall in Canada). These communities are thus on the low end of MWWWE producers in the country (Heinke, 1991; Daley, 2014).

Typical wastewater management systems in Nunavut include the hauling of wastewater via truck from the residences, businesses, and industries in the community to a Wastewater Stabilization Pond (WSP). These WSPs, also referred to as wastewater lagoons, are typically constructed on the outskirts of the communities, at a distance of approximately 0.5 – 1 km. There are twenty-five hamlets located in Nunavut, of which twenty-one use a WSP and/or a tundra wetland treatment area (WTA) and three use a form of mechanical treatment (Wootton et al., 2008; Johnson et al., 2014). The WSPs can have a scheduled decant with a mechanical pump, or passively discharge effluent, into the receiving environment during the treatment season. Most of the WSPs can be defined as intermittent systems, discharging effluent on a limited amount of occasions (e.g., 1-4) over a calendar year to marine receiving water environments. There are a few exceptions to this wastewater management scenario, such as the continuously discharging wastewater treatment plants in both Iqaluit and Pangnirtung, but the majority of the systems throughout Nunavut follow a similar model to that described above.

1.2 RESEARCH OBJECTIVES

The Centre for Water Resources Studies (CWRS) at Dalhousie University has conducted a research program on wastewater treatment infrastructure in Nunavut, Canada from 2011

to 2016. The research program is a collaborative effort with, and funded by, the Community and Government Services (CGS) department of the GN.

A focus area of the broader infrastructure research program was to conduct risk assessments on the receiving water environments typical of communities in Nunavut. To date, there have not been any studies that assess the assimilative capacity of arctic receiving water environments and the water quality impacts associated with MWWWE discharges in these unique environments. An analysis of the typical MWWWE discharge scenarios throughout the territory was undertaken with the following research objectives:

- i) Characterize the MWWWE discharge scenarios and receiving water characteristics for several typical study sites in Nunavut;
- ii) Characterize the water quality and study the transport of the MWWWE in the vicinity of the discharge locations at these study sites;
- iii) Define the extents of initial mixing zones based off of applicable water quality criteria for several discharge scenarios at each study site; and
- iv) Identify the primary factors influencing mixing and initial mixing zone extents at these locations, and provide recommendations for complying with initial mixing zone spatial requirements as defined in similar jurisdictions.

CHAPTER 2. LITERATURE REVIEW

2.1 WASTEWATER DISCHARGE TO RECEIVING WATER

ENVIRONMENTS

Discharging liquid waste streams to nearby receiving waters and relying on the downstream environment's natural ability to assimilate and buffer any undesirable constituents has long been the approach used to manage wastewater in developed areas. Coastal communities discharge their wastewater (with varying levels of treatment) to the shoreline waters. In particular, the discharge of a relatively small quantity of water from smaller communities into a seemingly infinite receiving water volume provides a solution with a perceived minimal risk of impact to the receiving water environments while typically representing the least costly solution.

In North American systems that discharge to marine environments, there are several typical discharge systems that exist: 1) submerged pipe, 2) single port diffuser, 3) multi-port diffuser, and 4) surface discharges. The diffuser based systems are subsurface discharge systems that can differ in their port orientation, angle, size, and other parameters to ensure effective mixing of the wastewater as it leaves the structure, and are typical in larger discharge scenarios in order to ensure appropriate levels of mixing immediately surrounding the discharge location are achieved. Much research, documentation, and design information has been created for application in situations where wastewater discharges are occurring subsurface (e.g., Fischer et al., 1979; US EPA, 1985; US EPA, 1994; Doneker and Jirka, 1990). However, surface discharges are typical in the Far North of Canada likely due to the simplicity and lack of additional costs associated with this approach, and are the focus of this study.

2.2 ENVIRONMENTAL IMPACTS OF WASTEWATER DISCHARGE

Discharging wastewater to receiving water environments is not without consequences. The assimilative capacities of these systems can be overloaded, especially in cases where quantities of wastewater discharged are significant. The environmental impacts related to inappropriately treated (or untreated) larger MWWWE discharges are well documented,

and include short and long-term impacts to nearby shellfish populations, significant bacterial contamination, large quantities of contaminated sediments, and poor water quality with the potential cumulative effects to nearby flora/fauna. In addition to these environmental concerns, the aesthetic qualities of these areas are diminished; there are restrictions to potential recreational uses due to heightened levels of bacteria (i.e., restrictions on swimming, fishing, shellfish harvesting, etc.) and resulting negative public perceptions (EC, 1997; HRM, 2015). The cumulative effect of these impacts is not only significant from an environmental perspective, but also from both economic and social perspectives.

General considerations for environmental impacts to receiving water systems due to MWWWE discharges include:

- The depletion of oxygen in the receiving water environment;
- The transport of pathogens into these waters;
- The bioaccumulation of municipal wastewater contaminants in the food chain within receiving water environments;
- The deposition of solids and the related effects on the flora/fauna in the areas of deposition;
- The transport of heavy metals and other potentially carcinogenic constituents; and
- Eutrophication resulting from the input of nutrients.

2.3 WASTEWATER DISCHARGE REGULATIONS

There are typically three main considerations in the efficient management of wastewater: 1) collection and treatment/reduction at the source, 2) wastewater treatment, and 3) effective discharge of treated effluent into the environment (Fischer et al, 1979). The focus of this study is on the third component of the overall management of wastewater.

The first effective regulations regarding the discharge of wastewater to receiving water environments in North America were outlined in the United States in the Federal Water Pollution Control Amendments in 1972, where requirements for effluent treatment were defined, among other details. The key goals of the act were to ensure protection for

fishable and swimmable waters, and setting water quality standards based on ‘designated uses’ of the receiving waters. As a result, permits under the National Pollution Discharge Elimination System (NPDES) were required in the United States, including standards and monitoring requirements for effluent discharges. The act was further refined in 1977 under the Clean Water Act, where classification of pollutants was undertaken and further standards for compliance were provided. There are two criteria in place: one related to the effluent discharge itself based on the classification of the specific industrial facility, and one related to the ambient environment in which the discharge is entering. In the United States, the Clean Water Act, and specifically Section 301(g), states that Initial Mixing Zones (IMZs; also referred to as an ‘allocated impact zone,’ ‘regulatory mixing zone’, and other similar terms) can be permitted for discharges to receiving waters, unless the State prohibits IMZs. These IMZs, which represent a set spatial boundary surrounding a discharge location within which modified water quality criteria are applied, can include two parts: the toxic dilution zone, where more stringent criteria are placed on contaminants with higher environmental risks, and the overall mixing zone which provides the boundary within which modified water quality criteria are in place. Once the beneficial uses of the receiving water are defined, MWWWE discharge criteria are back-calculated to ensure limits based on the most appropriate (e.g., stringent) classifications are met (Minnow Environmental Inc., 2005). As with any dilution-based approach to mitigating the risks of contamination, there are still concerns related to long-term accumulation of pollutants within the receiving water environments. Additional water quality and biological testing is recommended in order to ensure that the effects of this long-term accumulation are minimal. Details surrounding IMZs are further summarized in Section 2.3.4.

IMZs play a key role in regulating discharges into receiving water environments and determining appropriate discharge criteria for any discharges with potential to have downstream environmental impacts, and their use in determining appropriate wastewater discharge criteria has since been adopted in Canada.

2.3.1 MWWE REGULATION IN CANADA

The federal Fisheries Act provides the foundation for the protection of fish and fish habitat throughout Canada, and as such provides the basis for requirements placed on works that have the potential to impact these resources. Under the Fisheries Act, any deposit of deleterious substances into fisheries waters or any work, undertaking, or activity that results in serious harm to fish or fish habitat is prohibited without authorization (Government of Canada, 2016).

In addition to complying with the provisions included within the Federal Fisheries Act, any provincial/territorial Regulation requirements must also be addressed as part of the approval of a work, undertaking, or activity (EC, 2006). It is typically written in provincial regulations (in particular, the provincial/territorial Acts related to Water Resource management) that any release of water containing deleterious substances that may cause harm or injury to any living organism is not permitted. These requirements are typically tied into the goals of preventing ‘adverse effects’ to the environment as a result of any proposed activities.

When governmental agencies define appropriate and feasible wastewater discharge criteria, considerations related to the receiving water environment can be included. The natural environment has the ability to buffer out a certain level of increases to unwanted substances within its connected ecosystems through physical, chemical, and biological processes – this ability for environments to manage increases without a related degradation of water quality/ecosystem health can also be described as its inherent ‘assimilative capacity’. In some cases, in defining required discharge quality criteria, this assimilative capacity can be considered through defining a zone surrounding the discharge location where modified water quality criteria are applied – typically referred to as an IMZ. In this approach, the definition of appropriate IMZs and end-of-pipe wastewater discharge quality criteria must consider the overall assimilative capacity of the downstream environment in order to ensure that irreversible damages to these environments do not occur. When setting IMZ parameters, it is critical to take the following into account (MoEE, 1994; CCME, 2008 and 2008b):

- Requirements set to minimize risk of long-term, irreversible environmental impacts within the zone;
- IMZ location to not interfere with health of nearby residents (e.g., recreational activities, drinking water supplies);
- Requirements to be set to minimize risk of adverse effects to aquatic life and overall ecosystem health.

Examples of long-term environmental impacts within the zone could relate to the change in trophic status of a receiving water (e.g., eutrophication) and contamination of sediments in the vicinity of the discharge, while an example of potential consequences to human health would include illness resulting from physical contact with waters with elevated bacteria concentrations.

As a result, the definition of appropriate IMZs is best approached on a site-specific basis. This is not always feasible, however, as the cost of completing such studies can be prohibitive, especially within municipalities. There is no one size fits all solution for defining the IMZ requirements. In addition, it must be noted that the intent of IMZs is not to represent an alternative to the implementation of reasonable treatment requirements.

In Canada, it is up to the individual provincial/territorial regulatory bodies to define the IMZ criteria that are used. In the development of regulations to govern the discharge of wastewater to receiving water systems, decisions must be made in determining what level of both acute and chronic toxicity can be accepted in a particular receiving water. In doing so, efforts also must be made to ensure that any resulting criteria are made using strong statistical support. For example, accounting for the variability of the receiving water environment, it is more realistic to define worst-case conditions using references to the cumulative frequency distributions for each input parameter (e.g., receiving water currents, ambient conditions, etc.). In doing so, considerations for the rarity of the overlap of absolute worst-case scenarios for each parameter can be included in analysis (US EPA 1985). Also worth considering is the end goals of the regulation - if chronic exposure is the risk being mitigated, and risk for acute toxicity is low, then the wastewater parameters used in analysis may be better suited to be based on longer term averages.

2.3.2 CCME CANADA-WIDE STRATEGY

With the introduction of federal wastewater discharge criteria through the WSER, there is a consistent end of pipe requirement across Canada to be met going forward. As the new discharge criteria are independent of any site-specific considerations (e.g., discharge flow rates, timing, location, etc.), they present challenges for legacy wastewater treatment systems and locations where additional difficulties related to meeting these requirements exist. As previously mentioned in Section 1, the CCME has identified that such challenges exist in the northern jurisdictions of Canada.

Effluent Discharge Objectives (EDOs) are the maximum concentrations of key parameters in wastewater to be met at the point of discharge. A standardized methodology for developing EDOs for discharges from municipal wastewater treatment facilities has been created as part of The CCME *Strategy*. The framework for the environmental risk assessment (ERA) approach, which includes development of EDOs, is illustrated in Figure 2.1 (CCME, 2008b).

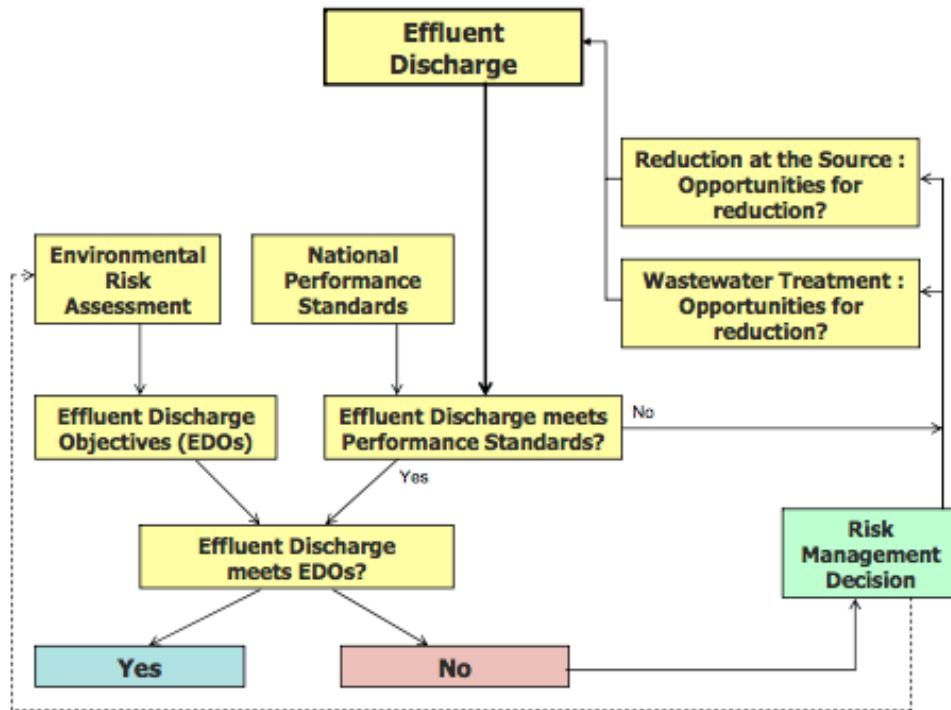


FIGURE 2.1. ENVIRONMENTAL RISK MANAGEMENT FRAMEWORK (SOURCE: CCME, 2008B)

These EDOs are set through determining appropriate Environmental Quality Objectives (EQOs), which are to be met at the boundary of the defined IMZ. EQOs are usually established as the Canadian Environmental Quality Guidelines (CEQGs) in conjunction, or replaced by, any related provincial or territorial water quality guidelines (CCME, 2008b). The intent of the EQOs is to define the upper limits for concentrations of substances of concern within a receiving water body.

The benefits to incorporating an environmental-risk based approach to developing MWW discharge criteria is that site specific inputs are considered, and as such, tailored solutions related to local risks to human and ecosystem health can be developed. This approach has the potential to be more cost-effective by allowing for focus to be placed on discharge scenarios with significant environmental risk, while still allowing for discharges with minimal predicted risk for impacts to be managed with an appropriate level of treatment and financial commitment. Under the CCME *Strategy*, southern

facilities for wastewater treatment and discharge must use an ERA approach in defining appropriate EDOs to be met for each facility (CCME, 2008b).

Of interest within the NPS is the decision to only require compliance with four set wastewater parameters: TSS, CBOD, Total Residual Chlorine (TRC), and Un-ionized Ammonia (NH₃). These wastewater parameters were chosen due to their commonality as pollutants within MWWE (CCME, 2009). Within the strategy for MWWE, there are additional EDOs that ‘should’ be achieved in order to “adequately protect human health and the receiving environment” (CCME, 2008b). These EDOs are to be set based on the results of site-specific environmental risk assessments. The environmental risk assessments and site-specific effluent discharge objectives were given an 8-year window to be completed for all facilities. Within this timeframe, the characterization of the MWWE discharged from wastewater treatment facilities is to take place over a yearlong period. Additional details can be found in The Strategy document (CCME, 2009). Further clarification has been provided by the CCME that EDOs developed as a result of completed ERAs are the responsibility of the jurisdiction, and are not a part of the WSER (CCME, 2014).

Treatment Requirements

As outlined in the WSER (Government of Canada, 2015), the following parameters have been defined as ‘deleterious’ and requiring treatment to set values:

TABLE 2.1. WSER EFFLUENT TREATMENT CRITERIA

Deleterious Wastewater Parameter	Wastewater Quality Criteria
Matter with Carbonaceous Biochemical Oxygen Demands (CBOD)	<25 mg/L average concentration
Total Suspended Solids (TSS)	<25 mg/L average concentration
Total Residual Chlorine (TRC)	<0.02 mg/L average concentration ¹
Un-ionized ammonia (NH ₃)	<1.25 mg/L as nitrogen (N) at 15°C ± 1°C

Notes: ¹ – If chlorine or a substance including it was used in treatment (Government of Canada, 2015)

Additional information related to how the averages and maximum in Table 2.1 above are to be determined can be found in the WSER document (Government of Canada, 2015).

In providing guidance for the management of wastewater discharges, the CCME has defined several treatment facility size categories for which different considerations may apply, as provided in Table 2.2 (CCME, 2008):

TABLE 2.2. MUNICIPAL WASTEWATER TREATMENT FACILITY SIZE CATEGORIZATION (SOURCE: CCME, 2008)

Size Category	Flow (m³/day)
Very Small ¹	≤ 500
Small ¹	500 – 2,500
Medium	2,500 – 17,500
Large	17,500 – 50,000
Very Large	> 50,000

As outlined by the CCME, the criteria as outlined in Table 2.1 may be different from the NPSs set for Canada’s Far North (CCME, 2008). The Strategy has outlined that all WWTP discharges should eventually achieve levels of treatment that correspond with effluent quality expected from secondary treatment (CCME 2008b). There is uncertainty related to the present level of treatment being obtained in the systems in Nunavut, and the

water usages, climatological parameters, and treatment scenarios can change significantly between communities.

The majority of Nunavut wastewater management facilities fall within the Small and Very Small categories defined in Table 2.2, and these categories will be the focus of the research of this thesis. One thing to note, however, is that the community of Iqaluit would be defined as a Medium or larger sized facility, and represents an outlier that is not explicitly considered as part of this research. For wastewater facilities that have been deemed as medium or larger such as Iqaluit, there may be additional considerations and treatment requirements. For example, a toxicity study on the effluent is required, as well as additional considerations related to the associated downstream environmental risks. Additional criteria relating to this requirement are outlined in the CCME *Strategy for the Management of Municipal Wastewater Effluent* (CCME, 2009).

2.3.3 NORTHERN CANADIAN CONTEXT

The '*Guidelines for the discharge of treated municipal wastewater in the Northwest Territories*' document (NWTWB, 1992) is used for guidance on wastewater discharges in Nunavut and the Northwest Territories (NWT). Another document titled the '*Guideline for the Discharge of Domestic Wastewater in Nunavut*' was prepared for the Nunavut Water Board, but this draft document is no longer applied in Nunavut and was not finalized (C. Zyla, personal communication, April 2014). The criteria presented in the now unused Nunavut Water Board document was very similar to those presented in the NWTWB (1992) document (Minnow Environmental Ltd, 2005).

The objectives of the NWTWB (1992) document are to provide guidance in complying with the Northern Inland Waters Act of 1972, and state that policy requires 'the treatment of all municipal waste by the best practical means before it is discharged into the receiving environment' (NWTWB, 1992). It is also outlined that while the guidelines provide a set of requirements license applicants must follow, there are additional considerations, such as any additional compliance requirements set by Federal, Territorial, or Municipal legislation. Most notably is the expectation that the criteria set by the Department of Fisheries and Oceans (DFO) will be followed, which includes

additional considerations for any discharge parameters that could be defined as ‘acutely lethal’ to the fish species of the area (NWTWB, 1992).

Within this document, the IMZ concept is applied, allowing for modified water quality criteria to be set within a defined area within the receiving water body. The following IMZ spatial definitions apply for marine discharges (NWTWB, 1992):

- An area bounded by 100 m radius surrounding the discharge location;
- Not occupying more than one-third of the cross-sectional area of a waterbody around a discharge point; and
- Not intruding on any drinking water intakes, shellfish beds, recreational areas, and biologically sensitive areas.

Measures to account for acute toxicity are also qualitatively present in the guidelines, with requirements to ensure that no fish kills or bioaccumulation of toxic wastewater constituents occur.

Water quality objectives are specified for outside the IMZ in terms of dissolved oxygen, residual chlorine, nutrients, coliforms, toxicity, suspended solids, floatable solids and scum, oil and grease, and metals (NWTWB, 1992). It should be noted that these are the minimum criteria to be applied – it is further outlined that more stringent criteria can be applied at the discretion of the Water Board and/or DFO on a case-by-case basis, including the addition of supplementary parameters to be met.

Due to the significant variability among environmental conditions, community water uses, effluent qualities, and receiving water environments, the NWTWB (1992) document states that final considerations will be made by the Water Board itself when considering any license. In addition, many of the considerations in the document (e.g., calculations of dilution) relate only to wastewater discharges to freshwater sources (e.g., rivers, lakes, estuaries). When considering marine discharges, it is stated that untreated discharges are permissible where ‘treatment is not practicable’, on the condition that floatables are removed and the discharge is free of larger particles. The document also provides additional guidelines related to the impacts that wastewater discharges may have

on commercial or recreational endeavours, such as fisheries, shellfish harvesting, and other hunting/gathering activities may be affected.

One of the key considerations of the NWTWB document is the need for flexibility when defining municipal wastewater discharge criteria. This is due to the significant variability observed among northern communities, as parameters of interest may differ significantly based upon considerations due to populations, industry, and geographic locations, as well as other considerations. For example, water quality criteria is provided for metals of concern, but is caveated with the focus for these parameters being mostly in scenarios where industrial and commercial discharges have the potential to contribute significant amounts of these parameters into community wastewater discharges. In addition, due to the large overall area within which these communities rest, there are diverse receiving water environments, with varying background concentrations of many of the parameters of interest. As such, considerations for the background levels of these parameters when determining effective water quality criteria are also included.

2.3.4 INITIAL MIXING ZONES

As explained above, an IMZ, which is also described in various guidance documents using similar language (including allocated impact zone, regulatory mixing zone, or simply a mixing zone), can be defined as a specific defined area surrounding a discharge location within which initial mixing of the effluent discharge occurs and where modified water quality criteria are applied. These mixing zones are defined by the regulatory body, and represent the spatial boundary outside of which water quality criteria must be met.

In Canada, it is up to the individual provincial/territorial regulatory bodies to define the IMZs that are used. Therefore, regulatory decisions must be made in determining what level of both acute and chronic toxicity can be accepted in a particular receiving water. In doing so, efforts also must be made to ensure that all combinations of concentrations of pollutants, discharge magnitudes, and environmental conditions are assessed for worst-case scenarios. When setting IMZ parameters, it is critical to minimize the risk of long-term environmental impacts to aquatic life and the ecosystem, and ensure the health and safety of humans in the vicinity (MoEE, 1994; CCME, 2008).

Water Quality/Use Considerations

In defining appropriate IMZs, several considerations related to potential local and downstream water uses must be made. Not only are considerations for potential environmental impacts related to MWWWE discharges required, including potential impacts to ecologically sensitive areas, but also considerations to the additional uses of the receiving water body. For example, considerations for proximity to drinking water supplies, recreational (e.g., fishing, swimming, boating) areas, and any commercial/industrial uses (e.g., commercial fishing, industrial water withdrawal locations).

In addition, there are substances that are not typically permitted as part of any mixing zone analysis (i.e., toxic, non-degradable, persistent/bioaccumulative substances). It has been written that mixing zones are only to apply for ‘conventional’ pollutants, and not to be applied to pollutants defined as hazardous substances that may be acutely toxic (MoEE, 1994, CCME, 2008). Mixing zone considerations relate to degradable substances that can be managed by the assimilative capacity of the receiving watercourse/water body.

As such, the defined IMZs represent a boundary within which modified water quality criteria are applied. As outlined in Section 2.3, two of these boundaries may be defined: 1) the acute IMZ, and 2) the chronic IMZ. In cases where both acute and chronic IMZs have been defined, the intent is to minimize the area affected by discharges that include constituents of greater concern. As outlined in the documents supporting the CCME *Strategy* (e.g., CCME, 2008), one general IMZ is recommended to be defined for use going forward.

Of particular concern is the settling of material in the IMZ and the effects of this settling on the sediments, flora, and fauna (benthic invertebrates) in these areas. As part of the overall Nunavut research project undertaken by Dalhousie, an analysis into the impacts on the benthic invertebrate populations surrounding the wastewater discharge locations at several study sites in the territory was completed, with results found in detail in Krumhansl et al. (2014). To summarize, the benthic invertebrate communities downstream of five different treatment facilities were studied to understand the impacts

of MWWWE discharges on the health of these invertebrates. Minimal impacts to benthic invertebrate communities were detected in four of the five Nunavut communities studied, with the larger discharge scenario in Iqaluit being the outlier.

In determining potential IMZ extents under The Strategy, EQOs are to be defined. Three separate approaches for determining EQOs are outlined in the CCME Technical Guidance document (CCME, 2008): 1) Setting quantitative exceedance criteria related to parameters of concern in the receiving waters, 2) A toxicological approach using ‘Whole Effluent Toxicity (WET)’, which defines MWWWE discharge limits based on an analysis of potential downstream toxicity, and 3) Criteria set off of biological assessment. A combination of the above methodology is also appropriate in setting site specific EQOs. As generic EQOs are of interest in this research, the physical/chemical/pathogenic criteria approach of 1) is the focus of this study. Additional information related to 2) and 3) can be found in the Technical Supplement 3: Standard Method and Contracting Provisions document (CCME, 2008).

Spatial Considerations

When determining appropriate spatial considerations for IMZs for estuaries and coastal marine waters (which are typical in Nunavut), an understanding of the variability, frequency, and magnitude of the ambient conditions that affect mixing within the receiving waters is required. In particular, design of appropriate spatial extents for IMZs should consider the ‘worst-case’ conditions in receiving waters. This is typically where minimal dilution/maximum transport is observed, and is important to understand in defining appropriate IMZ extents. There are several variables that must be considered in order to understand the potential mixing conditions in marine/estuary discharge cases, including (Jones et al., 1996; Jacques Whitford and Natech, 2003; Doneker and Jirka, 2007; CCME, 2008b):

- Differences in discharge and ambient water densities (e.g., freshwater/MWWWE and saltwater);
- Stratification of the receiving water body;
- Wind speed and direction;

- The speed and direction of ambient currents, with considerations for what depth is most appropriate for measurement (e.g., subsurface discharges);
- Tidal effects; and
- Others as identified during receiving water characterization (e.g., receiving water bathymetry, analysis of substrate, etc.).

As there is typically a level of variability in relation to the concentrations of different parameters within MWWE, it is also important to consider this variability when determining appropriate IMZ boundaries. It is recommended that statistical analysis of MWWE be undertaken in order to determine the likelihood related to meeting potential discharge criteria at the desired spatial extents (CCME, 2008). The results of this analysis can inform decision-making through providing supporting information associated with the probability of exceeding specific wastewater criteria.

As IMZs have been set to be typically on the range of 10 – 10² m from the location of discharge into a receiving water body, the resulting time-scales that these zones would have increased concentrations of potential pollutants would be on the order of minutes for intermittent discharges, which has been reported to be observed in the United States (US EPA, 1985). By focusing on discharge criteria that limit both the time and space permitted for increases to parameters of concern within the receiving water environments, it is assumed that the biological organisms in the vicinity of these discharges would have minimal contact and exposure to these raised levels prior to travelling through the impacted area (US EPA, 1985).

Depending on the jurisdiction and discharge situation, mixing zones dimensions typically have been set using a cross sectional area, surface area, or radius/width measurement (US EPA 1984).

2.4 EFFLUENT DISPERSION

Knowledge of the processes occurring during the transport and dilution of effluent within receiving water environments is required in order to determine the extent of disturbance resulting from discharges of deleterious substances to these environments. Much research

has been completed into characterizing these transport and dispersion processes, resulting in the creation of several different empirical and analytical/theoretical approaches.

2.4.1 MIXING PROCESSES

When wastewater is discharged into receiving water environments, a wastewater plume is typically formed. This plume exists due to the differences between the characteristics of the wastewater and the ambient receiving waters, particularly differences in their concentrations, buoyancies, and momentums. The mixing and transport processes that typically occur as the wastewater jet scatters within the ambient environment include diffusion and dispersion. As typical wastewater discharges are based in fresh water, they are less dense than the marine receiving waters and so are positively buoyant (US EPA, 1985). Therefore, when wastewater is discharged at a subsurface location (i.e. at depth), the resulting plume typically entrains and mixes with the ambient water and eventually rises to either a level of neutral buoyancy or the surface of the receiving waters if neutral is not attained. This mixing is driven by the differences in buoyancy and momentum between the wastewater jet and the ambient environment, leading to entrainment of the ambient water within the plume as turbulent diffusion processes dominate in the vicinity of the discharge location, and then to convective processes associated with the vertical transport of the plume within the water column due to buoyancy differences. Advective transport of the plume due to the bulk motion of the receiving water is also observed. Initially, plume transport will be dominated by the momentum of the jet, but buoyancy is shown to dominate mixing in many typical discharge cases. It has been reported that mixing from entrainment processes significantly slows once discharges reach neutral buoyancy (US EPA, 1985).

As differences in velocities typically exist between discharges and the receiving water environment, shear stresses causing turbulent mixing at the wastewater jet flow boundaries also typically exist until the velocities of the discharge match the ambient environment. Subsurface buoyant jets are detailed widely within literature, and there are various approaches and methods available for the design of systems for this particular discharge case. Designers can modify the characteristics of the jets in order to understand

the potential impacts of jet depth, orientation, port size, and quantity of ports on the resulting wastewater effluent plume (Fischer et al., 1979).

When wastewater is discharged at surface, the wastewater acts as a buoyant surface jet in the receiving water environment (US EPA, 1985; Jones et al., 1996). The MWWWE discharge scenarios typically observed in Nunavut involve these surface discharges, and are the focus of this research. Examples of typical buoyant jet discharge plumes are given in Figure 2.2.

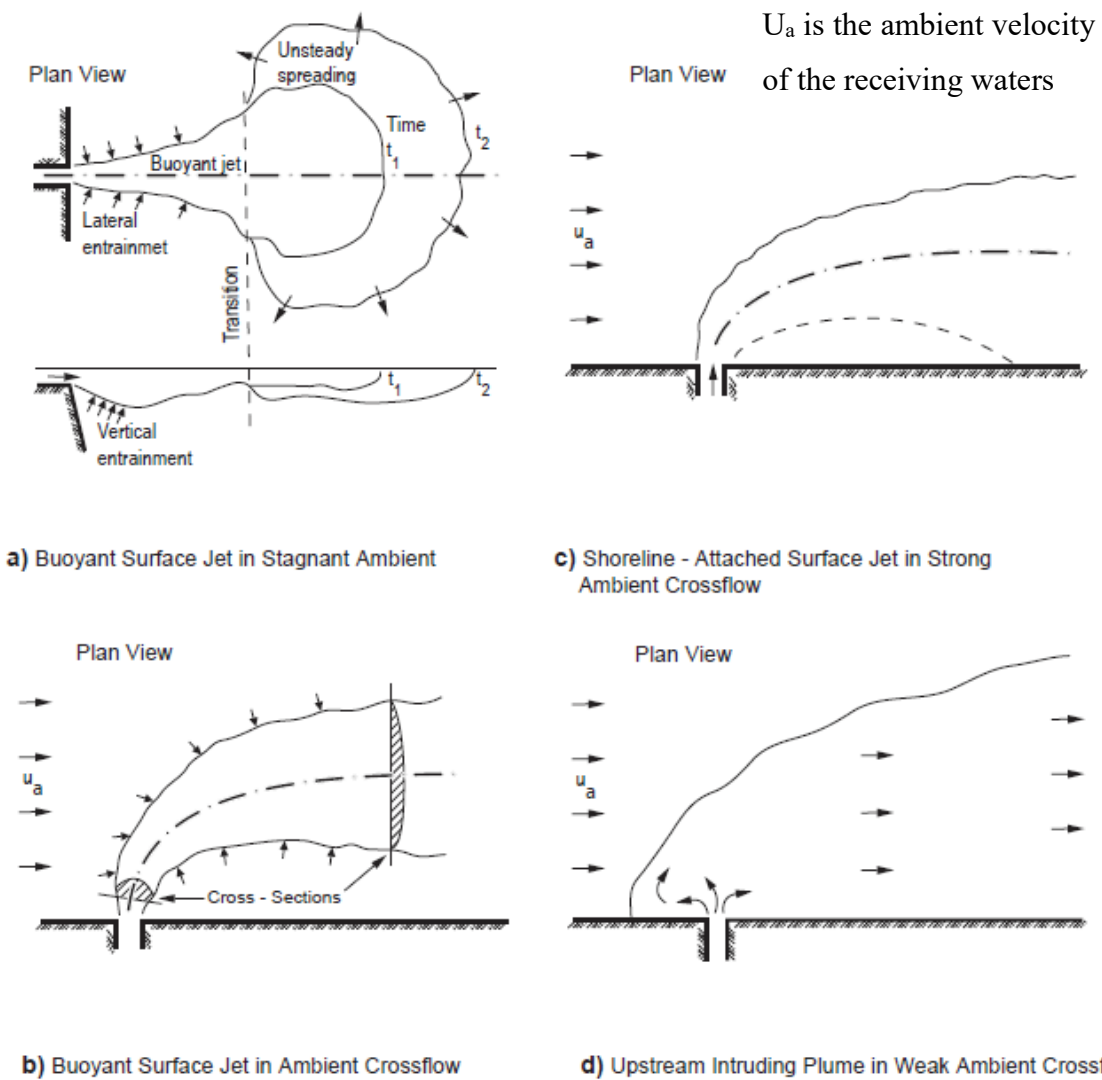


FIGURE 2.2. TYPICAL BUOYANT SURFACE JET MIXING FLOW PATTERNS UNDER STAGNANT OR FLOWING AMBIENT CONDITIONS (SOURCE: DONEKER AND JIRKA, 2007).

The mixing characteristics of effluent discharge in receiving water environments is a function of both the characteristics of the discharge, and those of the waters into which it is discharged. Several terms have been defined for use when referring to the mixing processes that dominate the different stages of wastewater dispersal in the receiving waters (Doneker and Jirka, 2007; MixZon, 2014; Zhao et al., 2011):

- Near-Field Mixing: Defined as the mixing processes that are dominated by the wastewater jet characteristics (i.e. momentum, buoyancy, and mass fluxes). Typically occur over small spatial ($< 10^2$ m) and time (minutes) scales.
- Far-Field Mixing: Defined as the mixing processes that are dominated by the characteristics of the ambient receiving water environments. Typically occur over greater spatial ($> 10^2$ m) and time (hours) scales, such as advection by ambient currents and natural lateral diffusive processes.

Under these definitions, it can be assumed that near-field mixing typically occurs within the IMZ, but the two terms are not interchangeable. Regulatory requirements are not bound to either the near or far field, and as such an understanding of both may be required (Jones et al., 2007). A figure outlining near and far-field locations is provided as Figure 2.3 below.

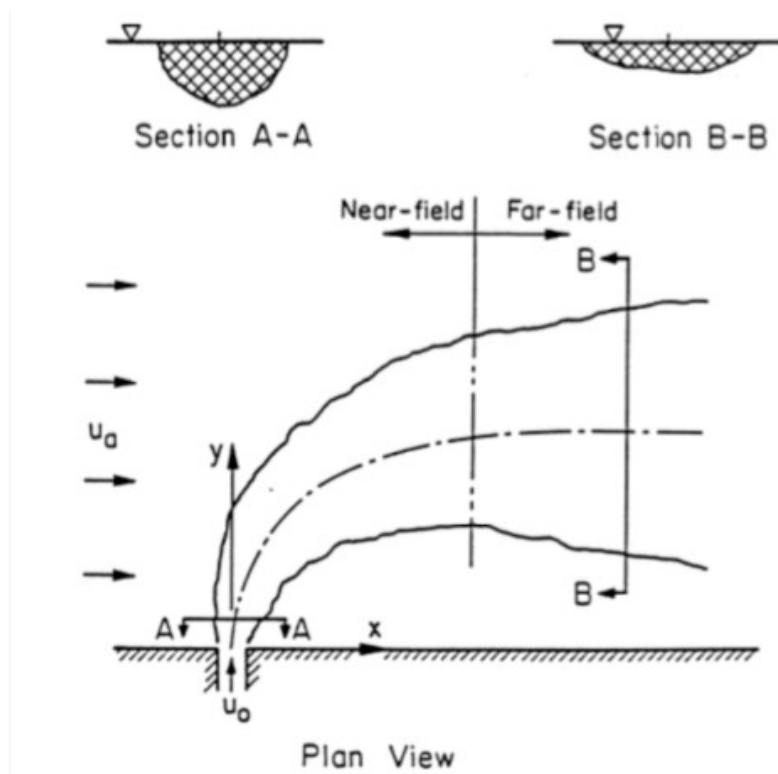


FIGURE 2.3. NEAR AND FAR-FIELD REGIMES FOR A TYPICAL BUOYANT SURFACE JET (SOURCE: JONES ET AL., 1996).

Near-field processes refer to those processes that are dominated by the characteristics of the jet discharge – i.e. the jet’s momentum, buoyancy, and mass flux. Through

calculation of the densimetric Froude number (which describes the ratio of inertial to gravitational forces), determinations can be made regarding whether the plume has characteristics similar to a momentum jet (higher initial Froude number), or to a buoyant plume (lower initial Froude number; US EPA, 1985). It has been noted that the plumes resulting from both of these classifications are similar in appearance, but have differing dilution and transport characteristics (US EPA, 1985).

It has also been noted that horizontal dispersion occurs at a much faster (i.e., an order of magnitude) rate than vertical mixing (Jacques Whitford and Natech, 2003). In cases of surface discharges, this corresponds to effluent ‘spreading’ over a large area on the surface of the receiving water prior to the effluent becoming fully mixed in the receiving water environment (Doneker and Jirka, 2007).

There are several parameters specific to the ambient environment that can affect the mixing characteristics of a discharge. Stratification within marine environments is commonplace, leading to large temperature and density differences through the water column that may affect mixing, particularly in discharges made subsurface. With surface buoyant discharges, complications associated with the atmospheric boundary can be more significant, such as the effect of wind and potentially air temperature on the surface layer. Flow boundaries, such as the interactions of discharges with both ocean bottom and shorelines, require consideration. Tidal effects also represent a significant consideration in mixing within marine environments, especially when dealing with surface discharges.

The presence of tides can significantly affect several key mixing parameters, such as the exact discharge location, available water depths, and ambient currents, while influencing turbulence (and as a result, turbulent mixing of the waters at the channel bottom) through the advancing tide’s contact with the bottom of the waterbody. Considerations for tidal reversals (i.e., when tide shifts from incoming to outgoing or vice-versa) are also important, as these occurrences can reflect worst-case scenarios through the pooling and recirculation of previous discharge plumes in proximity to the discharge location (Nash and Jirka, 1996; Jacques Whitford and Natech, 2003). With buoyant surface discharges in particular, the turbulent mixing occurring the discharge/ambient layer interface due to the movement of the more dense liquid below the discharge layer may not occur as

significantly without tidal influences, and as such less mixing can be expected (Fischer et al., 1979). As such, additional considerations must be made when designing for situations with significant tidal ranges. When determining the scale of tides to be used in analysis, knowledge of the characteristics of currents on timescales relative to the processes taking place (e.g., seconds, minutes) is required for proper analysis to take place. In other words, having an idea of maximum current velocities based on small-scale measurements is desired, in addition to information related to high-level current averages for the area of interest (Fischer et al., 1979; Nash and Jirka, 1996; Doneker and Jirka, 2007; Jacques Whitford and Natech, 2003).

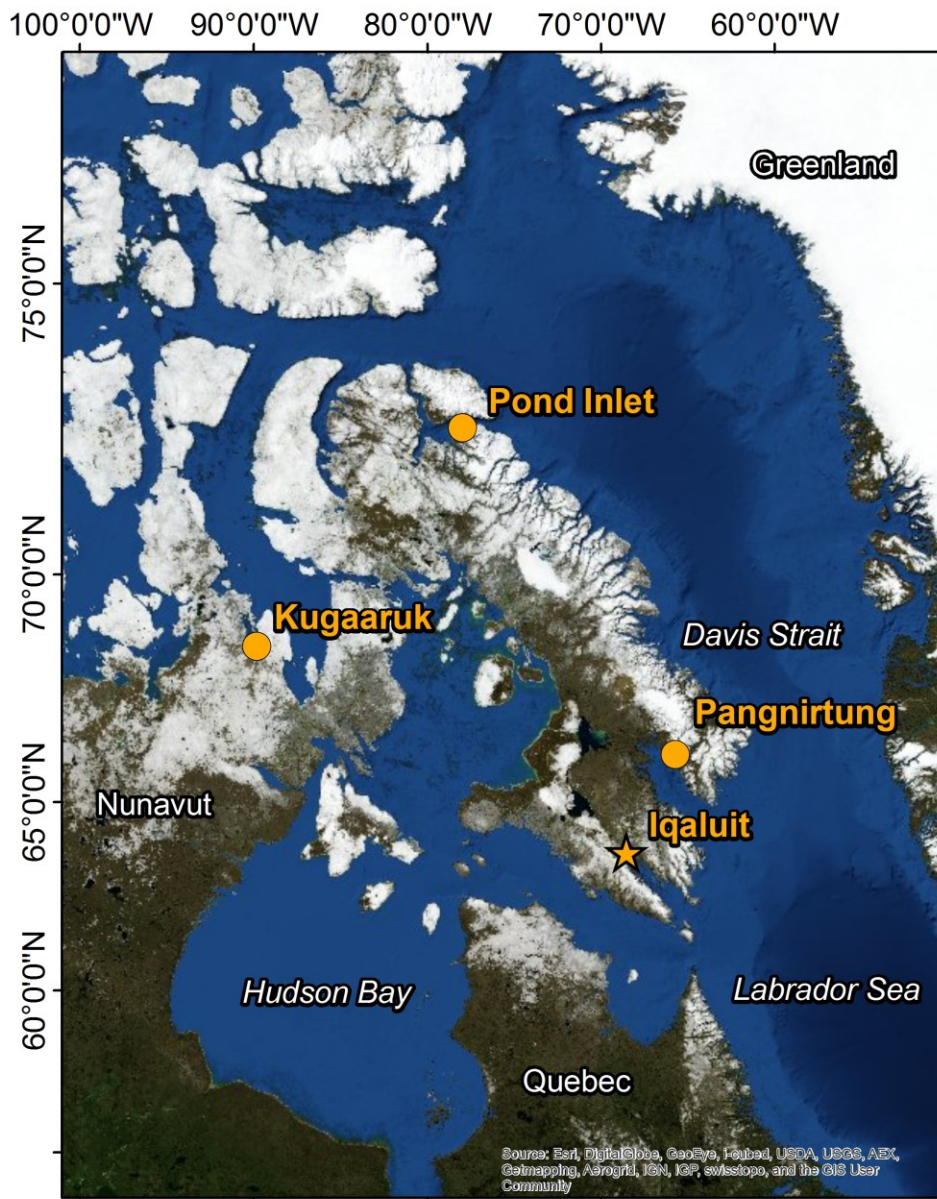
Far-field processes refer to those mixing processes that are dominated by the ambient receiving water environments. These processes are generally independent of any controls the outfall designer or WWTP operator may have on mixing, and occur after a transition phase between the mixing processes dominated by the jet characteristics to the processes dominated by the ambient conditions that can be referred to as the 'intermediate field'. Far-field processes include buoyant discharge spreading processes followed by slower passive diffusion processes (Fischer et al., 1979; Jones et al., 1996). Wastewater plumes are also transported through advection in the far-field, with ambient currents playing a large role in the transport of plume within the receiving waters. Mixing and transport in the far-field can also be influenced by wind conditions and large-scale ocean currents. A comprehensive description of the mathematical considerations of advection-diffusion processes in terms of effluent mixing can be found in Fischer et al. (1979), and a description of different calculation methodologies for far-field processes found in Zhao et al. (2011).

As the mixing processes associated with surface buoyant jets were understood to be complex, the data collection strategy for this study was focused to collect information to aid in understanding the most dominant processes and significant factors affecting mixing.

CHAPTER 3. METHODOLOGY

3.1 STUDY SITES

Study sites were chosen to capture data for several typical cases of wastewater management scenarios across the territory. Sites previously analyzed as part of the overall Nunavut wastewater research project were the focus, as this allowed for data sharing between projects and increased understanding due to focusing on Sites that represent analogues for other Nunavut communities. Communities with marine receiving environments and some form of treatment pre-discharge were considered, as this represents the most common type of discharge scenario in Nunavut. In addition, considerations were made for choosing locations with different geographic localities to represent the climatic variability across Nunavut. With these considerations in mind, the study sites chosen were: 1) Kugaaruk, 2) Pangnirtung, and 3) Pond Inlet, and are shown with the project Water Quality lab in Iqaluit on Figure 3.1.



Map showing site locations in Nunavut

Legend

- Study sites
- ★ Water quality laboratory

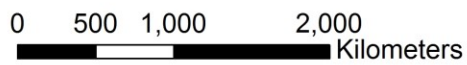


FIGURE 3.1. STUDY SITE MAP

A description of the study sites, a summary of their wastewater management, and descriptions of their receiving water environments are given both in Table 3.1 and further described in Sections 3.1.1 - 3.1.3 below.

TABLE 3.1. CHARACTERISTICS OF WASTEWATER TREATMENT SYSTEMS AND RECEIVING ENVIRONMENTS AT EACH STUDY SITE

Location	Pop. ³	Treatment type	Discharge timing	System Effluent Sampling Location	Receiving Environment	Max Tidal Range (m)	Exposed sediment at low tide
Kugaaruk 68°20'29.04"N, 90°14'25.80"W	771	WSP, decant cell and wetland	Annual – end of summer	End of wetland	Marine - small cove	2.9 ¹ – 3.3 ²	10 m rocky intertidal area
Pond Inlet 72°42'00.42"N 77°57'30.72"W	1549	WSP	Annual - end of summer	End of discharge channel	Marine - open channel	2.3 ¹	<5 m rocky intertidal area
Pangnirtung 66°08'47.61"N, 65°42'04.38"W	1425	Mechanical treatment (activated sludge)	Continuous - year round	Discharge pipe	Marine - medium size bay, high tidal range	6.7 ¹ – 7.6 ²	200-300 m - sandy sediment

Notes: ¹ – based on recorded data available from the Canadian Tides and Water Levels Data Archive (DFO, 2015)

² – based on tidal prediction data available from the Department of Fisheries and Oceans (DFO, 2015b)

³ – “Pop.” is population size according to 2011 census (Statistics Canada, 2012)

It should be noted that the values of maximum tidal ranges reported represent both the largest tidal range recorded in the available datasets (DFO, 2015) and those reported as part of the available tidal prediction charts (DFO, 2015b). As a result of the minimal quantity of recorded data available, there are discrepancies between the values obtained through either method, as illustrated in Table 3.1. As they represent the most up to date

values available, the values provided in the tidal prediction tables (DFO, 2015b) will be used through the remainder of this study.

3.1.1 PANGNIRTUNG

The hamlet of Pangnirtung is located on Baffin Island (66° 08' 48" N, 65° 42' 04" W) and has an estimated population of 1425 (Statistics Canada, 2012). Data collection for the characterization of the receiving water environment was conducted from July 23, 2013 to July 29, 2013. Average air temperatures measured from the nearest meteorological station with Climate Normals data (Iqaluit, NU) range from -23°C to -31°C in January, and from 4°C to 12°C in July. Values for average annual precipitation include 197 mm as rainfall, and 2293 mm as snow, for an annual total of 404 mm of precipitation (Government of Canada, 2015a).

Approximately 136 m³/d (50,000 m³/year) of primarily domestic municipal wastewater is generated in Pangnirtung (NWB, 2011a). MWW is collected via truck and transported to a wastewater treatment facility approximately 300 m from the community boundary. This wastewater treatment facility consists of a mechanical treatment plant (using an activated sludge process); which at the time of this report was being replaced with a new membrane bioreactor system. MWW is discharged from Pangnirtung continuously throughout the entire year, at a rate of approximately 105 – 260 m³/day.

Effluent is released through a small grassy channel over a distance of approximately 120 m prior to eventual discharge to the intertidal receiving water environment (Pangnirtung Fiord) as shown in Figure 3.2a. This area has a maximum tidal range of 6.94 m, which at low tide exposes a gently sloping tidal flat that is mainly composed of sand, with some patches of gravel and small boulders (Figure 3.2b). Ambient receiving water currents with varying magnitudes and direction were observed on-site, depending on the active tidal regime. In general, currents observed on-site were in an N-NE direction.

Traditional activities occur in the intertidal zone of the shoreline in the vicinity of Pangnirtung, with the practice of shellfish harvesting in the intertidal zone near the community of note. Due to the location of the wastewater treatment plant and lack of signage, there is the possibility of shellfish harvesting near to the MWW discharge area.

Migratory fish passage, hunting, and fishing are also possibilities beyond the intertidal zone.

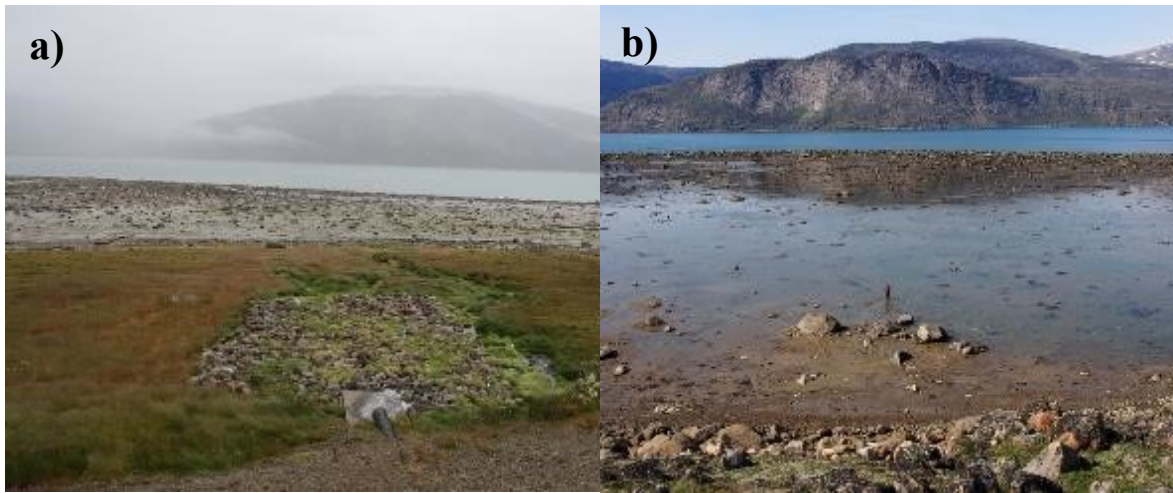


FIGURE 3.2. A) DISCHARGE CHANNEL AND THE EXPOSED TIDAL FLATS AT LOW TIDE ON AUGUST 22, 2012; AND B) SHALLOW RECEIVING WATER ENVIRONMENT NEAR LOW TIDE IN PANGNIRTUNG ON JULY 28, 2013

3.1.2 KUGAARUK

The hamlet of Kugaaruk ($68^{\circ} 32' 05''$ N, $089^{\circ} 49' 29''$ W) has an estimated population of 771 (Statistics Canada, 2012). The site-specific study on the receiving water quality was conducted in Kugaaruk from August 21, 2013 to August 28, 2013. Average air temperatures range from -30°C and -37°C in January, and from 5°C to 14°C in July. Values for annual average precipitation include 117 mm as rainfall, and 1460 mm as snow, for an annual total of 261 mm of precipitation (Government of Canada, 2015b).

Approximately $76\text{ m}^3/\text{d}$ ($28,000\text{ m}^3/\text{year}$) of primarily domestic municipal wastewater is generated daily (NWB, 2011b). Pump trucks are used to transport the wastewater from individual houses and establishments to the wastewater treatment facility located approximately 1 km south of the hamlet. The facility consists of a single-cell WSP with a decant cell and wetland treatment area. The WSP is typically decanted with a pump and generator into the decant cell twice for a period of several days between July and October. During the study period of 2013, effluent discharge rates into the receiving environment ranged from approximately 10 to $48\text{ m}^3/\text{d}$. The variability in discharge rates

can be attributed in part to a breakdown of the WSP pump during the site visit, which caused lower flow rates through the treatment systems.

The effluent discharges from the WTA into a coastal marine receiving environment, Pelly Bay, which has a maximum tidal range of approximately 3 m. Ambient receiving water currents were both cross-shore and inshore in direction, with the prevalent current direction being in the N-NE direction. The receiving environment is characterized by a moderately sloping boulder field that transitions to soft sediments ~50 m from shore (Figure 3.3a). Only a small rocky area is exposed at the lowest tides as shown in Figure 3.3b.

Traditional uses of the receiving waters near the discharge location of the MWWE include fishing, hunting, and boating. Tourism activities are also popular in the area such as kayaking and boating.

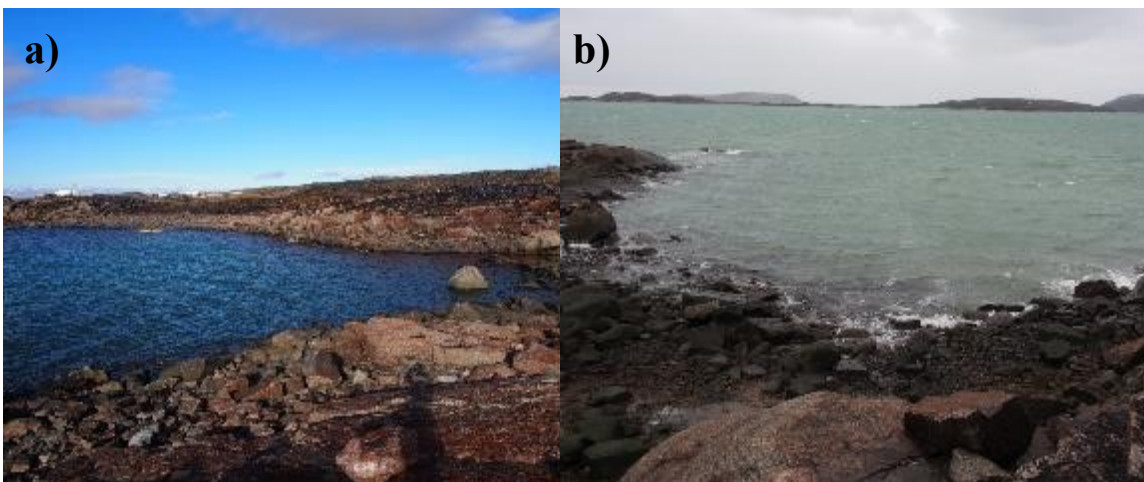


FIGURE 3.3. A) THE RECEIVING ENVIRONMENT IN KUGAARUK, NU ON AUGUST 23, 2013; AND B) ROCKY INTERTIDAL ZONE WHERE DISCHARGE POINT IS LOCATED IN KUGAARUK, NU ON AUGUST 26, 2013.

3.1.3 POND INLET

The hamlet of Pond Inlet is located on Baffin Island ($72^{\circ} 42' 00''$ N, $77^{\circ} 57' 31''$ W) and has an estimated population of 1549 (Statistics Canada, 2012). The site-specific study on the receiving water environment in Pond Inlet was conducted from September 11, 2013 to September 18, 2013. Average air temperatures range from -30°C to -37°C in January, and from 3°C to 11°C in July. Values for annual average precipitation include 91 mm as

rainfall, and 1319 mm as snow, for an annual total of 189 mm of precipitation (Government of Canada, 2015c).

Approximately 114 m³/d (42,000 m³/year) of primarily domestic municipal wastewater is generated in Pond Inlet (NWB, 2014). Pump trucks are used to transport the wastewater from individual houses and establishments to a single-cell WSP. The WSP is located approximately 1.4 km to the east of the hamlet. A manual decant of the WSP is performed annually for a period of three weeks in September or early October. A pump powered by a generator is used to lift the wastewater from the WSP to the discharge channel, at a rate measured within a range of approximately 1300 – 2400 m³/d, over 12 hours each day during the discharge period. The range of flows observed can be attributed to pump issues and periodic pump outages due to refueling requirements. The discharge channel is rocky and approximately 275 m in length and descends steeply from the berm of the WSP in a perpendicular direction towards the marine receiving environment (Figure 3.4a).

Primary treated effluent discharges into Baffin Bay. The nearshore environment is characterized by a shelf (1 – 8 m depth) composed mainly of rocks and gravel interspersed with patches of sand that extends ~200 m from shore before dropping off to deeper depths (Figure 3.4b). This area has a maximum tidal range of 2.5 m, and very little of the shelf is exposed at low tide. The ambient receiving water currents were generally in a cross-shore direction, though the direction of these currents (predominantly in either the E or W directions) was observed to change between studies, and were observed to be dependent on the tidal regime.

Traditional uses of the receiving waters include boating, fishing, and hunting. There is a known migratory route of Arctic Char during the summer period past the effluent discharge point. Timing of the discharge of MWW into the receiving environment attempts to avoid release of effluent during the migratory fish passage. In addition, hunting of Narwhal in the nearshore environment surrounding the community was observed during several September field studies.

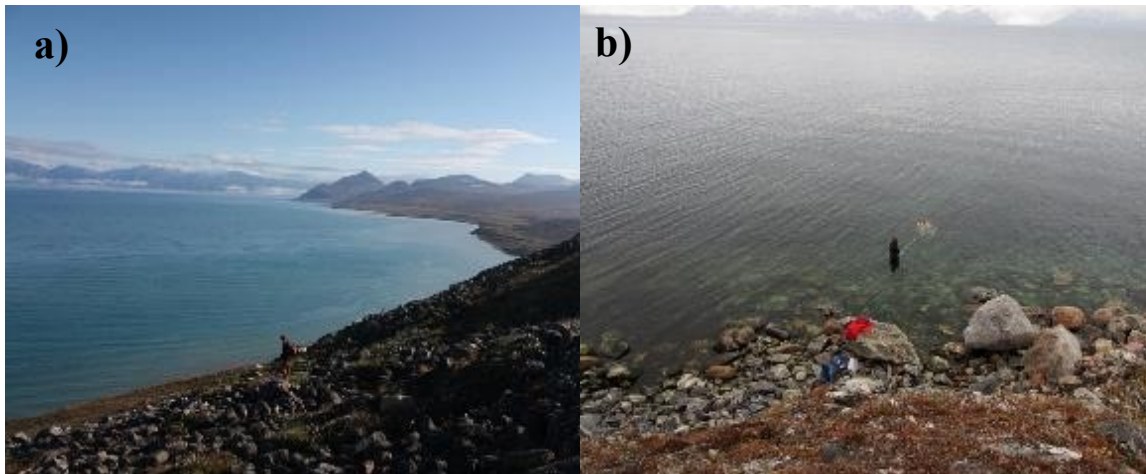


FIGURE 3.4. PHOTOGRAPHS OF POND INLET, NU SHOWING THE: A) STEEP SLOPE OF DISCHARGE CHANNEL ON AUGUST 19, 2010; AND B) RECEIVING ENVIRONMENT ON SEPTEMBER 12, 2013.

3.2 DATA COLLECTION STRATEGY

In order to provide analysis of the discharge scenarios for typical Nunavut MWWWE, the completion of various receiving water assessments at the chosen Site locations was completed. This fieldwork was completed on three separate trips during July – September 2013. The dates of the visits and the quantity of receiving water assessments completed is summarized in Table 3.2 below.

TABLE 3.2. SCHEDULE OF SITE VISITS

Location	Arrival	Departure	Number of Days	Tracer Studies	Water Quality Sampling Events
Pangnirtung, NU	23/07/2013	29/07/2013	7	8	8
Kugaaruk, NU	21/08/2013	28/08/2013	8	5	2
Pond Inlet, NU	11/09/2013	18/09/2013	8	5	4

One of the most significant logistical considerations when working in the Canadian north is transportation, which depends heavily on the weather systems in the region, which are highly variable and prone to change within short timeframes. For example, the trip to Kugaaruk required changes to the trip itinerary after flights into the hamlet were unable to land until two days after our original start date.

As these study sites all have marine discharges, considerations for the various tidal scenarios were required. As a result, the study approach focused on obtaining information from various tidal regimes: 1) Incoming Tide (IT), 2) High Tide (HT), 3) Outgoing Tide (OT), and 4) Low Tide (LT). The main objectives related to the receiving water assessments consisted of:

- Physical characterization of receiving water environment;
- Physical characterization of MWWWE discharge scenario; and
- Effluent dispersion characterization under several tidal regimes.

When determining the best approach for water quality sampling, a focus on the collection of physical/chemical parameters was chosen in lieu of the other water quality evaluation tools outlined by the CCME (e.g., Bio-assessments, WET testing), as the vast majority of wastewater treatment facilities in Nunavut fall under the small or very small categories defined by the CCME (Table 2.2).

3.3 TRACER STUDY

Tracer studies can be used to quantify the dispersion/transport of effluent in receiving water environments, conducting studies related to travel time through systems, and tracking of karst groundwater systems (Smart and Laidlaw, 1977, Jacques Whitford and Natech, 2003), among other applications. In this case, the completion of tracer studies was desired to both quantify effluent dispersion characteristics and to provide visual indications of wastewater plume transport within receiving waters under different discharge scenarios. The methodology for conducting the tracer tests was primarily designed based on the report titled *Revised technical guidance on how to conduct effluent plume delineation studies* (Jacques Whitford and Natech, 2003), supplemented with

additional information from various tracer studies documented in peer-reviewed literature (e.g. Pecly and Roldao, 2011; Carvalho et. al, 2002; Clark et al., 2007). Literature regarding tracer studies in smaller scale MWWWE discharges is not as readily available as that for larger municipal discharges, and as such, methodologies outlined in previous studies were modified in order to be both applicable and feasible in the remote northern study sites chosen. It is important to note that since the conditions occurring during the tracer tests are based on a fairly site-specific set of environmental conditions, the methodology used to complete the tracer tests differed slightly from what was suggested in literature, as well as on a site-by-site basis, as further described in the following sections.

3.3.1 TRACER DYE SELECTION

Several different types of tracer dyes are available and have been used in tracer studies, each with unique characteristics that are suited for various applications depending on overall study goals. Typical tracer dyes include Fluorescein, Rhodamine WT, Amidorhodamine G Extra, Sulfur Hexafluoride, and Sodium Bromide, among others.

The properties desired in an ideal tracer are as follows (Jacques Whitford and Natech, 2003; Allaire-Leung et al., 2000):

- Highly conservative (i.e., minimal loss of dye through decay through the study period);
- Low toxicity to the environments in which it is discharged;
- Minimal significant background levels within the receiving water environments;
- Measuring equipment that is both available and reliable; and
- Readily mixes within both the discharge streams and receiving water environments.

Rhodamine WT (RWT) is a commonly used tracer that has been reported to meet the criteria required. It has commonly been used in surface water studies, including those for effluent dispersion, hydraulic retention time, and other time of travel studies. It is recommended by both Environment Canada and US EPA (Jacques Whitford and Natech, 2003; US EPA, 2013), and was chosen as the tracer dye to be used in the receiving water

assessments as part of this study. The highly visible characteristics of RWT dye were desired in order to be able to visually observe the dye (and thus the wastewater discharge) through the receiving water environment. The characteristics of the RWT solution commonly used is approximately 20% RWT by weight, or a concentration of approximately 200 g/L (Keystone Aniline Corporation, Inman, South Carolina, United States). The specific concentrations of RWT used within these studies was reported at between approximately 21 and 21.57% RWT by weight, or approximately 210 g/L.

The limitations reported with RWT, such as its tendency to biodegrade and adsorb within highly organic environments, to degrade through exposure to light (photolysis), and its sensitivity to temperature (Smart and Laidlaw, 1977; Headley and Kadlec, 2007), were considered prior to dye selection. As the degradation of RWT is reported to be of concern in studies with longer time scales than what was considered in this Study (i.e., weeks vs. hours), this limitation was deemed acceptable given the other benefits that RWT provides. In addition, the sensitivity to temperature was assumed to be negligible, as the RWT measurements taken have been used to define dilution factors for each individual study (e.g., compared against similar temperatures), thus removing any requirement to consider changes in temperature between studies.

As RWT is a very bright red-pink colour when mixed in water, it was also necessary to notify the communities and relevant regulation authorities to its use. By doing so, any potential community concerns were more easily mitigated prior to study commencement. Notification letters regarding the planned tracer studies were sent to the Senior Administrative Officer (SAO) for each of the Kugaaruk, Pangnirtung, and Pond Inlet communities, as well as to Aboriginal Affairs and Northern Development Canada (AANDC), the Nunavut Water Board (NWB) and DFO. None of the contacted stakeholders expressed any concerns related to the proposed works.

3.3.2 TRACER INJECTION METHOD

There are two options for injection of the tracer dye typically used in tracer studies: slug injection (also referred to as ‘pulse’ or ‘impulse’ injection), or continuous injection. Slug injection allows for the development of a dye ‘cloud’ that can be then measured as it is

transported through the receiving water environment. Continuous injection, on the other hand, forms a more developed dye plume.

Through considerations of site locations, material transport, and study objectives, it was determined that the slug injection method of tracer injection was the most appropriate methodology for this study. As the study is concerned with characterizing the transport and dispersion of the effluent discharge in the receiving water environment, the information given by completing a study using slug injection is able to provide sufficient data to support this analysis, while allowing for minimal transport of RWT and equipment to each community. As all the equipment required to complete each tracer study equipment had to be transported to remote communities, minimizing the quantity of RWT and the associated equipment required to perform the tracer study that required transported was mandatory. In addition to these logistical challenges, certain study sites required that the equipment needed to complete the tracer study be carried over a significant distance of challenging terrain (e.g., over the bluff in Pond Inlet). As a result, the careful planning of what equipment was required in each tracer study was critical to ensure both the feasibility of the study and the health and safety of the field crew.

3.3.3 TRACER INJECTION LOCATION

While the location of the tracer injection differed between each study site due to site-specific restrictions that were observed, in each case the injection location was chosen to: 1) be consistent throughout the tracer studies undertaken at a particular site, 2) to allow for complete mixing of the dye throughout the discharge channel prior to discharge, and 3) to minimize travel time prior to discharge to the receiving water environment. The minimizing of travel time was desired due to several factors. As the tracer study team was generally small, having a dye input close to the receiving water area was desired in order to ensure team members were in proper locations prior to dye entrance in the receiving water environment. In addition, the amount of undesired dying of the wastewater discharge channel and surrounding area could be minimized, which was desired while ensuring complete mixing of the dye-wastewater mixture prior to entry in the receiving water environment.

Kugaaruk

As the effluent discharge pathway in Kugaaruk ends with subsurface transport through a cobble/boulder matrix, the dye injection location at Kugaaruk was challenging to select. In order to ensure that the largest percentage of the total effluent discharge was dyed and able to be tracked in the receiving water environment, a location was chosen upstream of the main discharge channel reaching the rocky shoreline zone. As a result of this and the mentioned subsurface flow, the dyed effluent had a larger distance to travel than the other study sites, with a much longer travel time. Compounded by the inability to follow the effluent stream through the subsurface rocky area, both the selection of an appropriate quantity of the dye and determination of dye input in the receiving waters was challenging to predict at this study site.

Pangnirtung

At the Pangnirtung study site location, the grassy effluent discharge channel was fairly well defined at the time of our study, allowing for consistent and straightforward dye injection for each tracer study undertaken. The location selected was approximately 5 m upstream from the observed high tide water level, and was kept consistent throughout the studies in order to factor in the effect of the tidal beach into the analysis, as the tidal influences at this location are significant (as outlined in Section 3.1.1).

Pond Inlet

As the effluent discharge travels over the bluff towards the receiving water, the discharge channel separates into two larger flow paths in the same vicinity, and one smaller flow path that meanders further away. At first attempt, the tracer dye was injected in the effluent stream before the stream splits. Due to the non-uniform discharge pathway through several types of channel materials (grassy channels, tundra, rock deposits), the dye entered the receiving water more like a continuous dye injection, and less of a slug injection as desired. As a result of this first attempt, the dye injection location was moved to the bottom of the bluff, approximately 5 m upstream of the entry of the effluent discharges into the receiving water environments. In order to best analyze the majority of the effluent discharges, two simultaneous dye injections were made in the two larger effluent discharge streams, which were located approximately 3 m apart.

3.3.4 TRACER INJECTION QUANTITY & MEASUREMENT

The RWT sensors were both calibrated prior to each field study, using 2-point calibration using RWT solutions prepared in the Dalhousie water quality lab, using ‘True Dye’ measurements as outlined in the technical support documents (YSI, 2005). The quantity of RWT used at each location varied slightly, as attempts were made to ensure that RWT was both visible within the receiving water environment and remained within the specifications of the RWT sensor. The RWT sensors used on both probes deployed as part of the study were YSI 6130 Rhodamine Sensors, with specifications as reported found in Table 3.3 (YSI, 2006).

TABLE 3.3. YSI 6130 RHODAMINE SENSOR SPECIFICATIONS

Measured Parameter	Range	Resolution	Accuracy
Rhodamine	0 – 200 µg/L	0.1 µg/L	±5% reading or 1 µg/L, whichever is greater

Notes: ¹ – Table recreated from YSI Rhodamine Sensor specification (YSI, 2006)

As such, attempts were made to ensure that the maximum concentrations measured within the receiving water environment were around 200 µg/L, as this allowed for strongest dye visibility as well as a large range of dilutions to be measured.

3.3.5 TRACER STUDY PROCEDURES

The tracer studies completed were based on a fairly specific (and unique) set of environmental conditions that were aligned with the discharge conditions encountered at each study site. As a result, the methodology used to complete the tracer studies not only differed from what has been previously presented in the literature, but also on a site-to-site basis. That being said, the tracer studies generally consisted of the following, as described visually in Figure 3.5:

- The measurement of a quantity of RWT dye, at approximately 20% RWT by weight (as described in Section 3.3.1), diluted in a holding vessel (e.g., bucket,

large bottle, etc.) at a ratio of at least 10 parts wastewater to 1 part RWT in order to ensure matching densities prior to discharge;

- Injection of the resulting dye mix to the wastewater stream at a location far enough upstream of the receiving water in order to ensure mixing throughout the discharge channel prior to discharge;
- Measurements of the dispersion of the RWT dye within the receiving water environment through the use of two YSI water quality sondes (WQ sondes) with RWT sensors;
- The use of Global Positioning System (GPS) equipment to track the location of the WQ sondes as transects are made through the observed dye cloud in the receiving water environment; and
- Water quality samples taken at various points both within, and at the edges of, the observed dye cloud and analyzed as described in Section 3.4.

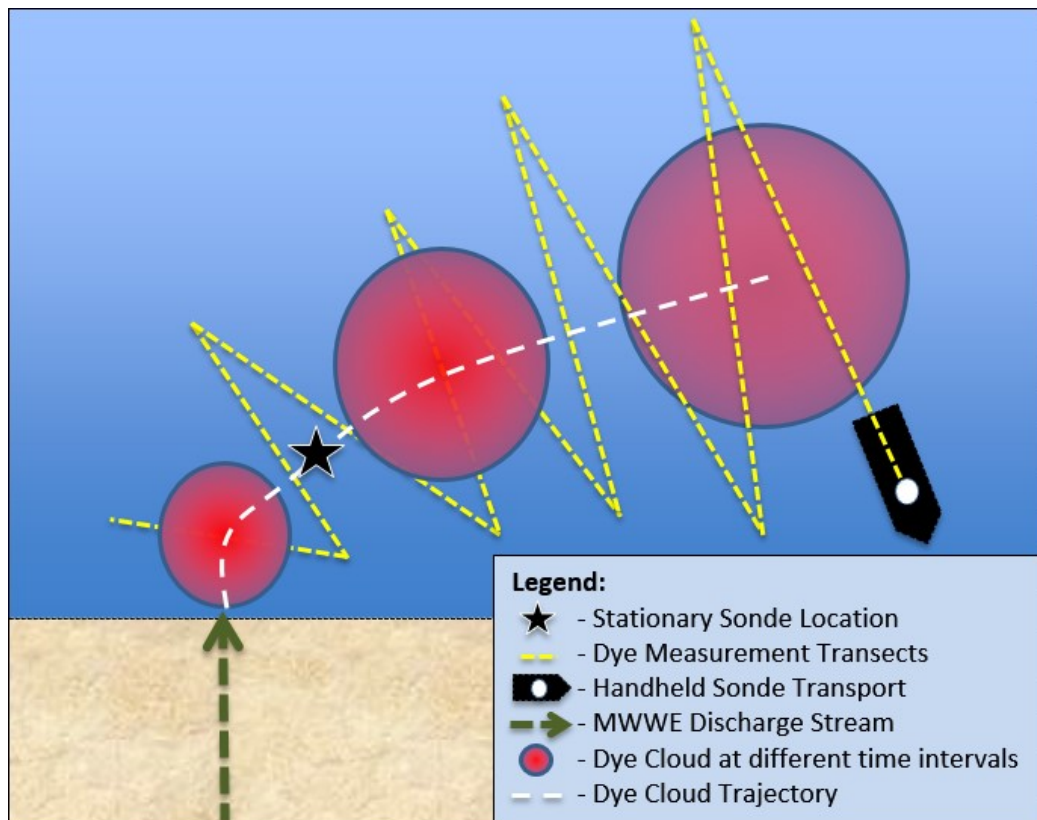


FIGURE 3.5. CONCEPTUAL DIAGRAM OF THE TRACER STUDY MONITORING TECHNIQUE.

As outlined in Figure 3.5, one WQ sonde was installed at the predicted centerline of the dye cloud prior to the injection of the RWT to gather data associated with the transport of the dye through the receiving waters as collected from a set location. The other WQ sonde was handled by a member of the field crew and was pulled through the receiving waters in transects through the dye cloud as it travelled through the receiving waters.

As transects were being completed, information regarding the timing of the start and end of each pass were reported, in cases where this information was necessary (i.e. cases where multiple passes of the same location were completed, cases where transects were not completed in a successive order, etc.). At sites where doing so was feasible, a weather resistant camera was set up to take photos of the dye transport surrounding the discharge location at a set time intervals was used to provide additional data for further analysis. These interval photos were supplemented by additional photos and videos of the transport of the dye through the receiving water environment.

These tracer studies were completed for various tidal regimes for each of the Kugaaruk, Pangnirtung, and Pond Inlet study sites. Each community had specific receiving water conditions that required changes to the method of completing transects, such as differences in nearshore bathymetry, significance of tidal influences on nearshore conditions, and significance of wave action in the nearshore environment. As some conditions were inappropriate for the use of boats, a level of flexibility in determining appropriate collection strategies was required. The information collected within these tracer tests was then used to create ‘dilution zones’ to illustrate the observed wastewater mixing zones at these study sites. These dilution zones also provided insight into the transport characteristics at each location.

GPS

In order to define zones related to various dilutions surrounding the discharge location, a knowledge of the locations of both the RWT measurements and the water quality samples was required. Due to differences in site and sampling locations and field conditions, two different methods of collecting this data were used throughout the study. When feasible, location data was collected using Real Time Kinematic (RTK) GPS equipment. In doing so, the Topcon HiPer Ga Real Time Kinematic GPS System (RTK) was used, which

provides corrected location data with reported accuracies of approximately 1 cm in the horizontal and 1.5 cm in the vertical axes (Topcon, 2007). The use of the RTK system was not always feasible, depending on the site specific conditions and transect collection strategy required for each site. For example, it was not feasible to install and operate the RTK equipment on the kayak in Kugaaruk, nor was it feasible to use the RTK in Pond Inlet, as the significant bluff at this location made the distance between the location of the base station and the remote unit too large. As a result, handheld Garmin eTrex 20 and Garmin Montana 650 units were used when use of the RTK system was not feasible. These devices provided a horizontal accuracy of approximately $\pm 2 - 5$ m, which although far less accurate than the RTK system, was deemed to be acceptable for the level of detail required in the study.

Kugaaruk Specific Considerations

In the case of Kugaaruk, the discharge scenario observed at this location required changes to planned tracer study events and procedures. Once on site, discharge into the sub-surface flow area (as described in Section 3.1.2 and in Figure 3.3b) was observed to significantly dampen the already minimal flow prior to discharge to the receiving water environment. As a result, a test run of RWT injection was completed prior to water quality sampling studies taking place in order to gain an understanding of the transport timing and extents. Observations from this test run suggested that the pulse injection of RWT at this location was responding much differently than the other sampling locations, in that the time of release of RWT into the receiving water was significantly stretched, making a dye cloud that was much less defined in shape and location than the other studies. As a result, the tracking and measurement of the maximum dye cloud extents was not possible.

In order to provide the most opportunity for future analysis of measurement data, the sampling procedure was modified to include two separate transect measurements to occur concurrently, one in the nearshore approximately 1 – 10 m from the discharge location by a field staff walking wearing fishing waders, the other taking measurements of the values at surface at locations 10 – 100 m from the discharge location via kayak. In order to

complete these concurrent measurements, the stationary WQ sonde set up in other studies was used by the nearshore staff member instead.

When taking water quality samples, locations representing the approximate edges and centerline of the MWW discharge plume were chosen based on dye observations. As the dye concentrations (and thus fluorescence) was affected by the subsurface flow, the selection of these locations were based on both dye observations and previously obtained readings at each location, with sampling distances from the discharge location chosen to provide approximately equal distances between measurements.

Pangnirtung Specific Considerations

As described in Section 3.1.1, Pangnirtung has an extensive tidal beach, which extends approximately 300 m from the high water mark at low tide (Figure 3.2a). Based on RTK surveying, the nearshore bathymetry at this location includes a slope of approximately 7.5% from the HWL measured at HT to approximately 30 m into the tidal beach, which reduced to a slope of approximately 1.3% prior to reaching the rock wall observed at the edge of the tidal beach. As a result of this tidal beach slope, the minimal observed transport at this location, and the rapidly changing tidal conditions observed, the use of a boat to conduct this study was not feasible.

Prior to first tracer study attempts, a rough characterization of the receiving water currents was undertaken through the deployment of several drifter oranges (following the approach of US ACE, 2002). The receiving waters were observed to be stagnant, and thus the collection of both dye measurements and water quality parameters by field staff in waders was deemed the most appropriate approach. Although the currents did increase in several of the tracer study sampling events, the collection using this approach remained appropriate. In each tracer study, one field staff was outfitted with the RWT Sonde and a backpack containing the RTK receiver to collect positional data, with another field staff charged with the collection of water quality samples at the locations defined by both RWT measurements and dye fluorescence observations.

Pond Inlet Specific Considerations

The much larger MWW discharge rate observed at this location, combined with a sufficiently deep nearshore bathymetry near the discharge, enabled the use of a boat for the collection of transect information at this site. A boat rental was procured from a local resident to be used in performing the desired receiving water transects. Out of the sites of study, this location followed the planned conceptual tracer study most closely. The addition of a boat allowed for much simpler deployment of the drifter drogue, the stationary WQ sonde, and allowed for the most effective collection of RWT transect measurements.

One field staff was located at the bow of the boat, and held the RWT sonde within the top ~15 cm of the receiving water. There was another field staff in the back of the boat that filled the required bottles at each desired sampling location.

One additional consideration at Pond Inlet was the degree of wave action observed during the field trip. Two planned tracer study days had to be cancelled due to the marine conditions, as the boat captain did not feel comfortable operating the vessel. As a result, tracer was injected into the MWW on these days, but only observations and measurement of parameters in areas that could be accessed safely by field staff wearing waders could be obtained.

3.4 WATER QUALITY MONITORING

As shown in Table 2.2, the majority of municipal wastewater treatment facilities found in Nunavut are categorized under Small or Very Small. As a result, the following substances have been defined by the CCME *Strategy* as being of potential concern:

TABLE 3.4. LIST OF POTENTIAL SUBSTANCES OF CONCERN FOR MWWE – VERY SMALL AND SMALL FACILITIES (SOURCE CCME, 2009)

Test Group	Substances
General Chemistry/ Nutrients	<ul style="list-style-type: none"> • Total Suspended Solids (TSS) • Carbonaceous Biochemical Oxygen Demand (CBOD₅) • Total Residual Chlorine (TRC) if chlorination is used • Total Ammonia Nitrogen • Total Kjeldahl Nitrogen (TKN) • Total Phosphorus (TP) • pH • Temperature
Pathogens	<ul style="list-style-type: none"> • <i>E. coli</i> (or other pathogen, as directed by the jurisdiction)

These parameters were considered when determining a final suite of water quality parameters to be analysed as part of the field studies completed.

3.4.1 DISCRETE SAMPLING

Water quality samples were taken while tracking the tracer dye plume movement and were analyzed for various water quality parameters, using both the List of Potential Substances of Concern for MWWE for Very Small and Small facilities (i.e. < 2,500 m³/day MWWE discharge as outlined in Table 2.2; CCME, 2008) and taking into consideration potential other substances of interest. The parameters analysed are outlined below:

- Total Suspended Solids (TSS);
- Total Nitrogen (TN);
- Total Ammonia Nitrogen (TAN);
- Un-ionized ammonia (NH₃);
- Total Phosphorus (TP);
- Metals;
- Chlorophyll-a (Chl-a);

- *Escherichia Coli (E. coli)*; and
- *Enterococci*.

As chlorine was not part of the treatment process at the locations that were a part of the field program, no considerations for TRC sampling were made. *E. coli* and enterococci were used as indicator organisms to assess microbial fate and transport in receiving waters. Enterococci was added to the list of parameters to be sampled as it is reported to potentially represent a better indicator of contamination in marine environments than fecal coliforms, due to its ability to survive in these saltwater environments (Hydromantis Inc et al., 2005). When possible, both *E. coli* and enterococci were measured. However, there were several occasions where the laboratory conducting the analysis was only able to analyze for one of the indicator organisms. Water quality criteria of 35 MPN/100 mL for enterococci and 200 MPN/100 mL for *E. coli* were used to characterize human health exposure risk (HC, 2012). These numeric criteria are based on the geometric mean sample criteria proposed by Health Canada, as this represents a more stringent requirement than the single sample maximum concentration criteria (HC, 2012). Samples were analyzed for the broader fecal streptococcus group and not solely the subgroup enterococcus at the Kugaaruk location by the laboratory used; these results have been assumed to be analogous to the other enterococcus results to be conservative.

The locations where water quality samples were collected were determined based on the tracer study results. For example, samples were collected where the dye tracer study concentrations were the highest, as well as along the dye tracer plume boundaries (based on both dye observations and RWT measurements). Reference sites were also selected at each of the three study communities. These reference sites were representative of areas of the marine environment where there are no human impacts from MWWWE affecting the ambient water quality. These reference sites were selected to have similar site conditions as the receiving water site, and were located 2.5 – 5.0 km from the effluent discharge point. The locations of the reference sites in relation to the MWWWE are shown in Figure 4.2 to **Error! Reference source not found.**

Samples were either collected in 1 L or 500 mL sterilized Nalgene water quality sampling bottles, or in 25 mL falcon tubes. Sample bottles were rinsed three times in the receiving

water prior to sample collection. These samples were then placed in coolers surrounded with ice packs in order to ensure temperatures below 4°C, and shipped through air cargo at the earliest opportunity in order to ensure arrival to the target laboratory. For both the Pangnirtung and Pond Inlet field events, samples were shipped to the Dalhousie University Northern Water Quality Laboratory located at the Nunavut Research Institute (NRI) in Iqaluit, Nunavut. Samples were then stored in a refrigerator at 4°C prior to analysis. In the case of Kugaaruk, samples for all parameters except metals and TN/TP were sent to the Taiga Environmental Laboratory in Yellowknife, NWT (NWTENR, 2015) in order to ensure that the proper sample holding times could be met, as shipments to Iqaluit were not feasible from this location. The metals and TN/TP samples were preserved using nitric acid and Sulphuric Acid/HCl, respectively and processed in Halifax.

The organization and scheduling of each field trip considered the shipping of samples back to the water quality lab in Iqaluit in order to have samples analyzed within their respective holding times specified by the manufacturers and/or APHA (2012). This often involved changing desired field plans in order to ensure synergy with local aircraft schedules.

Sampling Procedures

CBOD₅, and TSS were analyzed according to Standard Methods (APHA, 2012). *E. coli* was analyzed using IDEXX Colilert[®]-18 and Quanti-Trays[®] according to the manufacturer's procedure in IDEXX Laboratories, Inc. (2012).

TN was analyzed using Hach[®] TN Test 'N Tubes[™] (0.5 to 25.0 mg/L N), according to the manufacturer's procedure in Hach (2012a). Low concentration TP samples were analyzed using the Hach[®] TP Test 'N Tube[™] for low range samples ranging from 0.06 to 3.50 mg/L PO₄³⁻ according to Hach (2012b). High concentration phosphorus samples, Hach[®] TP Test 'N Tube[™] for high range samples ranging from 1.0 to 100.0 mg/L PO₄³⁻ were analyzed according to Hach (2008).

Total ammonia was analyzed using a Thermo Fisher Scientific High Performance Ammonia Ion Selective Electrode (ISE) according to Thermo Fisher Scientific (2007).

The ISE probe was used in conjunction with a Thermo Scientific Orion Star™ and Star Plus Meter (Thermo Fisher Scientific, 2010).

Finally, an understanding of the volumetric mass density of the samples taken along the measured path was desired in order to collect information regarding the differences between the MWW and ambient receiving waters. This information was measured using simple mass/volume measurements.

3.4.2 *IN-SITU* SAMPLING

Additional water quality data, including temperature, dissolved oxygen, pH, and electrical conductivity was collected *in-situ* through the deployment of a YSI 6-Series Multiparameter water quality monitoring Sonde (YSI, 2011). This information is desired to help characterize the receiving water environment in order to better evaluate the potential for MWW impacts. For example, both the temperature and pH within the IMZ influences the conversion of ammonia (NH_4^+) to un-ionized ammonia (NH_3), which represents a more toxic form of further concern. In addition, the temperature within the receiving water environment influences the mixing characteristics of the MWW discharge through differences in their buoyancies, as described in Section 2.4. In addition, the measurements of electrical conductivity can validate the density measurements that were measured in the lab.

3.5 CURRENT MEASUREMENTS

Information regarding the ambient currents is required as supporting information for transport analysis. After analysis of several different methods of measuring surface currents, the use of a drifter was deemed the most feasible for this project from a perspective of cost feasibility, ease of transportation, and level of quality of the data desired.

3.5.1 DAVIS CODE DRIFTER DESIGN

Considering the depths expected at the discharge locations, the anticipated field conditions, and equipment transport requirements for each study Site, a robust, inexpensive, and repairable drifter design was desired. After researching suitable current

measurement devices for this purpose, a Davis CODE drifter design was selected, similar to what is described in “*An Inexpensive Drifter for Surface Currents*” by Davis (1982, 1985). This basic design has been modified and deployed in many nearshore applications by different research agencies (NOAA, 2010; MSS, 2014), and systems are available from several manufacturers (Brightwaters, 2015; MetOcean, 2015). In order to ensure that the drifter could be easily repaired if needed, it was constructed using the outline provided by the Northeast Fisheries Science Centre (2015), using parts readily available at local hardware stores. The drifter used in this study was designed to have similar submerged sail area (approximately one square meter cross section), materials, and design to minimize above water device area to reduce potential wind effects on drifter movement (Figure 3.6).

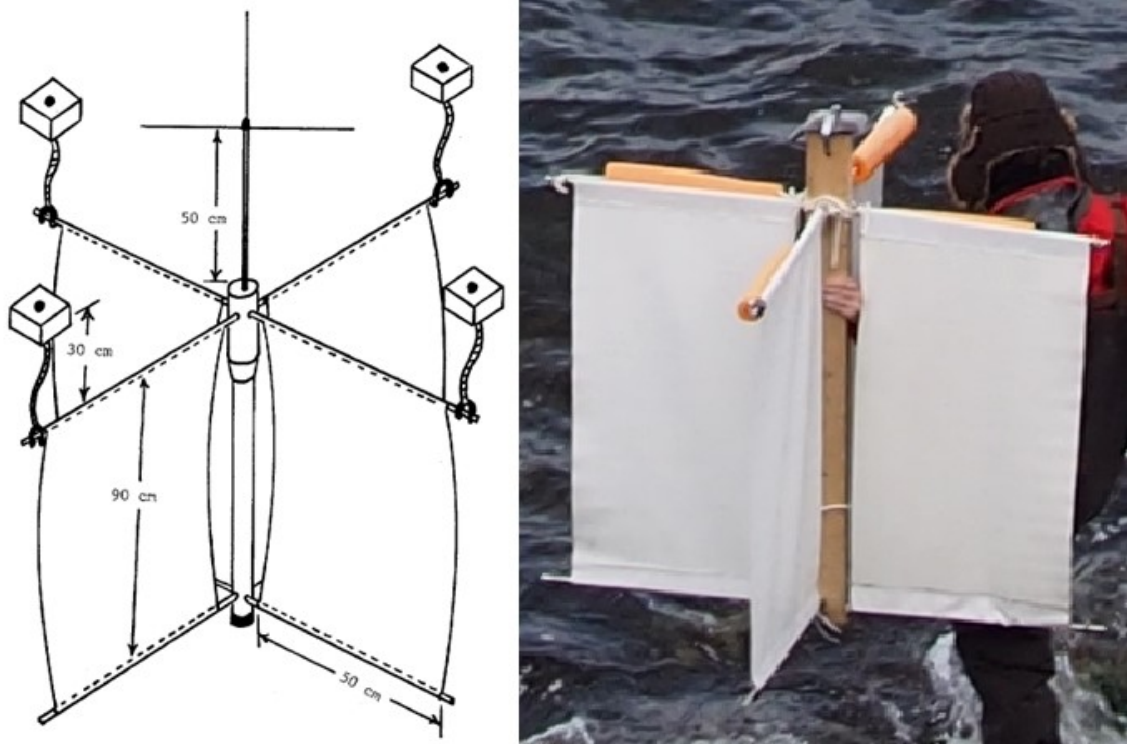


FIGURE 3.6. LEFT PHOTO: TYPICAL DAVIS CODE DRIFTER DESIGN (SOURCED FROM DAVIS, 1982). RIGHT PHOTO: MODIFIED DRIFTER USED THROUGHOUT ANALYSIS.

In order to determine the receiving water currents, a handheld GPS (Garmin eTrex 20) was then attached to the mast of the drifter device in a waterproof bag and set to track

locations every second. The GPS then tracked the movement of the drifter, with the current speed and direction logged for future analysis.

With these design parameters, a drifter was constructed and tested in several water bodies (both fresh and saltwater) prior to its deployment at the Study Sites. As a result of the findings of these test deployments, a second revision of the drifter was constructed, with additional considerations for durability and potential changes in receiving water buoyancy in mind. Since the potential for the drifter to be damaged through transport/deployment was thought to be fairly high due to the overall difficulty of travel between northern communities, the second revision of the drifter was constructed out of materials with the largest likelihood of being available on Site if needed.

Current measurement was through deploying the drifter in the nearshore environment at a depth where it was observed that the drifter would not have contact with the ocean floor, in areas observed to have the least amount of potential obstacles. The drifter was deployed either by an individual wearing chest waders (Kugaaruk, Pangnirtung, when possible at Pond Inlet), or by boat (Kugaaruk, Pond Inlet). The drifter was then left to travel without disturbance, with the time of deployment and the time of pick up noted for future GPS analysis.

In order to validate these current readings, the drifter was deployed in the Musquodoboit River in Nova Scotia prior to field trips taking place and additional current measurements were taken using a pygmy style velocity measurement system (625DF2N digital pygmy meter, Gurley Precision Instruments, Troy, New York, United States) at different locations and depths at different cross-section locations throughout the river. Although difficult to compare exact velocities using this methodology, it provides insight into the high-level reliability of the drifter's measurements. The differences observed between the average readings of the GPS on the drifter device and the spot readings using the pygmy system were observed to be in the $\pm 50 - 100\%$ range, depending on the overall magnitude of current recorded (i.e. larger percent difference in smaller current magnitudes). As current information inherently has a large variance, and the data is being used to determine bounds as part of the overall analysis, this level of accuracy is acceptable.

It must be noted that while the deployment of the drifter provides adequate current velocity and direction data for high-level analysis, it represents only the current conditions at that particular time and location.

3.5.2 ADDITIONAL METHODS

As the extent of MWWWE transport in the receiving water environments was unknown prior to the completion of any field characterizations, additional methodologies for obtaining rough estimates of surface currents were devised prior to the field activities taking place. The use of oranges as an inexpensive high level drifter for observational purposes was reported, and thus considered as a way to obtain additional high level current data where needed (US ACE, 2002). In the cases of orange deployment, observations based on movement of the orange through the receiving water environment and the related times taken were recorded.

DFO was contacted in order to obtain any current data that may be useful for consideration in analysis. Evaluation of the data they provided determined that the current measurements available were not at an appropriate scale and measured depth for comparison with data measured using the techniques above.

3.6 RECEIVING WATER CHARACTERIZATION

3.6.1 ADDITIONAL DATA COLLECTION

Discharge Measurement

Knowledge related to the quantity of MWWWE discharges to the receiving water environments is critical when developing an understanding of transport within receiving waters and overall potential environmental impacts. As such, measurements of MWWWE discharge prior to the release point in the receiving waters was completed at each study Site using a pygmy system comprised of a 625DF2N digital pygmy meter, a 1100 model digital readout, and 2 m wading rod. Measurements of both depth and overall channel width at each measurement location were also recorded as part of the discharge measurement. The discharge channels in all cases were relatively small (i.e. less than 20 cm depth and 40 cm width in all cases across all study Sites). Wherever possible,

attempts were made in all cases to measure at 60% of the depth as measured from the water surface, as standard for depths less than 0.75 m when stream gauging (Dingman, 2002).

The total discharge, Q (m³/s), for each discharge measurement location was calculated using:

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

Where:

v_{avg} = the average current velocity at that location based on at least three measurements (m/s), and

A_i = estimated area of each section based on width, depth, and observations (m²).

At each study Site, discharge measurement locations were chosen based on which locations represented the best opportunity to characterize the total discharge to the receiving waters, and as such changed on a site-to-site basis. In Kugaaruk, discharge measurements were taken at a combination of locations in order to best characterize both the discharge leaving the wetland and the discharge entering the receiving water environment. The largest discharge observed entering the receiving water environment has been used in this analysis, as observations on Site showed this to be the only measurable discharge stream visible during analysis. In Pagnirtung, discharge measurements were taken at the same location as both the wastewater water quality samples, as the MWW discharge channel was well defined and free of obstruction at this location. In Pond Inlet, the measurement of MWW discharge was taken just after the discharge pipe from the WSP within the riprap channel. This location was chosen due to the difficulties in measuring discharge closer to the receiving waters release point at Pond Inlet; as the discharge travels over the bluff towards the receiving water, the channel passes through a vegetated, rocky tundra, where it is near impossible to track and measure. As a result, the only feasible location for discharge measurement occurs prior to

the bluff section. The measurement of small discharges, such as those observed at each site covered in this study, is difficult to complete with a large level of certainty. The measurement of flow velocities at small depths is difficult, and may not be completed at the required 60% depth criteria. As a result, the values collected represent an order of magnitude measurement of expected flowrates from these systems.

Bathymetric Data

Nearshore bathymetry was collected for the Pangnirtung site through the use of the RTK unit, as the significant tides in the area allowed for the straightforward collection of this data during LT. Attempts were made to collect this information in Kugaaruk, but the significant wave action typically observed at this location made measurement difficult. As a result, only the nearshore tidal area data was collected at this location. In Pond Inlet, the significant wave action paired with the fairly deep nearshore environment made taking RTK measurements of this area unfeasible. As a result, measurements of overall water depth were taken using a depth sounding device in order to collect high level data to be used in determining nearshore bathymetric characteristics.

For the Kugaaruk and Pond Inlet study Sites, additional bathymetric data was provided by the Canadian Hydrographic Service.

Weather and Wind Data

Meteorological information was collected on Site using a portable weather station. The Kestrel 4500 Weather Meter (Nielsen-Kellerman, 2015) was used to measure wind speed, direction, air temperature, relative humidity, and other meteorological parameters. Additional details regarding the meter can be found in the Kestrel 4500 specifications (Nielsen-Kellerman, 2015).

3.7 DEVELOPMENT OF DILUTION ZONE MAPS

The plumes of the MWWE in the receiving water were delineated to determine approximate zones of dilution using the RWT concentrations measured within the nearshore environment. These zones of dilution represent the boundaries of: 1) minimal dilution, where less than 3x dilution of the concentration of RWT occurred (e.g., 1 mg/L diluted to 0.33 mg/L); and 2) increased dilution, which has between 3 and 15x dilution of

the concentration of RWT (e.g., 1 mg/L diluted to between 0.33 – 0.07 mg/L). These two dilution zones were arbitrarily selected as 3 and 15x dilutions through an analysis of the measured field data and are presented to provide a visual illustration of the plume behavior in the plume delineation maps that were generated for each site, with the intent of providing information to support analysis of the water quality results (See Section 5). The approaches used to delineate these dilution zones are described below.

In general, the following steps were completed when developing the dilution zone maps (Figures found in Section 4):

- GPS tracks collected during the tracer study were downloaded to the computer using Garmin Basecamp software and edited into collections for each specific tracer study based on start and stop times written in field notes. This was completed in Garmin Basecamp as this software allows for detailed viewing of time-related information stored within the GPS data;
- The edited GPS tracks were imported into the ArcMap GPS, and critical points were identified within the GPS tracks (e.g., water quality sampling locations, high water levels (HWLs), other data of interest) based on the field notes taken;
- The GPS track data are then combined with the records of RWT measurements in order to tie the RWT data to locations throughout the receiving waters based on the appropriate timestamps;
- These combined datasets were examined manually or with additional analysis using Surfer software (for Kugaaruk, further described in Section 3.7.2 below), and dilution zones based off of the RWT concentration reductions observed from the highest measurements in the nearshore environment were developed; and
- As several tracer studies were completed, the dilution zones were created through superimposing the dilution zones for each tracer study over each other and connecting the zones of dilution defined within each separate tracer study event.

Dilution used in determining dilution zones was calculated using the following calculation:

$$Dilution = \frac{(C_{eff} - C_{amb})}{(C_{plu} - C_{amb})} \quad [3.2]$$

Where:

C_{amb} = Concentration of parameter of interest in ambient receiving waters (i.e., background concentration)

C_{eff} = Concentration of parameter of interest in effluent prior to entering receiving waters

C_{plu} = Concentration of parameter of interest in RWT plume (i.e., at particular distance within the receiving waters)

In defining C_{amb} , the concentrations associated with the reference locations were used. The concentrations of effluent were taken at the point closest to the discharge location along the treatment train (i.e., at the end of any treatment train that exists). For RWT, values for C_{eff} were taken as the highest measured RWT values in the nearshore environment. This approach for calculating RWT dilution was selected after comparison of these results against the dilution factors calculated based on water quality results, as this approach provided results much more comparable than an approach based on the RWT pulse injected in the MWWWE upstream of the discharge location.

One of the critical components of both the successful completion and effective analysis of tracer studies is ensuring that all equipment (e.g., GPS, team watches, WQ sondes, etc.) have their times synchronized. This is critical in ensuring that all data collected is consistent and comparable with all other data, and is of the utmost importance in effective processing of the data. As a result, efforts were made prior to every sampling event to ensure that all times were within ± 1 second of each other.

Additional considerations were required on a site-by-site basis and are further described below.

3.7.1 PANGNIRTUNG

The dilution zone map prepared for Pangnirtung outlines the study boundaries (i.e., the area within which transects were completed) with zones of minimal dilution (i.e., <3 times dilution) identified. Efforts were made to ensure that the entirety of the RWT plume was covered by the tracer study transects in all tidal scenarios. For the OT study, both a lack of boundaries and the velocity of the outgoing tides made it difficult to track the RWT as it travelled through the tidal beach area – as a result, tracking all branches of the discharge plume was not feasible. Field observations, in combination with the processed data, suggest that several additional branches of RWT existed to the E and W of the main branch followed and covered within Figure 4.2. As such, the overall extent of MWE transport in the OT case may be underrepresented on Figure 4.2, but provides an adequate approximation of the travel extents of the most extensive plume observed.

From a comparison against the dilution numbers produced by the water quality sampling events, the dilutions calculated for RWT have good correlation while being conservative. In particular, in the OT case, dilutions based on RWT concentrations were observed to be significantly less than those observed through the water quality results. As a result, zones of minimal dilution have been based on RWT values alone to ensure conservative values have been presented.

3.7.2 KUGAARUK

In addition to the general steps outlined above, additional analysis was completed at Kugaaruk to account for site-specific tracer study details:

- The case of Kugaaruk differs from the rest of the analyses in that the subsurface flow required a modification in the tracer study approach as outlined in Section 3.3.1.2.
- As in this case it was extremely difficult to know whether or not the ‘centre’ of the plume was being measured (i.e., highest plume RWT concentrations), several transects over the same dyed locations were made in order to capture an increased set of data.

- In order to determine approximate dilutions for this study, the combined GPS – RWT data from the several transects that took place were input into the Surfer program (Golden Software, 2014), and a 1 by 1 m gridded overlay of the study area was created using the combined GPS – RWT data.
- This grid was developed in the Surfer software to provide only the maximum RWT concentrations at each grid location in order to provide a composite look at the RWT measurements taken during the tracer studies.
- This data was then used in determining approximate values for RWT dilutions throughout the study area.

As there were inconsistencies observed between the RWT measurement transects taken by the two WQ sondes, dilution zones were based off of the measurements of the WQ sonde that provided the most amount of spatial data. In particular, in the case of the August 26th tracer study, it was observed that the measurements of RWT obtained by the nearshore transects was very low compared against the values obtained by those taken further from the discharge location. As TSS values in the receiving waters were observed to be quite high during the round of sampling (68 – 120 mg/L), it is possible that this increased TSS interfered with the readings of the RWT (Edwards et al., 2010). As a result, values from the more consistent measurement WQ sonde were used in delineating dilution zones for this particular tracer study.

Dilution zones developed through RWT analysis were then compared against dilution factors calculated using the water quality results, with reasonable agreement. Through this analysis, it was observed that typical dilution numbers associated with the water quality parameters of concern occurred at a greater rate than what was measured in RWT, and as such, dilution zones based on RWT values provide a conservative illustration of potential dilution surrounding the discharge location.

In addition, it must be noted that while completing this field work, the pump from the WSP to the decant cell failed. As the decant cell was still full, there was discharge from the system into the treatment wetland and receiving waters, but this flow rate may have been reduced as compared against typical discharge events.

3.7.3 POND INLET

Due to equipment failure during several sampling runs completed at Pond Inlet, a slightly modified approach had to be undertaken when developing dilution zones. During several of the tracer studies, the WQ sondes did not record information due to communication failures with the handheld readout units. As a result, continuous RWT measurements are unavailable for several of the tracer studies completed at this location. In order to develop dilution zones, measurements of RWT were completed using the collected water quality samples after sampling runs were completed. Dilution zones were then created through a comparison of these results against field observations, and by linearly interpolating distances between samples based on the observed reduction in RWT concentration. As a result, the dilution zones have lower resolution for the Pond Inlet location.

As further described in Section 4.1.3, RWT travelled in a thin layer on the surface of the receiving waters in several of the tracer studies completed at Pond Inlet. An effect of the thin RWT layer was that it became hard to visually track in the receiving waters as the plume moved away from the discharge point. As a result, the centerline of the plume was difficult to identify and follow. In the largest transport scenario where the plume travelled W, RWT measurements after tracer study completion showed that the transects were off center, and true plume width was likely larger and spreading in a northern direction.

When compared against the dilution values obtained through water quality sampling, there is fairly good agreement with several of the parameters against the values obtained through RWT. In all cases, RWT values represented a conservative estimate of observed dilution within the travelling plume, and has been used in defining the zones represented in Figure 4.6.

CHAPTER 4. RESULTS

4.1 DILUTION ZONES

4.1.1 PANGNIRTUNG

Tracer study monitoring was conducted by wading in Pangnirtung due to the shallow receiving waters. Figure 4.1 shows the dye study during incoming (a) and high tide (b) in Pangnirtung. Overall, the MWWWE plume boundaries observed in Pangnirtung extended a maximum of approximately 150 m from the discharge channel. The large tidal range observed in Pangnirtung caused substantial differences in plume characteristics between tidal regimes. As shown in Figure 4.2, the results of the tracer tests show major differences in tracer dye plume boundaries between the HT, IT, OT, and LT scenarios.



FIGURE 4.1. A) MONITORING THE TRACER STUDY AT IT ON JULY 27, 2013 IN PANGNIRTUNG, NU; AND B) DISCHARGE POINT AND DYE INJECTION POINT AT HT ON JULY 26, 2013.

Pangnirtung had a unique low-tide discharge scenario, and was observed to have different effluent transport characteristics than the other study sites. During the low tide, there was a significant exposed intertidal beach zone, extending approximately 300 m from the high tide water level. As a result, wastewater discharged during this tidal regime traveled

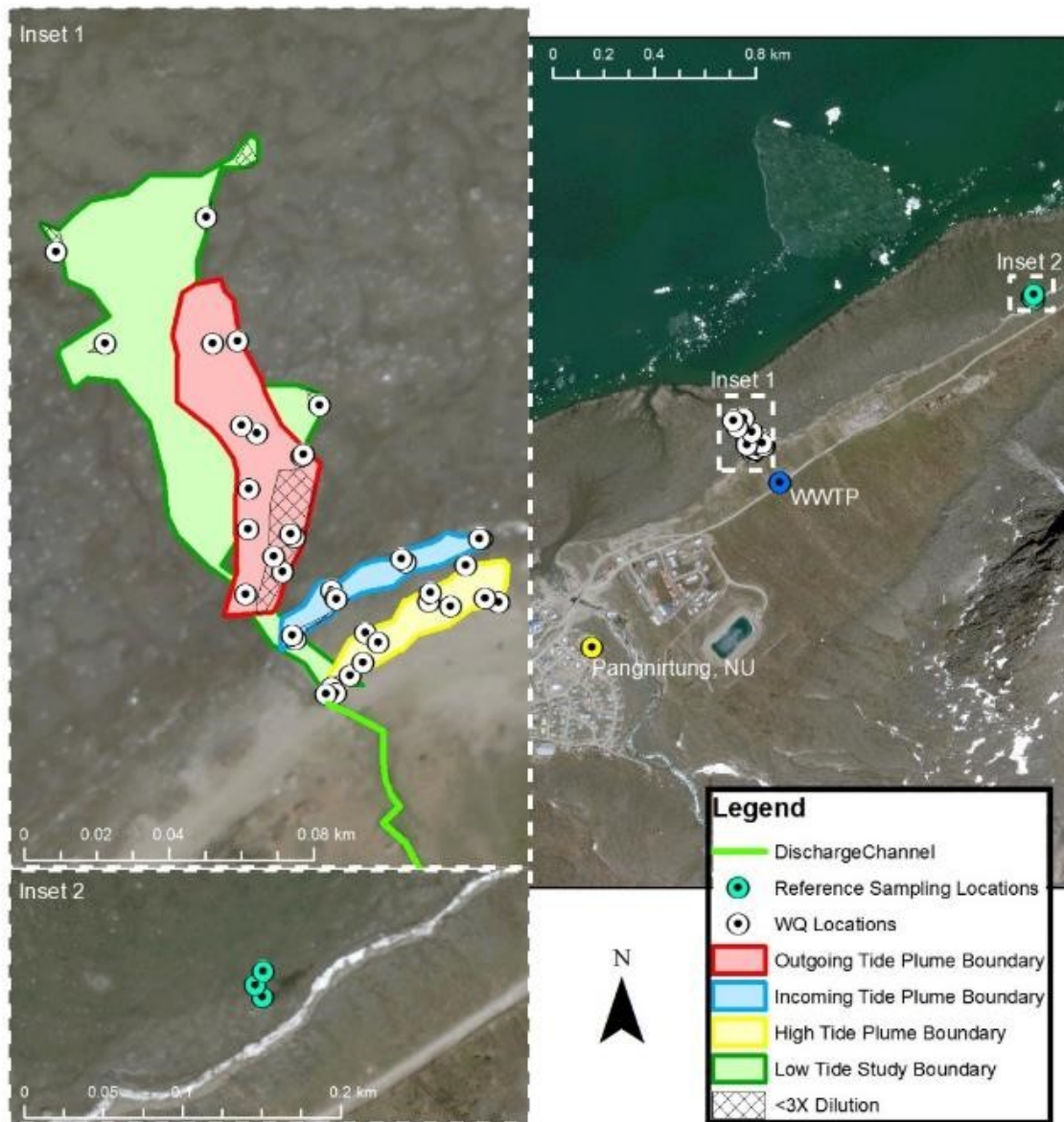


FIGURE 4.2. DELINEATION OF THE DILUTION ZONES AND WATER QUALITY SAMPLES IN PANGNIRTUNG, NU.

across the intertidal zone braided into several small channels, with the majority of effluent ponding before reaching the receiving water environment (ponding at approximately 170 m from the observed high water level). Figure 4.2 shows the LT boundary which represents the extents of wastewater transport in these channels, before all channels were observed to be ponded and stagnant.

In addition, the OT study showed a significant amount of transport of RWT across the exposed intertidal zone as the water level receded. The timing of the recession of the tides was fairly quick (approximately 0.02 – 0.03 m/s based off of both drifter measurements and high water level measurements at beginning and end of study), which transported both the effluent and associated RWT along the beach as the tide receded. For comparison, in the incoming tide case, currents were observed to be more significant, (0.03 – 0.08 m/s), but were predominantly in a cross-shore direction. With the minimal discharge quantity (measured at $\sim 0.001 - 0.003 \text{ m}^3/\text{s}$), velocity (measured at 0.2 – 0.3 m/s), and thus total discharge energy, observable near field mixing processes were limited at this particular study site. The majority of transport observed was influenced by both the ambient currents and the large tidal effects at this location (e.g., incoming and outgoing tidal currents). As such, when discharging to stagnant conditions, the MWWE was observed to have very minimal movement surrounding the discharge location.

With the minimal wave action observed at the time of performing many of the studies, there was limited observable turbulent mixing processes taking place. As a result, the effluent was observed to form a small layer on the receiving water surface in several of the sampling events, with limited vertical transport in the nearshore environment measured through RWT samples or in observations.

A separate site visit was completed on May 6, 2014 by GN representatives in order to collect information related to the winter discharge conditions observed in Pangnirtung. The purpose of the site visit was to assess the winter discharge conditions since Pangnirtung discharges effluent on a near-continuous annual basis. From this visit, it was difficult to determine transport characteristics, as ice cover extended to the approximate high water level previously observed on site, leaving the wastewater discharge hidden as it entered the intertidal zone. As a result of the observations made during this visit, it is

unfortunately not feasible to make any conclusions regarding the transport characteristics of the Pangnirtung wastewater discharge during periods of ice cover. To postulate, depending on the depth of ice cover experienced, the discharge would likely interact with the receiving waters in a similar manner to what was observed in ice-free scenarios in most tidal cases. The main cases for concern under ice cover would likely be the incoming and high tide cases, where ice cover has the potential to restrict tidal amplitudes, impeding transportation of the MWWWE and potentially causing localized areas of higher concentrations than otherwise would be observed.

4.1.2 KUGAARUK

Tracer monitoring was performed from a kayak in Kugaaruk due to predominately nearshore observed plume transport characteristics, as shown in Figure 4.3a. In Kugaaruk, the effluent entered the receiving water environment with minimal energy. This was due to both the dampening of discharge velocities through the treatment system, and the subsurface infiltration of the flow into a rocky tidal area prior to discharge into the marine environment. Discharges were observed to be transported with the ambient current along the shoreline in a north-eastern direction (Figure 4.3b). The highest concentrations of RWT were localized within a narrow banded area which extended approximately 60 m from the location where the wastewater entered the receiving waters (Figure 4.3c and Figure 4.4). The measured maximum extent of the observable and measurable plume boundaries from the discharge location was approximately 100 m.

The dye plume attached to the downstream shoreline in all of the tracer study cases undertaken (Figure 4.3d). This is likely due to the absence of energy in the discharge, leading to minimal jet-like properties in the nearshore environment. As such, the flow at this location could be deemed either shoreline attached or upstream intruding (as illustrated in Figure 2.2).

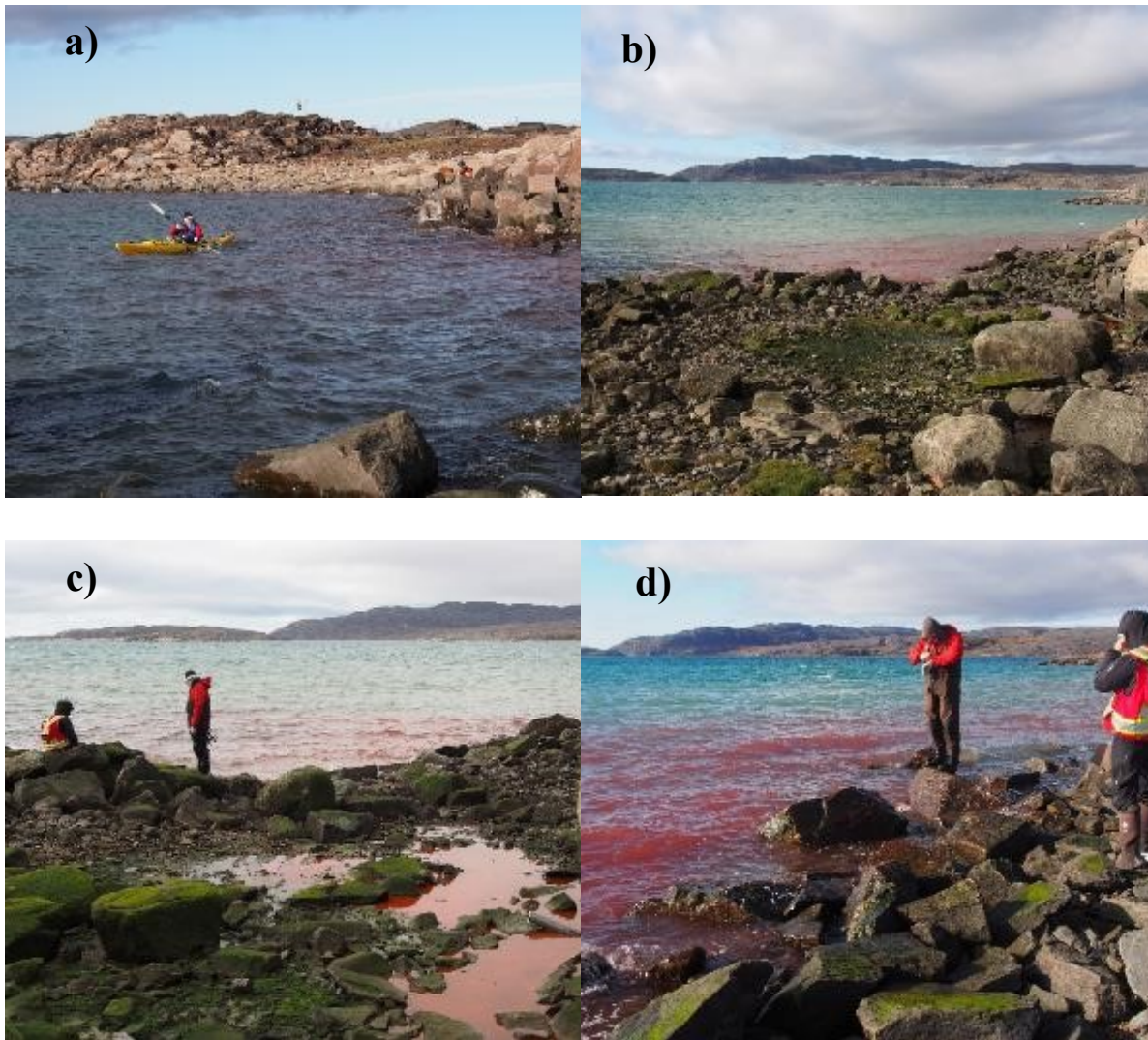


FIGURE 4.3. PHOTOGRAPHS OF THE TRACER STUDY IN KUGAARUK, NU ON AUGUST 23, 2013 SHOWING THE: A) MONITORING OF THE DYE PLUME IN A KAYAK; B) PLUME BOUNDARY TAKEN FROM THE NORTH-EASTERN PERSPECTIVE; C) DISCHARGE POINT JUST DOWNSTREAM OF WHERE DYE TRACER WAS INJECTED AND ENTERING THE ROCKY ZONE OF SUBSURFACE FLOW; AND D) SHORELINE ATTACHED BEHAVIOR PLUME BOUNDARY.

The measurement of accurate surface currents at the discharge location proved to be a challenge throughout the field studies. As the receiving water environment was fairly enclosed, had prevalent wave action, and fairly shallow nearshore bathymetry, the deployment of the drifter in the area surrounding the discharge was not feasible. As a result, the drifter was deployed at deeper locations a bit further removed from the discharge location, and was supplemented by the use of oranges in the nearshore environment in order to develop an understanding of the currents. These measured



FIGURE 4.4. DELINEATION OF THE DILUTION ZONES AND WATER QUALITY SAMPLES IN KUGAARUK, NU.

currents provided further evidence that transport along the shoreline can be expected at this location, as both drifter and oranges were transported towards the shoreline and then in a NE direction. The consistency of these findings across several studies may be attributed to the geography of the discharge location – Kugaaruk is found within Pelly Bay, and circulation within this area may be consistent. On a smaller scale, discharge occurs in a sheltered cove, which also may promote alongshore coastal currents when significant winds are observed (as was experienced during our field studies). Another consideration at this location is the potential influence of the freshwater stream that enters the receiving waters to the north of the community (Figure 4.4). RWT was observed to collect in pockets along the shoreline in eddies that existed in-between the larger rocks. As a result, higher pockets of RWT measurements are observed in the data wherever these pockets are found, providing further evidence for the delineation of the area of minimal dilution (<3 times dilution) shown in Figure 4.4.

Moderate and persistent winds (e.g., average of 27 km/hr measured during August 26th sampling event) generated wave action in the nearshore environment, which mixed the water column in the nearshore region. There was also slight density stratification along the nearshore travel path of the RWT, with values of specific conductivity at surface of approximately 11500 $\mu\text{S}/\text{cm}$, where bottom specific conductivity values were approximately 20200 $\mu\text{S}/\text{cm}$. As the values measured at surface have similar temperatures to those measured at the bottom, it is likely that this measured difference in salinity is due to the discharged effluent. The differences in salinity at surface were also shown to have a declining trend as measurements were taken further from the discharge location. This further suggests that the effluent is mixing throughout the water column within a fairly small distance from the discharge location.

4.1.3 POND INLET

Tracer study monitoring in Pond Inlet was conducted by wading at the shore, as well as from a boat when weather conditions permitted (Figure 4.5a). Pond Inlet's wastewater management system includes a rocky discharge channel that transports the discharge along the northern boundary of the WSP, prior to being conveyed through a grassy surface/subsurface channel over the bluff and into the receiving water environment. As a

result of this and the existing discharge schedule (once a year, occurring over approximately three weeks), the discharge flow rate and velocity are significantly higher than the other tracer study sites (velocities measured at 0.35 – 0.60 m/s, flow rates of 0.03 – 0.06 m³/s).



FIGURE 4.5. PHOTOGRAPHS OF THE RECEIVING ENVIRONMENT IN POND INLET, NU SHOWING THE: A) TRACER STUDY MONITORING FROM A BOAT ON SEPTEMBER 17, 2013; B) INITIAL MIXING OF THE DYE AT THE DISCHARGE LOCATION ON SEPTEMBER 12, 2013; C) NEAR-SHORE MONITORING OF THE DYE PLUME ON SEPTEMBER 15, 2013; AND D) LONG-RANGE TRANSPORT OF THE DYE ON SEPTEMBER 15, 2013.

Figure 4.6 shows the extent of the dilution zones in Pond Inlet. At this site, initial dye dilution through spreading on the surface of the receiving waters occurred once effluent reached the nearshore marine environment (Figure 4.5b and c). Minimal dilution was observed once the ambient current became the dominant transport process, as strong currents carried the plume large distances while slower ambient processes dominated

mixing. Several of the tracer tests completed at this location had conditions where wave action was minimal but currents remained strong (0.17 – 0.25 m/s). Under these scenarios, the dye was observed to remain in a small layer (approximately 5 – 15 cm) on the surface of the receiving waters as it was transported through the receiving water environment. This resulted in long-range transport of elevated tracer dye concentrations, measured as far as ≥ 450 m during one of the tracer tests (Figure 4.5d). Minimal turbulent mixing processes were observed to occur when receiving water conditions were calm, with slower mixing processes based on buoyancy differences and passive diffusion likely dominating in these cases.

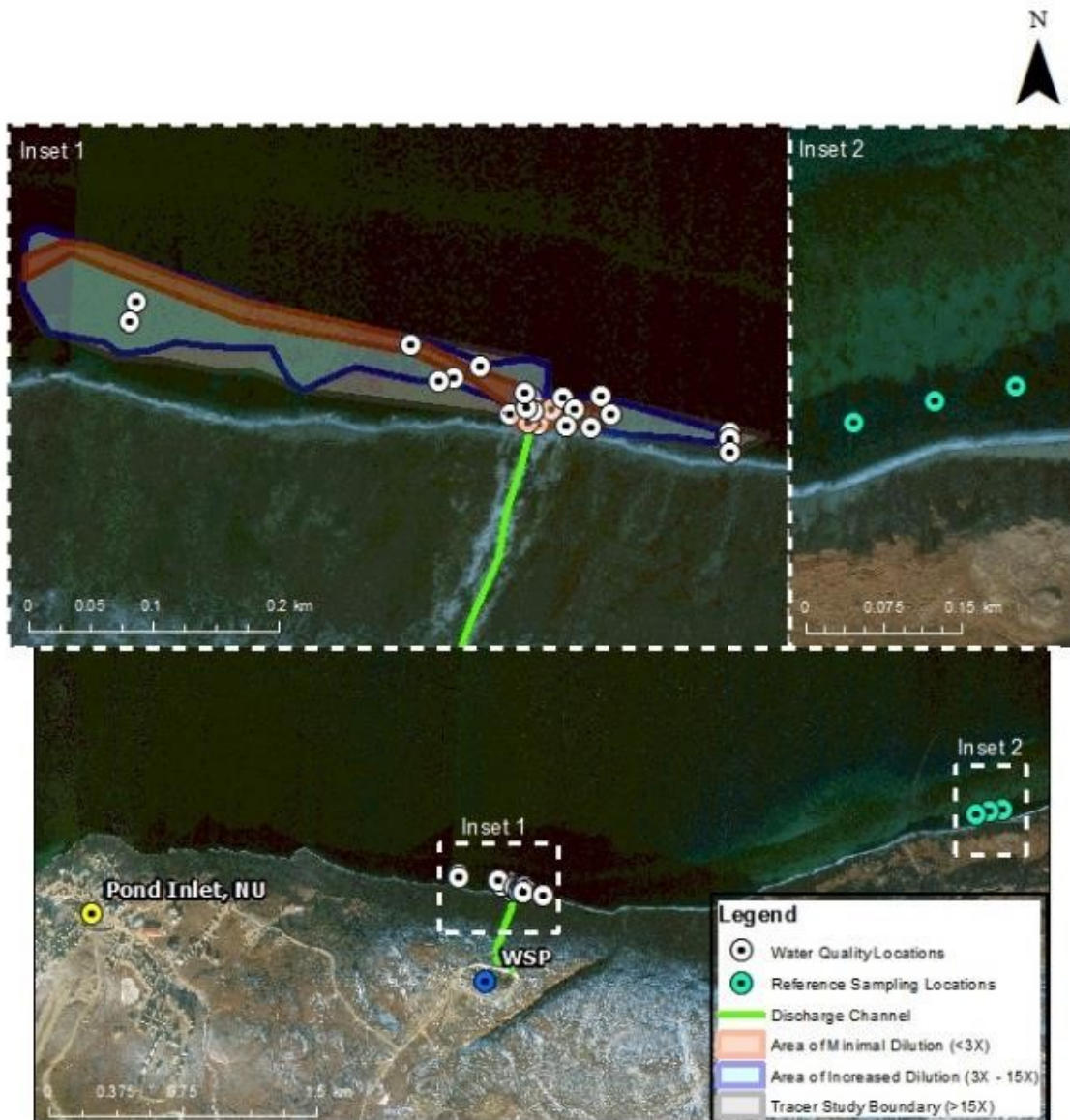


FIGURE 4.6. DELINEATION OF THE DILUTION ZONES AND WATER QUALITY SAMPLES IN POND INLET, NU.

In addition to the conditions outlined above, discharge scenarios accompanied by significant wind and wave action were observed during sampling events. Unfortunately, during these periods the boat operator was not comfortable operating the vessel, so minimal data could be obtained related to these events. Visual RWT observations were still made, however, and in one case samples were taken by wading during a period of

strong wave action in the discharge location. These results indicated that the increased wave action in the nearshore environment significantly increased the mixing occurring in these scenarios, with dilution occurring at a rate much faster than the calm scenarios.

With the greater velocity and energy behind the discharge, there is evidence through observations made on site that free surface buoyant jet-type near field flow patterns occur at this location before leading to transport driven by ambient conditions (e.g., advection and passive diffusion, ambient current u_a ; Figure 4.7). In addition, shoreline attached flows downstream of the discharge location were observed during certain discharge scenarios, likely a result of tidal currents. Several dye injections were completed outside of the tracer studies summarized in Figure 4.6 to build a greater understanding of the transport at this location. As our time with the boat was limited, these additional dye injections were completed in order to obtain as much information as possible surrounding potential plume transport scenarios for this Site. Based on observations related to dye movement in the receiving waters, plumes from these shoreline studies had similar travel extents to those captured by boat (Figure 4.6), with potential for the large transport event shown travelling to the W of the discharge location to occur travelling to the E as well.

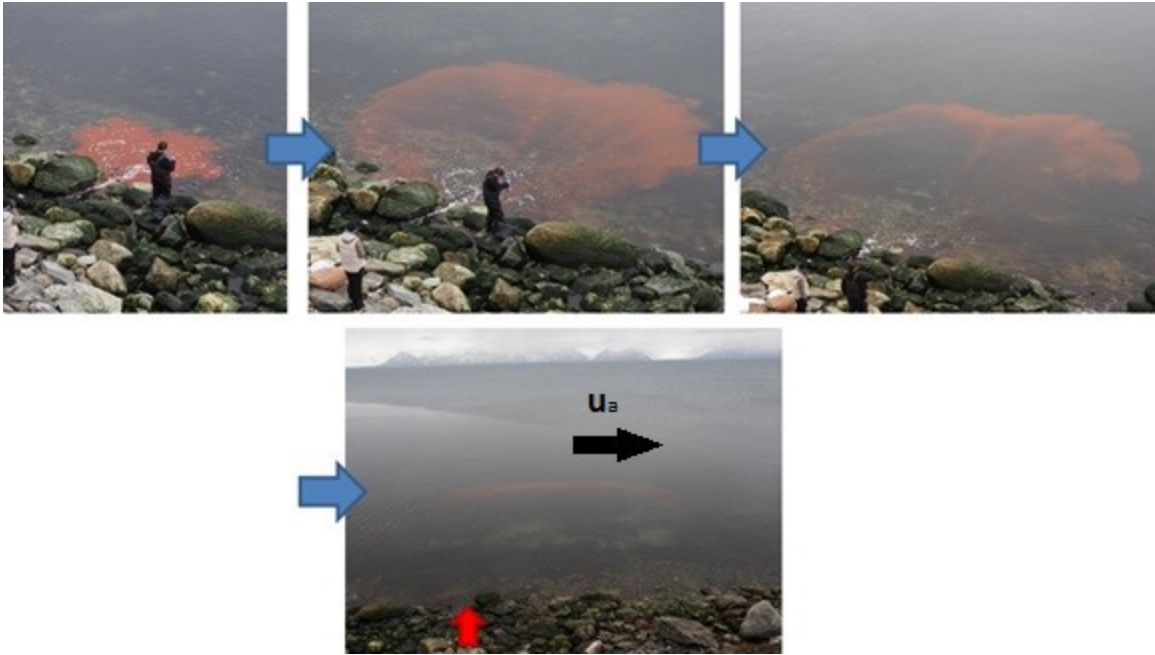


FIGURE 4.7. POND INLET OBSERVED NEAR-FIELD PROCESSES

4.2 MWWE WATER QUALITY

Samples of MWWE were collected prior to discharge to the receiving water environments in order to characterize the quality of the discharge prior to reaching the receiving water environment. In addition to the samples collected as a part of this study, sampling has occurred through the 2010 – 2013 treatment seasons. In order to provide a more thorough insight into the MWWE quality achieved at each study Site, the results of an analysis into this extended dataset previously reported (CWRS, 2014) has been summarized and presented in Table 4.1. Overall, the MWWE entering the receiving environment varies in quality from levels typical of primary to secondary treated wastewater. According to the Droste (1997) classification, the Pangnirtung system obtains primary treatment levels with <50% removal of CBOD and TSS; the Kugaaruk system achieves secondary treatment levels with >90% removal of CBOD and TSS; and the Pond Inlet system can be classified as an advanced primary level of treatment with between 50 – 90% removal of CBOD and TSS. Therefore, Kugaaruk and Pangnirtung had the best and worst MWWE quality, respectively. Additional details, including a summary of both WSP and MWWE sampling, can be found in the CWRS (2014) document.

TABLE 4.1. SUMMARY OF AVERAGE WASTEWATER SYSTEM EFFLUENT QUALITY FROM THE STUDY SITES FROM SAMPLES TAKEN DURING THE TREATMENT SEASONS FROM 2010 - 2013.

Site name	Parameter	CBOD ₅ (mg/L)	TSS (mg/L)	NH ₃ (mg/L)	<i>E. coli</i> (MPN/ 100 mL)	Enterococci (MPN/100 mL)	TN (mg N/L)	TP (mg P/L)	pH	DO (mg/L)
Pangnirtung	Average	104	253	0.80	6.0E5	1.8E5	66	8.9	7.8	7.5
	# of samples	6	6	6	6	3 ^a	3	3	6	6
Kugaaruk	Average	12	3	0.42	4.5E3	1.0E2	53	4.4	7.5	4.6
	# of samples	2	3	3	3	1 ^a	3	3	3	3
Pond Inlet	Average	47	77	1.85	9.1E5	1.5E4	80	5.4	8.2	15.1
	# of samples	4	8	7	7	4 ^a	7	8	3	3

^a – Taken from discharge stream just prior to entry into receiving water in 2013 samples, as identified as a parameter of interest primarily for this study alone. Represents a high level estimate.

4.3 MARINE RECEIVING WATER BACKGROUND

Establishing an understanding of the background concentrations within the receiving water environments is important in developing an understanding of appropriate concentrations at the IMZ boundary. Water quality grab samples were taken at the reference locations at each study Site as marked on Figure 4.2 – Figure 4.6 and analysed for the suite of water quality parameters outlined in Section 3.4.1 in order to provide a snap shot into expected receiving water quality at each study Site. In addition, values for DO and pH taken at the maximum extents of the study have been summarized for additional information. These results are found in Table 4.2 below. Many of the parameters were below the detection limit, with the exception of TSS. TSS was elevated at the reference sites likely as a result of wave action, which suspends sediment in the water column. This was especially prevalent at Kugaaruk where background concentration of TSS averaged 27 mg/L.

4.4 RECEIVING WATER QUALITY

4.4.1 MAXIMUM CONCENTRATIONS

The maximum values for various parameters obtained from the water quality sampling program undertaken in the receiving water for the Kugaaruk, Pangnirtung, and Pond Inlet study sites illustrate the differences in both mixing and effluent characteristics occurring at these sites (Table 4.3). The distance from the discharge point to where these values were observed has also been included. Please note that while efforts were made to ensure accuracy of water quality locations through collection of GPS data, uncertainties in GPS accuracy, non-static sampling locations (i.e., taken from moving boat), and other considerations require that the location data provided in Table 4.3 and Table 4.4 be treated as approximate.

Out of all the sites, Kugaaruk had the lowest maximum concentrations of all the parameters. The overall concentrations were low in Kugaaruk due to the relatively lower MWW concentrations, which can be attributed to the polishing treatment capacity of the wetland area. Also, greater mixing and associated dilution was observed in Kugaaruk compared to the other sites.

TABLE 4.2. RECEIVING WATER QUALITY RESULTS FOR THE REFERENCE SITES.

Site Name	Parameter	TSS (mg/L)	TAN (mg/L)	<i>E. coli</i> (MPN/100 mL)	Enterococci (MPN/100 mL)	TN (mg/L)	TP (mg/L)	pH	DO (mg/L)
Pangnirtung	Average Value	11	0.020	1.5	<1	0.4	<0.01	7.8	10.9
	# of Samples	3	3	3	3	3	3		
Kugaaruk	Average Value	27.3	<0.005 ^b	1	3.4	<2 ^b	0.12	7.8	13.9
	# of Samples	3	3	3	3	3	3		
Pond Inlet	Average Value	5.5	0.032	<1	36.3	<2 ^b	0.06	7.9	13.9
	# of Samples	2 ^a	3	3	3	3	3		

^a – With one outlier (32 mg/L).

^b – Detection limit.

TABLE 4.3. RECEIVING WATER TRACER STUDY WATER QUALITY SAMPLING RESULTS – MAXIMUM CONCENTRATIONS AND LOCATIONS.

Site Name	Parameter	TSS (mg/L)	TAN (mg/L)	<i>E. coli</i> (MPN/100 mL)	Enterococci (MPN/100 mL)	TN (mg/L)	TP (mg/L)
Pangnirtung	Maximum Value	383	85.8	2.20E3	<1.55E3	96.8	12.8
	Distance from Discharge (m)	140	130	50	50	140	140
	# of Samples	72	73	7	6	72	73
	Tidal Regime	LT	LT	OT ^b	HT ^c	LT	LT
Kugaaruk	Maximum Value	120 ^d	9.7	6 ^a	36.4	17.1	1.39
	Distance from Discharge (m)	10	5	5	8	5	5
	# of Samples	19	19	19	19	18	17
	Tidal Regime	OT/LT	LT/IT	LT/IT	LT/IT	LT/IT	LT/IT
Pond Inlet	Maximum Value	208 ^d	20.8	5.90E4	1.9E4	21.9	2.01
	Distance from Discharge (m)	10	5	10	15	40	60
	# of Samples	41	41	29	33	41	41
	Tidal Regime	HT	HT	OT	LT	IT	IT

^a - Fecal Coliform in CFU/100 mL.

^b - Based on only OT data.

^c - Based on only HT data.

^d – Values influenced by background concentrations at time of sampling, represent concentrations larger than corresponding MWWWE samples.

Pangnirtung and Pond Inlet distances to discharge based on HT Water Level Discharge locations.

Tidal Regimes: IT - Incoming Tide, LT - Low Tide, OT - Outgoing Tide, HT - High Tide.

TABLE 4.4. RECEIVING WATER TRACER STUDY WATER QUALITY SAMPLING RESULTS – LOCATIONS WHERE WATER QUALITY CRITERIA WERE MET.

Citations for water quality guideline		CCME (2015).	B.C.MoE (2001).	HC (2015).	HC (2015).	Background ^c	Background ^c
Site Name	Parameter	TSS (mg/L)	TAN (mg/L)	<i>E. coli</i> (MPN/100 mL)	Enterococci (MPN/100 mL)	TN (mg/L)	TP (mg/L)
Pangnirtung	Water Quality Guideline	16 ^b	7.5	200	35	<2	0.01
	Distance from Discharge (m)	>150 ^d	>150 ^d	>150 ^d	>150 ^d	>150 ^d	>150 ^d
	Tidal Regime	LT	LT	LT	LT	LT	LT
Kugaaruk	Water Quality Guideline	- ^a	7.5	200	35	<2	0.12
	Distance from Discharge (m)	-	<5	<5	<10	50	30
	Tidal Regime	-	LT/IT	LT/IT	LT/IT	LT/IT	LT/IT
Pond Inlet	Water Quality Guideline	10.5 ^b	7.5	200	35	<2	0.05
	Distance from Discharge (m)	>330 ^d	115	>330 ^d	>330 ^d	165	>330 ^d
	Tidal Regime	HT	HT	HT	HT	LT	HT

^a – Background concentrations much larger than MWWWE results at this location.

^b – Based on CCME WQAL marine requirements of 5 mg/L above background levels for longer term exposures (e.g., inputs lasting between 24 h and 30 d).

^c – Guideline value based on maximum concentrations observed in the reference samples, as no CCME water quality guideline for protection of aquatic life exists for these parameters.

^d – Values given as ‘greater than’ as sample results exceeded criteria at sampling boundary.

Pangnirtung had the highest maximum observed concentrations in terms of TSS, TN, NH₃-N, and TP. Furthermore, the location of maximum concentrations in Pangnirtung were further away from the discharge location than the other two sites (e.g., ~140 m vs. 10 m).

Effects of the tidal cycle on the maximum observed concentrations of parameters varied between the sites and parameters of interest. In Pangnirtung, many of the maximum observed concentrations collected were observed during LT, as a result of the tidal beach and lack of any capacity for dilution as the wastewater was transported. In contrast, maximum concentrations were observed to generally occur in conjunction with the LT/IT scenario in Kugaaruk, although these differences are not significant or likely directly attributable to the tidal scenario. In Pond Inlet, it is difficult to tie increased concentrations just to tidal cycles, as differences related to concentrations within different studies cannot be solely attributed to the tidal scenario with confidence.

4.4.2 INITIAL MIXING ZONES

The IMZs, according to available guideline levels, were determined for the study sites. These were determined as the distance from the discharge point to the boundary where water quality parameters were below applicable guidelines, as shown in Table 4.4. This table has been developed by taking the worst-case scenarios of the distance from discharge point to where applicable water quality criteria were met, based on the sampling information available. This provides a level of insight into what IMZ sizing definitions would be required in order to meet existing criteria based on the results of several discharge scenarios. There is a level of uncertainty in performing this analysis, however, as typically the value associated with the water quality criteria is found somewhere between two sampling locations. In these events, a linear decrease in concentrations was assumed to occur between the two points, and approximate distances to these interpolated locations have been provided. In addition, distances from discharge have been provided as straight line distances except for Kugaaruk, where distances account for the path requiring to exit the enclosed bay at low tide.

Based on this analysis, Kugaaruk had the smallest IMZs for all parameters of equal to or less than 50 m. Pangnirtung had an IMZ of approximately 150 m in all cases, which was due to exposed intertidal zone of the nearshore environment. This intertidal zone was periodically exposed resulting in minimal dilution during low tides. Pond Inlet had the worst cases for IMZs due to the characteristics of the receiving environment, with minimal mixing and strong currents. For instance, some of the parameters of interest did not meet the guideline values within the water quality sampling boundary (i.e., 330 m from the discharge location) during worst-case scenarios in Pond Inlet.

Further details relating to the selection of EQO criteria is found in the sections below.

Ammonia

Of particular note are total ammonia concentrations, whose toxicity is dependent on both pH and temperature measurements (Hydromantis Inc. et al., 2005). As IMZs are not appropriate for the management of acutely toxic, persistent, and bioaccumulative substances, end-of-pipe requirements for substances that fall into these three categories exist. However, should ammonia be the cause of end-of-pipe toxicity and below-toxic levels can be achieved at the boundary of the defined IMZ, then it may be an exception to this rule (CCME, 2008).

As there is no un-ionized ammonia water quality guideline for the protection of aquatic life available for marine conditions, guidelines presented by the Government of British Columbia (B.C. MoE, 2001) have been used in Table 4.4. These guidelines are presented as Total Ammonia Nitrogen concentrations. In using these guidelines, the following parameters have been chosen:

- Pond Inlet – Temperature of 0 °C, pH of 7.8, density of 1030 kg/m³
- Kugaaruk – Temperature of 0 °C, pH of 7.8, density of 1030 kg/m³
- Pangnirtung - Temperature of 5 °C, pH of 7.8, density of 1030 kg/m³

TSS

Criteria for TSS were set through following the CCME water quality guidelines for marine aquatic life for long-term exposure (e.g., inputs lasting between 24 hours and 30 days; CCME, 2015). In the case of Kugaaruk, the elevated TSS values observed for the

samples in the receiving water environment are due to the receiving water location that the discharge enters, as the results of the MWWWE discharge sampled just prior to entry into the nearshore were both significantly less (<3 mg/L).

Nutrients

As there are no CCME water quality guidelines for the protection of aquatic life for TN and TP, results were compared against the results of the reference location samples. In these cases, the maximum values obtained in the reference sampling results have been used. In addition, the Trophic Index for Marine Systems defined in CCME’s Canadian Guidance Framework for the Management of Nutrients in Nearshore Marine Systems Scientific Supporting Document (CCME, 2007), as outlined in Table 4.5 below, was used for context.

TABLE 4.5. CRITERIA FOR EVALUATING TROPHIC STATUS OF MARINE SYSTEMS (SOURCED FROM CCME, 2007)

Trophic Status	TN (mg/m³)	TP (mg/m³)	Chlorophyll a (µg/L)	Secchi (m)	Depth
Oligotrophic	<260	<10	<1	>6	
Mesotrophic	260-350	10-30	1-3	3-6	
Eutrophic	350-400	30-40	3-5	1.5-3	
Hypereutrophic	>400	>40	>5	<1.5	

Pathogen Indicators

E.coli and *Enterococci* were both sampled as part of the water quality program in order to identify human health risks associated with the discharge of MWWWE in locations that may have overlaps with recreational uses that may include human body contact (e.g., fishing). Guidelines from Health Canada for Recreational Water Quality (HC, 2012) presented for marine waters have been used. In this case, Health Canada provides a guideline for *Enterococci* of 35 MPN/100 mL geometric mean concentration guideline for marine environments, which has been used here. As outlined in Section 4.4.6, there were several cases where laboratory issues and travel logistics caused a level of uncertainty in the pathogen water quality results – in some cases and as described in

Section 4.4.6, *E. coli* values have been compared against the Health Canada guidelines of 200 MPN/100 mL geometric concentration guideline for context.

4.4.3 TOTAL SUSPENDED SOLIDS

In terms of TSS, Pangnirtung showed variable water quality depending on the tidal cycle, as shown in Figure 4.8. The worst-case scenario in Pangnirtung for TSS coincided with the low tide where concentrations close to 50 mg/L were observed approximately 150 m from the discharge point. The MWWE entering the receiving environment in Pangnirtung had much higher TSS concentrations than Pond Inlet (e.g., up to 650 mg/L in Pangnirtung compared to 50 mg/L in Pond Inlet). Comparatively, Pond Inlet had the worst-case scenario for TSS with concentrations of roughly 50 mg/L up to 330 m from discharge point during the HT sampling event (Figure 4.9). It should be noted that values from this tidal regime were observed to have very high TSS values when compared to the TSS observed in the MWWE discharge and the results from the other tidal regimes (Figure 4.9). Therefore, it is possible that the ambient TSS levels were high due to mobilization of bottom sediments through wave and current ambient conditions instead of from the MWWE.

4.4.4 ENTEROCOCCI AND *E. COLI*

Pond Inlet had the worst case for long-range transport of enterococci with concentrations of 1×10^2 to 1×10^3 MPN/100mL up to 330 m from the discharge location during high tides (Figure 4.10). This was due to strong currents and limited mixing and dispersion of the effluent plume. Kugaaruk had the best case for enterococci concentrations with concentrations of less than 100 MPN/100 mL entering the receiving waters and a maximum distance of influence of 50 m (Figure 4.11). Generally, all of the parameters monitored in Kugaaruk followed this trend of having measurable concentrations localized to a small area. Figure 4.12 shows concentrations of 1×10^2 to 1×10^3 MPN/100mL for *E. coli* observed at 60 to 80 m from the discharge point in Pangnirtung.

4.4.5 AMMONIA

In terms of TAN, the best-case discharge scenario was observed in Kugaaruk with a localized area of less than 50 m from the discharge point where measureable

concentrations were observed (Figure 4.13). In Pond Inlet, concentrations above the water quality criteria of 7.5 mg/L for TAN were observed in the effluent plume for distances of up to 115 m from the discharge location (Figure 4.14). As a result, long-range transport of TAN (i.e., greater than 150 m) was noted in Pond Inlet and in the LT tidal regime in Pangnirtung. Whereas, Kugaaruk was below the *Ambient water quality criteria for ammonia to protect marine aquatic life* (B.C. MoE, 2001) defined in Table 4.4 within 5 m of the discharge location.

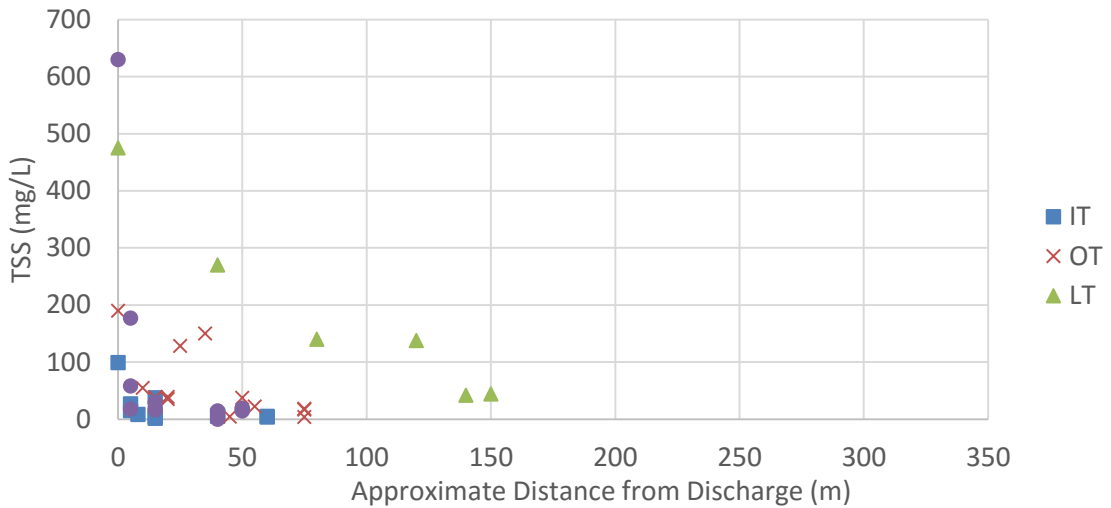


FIGURE 4.8. TSS IN THE RECEIVING ENVIRONMENT OF PANGNIRTUNG, NU.

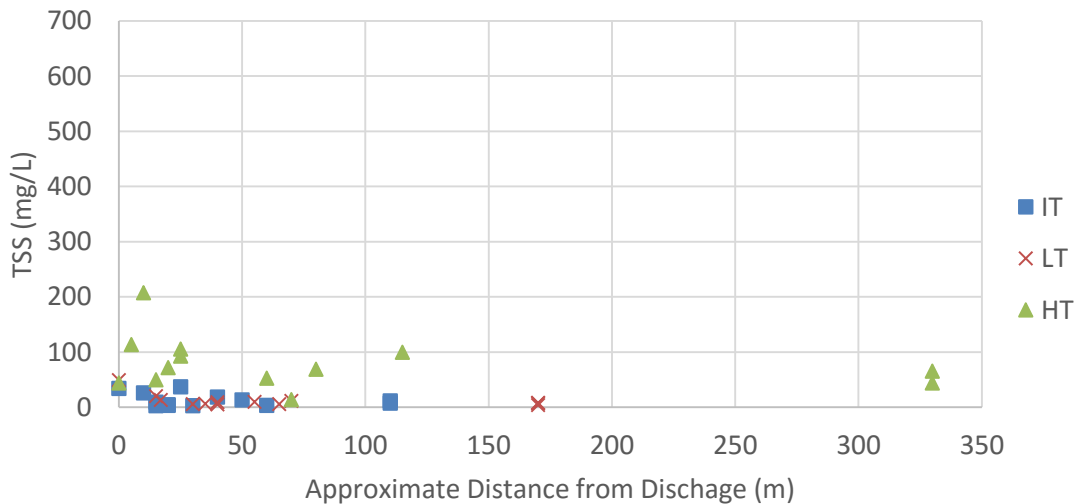


FIGURE 4.9. TSS IN THE RECEIVING ENVIRONMENT OF POND INLET, NU.

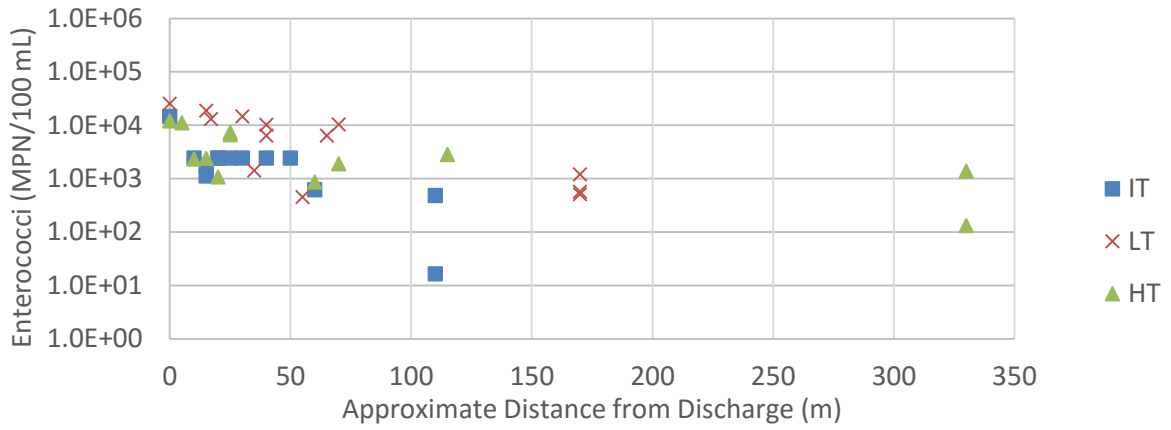


FIGURE 4.10. ENTEROCOCCI IN THE RECEIVING ENVIRONMENT OF POND INLET, NU.

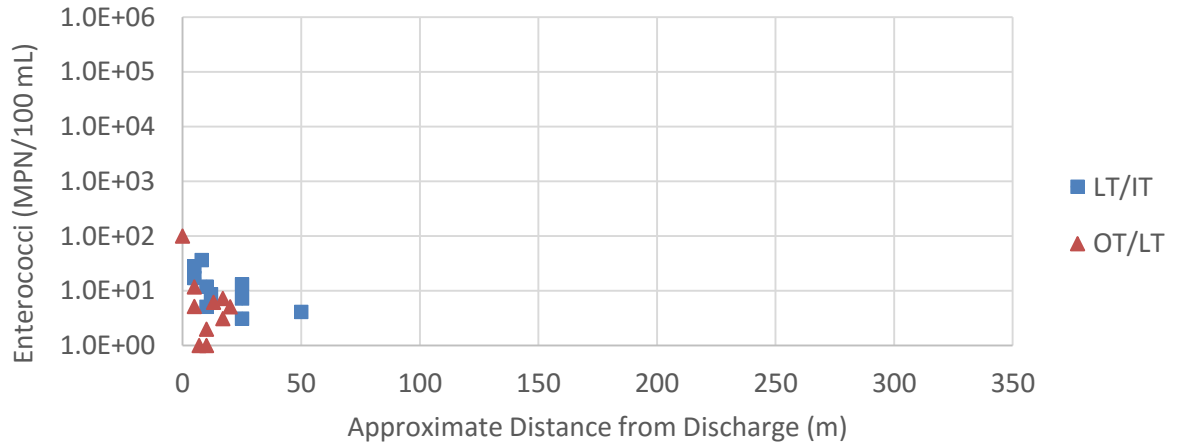
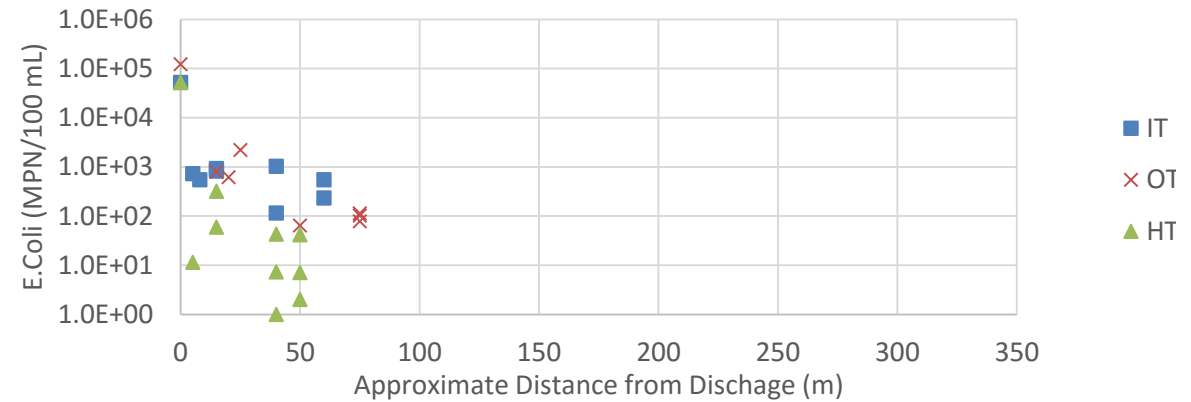


FIGURE 4.11. ENTEROCOCCI IN THE RECEIVING ENVIRONMENT OF KUGAARUK, NU.



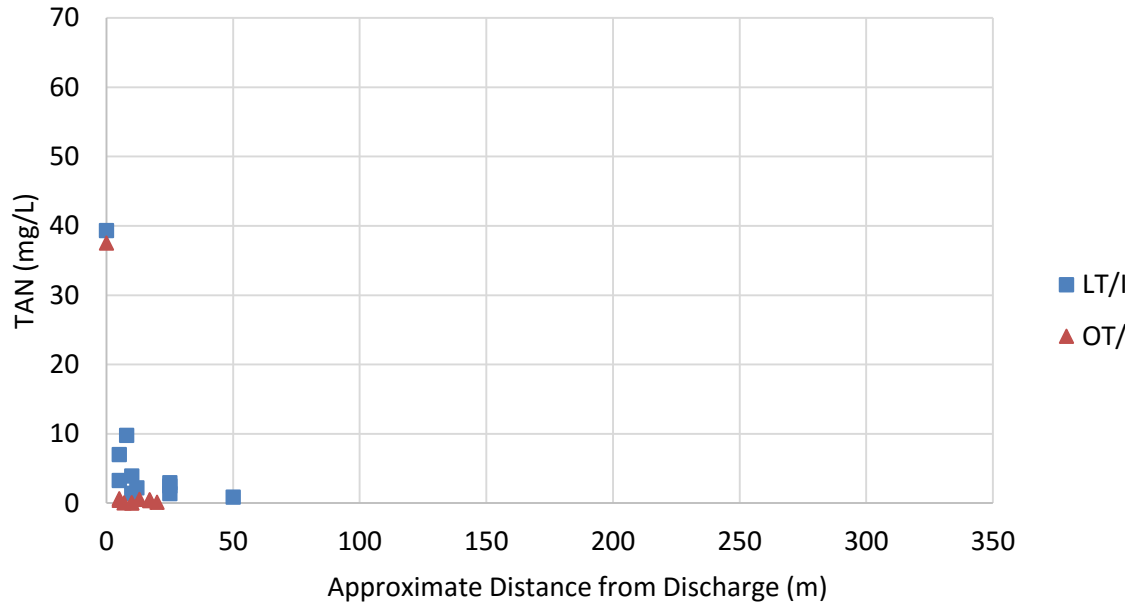


FIGURE 4.13. TAN IN THE RECEIVING ENVIRONMENT OF KUGAARUK, NU.

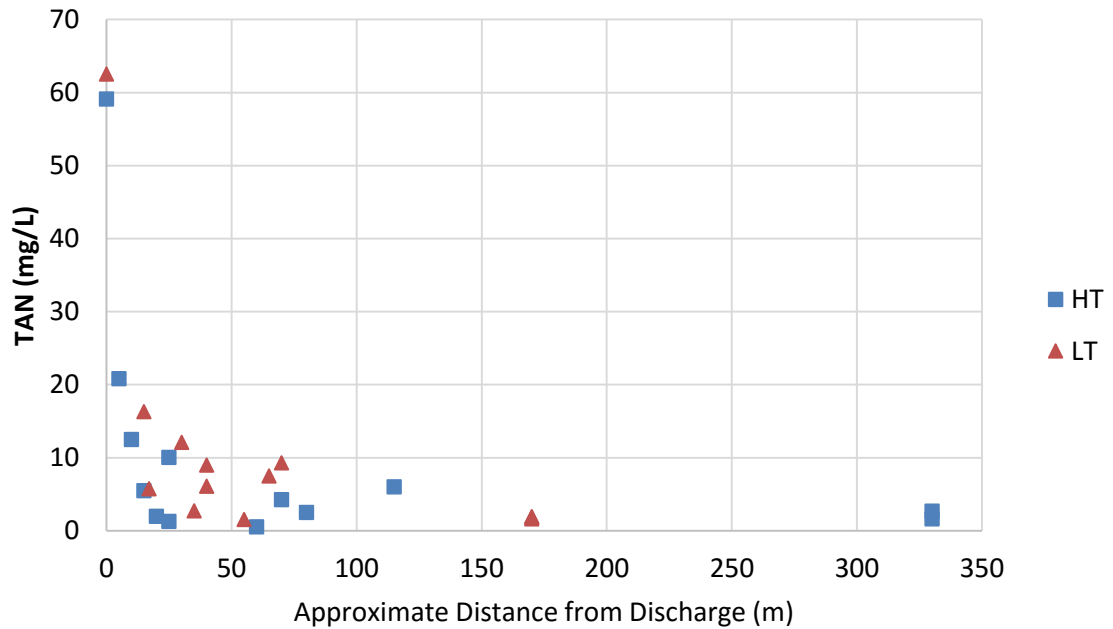


FIGURE 4.14. TAN IN THE RECEIVING ENVIRONMENT OF POND INLET, NU.

4.4.6 WATER QUALITY SAMPLE UNCERTAINTIES

The logistics and execution of fieldwork in remote northern communities comes with unique challenges relating to equipment, timing, transport, and other critical project parameters. As previously outlined, efforts were made through planning and logistics to ensure that hold times for particular parameters were met, and that proper analysis using standard techniques were followed. Unfortunately, there were certain circumstances throughout the field data collection periods where this was not feasible. As a result, only certain samples have been considered as part of the final water quality analysis. In addition, it was observed that walking through the nearshore environment (particularly the tidal beach area in Pangnirtung) for the purposes of the tracer study and water quality sampling mobilized sediments into the water column, which may have affected water quality results. As the bathymetry of this area was very shallow and the transport of RWT (and thus discharge) localized in many of the sampling events, there was no feasible way to avoid this occurrence.

As a result of some undesirable weather systems in Kugaaruk, there were issues with sending samples on August 27, 2013 as no flights could exit the community. As a result, the samples for TSS, NH₃-N, enterococci, and *E.coli* were obtained by Taiga labs 1 day after what was planned, with enterococci, *E.coli*, and NH₃-N all exceeding their holding times. It is understood that there is additional uncertainty associated with these results, but the value of these results in observing trends in the receiving water remains. These results have thus been included.

CHAPTER 5. DISCUSSION

5.1 FACTORS INFLUENCING SITE SPECIFIC IMZs

The discharge scenarios as part of this study were analyzed within the frame of meeting the water quality guidelines outlined in Section 4.4 in the 100 m radius IMZ, with considerations for overlaps with human uses of those areas, as defined by the NWT guidelines (NWTWB, 1992). Through an analysis of the water quality results collected in the nearshore environment, it was observed that Kugaaruk was the only location that met the water quality requirements at a distance of 100 m or less from the discharge location during each sampling event. There are a variety of factors that influence the mixing of MWW and resulting water quality in the receiving environment. As such, it is difficult to draw definite conclusions related to IMZ sizes at the study sites based solely off of the field studies that occurred, as the definition of IMZs requires an understanding of the worst-case conditions that can be expected at each location. However, in addition to providing insight into IMZ extents based off of several sampling events for each location, several key factors that contribute to the extents of the IMZs in these discharge scenarios were identified as a result of this study:

- ambient characteristics of the receiving environment;
- discharge rates;
- timing of the discharges; and
- water quality of the effluent.

5.1.1 AMBIENT CHARACTERISTICS OF THE RECEIVING ENVIRONMENT

The relationship between the receiving environment bathymetry and tidal ranges was observed to be an important characteristic influencing the water quality impacts associated with the MWW discharge. This was best demonstrated in Pangnirtung as a result of the intermittently exposed intertidal area where the effluent was discharged. The least favorable water quality scenarios for this system coincided with low and outgoing tides because the intertidal zone was exposed with little to no dilution of the effluent for up to 150 m from the discharge. The other sites did not include exposed intertidal zones

and had steeper nearshore bathymetry, which helped to dilute the effluent within a much shorter distance from the discharge location.

The near-field mixing and dispersion processes were generally not observed to be significant, which was deemed to be a result of the low energy discharge rates at two of the sites (e.g., <150 m³/d) and the discharges to surface. By discharging to surface, the increased mixing observed through differences in buoyancy that are typically observed in subsurface discharges are not obtained. Therefore the characteristics of the ambient environment were more important for transport and mixing of the effluent plume. Additionally, the buoyancy differences between the MWW discharge and ambient receiving water affected the mixing of the effluent plume. This is particularly important because the ambient current conditions were observed to be both highly variable based on site specific conditions (e.g., wind, tides), as well as varying on a site to site basis. For example, the ambient currents were not observed to be as strong in Kugaaruk (e.g., predominantly 0.02 – 0.05 m/s). These ambient currents combined with the receiving water features resulted in shoreline-attached flows at all times in Kugaaruk. This contributed to a relatively small area of influence of the effluent plume compared to the other sites. There would be concern, however, if there were human activities occurring along the shoreline close to the effluent discharge location. This contrasted with Pond Inlet where strong ambient currents (e.g., consistently measured at 0.17 – 0.25 m/s) transported the effluent plume long distances with limited dilution (i.e., > 330 m in one scenario). As near-field mixing processes were observed to be very limited in the majority of discharge scenarios, the ambient conditions will govern the mixing processes. Therefore, developing an understanding of the magnitudes, directions, and factors influencing ambient currents is very important in assessing the water quality impacts of MWW discharge in these small communities.

It must be noted that in the development of IMZs and the analysis of the differences between sites, that no considerations have been made to differences between sites in specific mechanisms related to removal of pollutants, such as particle settling, chemical transformations, and adsorption/absorption. This assumption was made due to the small

time scale associated with discharges associated with either having unobservable dye levels or exiting the 100 m radius IMZ.

5.1.2 DISCHARGE RATES

Many communities in Nunavut are classified as very small to small according to the WSER. These systems have daily discharge rates ranging from approximately 10 to 2400 m³/d (with exception of Iqaluit). Therefore the volume of wastewater entering the receiving environment is small relative to many other systems across Canada. In this study, Kugaaruk had the lowest daily discharges ranging from 10 to 48 m³/d, in comparison to Pangnirtung which had discharges of 136 m³/d. Whereas, Pond Inlet had daily discharges of up to 2400 m³/d over a short time frame (few weeks). These discharges are all relatively small in magnitude, which results in low energy discharges when compared to larger treatment systems common in more southern jurisdictions. Therefore the ambient conditions in the receiving environment quickly become the controlling factor for mixing and dispersion. As a result of the similarity in discharge magnitudes, the differences in mixing and dispersion of the effluent plume were attributed mainly to ambient conditions. As many communities throughout Nunavut share similar facility size classifications and discharge scenarios, this implies that ambient conditions will likely drive dispersion of MWWWE across these communities within the territory.

5.1.3 TIMING OF DISCHARGES

For these studies, the timing of discharge has been identified as an important management parameter. This significance is best illustrated in Pangnirtung, where discharges that coincided with LT conditions experienced minimal dilution as a result of the effluent being discharged onto the exposed tidal flats. This scenario (as well as OT scenarios) also increased the overall extent of MWWWE transport within the receiving water environment.

In addition, long-range transport of weakly diluted effluent occurred when discharge timing coincided with strong ambient currents with minimal observed wave action in Pond Inlet. These scenarios observed may allow for a level of mitigation to be

implemented through limiting the timing of discharges to periods when these scenarios could be avoided/minimized. The majority of the tracer studies completed also showed ambient currents and resulting dye cloud transport towards the community. Through an examination of these results and their corresponding tidal cycles, it is difficult to correlate this transport solely to the tidal scenario in which they were discharged. Further study to understand the range of current velocities and direction at this location may provide further insight into ties with tidal cycles. This all being said, the conditions observed in Pond Inlet may be too variable to implement feasible management changes (i.e., unpredictable currents and wave influenced mixing).

5.1.4 WATER QUALITY OF THE EFFLUENT

The water quality of the MWWWE entering the receiving environment varied depending on the upstream treatment system. Kugaaruk had the best MWWWE water quality, and Pond Inlet and Pangnirtung had similar MWWWE water quality entering the receiving waters. The Kugaaruk treatment system consists of the wetland and WSP in combination; whereas the Pond Inlet system uses only a WSP; and Pangnirtung relies on a mechanical treatment plant, which at the time of the study was not providing significant treatment. The differences in MWWWE quality likely contributed at least in part to better overall results for Kugaaruk in terms of IMZs and distances to maximum observed concentrations.

5.2 RECEIVING WATER USES

In addition to water quality considerations, the uses of the waters surrounding a discharge location also contribute to the delineation of appropriate IMZs. As a result, the water quality and IMZ results are discussed in the context of potential water uses at a high level below.

The community uses of the receiving water were observed to be different between the study sites. For example, the sites with the highest observable human risk in terms of body contact with elevated bacteria concentrations through receiving water uses were at Pangnirtung and Pond Inlet. Shellfish harvesting occurs in the intertidal zone near the community in Pangnirtung, and there is presently uncertainty associated to the

geographic extents of this activity. Due to the large observed transport of undiluted MWWWE during certain tidal scenarios at this location, further analysis into determining potential overlaps at this location is recommended. Through observations made during the study activities at Pond Inlet, boating, hunting (e.g., Narwhal), and migratory fish passage may overlap with the IMZ at this location, as it was observed that elevated dye concentrations were measured ≥ 420 m from the discharge location in the direction of the community. Water quality results from this location also indicate that enterococci samples exceed the criteria set by Health Canada associated with potential body contact with such waters, which may occur in hunting and fishing activities. As a result, these and other communities with similar overlaps of water uses and transport of contaminants should have additional requirements when setting appropriate IMZs and related discharge objectives.

5.3 OPTIONS FOR RISK MITIGATION

There are simple and more complex techniques that may be used to mitigate the risk of impacts to human health and the environment associated with MWWWE discharges to receiving water environments. The choice of the appropriate technique depends on the likelihood of occurrence and severity of consequences to human health and the environment. Some of the simpler options to mitigate risks of impacts to human health and the receiving environment identified in this study include strategic timing of discharges and community education. Advanced technical solutions may also be worth consideration depending on the final level of risk associated with the wastewater treatment systems with these and other communities – the options below aim to cover the specific scenarios encountered at the study sites, but may be applicable to other similar communities as well.

The goal for strategic timing of discharges would be to avoid low tide discharges at sites that have exposed tidal flats or similar shallow bathymetries that have potential for increased transport and minimal dilution. By managing discharge timing to avoid these scenarios, situations where undiluted effluent is able to have extended contact with bottom sediments and exists in high concentrations at potentially much larger distances from the discharge location can be avoided.

Community education would aim to engage the local residents and users of the receiving waters to communicate the location of the wastewater discharge and extent of the IMZs. Recreational activities can be avoided in the areas where the plume is suspected to be during discharge periods. Indigenous knowledge on the timing of fish and mammal passage can be gained through community consultation and the timing of discharges can be planned to avoid interference with the aquatic life. Installation of signage near the discharge point would be useful to warn of the presence of municipal wastewater. As shoreline attached plumes were observed to occur at all study sites, signage both at the discharge location and at set distances along the shore reflecting the extents of the IMZs surrounding this discharge location would provide an indication to residents of potential areas of concern related to MWWWE discharges.

More complex risk mitigation solutions involve improving upstream wastewater treatment systems to improve MWWWE water quality; which would likely have significant capital costs. In some scenarios, it may be more cost-effective to consider using submerged discharge pipe and diffuser system to increase mixing and dispersion of the MWWWE in the water column. The design and installation of a single or multi-port diffuser system to promote mixing and dispersion of the MWWWE in the receiving waters would provide benefits compared to buoyant surface discharges, as wastewater jets with buoyancy are observed to have significantly more dilution (Fischer et al., 1979). This would also allow for communities to have a level of control over the mixing observed at each location, as at present ambient, minimal near-field mixing processes were observed through this study. For example, this may be an option in Pond Inlet to help mitigate long-range transport of the minimally diluted effluent plume. The implementation of an outfall diffuser system would allow for a level of control over the near field mixing processes taking place at the discharge location through the control of the factors associated with the wastewater jet (e.g., jet shape, orientation, depth), and would provide opportunities for increasing the rates of mixing within the IMZ. Having a designed piped outfall system may also allow for increased separation between these discharges and the communities, which represents additional benefits to their implementation.

In cases where transport of elevated concentrations of bacteria is seen to be a risk, as was observed in Pond Inlet, a study into the feasibility of additional treatment processes specific to the removal/reduction in bacteria concentrations would allow for options to mitigate this specific risk to be identified. Evidence also suggests that the existence of a wetland treatment system area upstream of the discharge location (e.g., Kugaaruk) allows for both improvements to MWW quality and dissipation of the discharge energy prior to entering the receiving water environment (Hayward et al., 2014).

CHAPTER 6. CONCLUSIONS

This study focused on understanding the MWWWE discharge scenarios, receiving water characteristics, and eventual mixing and transport of MWWWE within the receiving water environment for several study sites in Nunavut. This understanding was then used to delineate IMZs within the receiving waters where appropriate water quality guidelines were met, and to identify the primary factors that influenced mixing at each location to provide several high-level recommendations for mitigation of risks that were identified. This was facilitated with a field program that included dye tracer tests and water quality monitoring in three marine receiving water environments downstream of MWWWE treatment facilities in different locations throughout the territory. Findings from the studies highlighted that there are a variety of environmental factors to be considered to adequately assess the water quality impacts to the receiving environments, and that a range of different IMZ sizes could be expected throughout the communities with similar discharge scenarios in Nunavut.

In general, MWWWE facilities in Nunavut fit within the Small to Very Small classifications outlined by CCME (2008), as the vast majority of the communities within the territory are of a population of 2,000 or smaller (22 of 25 reported; Government of Nunavut, 2014). Treatment facilities and discharge scenarios can vary from being very minimal continuous discharges ($<0.01 \text{ m}^3/\text{s}$), to having intermittent discharges at larger flowrates ($<0.1 \text{ m}^3/\text{s}$) that still represent smaller values than typically studied. The nearshore environments of each study site differed in their bathymetries, tidal magnitudes and influences, and in the variation of their ambient currents. Although efforts were made to ensure the three study sites provided data that could be related to the most typical MWWWE discharges in Nunavut, it was not feasible to cover all the potential discharge scenarios expected throughout the territory. For instance, the type of receiving environment was marine in all three cases; therefore freshwater receiving environments (e.g., Coral Harbour) were not assessed. In addition, the collection of data to support this study was challenging, as not only were the sampling conditions difficult, but the small scale and variability of certain measurements (e.g., MWWWE discharges, MWWWE samples in shallow transport layers) makes these results more appropriate for use in identifying

general IMZ extents and scenarios that carry higher risk of not meeting water quality guidelines within typical IMZ spatial extents.

Several key site-specific considerations observed to drive MWWWE mixing and transport in different discharge scenarios were identified. Near-field mixing processes associated with the discharges were observed to be minimally occurring at each study site, with the characteristics of the receiving environment — specifically the bathymetry, tidal cycle and ambient currents — shown to be an important consideration in relation to the IMZs associated with each system. Pangnirtung and Pond Inlet both had results where concentrations of water quality parameters of interest exceeded applicable criteria in an area both larger than outlined in accepted guidelines (NWTWB, 1992) and in locations that may represent a potential risk to human health based on community uses of those receiving waters. Pangnirtung's receiving environment was characterized by a shallow intertidal area where discharge during low tides led to exposed and undiluted effluent on the tidal flats observed up to 150 m from the discharge location. In Pond Inlet, higher discharge rates and strong ambient currents facilitated long-range transport of minimally diluted effluent observed over 330 m from the discharge point. The plume was transported in a thin layer on the surface of the water as a result of limited mixing and dispersion due to strong buoyancy differences between the MWWWE and ambient waters and lack of processes to promote vertical mixing (e.g., wave action). Both of these observed scenarios, when compared against potential community water uses at these locations, may represent a human health risk for body contact with elevated bacteria concentrations (e.g., fecal indicators levels above the recreational water quality guideline of 35 MPN/100mL for enterococci).

In terms of extent of water quality impacts in the receiving environment, the best-case scenario was observed in Kugaaruk. This was due to several factors, including the semi-enclosed shoreline morphology, significant wave action observed in the receiving environment, the relatively better water quality of the MWWWE, and the dampened discharge rates. As a result, the discharging effluent required less dilution to reach desired water quality criteria and was mixed well within relatively short distances from the discharge point (IMZs observed to be generally less than 30 m).

Each study site was different and had a unique set of factors that influenced the overall water quality impacts associated with MWWWE discharge into the receiving environment. That being said, the findings from this study can be used to identify other communities with higher risk discharge scenarios, such as discharges to tidal beaches and those with larger discharge rates and significant ambient currents. In addition, the importance of characterizing the receiving water environments that were observed to dominate the mixing and transport processes was identified.

A few of the factors that influence the associated human health and environmental risk in the Far North are unique from the rest of Canada. For instance, the discharge rates are generally very small to small (10 – 2400 m³/d), many discharges are intermittent (period of few weeks at the end of summer), the water treatment systems are varied, and in many cases passive. These treatment systems have variable effluent water quality that typically does not consistently meet southern WSER quality standards due to practical limitations.

These differences should be considered during the formation of a risk assessment framework for the Far North. Of particular importance is the consideration of relatively simple and practical solutions for mitigating the severity of the water quality impacts associated with MWWWE discharges, such as strategic timing of discharges around tidal cycles, community engagement and education, and implementing changes to discharge outfalls to promote additional mixing.

6.1 RECOMMENDATIONS FOR FUTURE RESEARCH

The research outlined in this study provides insight into the mixing processes and receiving water quality at three study sites with discharge conditions typical to communities throughout Nunavut. With that being said, there are limitations to the results, and additional study is required to support these findings in determining the most effective approach to discharging effluent in northern communities, including:

- Further characterization of discharge scenarios throughout the communities of Nunavut, including the identification of those with large tidal effects, the characterization of effluent water quality for the remaining northern communities, the measurement of discharge quantities and documentation of discharge timing,

and the identification of systems with treatment wetlands. An understanding of these key components will allow for high-level screening of higher risk discharge scenarios;

- Further analysis into the effectiveness of northern treatment wetlands for treatment of effluent, with a focus on understanding the critical design criteria that drives treatment processes in typical discharge and climatological conditions;
- Characterization of ambient conditions surrounding discharge locations in northern communities, including establishing a more detailed understanding of the range of ambient current speeds and directions, wind speeds and direction, density stratification in the nearshore environment, and background water quality at appropriate reference locations to aid in identifying areas at higher risk for long-range effluent transport;
- Detailed analysis into community uses in the vicinity of wastewater discharge locations, including delineation of areas of frequent use in order to fully understand level of human health risk.

The data collected as part of this study represents snapshots in time for different discharge scenarios typical for several communities and scenarios in Nunavut. While as many of these 'snapshots' were taken as possible during the field studies completed, it is not feasible to collect information for all potential discharge scenarios and ambient environments. The mixing processes occurring in one particular study may not be reflective of the mixing that occurs at a different time with a different suite of discharge and ambient conditions that may affect the level of turbulence and other mixing processes taking place. There is a need to test 'worst case scenarios' that could potentially occur at these discharge locations. Methods for determining worst case scenarios are covered in literature, and can include modelling conditions based off of using the 10 percentile cumulative frequency distribution values for inputs affecting initial dilution (US EPA, 1985). Additional considerations for modelling scenarios for potential use can be found in US EPA, 1985. The development of models represents a strong opportunity to further examine potential discharge scenarios, using data collected from field studies for calibration/validation.

By further developing an understanding of the discharge scenarios for the communities in Nunavut as outlined above, the development of a risk characterization matrix to use in selecting high-risk sites through comparison of tidal scenarios, effluent water quality, receiving water characterization, and proximity to human activities could be effectively undertaken. Once this exercise is complete, a cost-benefit analysis into potential solutions for each site (with a potential focus on those high risk sites) could be completed (e.g., design and analysis of single or multi-port diffusers to promote near-field mixing, analysis into the feasibility of implementing timed discharge depending on tidal conditions). This approach could represent a methodology to identify and address those sites that represent the largest risks to environmental and human health.

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