EFFECT OF WATER DISTRIBUTION SYSTEM ON DRINKING WATER QUALITY IN FIRST NATIONS COMMUNITIES IN MANITOBA, CANADA

by

Geethani Samanthika Amarawansha Eragoda Arachchilage

A Thesis Submitted to the Faculty of Graduate Studies of

The University of Manitoba

in partial fulfilment of the requirements of the degree of

DOCTOR OF PHILOSOPHY

Department of Soil Science

University of Manitoba

Winnipeg

ABSTRACT

Geethani Samanthika Amarawansha Eragoda Arachchilage. Ph.D., The University of Manitoba, October 2021. Effect of Water Distribution System on Drinking Water Quality in First Nations Communities in Manitoba, Canada. Co-advisors: Annemieke Farenhorst and Francis Zvomuya.

Approximately 50% of the homes on First Nation reserves in Manitoba, Canada, receive piped water from water treatment plants (WTP). Of the remaining homes, 31% are equipped with cisterns filled by a water truck. The overall objective of this thesis research was to compare the quality of tap water in homes with piped water versus those with cisterns. The study was in collaboration with three First Nations communities in Manitoba. Each community was predominantly equipped with belowground concrete cisterns, belowground fibreglass cisterns, or aboveground polyethylene cisterns stored in insulated shelters. Free residual chlorine concentration was significantly lower in water samples from homes with cisterns than piped water. The frequency and severity of *Escherichia coli* and total coliform contamination were significantly greater in drinking water from belowground concrete and fibreglass cisterns than in piped water samples in each community. The contamination of belowground cisterns by coliform bacteria was greatest in late spring. Data obtained under the Access to Information Act for 2014 and 2018 showed no clear evidence of improved water quality in any of the Tribal Councils linked to the three communities. Field and laboratory chlorine disappearance studies indicated a greater contribution of concrete cistern walls than bulk water to the disappearance, hence lowering free residual chlorine concentration in belowground cisterns. Treated water from the WTP in two of the First Nations communities frequently exceeded the Health Canada guideline of 100 µg L^{-1} total trihalomethanes (TTHMs), as did 75% of piped water samples and 70% of cistern water samples. Although the water distribution

system had no significant effect on TTHM concentration, sampling month significantly affected due to temporal changes in source water dissolved organic carbon content. Dissolved organic carbon removal before further water treatment might be an important process in effectively reducing TTHMs formation, and thereby decreasing the exposure of residents to high levels of TTHMs. Additional treatment at household level, regular cleaning, and maintenance of belowground concrete and fibreglass cisterns are important to reduce the risk of water-borne illnesses. Also, belowground cisterns must be replaced with aboveground cisterns to reduce coliform contamination, although piped water is the preferred option.

FORWARD

This thesis research was conducted in three First Nations communities from three Tribal Councils, Swampy Cree (Community A), West Region (Community B) and Island Lake (Community C). Communities A and B are in Treaty 4 territory, while Community C is in Treaty 5 territory. A community-based approach was followed to collect water samples from the communities. A project coordinator, graduate students, and their academic supervisors met with the Chief and Band Council in each community to discuss the nature of the research and relevant logistics such as data collection and sharing. A Band Council Resolution was developed and signed. Community representatives (a Community-based Water Quality Monitor in Communities A and B and an Elder in Community C) were appointed to work with graduate students and the broader team. The community representative identified the homes from where water samples were to be collected, as well as served as the point of contact between the community and university researchers. Data was shared with the communities through written reports and oral communications. The research was also presented at scientific conferences, with the community representatives and graduate students being co-authors. When requested by the Chief and Band Council, graduate students and their academic supervisors participated in meetings with government officials to discuss the research results and advocate for the need for improved water infrastructure.

The thesis was written in manuscript style following the guideline of the Department of Soil Science and the Faculty of Graduate Studies, University of Manitoba. The thesis consists of five chapters, including the introduction (Chapter 1), three stand-alone research chapters (2 to 4) and the overall synthesis (Chapter 5).

Chapter 2 has been published:

Amarawansha, G. E. A., Zvomuya, F., & Farenhorst, A. 2021. Water delivery system effects on coliform bacteria in tap water in First Nations reserves in Manitoba, Canada. Environmental Monitoring and Assessment, 193(6), 1-16.

Chapter 3 has been submitted to the Environmental Monitoring and Assessment journal. Submission ID: EMAS-D-21-03121.

Chapter 4 is currently being finalized for journal submission.

ACKNOWLEDGEMENTS

I would like to express my deepest appreciation to all those people who made my journey possible. First, my deepest gratitude goes to my supervisors, Dr. Annemieke Farenhorst and Dr. Francis Zvomuya for their invaluable guidance, great supervision, and encouragement given to me for the successful completion of my research and the thesis. My advisory committee members, Dr. Mario Tenuta and Dr. Greg Tomy helped me with valuable comments and suggestions to shape up the study, which I really appreciate. Special thanks go to Dr. Tomy for letting me use his laboratory for chemical analysis and Thor Holdorson for his help and guidance in the laboratory.

This work was supported by the Natural Sciences and Engineering Research Council of Canada (NSERC) through its Collaborative Research and Training Experience program [CREATE grant# 432009-2013 (CREATE H2O) and grant # 496906-2017 (CREATE SAFER)] and Discovery Grant (DG) programs. I am grateful for the financial support provided by the NSERC – PGS-D scholarship, the University of Manitoba Graduate Fellowship (UMGF), the John Dueck Graduate Entrance Scholarship and the Alfred Rea Tucker memorial scholarship. This helped me keep my focus on the studies.

I want to thank the Chief, Council, water plant and truck operators, water quality monitors and residents of the three First Nation communities collaborated for the ongoing partnership in this research. Special thanks go to Frank Cook, Audrey Brass and Elder Harper in the communities. Continuous support from Wendy Ross, program coordinator of the CREATE H2O program, with coordination and communication with the communities is highly appreciated.

I also want to thank my colleagues in the Department of Soil Science for their support and friendship. Special thanks go to summer students and graduate students Ruidong Mi and Mauli

Gamhewage, who helped me in the field and laboratory. Guidance with statistical analysis by Dr. Inoka Amarakoon and Dr. Ike Agomoh is also highly appreciated. I thank Rob Ellis and Ross McQueen for their technical support and assistance.

I wish to thank my parents and sisters and friends who were always supportive and inspired me towards my dreams. Lastly, I want to express my heartiest appreciation to my best friend, my husband Sumudu Alexander, my daughter Nevmi Binaya and my son Binul Aayuka, for always being with me, for your understanding, everlasting support, and love!

Dedication

For those who suffer all around the world with no access to safe drinking water…

TABLE OF CONTENTS

LIST OF TABLES

LIST OF FIGURES

LIST OF ABBREVIATIONS

ANOVA: Analysis of variance

DOC: Dissolved Organic Carbon

EHO: Environmental Health Officer

MAC: Maximum Acceptable Concentration

NOM: Natural Organic Matter

SUVA: Specific Ultra-Violet Absorbance

TTHMs: Total Trihalomethanes

UV254: Ultra-Violet Absorbance at 254 nm

WTP: Water Treatment Plant

1. INTRODUCTION

1.1. Drinking water quality in First Nations communities

Inadequate access to safe drinking water is a major issue for First Nations communities in Canada (Baijius and Patrick 2019; Neegan Burnside 2011a). The most recent national assessment was completed about a decade ago (Neegan Burnside 2011a) and revealed that 39% of water systems in First Nations communities are at high risk and require immediate corrective actions. Only 72% of First Nations households have direct piped water from a water treatment plant (WTP). The remaining homes have their drinking water delivered by trucks and stored in cisterns (13.5%), or rely on well water, or do not have running water (14.5%) (Neegan Burnside 2011a). Despite the initial intention of the Government of Canada to lift all the long-term drinking water advisories in First Nations communities by March 2021, as of January 31, 2021, there were at least 105 drinking water advisories, including 57 long-term advisories in effect in First Nations communities across Canada (First Nations Health Authority 2021; Indigenous Services Canada 2021a, 2021b). According to recent media reports (Aiello 2020), the Federal Government is asking for additional time to solve this drinking water crisis in Canada. First Nations communities receive 2.5 times more Boil Water Advisories from federal regulatory agencies than other communities in Canada (Patrick 2011). In addition to Boil Water Advisories, drinking water advisories include Do Not Consume Advisories and Do Not Use Advisories. Since drinking water advisories are implemented to protect public health from drinking water that could be contaminated (Health Canada 2015), the fact that violations in Boil Water Advisories occur more frequently in First Nations communities than in other communities in Canada is concerning.

Water management in First Nations communities is a shared responsibility between the First Nation

communities and federal government. Indigenous Services Canada, Health Canada, and the Department of Environment and Climate Change are the three federal government departments directly responsible for water governance in First Nations communities. Band councils are responsible for the design, construction, and management of water treatment plants and distribution systems. Indigenous Services Canada is primarily responsible for providing funding for construction and upgrades of water systems and 80% of operating and maintenance expenses and overseeing the design, construction, and management of the water systems. Health Canada is responsible for water quality monitoring within the communities while the Department of Environment and Climate Change is responsible for source water protection.

Some researchers have identified remote location of the First Nations communities as one of the main reasons for unsafe and inadequate drinking water in these communities (Galway 2016; Murphy et al. 2016). The location of First Nations communities is a result of colonialism and the Government of Canada forcing First Nations to live on reserves while granting large areas of fertile land to White European colonisers (Sarkar et al. 2015; Stewart and Mashford-Pringle 2019). Poor quality of reserve land (e.g., prone to flooding) and the contamination of water resources because of natural resource extraction, agriculture, and other industrial activities continue to be of concern to First Nations (Sarkar et al. 2015). A variety of other researchers have outlined factors contributing to Canada's First Nations drinking water crisis (Dyck et al. 2015; Murphy et al. 2015; O'Gorman and Penner 2018). These researchers highlight factors such as inadequate disinfection at the drinking water treatment plant or the distribution system; the inadequacy of current water treatment designs to meet the needs of small communities, including due to high operational and maintenance costs; lack of trained operators; and lack of funding for operation and maintenance of current systems. The absence of a regulatory framework, the lack of clarity regarding roles and

responsibilities of governments (Federal, Provincial, First Nations), and the lack of involvement of First Nations in decision-making processes are also barriers to providing safe drinking water to First Nations communities (Morrison et al. 2015; Murphy et al. 2016; Patrick 2011). Also, there is evidence that the use of chemicals in water treatment, particularly chlorine, can result in adverse tastes and odours of tap water, leading to some First Nations communities seeking alternative drinking water sources (EKOS Research Associates Inc. 2011).

Inadequate access to safe running drinking water in First Nations communities has been a persistent issue in Canada. Federal government funding was provided to develop and implement *First Nations Water Management Strategy* in 2003 to improve the quality of water and wastewater treatment in First Nations communities (INAC 2007). *The Plan of Action for Drinking Water in First Nations Communities,* funded in 2006 and *the First Nations Water and Wastewater Action Plan* funded in 2008, are some other government's attempts to improve access to safe drinking water for First Nations communities. *The Safe Drinking Water for First Nations Act Bill S-8* is one of the major legislative initiatives the federal government implemented to address drinking water issues on reserves (Ontario 2012) but Bill S-8 has been highly criticized for not providing adequate support to meet First Nations infrastructural needs, and for further eroding Indigenous self-governance (Assembly of First Nations 2012; Busby 2016). Budget 2016 invested $1.8 billion over five years to improve water infrastructure on First Nations reserves, end long-term drinking water advisories on the reserves, and prevent short-term advisories from becoming long-term. The government was able to lift 107 long-term drinking water advisories as of May 21, 2021, but 52 long-term drinking water advisories remain in effect in First Nations communities across Canada (Indigenous Services Canada 2021a). In terms of advancing Canada's success in addressing the drinking water crisis in First Nations reserves, factors such as ensuring leadership of First Nations in decision-making

processes, and respecting Indigenous knowledge are important considerations (Black and Mcbean 2017; Dyck et al. 2015; Patrick 2011).

Some First Nations communities have implemented a more holistic, multi-barrier approach to address their drinking water quality concerns (Dyck et al. 2015; Patrick 2011; Plummer et al. 2010). This approach includes implementing a community plan for: (1) source water protection to assure the quality and the quantity of water available to fulfil the needs of society and ecosystems, (2) water treatment processes to ensure the safety of water for human consumption, (3) proper water storage and distribution systems to avoid environmental and operational stresses which can cause water quality deterioration, (4) monitoring of water quality throughout the distribution system, and (5) implementing emergency response plans for each step of the multi-barrier approach. However, lack of sufficient financial, technical, legal, and human support is one of the main constraints for implementing and continuing the multi-barrier approach (Patrick 2011). Social, cultural, economic and political contexts within each community may also be challenging for applying new approaches (Dyck et al. 2015). Identifying financial constraints at each step in the multi-barrier approach and exploring various funding opportunities could improve opportunities for strengthening water treatment and distribution systems in communities.

1.2. Drinking water systems in First Nations communities

A national assessment that was completed about a decade ago (Neegan Burnside 2011a) showed that a majority of First Nations communities (98%) had water systems, whereas in the remaining communities (2%) relied on individual wells (Neegan Burnside 2011a). Most First Nations communities have conventional water treatment plants for which water treatment includes coagulation, flocculation, sedimentation, (granular media) filtration, and disinfection removes

sediments, pollutants associated with sediment, and microorganisms (Matilainen and Sillanpää 2010; Neegan Burnside 2011a). The source water supplying the water treatment plant is primarily groundwater (46%) and surface water (29%), with the remaining water systems relying on a Municipal Type Agreement (19%) or groundwater under the direct influence of surface water (6%).

The '*Protocol for safe drinking water for First Nations communities*' (INAC 2010) outlines water treatment requirements, as influenced by the type and quality of water sources and the size of the population to be served by the water treatment plant. Soil acts as a filter and groundwater has a smaller risk of becoming contaminated with chemical and microbial contaminants than surface water (Camarillo et al. 2014). A minimum of 0.2 mg L^{-1} free residual chlorine concentration should be present in the water distribution system, including water mains, reservoirs, pumping stations, valves, and water trucks at the time of delivery regardless of the water source (INAC 2010). When using groundwater as source water to a WTP, the minimum requirement is two-fold: 1) primary disinfection and 2) secondary disinfection. Primary disinfection aims to kill or inactivate pathogenic microorganisms possibly present in source water, whereas secondary disinfection is intended to maintain 0.2 mg L^{-1} of free residual chlorine in the water distribution system to prevent microbial regrowth. Primary and secondary disinfection together should be able to remove 99% of *Giardia* and *Cryptosporidium* cysts and viruses before entering the distribution system (INAC 2010). When using surface water as source water to a WTP, the additional requirement is filtration along with primary and secondary disinfection, which should ensure 99% removal of *Giardia* cysts, 99.9% removal of *Cryptosporidium* cysts and 99.99% removal of viruses.

1.3. Piped versus Cistern Systems on First Nations reserves

Cisterns are used to store drinking water by many homes in some First Nations communities with

no access to piped water supply from the water treatment plant (Baird et al. 2013). Cisterns are filled with water delivered by a water truck from the community water WTP. The document, *Guidance for Designing, Installing, Maintaining and Decommissioning Drinking Water Cisterns in First Nations Communities South of 60°*, published by Health Canada (Health Canada 2012), provides information to cistern installers, operators and residents in all aspects of cistern placement and use. Cisterns could be concrete or fibreglass cisterns located belowground, or polyethylene cisterns located in aboveground insulated shelters (Health Canada 2012). The choice of the type and location of the cisterns depends on funding availability and environmental factors such as temperature, soil conditions and drainage properties such as stoniness, bulk density of soil, groundwater table and slope, and potential sources of contamination. Often, the use of cisterns in some of the homes in First Nations communities is a necessity due to the unavailability of a piped distribution system. Cleaning of cisterns after installation and once a year is recommended by Health Canada (Health Canada 2012). The band council is responsible for cistern cleaning using certified professionals (with confined environment certification). However, shortages in funding and human resources are real challenges that interfere with the regular cleaning and maintenance of cisterns in First Nations communities.

The use of cisterns to store drinking water mainly poses three types of risks for contamination. First, during water transportation from the WTP to the cistern, physical contamination might occur through the entry of foreign matter like dust/soil directly into the truck or cisterns (Baird et al. 2013). Substantial total coliform and *E. coli* counts have been reported in cisterns and water delivery trucks in a First Nations community in Manitoba (Farenhorst et al., 2017). The presence of microorganisms belonging to the Burkholderiales order in these same samples and the fact that these bacteria are commonly found in soil and atmospheric dust suggested that water delivery

trucks can be contaminated with dust/soil when the trucks are filled at the water treatment plant. Secondly, cisterns can be contaminated with external water leaking through cracks or poorly installed material (Farenhorst et al. 2017; Wright et al. 2018). Belowground concrete, polyethylene or fibreglass cisterns sometimes show evidence of structural damage induced by soil and boulder movement and/or improper protection of the cistern. Organic material seeping in with external water can deposit at the bottom and on wall surfaces of the cisterns, causing deterioration of water quality over time (Artiola et al. 2012). Therefore, depending on environmental conditions, such as temperature and the intensity and frequency of rainfall, and soil conditions, such as permeability and water holding capacity, changes in water quality over the year can be expected (Mahler et al. 2000). Lastly, water quality can deteriorate during storage in cisterns due to processes occurring within the cistern itself (Baird et al. 2013). For example, loss of residual chlorine through decay via reaction with organic and inorganic matter, biofilms and walls and volatilization during prolonged storage can increase the risk of microbial regrowth (Graham and Vanderslice 2007; Xu et al. 2018). Increased contact time during water storage in cisterns may also facilitate the formation of total trihalomethanes (TTHMs), mainly when cistern water contains biofilms, dissolved organic matter (DOM), or when cisterns are contaminated with sediments (Abokifa et al. 2016; Artiola et al. 2012). Less frequent cleaning/disinfection of cisterns and water delivery trucks has exacerbated contamination problems (Baird et al. 2013; Bradford et al. 2018; Mi et al. 2019).

Even though pipe water distribution is the safest and most efficient method, substantial maintenance and management are necessary to ensure delivered water safety (Sadiq et al. 2004). Microbial (including pathogens) intrusion resulting from a malfunction of the system, e.g., due to low pressure and cross-connections, is one of the major ways by which treated water in the pipe distribution system becomes contaminated (Sadiq et al. 2004). Microbial regrowth is another

concern, especially in biofilms on pipe walls and low chlorine concentrations towards the end of the pipe distribution system (Holinger et al. 2014; Lechevallier et al. 1996; Simoes and Simoes 2013).

1.4. Drinking water quality monitoring in First Nations communities

First Nations communities rely on Community Based Water Monitors and Environmental Health Officers (EHOs) for sampling and testing water distribution systems (Health Canada 2017). Health Canada is responsible for providing guidance and training to Community Based Water Monitors on drinking water quality monitoring to test bacteriological parameters (Health Canada 2007). An EHO from Health Canada is responsible for testing all bacteriological, physical, chemical and radiological parameters and reviewing and interpreting all drinking water quality results according to the latest version of the Guidelines for Canadian Drinking Water Quality, including bacteriological, chemical, physical and radiological parameters. EHOs are also responsible for issuing water advisories, as needed. A recent Health Canada Departmental Results Report (Health Canada 2017) indicates that, on average, only 80% of public water distribution systems in First Nations communities met recommended sampling frequency in 2016-2017, highlighting the need for further interventions by Health Canada to meet the recommended sampling frequency for bacteriological parameters.

The growth and persistence of pathogens in water distribution systems depend on many variables, including temperature, pipe surface, nutrient levels, and type and concentration of disinfectant residuals (Berry et al. 2006; Ndiongue et al. 2005). Generally, it is not feasible or cost-effective to test drinking water for all the known pathogens (WHO 2017). Correlations between the presence of pathogens and fecal bacteria have been well-established (Leclerc et al. 2001; WHO 2017) and

as a result, water monitoring uses total coliforms and *E. coli* as indicators of the potential presence of pathogenic organisms. The Protocol for Safe Drinking Water in First Nations Communities (INAC 2010) recommends three types of water quality monitoring. First, operational monitoring which the water treatment plant operator conducts under the direction of the band council in order to verify water quality and system performance. This includes testing at the WTP of raw and treated water and testing samples from the piped water distribution system every week for microbial water quality and daily chlorine residual testing of treated water at the WTP and one sample per week from the piped distribution system. The number of samples required for microbial testing and the frequency of testing depend on the size of the water system (population served) and the type of source water (ground water vs. surface water). Where a truck delivery system is in use, the water truck operator must also monitor residual chlorine at the tank outlet (free residual chlorine should be greater than 0.2 mg L^{-1} at the tank outlet) at least once per day and maintain a record.

The second type of monitoring is quality assurance and quality control, in which 10% of all samples for microbiological parameters at the operational monitoring will be submitted to a (second) accredited laboratory for comparison. Third, compliance and third-party monitoring are conducted by Community Based Water Monitors or EHO under the Environmental Public Health Program implemented by Health Canada (Health Canada 2007; INAC 2010). This includes a periodic sampling of piped distribution systems, cisterns and public-access wells by EHO and analysis of microbial quality of drinking water. These data must be accompanied by the residual chlorine results for that time and location.

The Protocol for Safe Drinking Water in First Nations Communities (INAC 2010) does not provide any specific information on testing cisterns for microbial quality. Moreover, Health Canada recommends only quarterly sampling for cisterns (Health Canada 2007). Health Canada also

recommends that sampling and analysis of damaged or poor quality cisterns should not be conducted until the repairs are done or the cisterns are cleaned and disinfected. However, because of a lack of funding, households continue to rely on damaged cisterns but might not drink the water because of the fear of water-borne illnesses (Anderson 2020).

Microbial analyses are done chiefly using Colilert field kits and relevant equipment such as sterile vessels and incubators (Health Canada 2007). If these are not readily available within the community, samples must be sent to an accredited laboratory for testing. The large costs associated with testing samples in an accredited laboratory might mean that the community is limited to only being able to sample at a few locations (e.g., WTP, school, nursing station) or sample households less frequently. The maximum acceptable concentration for bacteriological quality of drinking water is no *E. coli* and coliform detectable per 100 mL (Health Canada 2019; WHO 2017).

Other routine water monitoring in First Nations communities includes turbidity in one raw water sample per month and a continuous system in each filter effluent line (INAC 2010). Turbidity of chemically assisted filtration should be less than or equal to 0.3 Nephelometric Turbidity unit (NTU) in 95% of samples or 95% of the year where turbidity should never exceed 1 NTU (Health Canada 2007). For slow sand and diatomaceous earth filtration, the turbidity should be $\leq$ 1 NTU in 95% of the sample or time and should never exceed 3 NTU.

For TTHMs, quarterly monitoring of treated water from the WTP and from the distribution system where there is the highest potential for TTHMs formation is recommended (Health Canada 2007). The EHO, in consultation with Indigenous Services Canada and Public Works and Government Services Canada, decides on sample frequency, sample locations and water sampling. Samples are to be analyzed by an accredited laboratory and it is the responsibility of the EHO to provide the

data to the First Nations community. The maximum acceptable concentration (MAC) of TTHMs in water samples is 100 µg L^{-1} (Health Canada 2019). However, if five years of TTHMs monitoring results never exceed 50% of the MAC, testing frequency can be reduced to once every three years (INAC 2010). In addition, once a year, analysis of at least one raw water sample and treated water sample from the water treatment plant and samples from water distribution systems is recommended for several chemical and physical water quality parameters such as heavy metals, alkalinity, hardness, dissolved organic matter, and total solid (Health Canada 2007). Sampling is done by the community EHO and analysis is typically done through an accredited laboratory.

1.5. Disinfection and chlorine chemistry

Microbial contaminants in drinking water are of great concern due to their detrimental effects on human health (Tallon et al. 2005). Disinfection, which inactivates biological pathogens, is imperative to prevent water-born illnesses due to drinking water consumption (Crittenden et al. 2012). The most widely used disinfectant is chlorine, which has been used for around a hundred years in drinking water treatment processes. Chlorine has a high oxidizing potential which leads to high disinfection effectiveness (Gopal et al. 2007). Use of chlorine is relatively easy and cost-effective, while the capital costs required for installing the treatment system are also low. Disinfection using chlorine can also provide a minimum amount of residuals for a reasonably prolonged period to protect against microbial regrowth. In water treatment processes, chlorine is generally applied as chlorine gas (Cl_2), sodium hypochlorite or calcium hypochlorite (Brown et al. 2011). Upon addition of chlorine to water, it forms two chlorine species: HOCl (electrically neutral) and the hypochlorite ion (OCl^-, electrically negative), which are collectively referred to as free chlorine (Brown et al. 2011) as described in Eq. 1, 2 and 3.

Gaseous chlorine:

$$Cl_2 + H_2O \leftrightarrow H^+ + Cl^- + HOCl \quad [1]$$

Sodium hypochlorite:

$$NaOCl + H_2O \leftrightarrow H_2O + OCl^- + Na^+ \quad [2]$$

Calcium hypochlorite:

$$Ca(OCl)_2 + 2H_2O \leftrightarrow 2HOCl + 2OH^- + Ca^{2+} \quad [3]$$

The hypochlorous acid (HOCl) generated in Eq. 1 and Eq. 3 will dissociate as follows (Eq. 4):

$$HOCl \leftrightarrow H^+ + OCl^- \quad [4]$$

Dissociation of HOCl is reversible and the HOCl:OCl^- ratio depends on the pH and temperature of the water. The pKa of HOCl is 7.53 at 25 °C where high pH favours the formation of OCl^- while low pH favours the formation of HOCl. At a constant pH, increasing temperature increases the dissociation of HOCl. However, maintaining a high HOCl concentration is important as HOCl is more reactive than OCl^- and is also a stronger disinfectant and an oxidant (White 1974). Free residual chlorine ranging from 0.2 to 4 mg L^{-1} in water distribution systems is recommended to prevent bacterial re-growth and to help protect treated water throughout the distribution system (Lechevallier et al. 1996).

Chlorine volatilization and reaction with various materials lead to chlorine decay, which reduces its effectiveness in suppressing microbial growth. Chlorine decay due to the reactions within the water is referred to as bulk chlorine decay (Powell et al. 2000a). Bulk chlorine decay can be affected by the types and amounts of organic and inorganic compounds, temperature, pH, and initial chlorine concentration of the water (Brown et al. 2011; Powell et al. 2000a). Chlorine can also

react with the biofilms and pipes of the distribution system or the walls of the storage tanks. This type of decay is known as chlorine wall decay. Wall chlorine decay is primarily affected by the mass transfer of chlorine from the bulk water to the walls, pipe diameter, type of pipe material, inner coating of pipe, age of the pipe wall and the presence of biofilms (Castro and Neves 2010; Xu et al. 2018; Zhang et al. 2017). The overall chlorine decay constant is defined as the sum of the bulk decay constant and the effective wall decay constant (Brown et al. 2011; Vasconcelos et al. 1997). Due to its non-selective oxidizing nature, chlorine can react with both inorganic and organic compounds in the water. Inorganic compounds at reduced valence state, such as bromide, ferrous, manganous, sulfide, and ammonium, are the most reactive compounds with chlorine in natural waters (Brown et al. 2011; Deborde and von Gunten 2008). The reaction rates of these reactions are high and can occur within seconds, even within the treatment process itself. However, the reactions between organic compounds and chlorine occur at slower rates and can proceed for up to three weeks, slowly producing disinfection by-products (Gallard and von Gunten 2002).

Chlorine decay in water distribution systems and source water has been commonly modelled according to first-order kinetics with respect to chlorine, where the rate of reaction is proportional to the concentration of chlorine (Chambers et al. 1995; Powell et al. 2000b). The first-order model (Eq. 5) has been shown to be the best fit to evaluate long-term chlorine decay and is simple to apply (Clark and Sivaganesan 2002).

$$C = C_0 \, exp(-k_b t) \qquad [5]$$

where C is the chlorine concentration (mg L^{-1}) at time t, C_0 is the initial chlorine concentration (mg L^{-1}), k_b is the bulk decay coefficient and t is the residence time in the pipe (d).

However, some researchers have raised concerns over the ability of the first order model to address

the initial fast decay and slow long-term decay phases (Clark and Sivaganesan 2002). Therefore, previous researchers tried different kinetic models to best describe chlorine bulk decay in water (Clark and Sivaganesan 2002; Haas and Karra 1984; Hua et al. 1999). In a study conducted to investigate chlorine decay in wastewater, Haas and Karra (1984) observed that the parallel first-order model (Eq. 6) outperformed all the other models, including the first-order model, because of its ability to address the initial fast component and the slow long-term chlorine residual decay component. Parallel first-order decay assumes two components to the reaction, each decaying according to a first-order law, but with different decay rate constants:

$$C = C_0 z\, exp\left(-k_{bfast} t\right) + C_0(1 - z) exp(-k_{bslow} t) \quad [6]$$

where z is the ratio of the fast to the slow reaction (dimensionless), k_{bfast} is the bulk decay constant for the fast reaction (h^{-1}), and k_{bslow} is the bulk decay constant for the slow reaction (h^{-1}).

Limited first-order decay (Eg. 7) assumes that a fraction of the initial chlorine concentration remains unchanged and only the remainder decays exponentially according to first-order kinetics. This can happen if the reaction is limited by other factors, such as insufficient organic matter content.

$$C = C^* + (C_0 + C^*) exp(-k_b t) \quad [7]$$

where C^* is limiting chlorine concentration (mg L^{-1}).

Second-order decay with respect to chlorine concentration and a second reactant is another commonly used and well-performing model. Clark and Sivaganesan (1998) initially developed the second-order chlorine decay model based on the concept that the reaction of chlorine can be summarized as a reaction between chlorine and all the organic and inorganic species that potentially

react with chlorine. This phenomenon was later applied to predict chlorine decay in drinking water by the same researchers as well as in several other studies and they used surrogates of organic matter such as total or dissolved organic matter and chemical oxygen demand as the second reactant (Al Heboos and Licskó 2017; Clark and Sivaganesan 2002; Powell et al. 2000b). Powell et al. (2000b) believe that this model would perform better for waters with low organic matter contents because chlorine is no longer the limiting factor. Al Heboos and Licskó (2017) found that the second-order model with chlorine and chemical oxygen demand as reactants performs better than the parallel first-order model. Second-order decay of chlorine residual with respect to chlorine concentration only is another possible decay model where the reaction rate is proportional only to chlorine concentration (Powell et al. 2000b). Power-law decay (n^{th} order) is also sometimes used where the reaction rate is proportional to the n^{th} power of chlorine concentration (Brown et al. 2011). Even though n^{th} order models perform well for chlorine decay in drinking water, they have only limited use since n is not related to measurable parameters and dimensions of the decay rate depends on the value of n, which makes it hard to define the relationship with measurable parameters (Powell et al. 2000b).

1.6. Trihalomethane formation and kinetics

Reactions between chlorine and natural organic matter (NOM) present in the water produce several harmful disinfection by-products, such as TTHMs and haloaceticacids (Rodriguez et al. 2003; Sadiq and Rodriguez 2004). To date, more than 600 disinfection by-products have been identified. Trihalomethanes are the most common disinfection by-products and are single carbon compounds with one to three halogens and they are volatile and slightly water-soluble (Kumari and Gupta 2015; Li and Mitch 2018). The regulated TTHMs (chloroform ($CHCl_3$), chlorodibromomethane

($CHClBr_2$), bromodichloromethane ($CHBrCl_2$) and bromoform ($CHBr_3$)) have been identified as potential carcinogens and can cause reproductive abnormalities by acting as teratogens and to a lesser extent as mutagens (Krasner 2009; Villanueva et al. 2014). Based on available toxicological information, governing agencies have set the maximum allowable concentration in treated water (Health Canada 2019; WHO 2017). In Canada, the MAC of TTHMs in treated water at 100 $\mu g\ L^{-1}$ (Health Canada 2019). The presence of TTHMs also indicates the presence of other chlorinated by-products (WHO 2005).

Trihalomethane formation depends mainly on the type of chlorine (or disinfection material) and initial chlorine concentration (Brown et al. 2011). Higher initial chlorine concentration results in higher TTHMs levels (Clark and Sivaganesan 1998). Free chlorine (Cl_2, sodium hypochlorite (NaOCl), and calcium hypochlorite ($Ca(OCl)_2$) produce higher TTHMs compared to chloramines and chlorine dioxide (Bougeard et al. 2010; Goslan et al. 2009; Hua and Reckhow 2007). TTHMs in the piped distribution system increase with increasing distance from the WTP, increasing residence time, and increasing water temperature (Liang and Singer 2003; Rodriguez et al. 2003; Wei et al. 2010). Drinking water pH is also a key factor that controls TTHMs formation. Higher pH levels increase the rate of trihalomethane formation, which increases the amount of TTHMs formed (Brown et al. 2011; Clark and Sivaganesan 1998). Inorganic compounds in the drinking water supply, mainly bromide concentration, are another important factor that controls trihalomethane formation (Brown et al. 2011; Gopal et al. 2007). Both HOBr and OBr^- react with NOM to form a mixture of brominated and mixed chloro-bromo derivatives. However, the formation of brominated trihalomethane depends on the Br_2 to NOM ratio and Br_2 to chlorine ratio (Brown et al. 2011). In a study examining the effect of contact time on trihalomethane formation, Liang and Singer (2003) observed that trihalomethane formed rapidly within the first few hours

after chlorine addition. After that, the trihalomethane formation rate decreased as chlorine and NOM concentrations decreased; however, it continued to produce trihalomethane over the period.

There are two types of models: (i) models developed based on empirical relationships and (ii) models based on kinetic reactions during chlorination. The complexity and uncertainty of reactions between chlorine and organic matter, make it hard to develop kinetic models for trihalomethane formation in water (Brown et al. 2011). However, many empirical mathematical models to estimate trihalomethane concentrations have been developed under laboratory conditions and full or pilot-scale field studies based on relationships among TTHM concentrations and parameters that affect their formation, such as temperature, NOM concentration, pH, and bromide concentration using linear, nonlinear or multivariate regression (Di Cristo et al. 2013; Ike et al. 2020). The applicability and suitability of these empirical models to specific water systems is questionable because the prediction performance of these empirical models has been observed to be poor and dependent on the operational conditions of the water treatment/distribution systems such as water treatment method, type of disinfectant and residence time of the treated water and water quality characteristics (Di Cristo et al. 2013). However, the performance of the empirical model can be improved by using field data instead of lab-scale data and by including parameters other than water quality measures, such as reaction time or time it takes for treated water to reach the consumer.

Li and Zhao (2006) found that the rate of trihalomethane formation is first-order with respect to chlorine, first-order with respect to total organic carbon, and the overall reaction is second order. Di Cristo et al (2013) applied first-order kinetics with respect to chlorine (HOCl) to several water systems and observed a better fit between predicted and observed concentration of TTHMs. However, like empirical models, the kinetic constants of these models also depend on water quality parameters and operational conditions and environmental conditions, which influence the chlorine

decay and the TTHMs formation. Therefore, additional studies are needed to investigate the suitability of empirical models and the calibration of kinetic models with the specific water system.

1.7. Research objectives

In First Nations communities in Manitoba, 51% of homes have direct piped water from WTPs and 31% of the homes have their drinking water delivered by truck and stored in cisterns. The remaining homes (18%) rely on well water or do not have running water (Neegan Burnside 2011b). Recent research confirms that water samples collected from households on First Nations reserves in Manitoba have unacceptable levels of total coliforms and *E.coli* (Farenhorst et al. 2017; Fernando et al. 2016; Mi et al. 2019). In a fly-in First Nations reserve in Manitoba, the water distribution method (piped, belowground concrete cisterns, storage containers etc.) influenced the frequency and levels of total coliforms detected in the tap water of homes (Farenhorst et al. 2017; Fernando et al. 2016). Previous studies conducted in a few First Nations communities also observed lower chlorine concentrations in cistern water compared to piped water (Farenhorst et al. 2017; Mi et al. 2019). Loss of residual chlorine through decay via reaction with organic and inorganic matter, biofilms and walls and volatilization during prolonged storage can increase the risk of microbial regrowth (Graham and Vanderslice 2007; Xu et al. 2018). Increased contact time during water storage in cisterns may also facilitate the formation of TTHMs, particularly when cistern water contains biofilms and/or NOM or when cisterns are contaminated with sediments (Abokifa et al. 2016; Artiola et al. 2012).

Cistern systems common in First Nations communities in Canada include belowground concrete cisterns, fibreglass cisterns and aboveground polyethylene cisterns. Therefore, this research included different types of cisterns to compare the bacterial quality and TTHM concentrations of

tap water in households relying on these cisterns with households in the same community relying on piped water. The researchers collaborated with three First Nations communities representing three Tribal Councils in Manitoba, each having a different predominant cistern system (belowground concrete cisterns, belowground fibreglass cisterns or aboveground polyethylene cisterns). Each community utilized surface water as source water for water treatment and sodium hypochlorite for disinfection. Specific objectives of this thesis were to (i) determine the effects of cistern use on drinking water bacterial safety relative to drinking water piped directly to homes from the WTP (Chapter 2); (ii) examine the effect of the water distribution system (piped vs. cistern) on the types and concentration of trihalomethanes and related water quality parameters in the water distribution systems in three First Nations communities (Chapter 3) and (iii) determine in-situ and in-vitro free residual chlorine decay kinetics and to evaluate the formation of trihalomethanes in drinking water stored in belowground concrete cisterns (Chapter 4).

1.8. Thesis outline

This thesis is written following the thesis guidelines of the Department of Soil Science and the Faculty of Graduate Studies, University of Manitoba. The thesis is formatted in manuscript style with five chapters, including the introduction (Chapter 1), three research chapters as given below (2 – 5) and the overall synthesis (Chapter 5).

Chapter 2: Water delivery system effects on coliform bacteria in tap water in First Nations reserves in Manitoba, Canada.

Amarawansha, G. E. A., Zvomuya, F., & Farenhorst, A. 2021. Water delivery system effects on coliform bacteria in tap water in First Nations reserves in Manitoba, Canada. Environmental

Monitoring and Assessment, 193(6), 1-16.

Chapter 3: Trihalomethanes in drinking water from three first nation reserves in Manitoba, Canada.

Amarawansha, E. A. G. S., Zvomuya, F., Farenhorst, A. 2021. Trihalomethanes in drinking water from three first nation reserves in Manitoba, Canada. This has been submitted to the Environmental Monitoring and Assessment journal. Submission ID: EMAS-D-21-03121.

Chapter 4: Effects of belowground cistern storage of drinking water on chlorine decay kinetics and trihalomethane formation

Amarawansha, E. A. G. S., Zvomuya, F., Farenhorst, A. 2021. Effects of belowground cistern storage of drinking water on chlorine decay kinetics and trihalomethane formation. Will be submitted to Circumpolar Health Journal.

1.9. References

Abokifa, A. A., Yang, Y. J., Lo, C. S., & Biswas, P. (2016). Investigating the role of biofilms in trihalomethane formation in water distribution systems with a multicomponent model. *Water Research*, *104*, 208–219. https://doi.org/10.1016/j.watres.2016.08.006

Aiello, R.. Trudeau won't commit to keeping promised timeline to end drinking water advisories. *CTV News.ca*, 2020, October 23, (Online) https://www.ctvnews.ca/politics/trudeau-won-t-commit-to-keeping-promised-timeline-to-end-drinking-water-advisories-1.5158005. Accessed on November 02, 2021

Al Heboos, S., & Licskó, I. (2017). Application and comparison of two chlorine decay models for predicting bulk chlorine residuals. *Periodica Polytechnica Civil Engineering*, *61*, 7–13. https://doi.org/10.3311/PPci.9273

Anderson, K. 2021. Perceptions of household drinking water across a variety of water distribution systems in three First Nations in Manitoba. MSc thesis, University of Manitoba, Winnipeg, Manitoba, Canada.

Artiola, J., Rock, C., & Hix, G. (2012). Water storage tank disinfection, testing, and maintenance. The University of Arizona Cooperative Extension. College of Agriculture and Life Sciences, University of Arizona, Tucson, AZ, USA.

Assembly of First Nations. (2012). *Assembly of First Nations Submission to the Standing Senate Committee on Aboriginal Peoples Bill S-8: Safe Drinking Water for First Nations Act About the Assembly of First Nations*. www.afn.ca. Accessed 18 September 2020

Baijius, W., & Patrick, R. J. (2019). "We don't drink the water here": The reproduction of undrinkablewater for First Nations in Canada. *Water (Switzerland)*, *11*. https://doi.org/10.3390/w11051079

Baird, J. M., Summers, R., & Plummer, R. (2013). Cisterns and safe drinking water in Canada. *Canadian Water Resources Journal*, *38*, 121–134.https://doi.org/10.1080/07011784.2013.780790

Berry, D., Xi, C., & Raskin, L. (2006). Microbial ecology of drinking water distribution systems. *Current Opinion in Biotechnology*, *17*, 297–302. https://doi.org/10.1016/j.copbio.2006.05.007

Black, K., & Mcbean, E. (2017). First Nations' water sustainability and Security Strategy: Tools and methodologies for community-driven processes for water treatment in Indigenous

communities. *Technology in Society*, *50*, 57–65. https://doi.org/10.1016/j.techsoc.2017.04.004

Bougeard, C. M. M., Goslan, E. H., Jefferson, B., & Parsons, S. A. (2010). Title: Comparison of the disinfection by-product formation potential of treated waters exposed to chlorine and monochloramine. *Water Research*, *44*, 729–740. https://doi.org/10.1016/j.watres.2009.10.008

Bradford, L., Waldner, C., McLaughlin, K., Zagozewski, R., & Bharadwaj, L. (2018). A mixed-method examination of risk factors in the truck-to-cistern drinking water system on the Beardy's and Okemasis First Nation Reserve, Saskatchewan. *Canadian Water Resources Journal*, *43*, 383–400. https://doi.org/10.1080/07011784.2018.1474139

Brown, D., Bridgeman, J., & West, J. R. (2011). Predicting chlorine decay and THM formation in water supply systems. *Reviews in Environmental Science and Bio-Technology*, *10*, 79–99. https://doi.org/10.1007/s11157-011-9229-8

Busby, K. (2016). Troubling waters: Recent developments in Canada on international law and the right to water and sanitation. *Canadian Journal of Human Rights*, *5*, 1–26.

Camarillo, M. K., Stringfellow, W. T., & Jain, R. (2014). Drinking Water Security for Engineers, Planners, and Managers: Integrated Water Security Series. *Drinking Water Security for Engineers, Planners, and Managers: Integrated Water Security Series*, *1*, 1–241. https://doi.org/10.1016/C2012-0-06924-4

Castro, P., & Neves, M. (2010). Chlorine decay in water distribution systems case study–Lousada network. *Electronic Journal of Environmental, Agricultural and Food Chemistry*, *2*, 261–266.

Chambers, V. K., Creasey, J. D., & Joy, J. S. (1995). Modeling free and total chlorine decay in potable water distribution systems. *Journal of Water Supply Research and Technology-Aqua*, *44*, 60–69.

Clark, R. M., & Sivaganesan, M. (1998). Predicting chlorine residuals and formation of TTHMs in drinking water. *Journal of Environmental Engineering*. https://doi.org/10.1061/(ASCE)0733-9372(1998)124:12(1203)

Clark, R. M., & Sivaganesan, M. (2002). Predicting Chlorine Residuals in Drinking Water: Second Order Model. *Journal of Water Resources Planning and Management*, *128*, 152–161. https://doi.org/10.1061/ASCE0733-94962002128:2152

Crittenden, J. C., Trussell, R. R., Hand, D. W., Howe, K. J., & Tchobanoglous, J. (2012). *MWH ' s Water Treatment Principles and Design* (3rd ed.). New Jersey: John Wiley & Sons, Inc.

Deborde, M., & von Gunten, U. (2008). Reactions of chlorine with inorganic and organic compounds during water treatment - Kinetics and mechanisms: A critical review. *Water Research*, *42*, 13–51. https://doi.org/10.1016/j.watres.2007.07.025

Di Cristo, C., Esposito, G., & Leopardi, A. (2013). Modelling trihalomethanes formation in water supply systems. *Environmental technology*, *34*, 61–70. https://doi.org/10.1080/09593330.2012.679702

Dyck, T., Plummer, R., & Armitage, D. (2015). Examining First Nations' approach to protecting water resources using a multi-barrier approach to safe drinking water in Southern Ontario, Canada. *Canadian Water Resources Journal*, *40*, 204–223. https://doi.org/10.1080/07011784.2015.1033759

EKOS Research Associates Inc. (2011). *Perceptions of Drinking Water Quality in First Nations Communities and General Population*. Ottawa, Ontario. https://www.ekospolitics.com/articles/015-11.pdf

Farenhorst, A., Li, R., Jahan, M., Tun, H. M., Mi, R., Amarakoon, I., et al. (2017). Bacteria in drinking water sources of a First Nation reserve in Canada. *Science of the Total Environment*, *575*, 813–819. https://doi.org/10.1016/j.scitotenv.2016.09.138

Fernando, D. M., Tun, H. M., Poole, J., Patidar, R., Li, R., Mi, R., et al. (2016). Detection of antibiotic resistance genes in source and drinking water samples from a first nations community in Canada. *Applied and Environmental Microbiology*, *82*, 4767–4775. https://doi.org/10.1128/AEM.00798-16

First Nations Health Authority. (2021). Drinking Water Advisories. *First Nations Health Authority*. https://www.fnha.ca/what-we-do/environmental-health/drinking-water-advisories. Accessed 20 February 2021

Gallard, H., & von Gunten, U. (2002). Chlorination of natural organic matter: kinetics of chlorination and of THM formation. *Water research.*, *36*, 65–74. https://doi.org/10.1016/S0043-1354(01)00187-7

Galway, L. P. (2016). Boiling over: A descriptive analysis of drinking water advisories in first nations communities in Ontario, Canada. *International Journal of Environmental Research and Public Health*, *13*, 505. https://doi.org/10.3390/ijerph13050505

Gopal, K, Tripathy, S. S., Bersillon, J. L., & Dubey, S. P. (2007). Chlorination byproducts, their toxicodynamics and removal from drinking water. *Journal of Hazardous Materials*, *140*, 1–6. https://doi.org/10.1016/j.jhazmat.2006.10.063

Goslan, E. H., Krasner, S. W., Bower, M., Rocks, S. A., Holmes, P., Levy, L. S., & Parsons, S. A. (2009). A comparison of disinfection by-products found in chlorinated and chloraminated drinking waters in Scotland. *Water Research*, 43, 4698–4706. https://doi.org/10.1016/j.watres.2009.07.029

Graham, J. P., & Vanderslice, J. (2007). The effectiveness of large household water storage tanks for protecting the quality of drinking water. *Journal of Water and Health*, 5, 307–313. https://doi.org/10.2166/wh.2007.011b

Haas, C. N., & Karra, S. B. (1984). Kinetics of Wastewater Chlorine Demand Exertion. *Journal (Water Pollution Control Federation)*, *56*, 170–173.

Health Canada. (2007). *Procedure Manual for Safe Drinking Water in First Nations Communities South of 60°.* First Nations and Inuit Health Branch, Health Canada, Ottawa, Ontario.

Health Canada. (2009). *Guidance for Issuing and Rescinding Drinking Water Avoidance Advisories in Emergency Situations*. Ottawa, Ontario. www.healthcanada.gc.ca. Accessed 18 September 2020

Health Canada. (2012). *Guidance for Designing, Installing, Maintaining and Decommissioning Drinking Water Cisterns in First Nations Communities South of 60°.* First Nations and Inuit Health Branch, Health Canada, Ottawa, Ontario.

Health Canada. (2015). *Guidance for Issuing and Rescinding Boil Water Advisories in Canadian Drinking Water Supplies.* Water and Air Quality Bureau, Healthy Environments and Consumer Safety Branch, Health Canada, Ottawa, Ontario.

Health Canada. (2017). *Health Canada Departmental Results Report 2016-2017.* https://www.canada.ca/content/dam/hc-sc/documents/corporate/transparency/corporate-

management-reporting/report-plans-priorities/2016-2017/2016-17-drr-final-eng.pdf. Accessed 22 September 2020

Health Canada. (2019). *Guidelines for Canadian Drinking Water Quality Summary Table*. Water and Air Quality Bureau, Healthy Environments and Consumer Safety Branch, Health Canada, Ottawa, Ontario.

Holinger, E. P., Ross, K. A., Robertson, C. E., Stevens, M. J., Harris, J. K., & Pace, N. R. (2014). Molecular analysis of point-of-use municipal drinking water microbiology. *Water research.*, *49*, 225–235. https://doi.org/10.1016/j.watres.2013.11.027

Hua, F., West, J. R., Barker, R. A., & Forster, C. F. (1999). Modelling of chlorine decay in municipal water supplies. *Water Research*, *33*, 2735–2746. https://doi.org/10.1016/S0043-1354(98)00519-3

Hua, G., & Reckhow, D. A. (2007). Comparison of disinfection byproduct formation from chlorine and alternative disinfectants. *Water Research, 41*, 1667–1678 https://doi.org/10.1016/j.watres.2007.01.032

Ike, I. A., Karanfil, T., Ray, S. K., & Hur, J. (2020). A comprehensive review of mathematical models developed for the estimation of organic disinfection byproducts. *Chemosphere*, *246*, 125797. https://doi.org/10.1016/j.chemosphere.2019.125797

INAC. (2007). *Summative evaluation of the First Nations Water Management Strategy*. https://www.rcaanc-cirnac.gc.ca/eng/1100100012016/1543929014466

INAC. (2010). *Protocol for Centralised Drinking Water Systems in First Nations Communities - Standards for Design, Construction, Operation, Maintenance, and Monitoring of Centralised Drinking Water Systems*. Gatineau, Quebec. http://www.ainc-inac.gc.ca/enr/wtr/index-eng.asp. Accessed 25 August 2020

Indigenous Services Canada. (2021a). Ending long-term drinking water advisories. *Indigenous Services Canada*. https://www.sac-isc.gc.ca/eng/1506514143353/1533317130660. Accessed 20 February 2021

Indigenous Services Canada. (2021b). Short-term drinking water advisories. *Indigenous Services Canada*. https://www.sac-isc.gc.ca/eng/1562856509704/1562856530304#wb-auto-6. Accessed 08 June 2021

Krasner, S. W. (2009). The formation and control of emerging disinfection by-products of health concern. *Philosophical Transactions of the Royal Society A: Mathematical, Physical and Engineering Sciences*, *367*, 4077–4095. https://doi.org/10.1098/rsta.2009.0108

Kumari, M., & Gupta, S. K. (2015). Modeling of trihalomethanes (THMs) in drinking water supplies: a case study of eastern part of India. *Environmental Science and Pollution Research*, *22*, 12615–12623. https://doi.org/10.1007/s11356-015-4553-0

LeChevallier, M. W., Welch, N. J., & Smith, D. B. (1996). Full-scale studies of factors related to coliform regrowth in drinking water. *Applied and Environmental Microbiology*, *62*, 2201–2211. https://doi.org/10.1128/aem.00794-10

Leclerc, H., Mossel, D. A. A., Edberg, S. C., & Struijk, C. B. (2001). Advances in the Bacteriology of the Coliform Group: Their Suitability as Markers of Microbial Water Safety. *Annual Review of Microbiology*, *55*, 201–234. https://doi.org/10.1146/annurev.micro.55.1.201

Li, X. F., & Mitch, W. A. (2018). Drinking water disinfection byproducts (DBPs) and human health effects: multidisciplinary challenges and opportunities. *Environmental Science and Technology*, *52*, 1681–1689. https://doi.org/10.1021/acs.est.7b05440

Li, X., & Zhao, H.B. (2006). Development of a model for predicting trihalomethanes propagation in water distribution systems. *Chemosphere*, *62*, 1028–1032. https://doi.org/10.1016/j.chemosphere.2005.02.002

Liang, L., & Singer, P. C. (2003). Factors influencing the formation and relative distribution of haloacetic acids and trihalomethanes in drinking water. *Environmental Science & Technology*, *37*, 2920–2928. https://doi.org/10.1021/es026230q

Mahler, B. J., Personne, J. C., Lods, G. F., & Drogue, C. (2000). Transport of free and particulate-associated bacteria in karst. *Journal of Hydrology*, *238*, 179–193. https://doi.org/10.1016/s0022-1694(00)00324-3

Matilainen, A., Vepsalainen, M., & Sillanpaa, M. (2010). Natural organic matter removal by coagulation during drinking water treatment: A review. *Advances in Colloid and Interface Science*, *159*, 189–197. https://doi.org/10.1016/j.cis.2010.06.007

Mi, R., Patidar, R., Farenhorst, A., Cai, Z., Sepehri, S., Khafipour, E., & Kumar, A. (2019). Detection of fecal bacteria and antibiotic resistance genes in drinking water collected from three First Nations communities in Manitoba, Canada. *FEMS Microbiology Letters*, *366*, 1–9. https://doi.org/10.1093/femsle/fnz067

Morrison, A., Bradford, L., & Bharadwaj, L. (2015). Quantifiable progress of the First Nations Water Management Strategy, 2001–2013: Ready for regulation? *Canadian Water Resources Journal*, *40*, 352–372. https://doi.org/10.1080/07011784.2015.1080124

Murphy, H. M., Bhatti, M., Harvey, R., & McBEAN, E. A. (2016). Using Decision Trees to Predict Drinking Water Advisories in Small Water Systems. *Journal - American Water Works Association*, *108*, E109–E118. https://doi.org/10.5942/jawwa.2016.108.0008

Murphy, H. M., Corston-Pine, E., Post, Y., & McBean, E. A. (2015). Insights and opportunities: Challenges of Canadian First Nations drinking water operators. *International Indigenous Policy Journal, 6*. https://doi.org/10.18584/iipj.2015.6.3.7

Ndiongue, S., Huck, P. M., & Slawson, R. M. (2005). Effects of temperature and biodegradable organic matter on control of biofilms by free chlorine in a model drinking water distribution system. *Water Research, 39*, 953–964. https://doi.org/10.1016/j.watres.2004.12.019

Neegan Burnside. (2011a). *National Assessment of First Nations Water and Wastewater Systems National Roll-Up Report*. Orangeville, Ontario

Neegan Burnside. (2011b). *National Assessment of First Nations Water and Wastewater Systems - Manitoba Regional Roll-Up Report*. Orangeville, Ontario

O'Gorman, M., & Penner, S. (2018). Water infrastructure and well-being among First Nations, Métis and Inuit individuals in Canada: what does the data tell us? *Environmental Science and Pollution Research*, *25*, 33038–33055. https://doi.org/10.1007/s11356-018-1258-1

Patrick, R. J. (2011). Uneven access to safe drinking water for First Nations in Canada: Connecting health and place through source water protection. *Health & Place*, *17*, 386–389. https://doi.org/10.1016/j.healthplace.2010.10.005

Plummer, R., Velaniškis, J., de Grosbois, D., Kreutzwiser, R. D., & de Loë, R. (2010). The development of new environmental policies and processes in response to a crisis: the case of the multiple barrier approach for safe drinking water. *Environmental science & policy, 13*, 535–548. https://doi.org/10.1016/j.envsci.2010.05.004

Powell, J. C., Hallam, N. B., West, J. R., Forster, C. F., & Simms, J. (2000a). Factors which control bulk chlorine decay rates. *Water Research, 34*, 117–126. https://doi.org/10.1016/S0043-1354(99)00097-4

Powell, J. C., West, J. R., Hallam, N. B., Forster, C. F., & Simms, J. (2000b). Performance of various kinetic models for chlorine decay. *Journal of Water Resources Planning and Management*, *126*, 13–20.

Rodriguez, M. J., Vinette, Y., Sérodes, J.-B., & Bouchard, C. (2003). Trihalomethanes in drinking water of greater Québec region (Canada): Occurrence, variations and modelling. *Environmental Monitoring and Assessment*, *89*, 69–93. https://doi.org/10.1023/A:1025811921502

Sadiq, R., Kleiner, Y., & Rajani, B. (2004). Aggregative risk analysis for water quality failure in distribution networks. *Journal of Water Supply Research and Technology-Aqua*, *53*, 241–261. %3CGo

Sadiq, R., & Rodriguez, M. J. (2004). Disinfection by-products (DBPs) in drinking water and predictive models for their occurrence: a review. *Science of the Total Environment*, *321*, 21–46. https://doi.org/10.1016/j.scitotenv.2003.05.001

Sarkar, A., Hanrahan, M., & Hudson, A. (2015). Water insecurity in Canadian Indigenous communities: Some inconvenient truths. *Rural and Remote Health*, *15*.

Simoes, L. C., & Simoes, M. (2013). Biofilms in drinking water: problems and solutions. *Rsc Advances*, *3*, 2520–2533. https://doi.org/10.1039/c2ra22243d

Stewart, S., & Mashford-Pringle, A. (2019). Colonial Trauma: Complex, continuous, collective, cumulative and compounding effects on the health of Indigenous peoples in Canada and beyond. *International Journal of Indigenous Health*, *14*, 5–6. https://doi.org/10.32799/ijih.v14i2.32958

Tallon, P., Magajna, B., Lofranco, C., & Leung, K. T. (2005). Microbial indicators of faecal contamination in water: a current perspective. *water, ai*, *166*, 139–166.

Vasconcelos, J., Rossman, L. A., Grayman, W. M., Boulos, P. F., & Clark, R. M. (1997). Kinetics of chlorine decay. *American Water Works Association*, *89*, 54–65.

Villanueva, C. M., Kogevinas, M., Cordier, S., Templeton, M. R., Vermeulen, R., Nuckols, J. R., et al. (2014). Assessing exposure and health consequences of chemicals in drinking water: Current state of knowledge and research needs. *Environmental Health Perspectives*, *122*, 213–221. https://doi.org/10.1289/ehp.1206229

Wei, J. R., Ye, B. X., Wang, W. Y., Yang, L. S., Tao, J., & Hang, Z. Y. (2010). Spatial and temporal evaluations of disinfection by-products in drinking water distribution systems in Beijing, China. *Science of the Total Environment*, *408*, 4600–4606. https://doi.org/10.1016/j.scitotenv.2010.06.053

WHO. (2005). Trihalomethanes in Drinking-water: Background document for development of WHO Guidelines for Drinking-water Quality. http://www.who.int/water_sanitation_health/. Accessed 8 December 2020

WHO. (2017). *Guidelines for Drinking-water Quality - fourth edition incorporating the first addendum*. World Health Organization, Geneva, Switzerland.

Wright, C. J., Sargeant, J. M., Edge, V. L., Ford, J. D., Farahbakhsh, K., Shiwak, I., et al. (2018). Water quality and health in northern Canada: stored drinking water and acute gastrointestinal illness in Labrador Inuit. *Environmental Science and Pollution Research*, *25*, 32975–32987. https://doi.org/10.1007/s11356-017-9695-9

Xu, J., Huang, C., Shi, X., Dong, S., Yuan, B., & Nguyen, T. H. (2018). Role of drinking water biofilms on residual chlorine decay and trihalomethane formation: An experimental and modeling study. *Science of the Total Environment*, *642*, 516–525. https://doi.org/10.1016/j.scitotenv.2018.05.363

Zhang, C., Li, C., Zheng, X., Zhao, J., He, G., & Zhang, T. (2017). Effect of pipe materials on chlorine decay, trihalomethanes formation, and bacterial communities in pilot-scale water distribution systems. *International Journal of Environmental Science and Technology, 14*, 85–94. https://doi.org/10.1007/s13762-016-1104-2

2. WATER DELIVERY SYSTEM EFFECTS ON COLIFORM BACTERIA IN TAP WATER IN FIRST NATIONS RESERVES IN MANITOBA, CANADA.

2.1. Abstract

About one-half of the homes on First Nations reserves in Manitoba, Canada, receive piped water from a water treatment plant (WTP). Many other homes (31%) are equipped with cisterns that are filled by a water truck. Our objective was to determine how the use of cisterns affects drinking water safety relative to drinking water piped directly to homes from the WTP. The study included belowground concrete cisterns, belowground fibreglass cisterns, and aboveground polyethylene cisterns stored in insulated shelters. Results showed that, across sampling times and communities, 20% of the homes with piped water and 55% of the homes with cisterns had positive detects for total coliforms. The frequency and severity of *Escherichia coli* and total coliform contamination were greater in drinking water samples from belowground concrete and fibreglass cisterns than in piped water samples in each community. The level of contamination of belowground cistern water by coliform bacteria was greatest in late spring relative to other parts of the year. Data obtained under the Access to Information Act showed no statistical differences in the percent of satisfactory samples (no detects) between 2014 and 2018, suggesting no clear indication of improved water quality in any of the Tribal Councils of which these three and other communities are a member. Our results point to the need for additional drinking water treatment in homes supplied by belowground concrete or fibreglass cisterns, regular maintenance and replacement of belowground cisterns with aboveground cisterns or piped water to reduce the risk of water-borne illnesses.

Keywords: Drinking water, Cisterns, Piped water, Chlorine, *Escherichia coli*

2.2. Introduction

First Nations reserves across Canada are frequently under a drinking water advisory, often due to unacceptable levels of coliform bacteria (Bradford et al. 2018). Previous publications have discussed a range of factors that hinder safe drinking water supplies in homes on First Nations reserves in Canada (Patrick 2011; Daley et al. 2015). Contributing factors might include the lack of financial and human support for small-scale water distribution systems; the absence of a regulatory framework; the lack of clarity regarding governments' roles and responsibilities and other factors. American Indians/Alaskan Natives populations living in small communities also experience poor water services (Thomas et al. 2016). As such, Indigenous Peoples of the Americas living in small communities suffer from infections caused by poor water quality at the household level (Howard and Bartram 2003; Stelmach and Clasen 2015; Thomas et al. 2016).

First Nations account for 10% of the population in Manitoba, Canada, and more than one-half of First Nations in Manitoba live in homes on First Nations reserves. Although more than two-thirds of the households on First Nations reserves across Canada are connected to a water treatment plant (WTP) by pipes, the Province of Manitoba has a much poorer performance where only ~50% of homes on First Nations reserves have piped water (Neegan Burnside 2011a, 2011b). A large (31%) portion of homes on First Nations reserves in Manitoba continue to rely on a water truck to deliver water from a WTP to a cistern at home. Health Canada (2007) recommends that no sample should have *E. coli*. However, the First Nations communities have a more flexible guideline for total coliforms in which no more than 10% of the samples or no consecutive samples collected from the same location should have total coliforms. Also, if the samples tested are < 10, no samples should be positive for total coliforms (INAC 2010). In a fly-in First Nations reserve in Manitoba, Canada, the system of water distribution influenced the frequency and levels of total coliforms detected in

tap water, with tap water from households equipped with cisterns being of a significantly poorer microbial quality than water from homes that received piped water (Farenhorst et al. 2017). A study conducted in a First Nations community in Saskatchewan also showed that the detection of total coliforms was higher in cistern water than in piped water (Bradford et al. 2018).

There are various pathways by which piped and cistern distribution systems can become contaminated with coliform bacteria. For example, the intrusion of microorganisms into broken pipes or cracked cisterns can occur through seepage of contaminated water and particulate matter (Sadiq et al. 2004; Artiola et al. 2012). Biofilms can form on the inner walls of pipes or cisterns to become a source of microorganisms (Farenhorst et al. 2017; Simões and Simões 2013), including coliform bacteria, and the type of pipe material has been shown to influence the rate at which biofilms form on cistern walls (Zhang et al. 2017).

A free residual chlorine concentration of at least 0.2 mg L^{-1} is generally considered the minimum requirement for preventing coliform bacterial contamination in piped water distribution systems (Lechevallier et al. 1996). The end of the pipe distribution system is typically lower in free residual chlorine, hence it is a location where coliform bacteria are most likely to occur (Lechevallier et al. 1996; Simões and Simões 2013). Chlorine can react with biofilms on pipe and cistern walls (Simões and Simões 2013) and pipe and cistern walls free of biofilms (Rossman et al. 1994). Due to its non-selective oxidizing nature, chlorine can react with both inorganic and organic compounds in the water, reducing disinfection efficiency in piped and cistern distribution systems (Brown et al. 2011). Additionally, since water sits in belowground or aboveground cisterns for several days after the water treatment process, chlorine decay and volatilization can reduce residual chlorine concentration in cistern water and its effectiveness to inhibit the growth of coliform bacteria (Graham and Vanderslice 2007).

Baird et al. (2013) emphasized the need for more research to understand better risks associated with cistern use in Canada, including how to manage risks. The types of cisterns commonly used on First Nations reserves in Canada include belowground concrete or fibreglass cisterns or aboveground polyethylene cisterns. Unlike direct piped water, water quality in different types of cisterns has not been adequately characterized.

This study focused on three First Nations reserves in Manitoba, each having a different predominant type of cistern system typically found on First Nations reserves in Manitoba: belowground concrete cistern, belowground fibreglass cisterns, or aboveground polyethylene cisterns stored in insulated shelters. The primary objective of this research was to determine how the use of cisterns affects microbial drinking water safety relative to drinking water piped directly to homes from the WTP.

2.3. Materials and Methods

2.3.1. Description of the water distribution systems

Three communities from the Tribal Councils of Swampy Cree (Community A), West Region (Community B) and Island Lake (Community C) were included in the study (Table 2.1). The key difference among the communities was that Communities B and C were equipped with belowground concrete and fibreglass cisterns, respectively, while Community A was equipped with aboveground polyethylene cisterns stored in insulated shelters. All three communities had conventional water treatment systems, which used a combination of coagulation, flocculation, sedimentation, filtration, and disinfection using chlorine (sodium hypochlorite solution). Additionally, Community B was equipped with a reverse osmosis system and a dissolved air floatation system for clarification (sedimentation), while Community C was equipped with an

ultraviolet disinfection system. In each community, tap water samples from homes with piped water and cisterns were periodically tested for the presence of total coliforms, with the number of homes sampled varying among communities and water distribution systems and across time. Communities B and C also tested for the presence of *E. coli* in these samples.

Table 2.1. Water treatment and distribution system characteristics in the three First Nations communities[a].

	Community A	**Community B**	**Community C**
Source water	River	Lake	Lake
Water treatment class	Level II	Level III	Level III
Distribution class	Level I	Level I	Level II
Population served	1137	1569	3993
Homes piped	170	60	151
Homes trucked (cistern storage)	34	151	69
No running water	0	0	300 (est.)
Distance from Winnipeg	600 km (Road)	410 km (Road)	1 h 20 min (Fly-in + boat ride)

[a] Data obtained from literature (Neegan Burnside 2011a) and discussions with community members.

2.3.2. Sample collection and analysis

During each sampling event, three samples each were collected from the WTP, the source water supplying the WTP, the water delivery truck, as well as the tap water from the kitchen sink of approximately ten homes with piped water and ten homes with cisterns. The water samples were collected at three different times in Communities A (October 2014, February 2015 and October 2015) and B (March 2015, June 2015 and October 2015) to investigate the temporal variation in water quality. Due to its remote location (accessible only by fly-in and motorboat ride), water samples were collected only once from Community C. At the time of sampling, free and total residual chlorine concentrations were measured onsite using the Hach DPD chlorine reagent and a Hach Chlorine Pocket Colorimeter™ II (VWR International, Mississauga, ON, Canada) following

the USEPA DPD Methods 8021 and 8167, respectively (Hach 2013). Dissolved oxygen concentration and pH were measured using a potable pH/DO meter (Extech DO700, Waltham, MA, USA) and turbidity was determined with an AQ4500 turbidity meter (Thermo Scientific™, Beverly, MA, USA). Samples for determination of dissolved organic carbon (DOC), total coliforms and *E. coli* were transported in coolers to the University of Manitoba, Winnipeg, on the same day of collection when possible or stored overnight in a refrigerator (4℃) before transportation the following morning.

For DOC, samples were collected into 250 mL polyethylene bottles. DOC in each sample was determined using APHA Method 5310C with a Fusion Total Organic Carbon (TOC) Analyzer (Teledyne Tekmar Inc., Mason, OH, USA) (APHA 2012). For total coliforms and *E. coli*, samples were collected and processed according to APHA Methods 9060A (sample bottle pre-treatment) and 9060B (preservation and storage) (APHA 2012). Within 30 h of sampling, total coliforms and *E. coli* in the water samples were enumerated using APHA Method 9222 (standard membrane filter procedure) (APHA 2012). The enumeration and culturing were done in duplicate. Field blanks and laboratory blanks with sterilized water were also included. A 100-mL subsample of each water sample was filtered through a 0.45 μm filter (VWR International Co., ON, Canada) to trap total coliforms, including *E. coli*. Filters were then placed on agar plates containing Brilliance *E. coli*/coliform medium (Thermo Fisher Scientific Inc., ON, Canada), which can distinguish *E. coli* and coliforms from other bacteria based on β-D-glucuronidase production by *E. coli* (resulting in purple *E. coli* colonies) and galactosidase production by a majority of coliforms (resulting in pink coliforms colonies) (Wohlsen 2011). Following incubation at 37°C for 24 h, purple and pink colonies were counted to quantify the colony-forming units (CFU) of *E. coli* and total coliforms, respectively. While Brilliance agar is commonly used for the detection of coliforms and *E. coli*

(Farenhorst et al. 2017; Fernando et al. 2016; Winterbourn et al. 2016), it is possible that this medium does not properly identify hemorrhagic *E. coli* strains because they are not efficient producers of β-D-glucuronidase (Fricker et al. 2010).

2.3.3. Monthly microbial data from the communities

In each community, tap water samples from homes with piped water and cisterns were periodically tested by the community water quality monitor for the presence of total coliforms using the Colilert method. However, the number of homes sampled varied among communities, between water distribution systems and across sampling times. From Community A, 307 piped and 85 cistern water samples, from Community B, 371 piped and 351 cistern water samples and from Community C, 388 piped water samples and 189 cistern water samples were collected and analyzed by the community water quality monitors. Communities B and C also tested for the presence of *E. coli* in the samples. Monthly data were obtained directly from Communities A (April 2014 to March 2015) and B (January to December 2014), while data for Community C (January to December 2014) were obtained by the community from Health Canada.

2.3.4. Yearly microbial data from First Nations communities – 2014 and 2018

Data were obtained from Health Canada under Canada's Access to Information Act (R.S.C., 1985, c. A-1). The data included the Colilert results for samples taken from drinking water supply systems across 62 First Nations reserves in Manitoba. These samples showed either a satisfactory (absence of *E. coli* or coliforms) or an unsatisfactory (presence of *E. coli* or coliforms) record. The data for 2014 were obtained because this was around the time that the experimental samples were taken, and the 2018 data were obtained to specifically evaluate the changes in the percentage of satisfactory samples compared to 2014. Using a portion of these data, we contrasted the results for the category "WTP/piped water" versus "cistern water", giving a total of 27,022 (in 2014) and

22,238 (in 2018) water samples included in the statistical analysis (see below).

2.3.5. Statistical analysis

Analysis of variance (ANOVA) was performed on water quality data (*E. coli*, total coliform, free and total residual chlorine concentrations, pH, turbidity and DOC concentration) using PROC GLIMMIX in SAS version 9.4 (SAS Institute 2013) with distribution system and sampling time as fixed effects and homes within a community as replicates. Statistical analyses were conducted separately for each community because cistern configurations (i.e., concrete vs. fibreglass vs. polyethylene) and setups (underground vs. aboveground) were not consistent among communities. Statistical power analysis based on our data indicated that a sample size of ten was large enough to provide the statistical power of at least 0.8, which is required to detect real treatment differences if they exist. Data for total coliforms were modelled as negative binomially distributed (DIST = NEGB in PROC GLIMMIX). Free and total residual chlorine, pH, turbidity, and DOC were modelled as normally distributed. One-sample t-tests were conducted using long-term means of WTP water DOC and turbidity to determine the effect of water treatment on turbidity and DOC. For monthly microbial data received from communities, ANOVA was conducted to determine the effect of the distribution system on the percentage detection of total coliform or *E. coli* over the period using the distribution system as a fixed effect (DIST = beta in PROC GLIMMIX). For each of the three Tribal Councils, PROC GLIMMIX was conducted with the beta distribution to investigate any differences in the percentage of total coliforms between 2014 and 2018 and whether the differences were the same for "WTP/piped water" versus "cistern water". For all statistical analyses, the Tukey multiple comparison procedure was used to compare the least square means at $\alpha = 0.05$.

2.4. Results and discussion

In Community A, the water treatment process significantly decreased turbidity by 88% relative to source water. However, DOC only numerically decreased by 3.6% (the granular carbon filtration system used might have removed the DOC in source water while introducing new DOC into the treated water). In Community B, relative to the source water, the water treatment process significantly decreased turbidity by 81% and DOC by 13%. In Community C (no statistical analysis performed because only one sampling was conducted), the water treatment process numerically decreased turbidity by 60% but increased DOC by 5% relative to the source water.

As a reference, data from six provinces and territories in Canada indicate that the concentration of free residual chlorine in water leaving the WTP typically ranges from 0.04 to 2.0 mg L^{-1} (Health Canada 2009b). This condition was met in all three communities and at all sampling times, except for samples collected from Community A in February, which had relatively high DOC levels in source water (17.6 mg L^{-1}) and WTP water (Table 2.2), resulting in lower chlorine concentration (0.25 mg L^{-1}). In Communities A and C, the treated water in the WTP and trucks always tested negative for total coliforms and *E. coli*. In Community B, WTP and water truck samples often showed positive detections, particularly in May when the treated water at the WTP exceeded the 1.0 NTU limit for turbidity (Health Canada 2020a) and when truck water had free residual chlorine concentrations < 0.2 mg L^{-1} (Table 2.2). No other treated water samples showed high turbidity, except that both WTP and truck samples in Community A showed turbidity levels > 1.0 NTU in October

Table 2.2. Chemical, microbial and physical properties of treated water in each community. Numbers refer to mean (standard deviation).

Community	Sampling time	Location	Free residual chlorine	Total residual chlorine	Total coliforms, *E. coli* (range)	Turbidity	DOC	pH
			mg L^{-1}	mg L^{-1}	CFU 100 mL^{-1}	NTU	mg L^{-1}	
A	1 (10/14)	WTP[a]	1.25 ± 0.02	1.90 ± 0.01	0, 0	2.00 ± 1.73	10.6 ± 0.15	7.96 ± 0.11
		Truck[a, b]	1.21	1.85	0, 0	1.33 ± 0.58	9.73 ± 0.15	7.98 ± 0.08
	2 (02/14)	WTP	0.25 ± 0.00	1.00 ± 0.06	0, 0	0.69 ± 0.04	18.2 ± 0.30	7.99 ± 0.04
		Truck	0.31 ± 0.01	0.96 ± 0.01	0, 0	0.60 ± 0.05	18.5 ± 0.06	7.84 ± 0.01
	3 (10/15)	WTP	1.60 ± 0.28	4.30 ± 1.13	0, 0	0.32 ± 0.07	9.70 ± 0.62	7.74 ± 0.00
		Truck	1.55 ± 0.07	3.20 ± 0.14	0, 0	0.35 ± 0.04	9.40 ± 0.26	7.88 ± 0.02
B	1 (03/15)	WTP	0.95 ± 0.08	1.69 ± 0.02	1-5, 1-5	0.21 ± 0.10		7.93 ± 0.15
		Truck	1.15	1.70	0, 0	0.12		7.95 ± 0.07
	2 (05/15)	WTP	0.93 ± 0.00	1.45 ± 0.01	1-13, 1-2	1.71 ± 0.03	10.5 ± 0.71	8.24 ± 0.01
		Truck	0.16	0.37	42-72, 9-12	2.38	10.6 ± 0.49	8.13 ± 0.08
	3 (10/15)	WTP	1.14 ± 0.03	1.40 ± 0.28	0-1, 0-1	0.21 ± 0.00	7.67 ± 0.72	7.71 ± 0.10
		Truck	0.90	1.36	0, 0	0.14	8.67 ± 0.47	7.97 ± 0.01
C	1 (06/15)	WTP	0.81 ± 0.11	1.02 ± 0.10	0, 0	0.46 ± 0.01	9.73 ± 0.84	7.19 ± 0.02
		Truck	1.33 ± 0.16	1.50 ± 0.03	0, 0	0.49 ± 0.01	9.60 ± 0.2	7.36 ± 0.20

[a]Water treatment plant (WTP) and truck water were collected as three subsamples for each sampling date.

[b]Only one truck water sample was analyzed for free and total residual chlorine and turbidity (measurements taken at the time of sampling) due to technical difficulties.

2014 (Table 2.2). In all communities, treated water was slightly to moderately alkaline (Table 2.2) and within the acceptable range of 7 to 10.5 for treated water at the WTP (Health Canada 2020a).

In all three communities, free and total residual chlorine concentrations were significantly lower in tap water samples from cisterns than those from piped homes (Table 2.3). Across the three communities and all sampling times, 55% of cistern and 20% of piped water samples tested positive for total coliforms, while 48% of cistern and 14% of piped water samples tested positive for *E. coli*. Eighty-five percent of all cistern samples had free residual chlorine concentrations < 0.2 mg L^{-1} and about 55% were positive for total coliforms (Table 2.4). Thus, 91% of cistern samples that tested positive for total coliforms had free residual chlorine concentrations < 0.2 mg L^{-1}. Forty-five percent of all piped samples had free residual chlorine concentrations < 0.2 mg L^{-1} and 37% of these piped samples were positive for total coliforms (Table 2.4). Thus, all piped samples that tested positive for total coliforms had free residual chlorine concentrations < 0.2 mg L^{-1}. Of the 15% of cistern samples with a free residual chlorine concentration ≥ 0.2 mg L^{-1}, 33% were positive for total coliforms, whereas only one piped water sample (3%) of the 65% piped samples with a free residual chlorine concentration ≥ 0.2 mg L^{-1} tested positive for total coliforms (Table 2.4). These results indicate that piped water distribution systems present a lower risk of bacterial contamination than cistern distribution systems, even if the free chlorine residual concentration in the system is low. The greater detection of total coliforms in cisterns versus piped water might result from biofilms on cistern walls. When a biofilm is present under low residual chlorine conditions (biofilm also reduces chlorine residue), total coliform regrowth could be substantial (Simões and Simões 2013). Cisterns may contain particulate matter that enters the cistern during filling or originates from the WTP during filling of the truck. Particulate matter can accumulate at the bottom of the cisterns or on wall surfaces (in biofilms), causing changes in water quality over

time, including reacting with chlorine and reducing its disinfection efficiency (Abokifa et al. 2016; Simões and Simões 2013).

Table 2.3. Least square means of free and total residual chlorine concentrations in each community as affected by water distribution system and sampling time.

		Community A		Community B		Community C	
Effect		**Free residual chlorine**	**Total residual chlorine**	**Free residual chlorine**	**Total residual chlorine**	**Free residual chlorine**	**Total residual chlorine**
		mg L^{-1}					
Distribution system	Piped	0.39a [a]	1.22a	0.54a	1.00a	0.18a	0.28a
	Cistern	0.13b	0.66b	0.08b	0.43b	0.02b	0.06b
Sampling time [b]	1	0.38a	0.89b	0.48a	0.70a		
	2	0.06b	0.53c	0.25a	0.92a		
	3	0.34a	1.39a	0.19a	0.52a		
		p-value					
Distribution system		0.0001	<0.0001	0.003	0.002	0.002	0.0001
Sampling time		0.0002	<0.0001	0.26	0.19	0.19	
Distribution system × Sampling time		0.07	0.48	0.54	0.47	0.47	

[a]Means within the same column and effect followed by the same letter are not significantly different according to the Tukey multiple comparison procedure ($p < 0.05$).
[b]In Community A, sampling time 1 = Oct 2014; 2 = Feb 2015; 3 = Oct 2015; In Community B sampling time 1 = Mar 2015; 2 = May 2015; 3 = Oct 2015; Community C only had one sampling time, June 2015.

Community A: From April 2014 to March 2015, Community A screened a total of 392 water samples for total coliforms using the Colilert method (Figure 2.1a), and total coliforms were detected in piped water samples in July, August, and November 2014, and in March 2015. Cistern water samples were periodically collected in 5 out of the 12 months and total coliforms were detected in September and October 2014. A total of 3% of the piped water samples and 5% of the cistern samples from Community A tested positive for total coliforms, within the range of data reported for piped and cistern water by the Swampy Cree Tribal Council (Figure 2.3), which

coordinates eight band councils, including Community A.

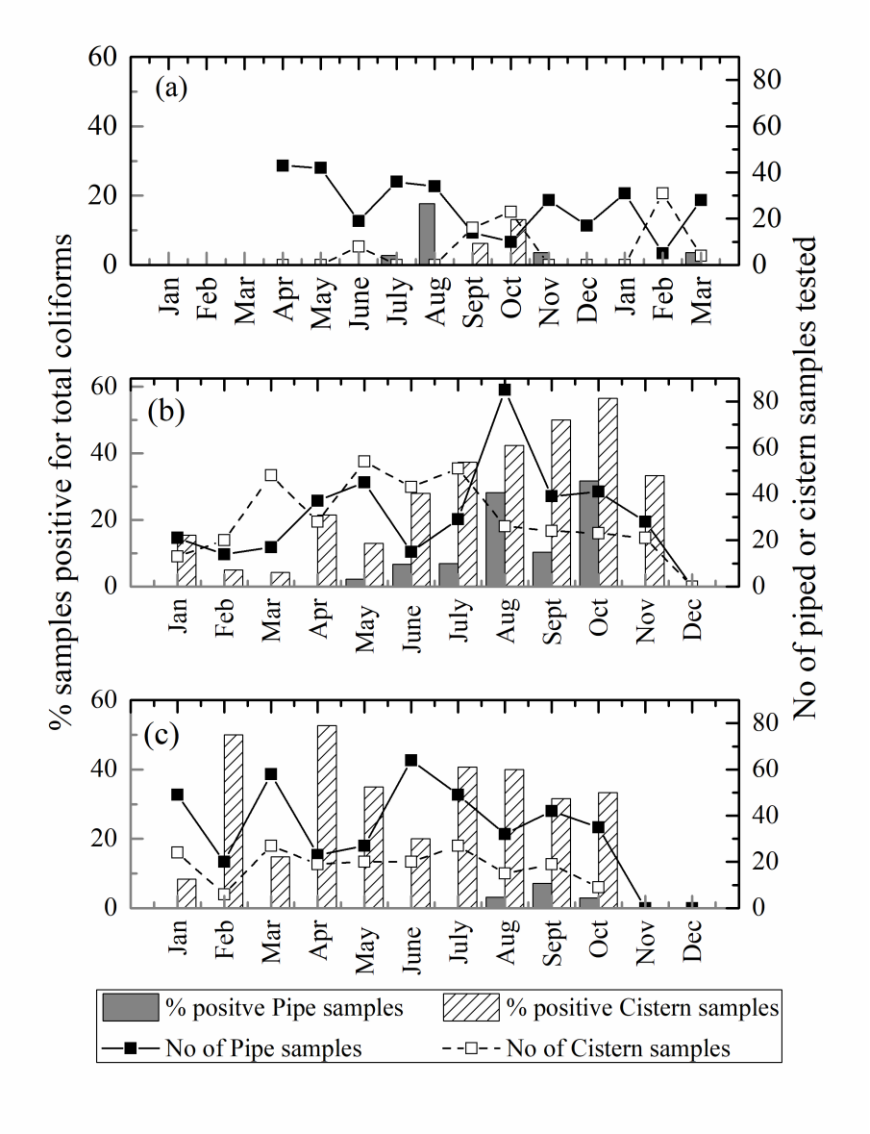

Figure 2.1. Frequency of detection of total coliforms in; (a) Community A (n = 722), (b) Community B (n = 380), and (c) Community C (n = 577) in drinking water samples collected in 2014.

Table 2.4. Range of free residual chlorine concentration, frequency of coliform detection, proportion of low chlorine (free residual chlorine below 0.2 mg L^{-1}) and adequate chlorine (above 0.2 mg L^{-1}) and percentage detection of total coliforms within the given free residual chlorine.

Community	Sampling time	Distribution system	Free residual chlorine range (mg L^{-1})	Frequency coliforms (%)	Low chlorine		Adequate chlorine	
					Frequency low chlorine (%)	Frequency coliforms (%)	Frequency high chlorine (%)	Frequency coliforms (%)
All	Overall	Piped	BD[a] – 3.4	20	45	37	65	3
		Cistern	BD – 0.80	55	85	55	15	33
A	Overall	Piped	BD – 1.20	0	38	0	62	0
		Cistern	BD – 0.80	30	75	33	25	33
	1 (10/14)	Piped	0.03 – 0.92	0	11	0	89	0
		Cistern	0.05 – 0.53	75	63	80	38	67
	2 (02/15)	Piped	0.04 – 0.27	0	89	0	11	0
		Cistern	BD – 0.05	25	100	25	0	0
	3 (10/15)	Piped	BD – 1.20	0	0	0	100	0
		Cistern	BD – 0.80	0	63	0	38	0
B	Overall	Piped	BD – 3.4	39	48	73	52	8
		Cistern	BD – 0.40	73	88	78	12	33
	1 (03/15)	Piped	0.04 – 1.28	50	38	100	63	20
		Cistern	BD – 0.07	90	100	100	0	0
	2 (05/15)	Piped	BD – 3.4	57	57	100	43	0
		Cistern	0.02 – 0.38	78	78	86	22	50
	3 (10/15)	Piped	BD – 0.82	12	50	25	50	0
		Cistern	BD – 0.31	43	86	50	14	0
C	Overall	Piped	0.06 – 0.46	30	70	43	30	0
		Cistern	BD – 0.09	62	100	63	0	0

[a] BD = detection limit of the Hach colorimeter, 0.02 mg L^{-1}

The number of cistern water samples collected by Community A was too small to allow for statistical comparison of the two water distribution systems. However, based on the comparison of tap water quality between households served via pipes versus cisterns for samples taken in October 2014 and in February and October 2015 using APHA Method 9222 (standard membrane filter procedure), all piped water samples were free from total coliforms, but 30% of the total cistern samples were positive for total coliforms (Table 2.4). In cistern water, *E. coli* and total coliform counts ranged from 1 to 17 CFU 100 mL^{-1}. Thus, despite the good physical condition of aboveground cisterns in Community A and the statistically similar turbidity and DOC levels of piped and cistern water samples (Table 2.5), households with cisterns are more likely to encounter microbial contaminants than households with piped water, perhaps because of the presence of biofilms in cisterns. As presented in Table 2.4, across all sampling times, approximately 75% of cistern water samples had free residual chlorine concentrations < 0.2 mg L^{-1} and one-third of these samples were positive for total coliforms. Of the remaining 25% of the cistern samples that had free residual chlorine concentrations > 0.2 mg L^{-1}, about one-third tested positive for total coliforms, indicating that aboveground polyethylene cisterns are often contaminated regardless of the level of free residual chlorine concentrations. In contrast, none of the 38% of piped samples that had free residual chlorine concentrations ≥ 0.2 mg L^{-1} or piped samples that had free residual chlorine concentrations < 0.2 mg L^{-1} (62%) were positive for total coliforms (Table 2.4).

Community B: From January to December 2014, Community B screened a total of 722 water samples for total coliforms using the Colilert method (Figure 2.1b) and the percentage detection of total coliforms was significantly greater in cistern water than in piped water. Total coliforms were detected in 12% of the piped water samples and 26% of the cistern water samples. The percentages of piped (12%) and cistern (26%) samples that tested positive for total coliforms were substantially

greater than reported for an average of eight band councils that form the West Region Tribal Council, including Community B (Figure 2.3). Total coliforms in cistern water samples were detected throughout the year, with higher detection frequencies in April through November. Total coliforms in piped water were also detected in May through November 2014 but not during winter months.

E. coli was detected in cistern water samples between April and September and to a much lesser extent in piped water samples in August and September (Figure 2.2). Consistent with observations obtained using the Colilert method, the APHA Method 9222 (standard membrane filter procedure) results showed that across all sampling times (March 2015, June 2015 and October 2015), 73% of cistern water samples but only 39% of piped water samples tested positive for total coliforms (Table 2.4). *E. coli* counts ranged from 1 to 110 CFU 100 mL^{-1} in cistern water, but from 1 to 11 CFU 100 mL^{-1} in piped water. *E. coli* and total coliform counts (Table 2.6), turbidity and DOC were significantly greater in cistern samples than piped samples (Table 2.5). Sampling time had no significant effect on free residual chlorine and total chlorine concentrations in Community B (Table 2.3). However, *E. coli* and total coliform counts, turbidity and DOC were significantly greater in May than in March or October (Table 2.6). Flooding from snowmelt and rainfall in April and May might have enhanced the seepage of bacteria, particulate matter and DOC into cracked belowground concrete cisterns. The leaching of bacteria and DOC from the soil surface to groundwater has been shown to increase during periods of high rainfall (Artiola 2012; Mahler et al. 2000).

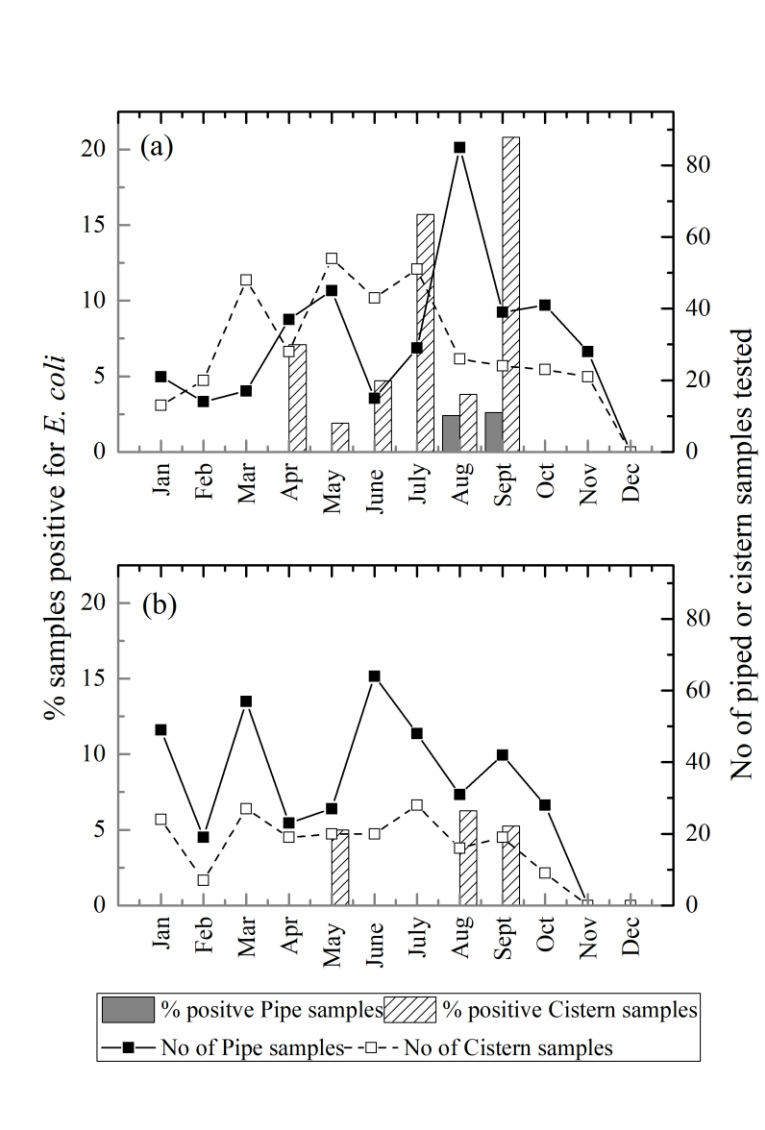

Figure 2.2. Frequency of detection of *E. coli* in (a) Community B (n = 380) and (b) Community C (n = 577) in drinking water samples collected in 2014.

Table 2.5. Least square means of chemical and physical properties of piped and cistern water in each community.

		Community A			Community B			Community C		
Effect		**Turbidity**	**DOC**	**pH**	**Turbidity**	**DOC**	**pH**	**Turbidity**	**DOC**	**pH**
		NTU	mg L^{-1}		NTU	mg L^{-1}		NTU	mg L^{-1}	
Distribution	Piped	0.66	12.2	7.82b [a]	0.56b	8.13b	7.87	0.56	9.29b	7.19b
system	Cistern	0.92	12.3	7.96a	0.89a	8.86a	7.98	0.62	10.32a	7.71a
Sampling time[b]	1	1.31a	9.88	7.90	0.58b		7.88			
	2	0.63b	17.14	7.96	1.28a	10.51a	8.07			
	3	0.43b	9.72	7.80	0.32b	6.48b	7.83			
Distribution	Piped - 1	1.00	10.02b	7.84ab	0.47		7.75bc			
system ×	Piped - 2	0.68	17.58a	7.92ab	0.98	10.24	8.14a			
Sampling time	Piped - 3	0.29	9.16b	7.69b	0.23	6.02	7.72c			
	Cistern - 1	1.62	9.75b	7.95ab	0.70		8.02a			
	Cistern - 2	0.57	16.7a	7.99a	1.57	10.78	7.99ab			
	Cistern - 3	0.56	10.27b	7.91ab	0.40	6.93	7.94abc			
						p-value				
Distribution system		0.11	0.96	0.01	0.045	0.04	0.03	0.37	0.002	<0.0001
Sampling time		<0.0001	<0.0001	0.07	<0.0001	<0.0001	0.001			
Distribution system x sampling time		0.17	0.01	0.49	0.51	0.58	0.007			

[a]Means in the same column followed by the same letter are not significantly different according to the Tukey-Kramer test ($p < 0.05$). Mean comparison letters are applied to the main effects only in the absence of a significant interaction.

[b]In Community A, sampling time 1 = Oct 2014; 2 = Feb 2015; 3 = Oct 2015; In Community B sampling time 1 = Mar 2015; 2 = May 2015; 3 = Oct 2015; In Community C only one sampling time June 2015.

Table 2.6. Least square means of *E. coli* and total coliform counts in Community B as affected by water distribution system and sampling time.

Effect		*E. coli*	Total coliforms
		CFU 100 mL^{-1}	CFU 100 mL^{-1}
Distribution system	Piped	0.85b [a]	1.80
	Cistern	15.5a	81.2
Sampling time	03/15	3.28b	9.40
	05/15	10.1a	38.4
	10/15	1.46b	5.00
Distribution system ×Sampling time	Piped - 03/15	0.12	0.12d
	Piped - 05/15	5.71	27.1b
	Piped - 10/15	0.87	1.84c
	Cistern - 03/15	17.0	703a
	Cistern - 05/15	17.8	54.4b
	Cistern - 10/15	12.3	14.3b
		p-value	
Distribution system		<0.0001	0.02
Sampling time		0.04	<0.0001
Distribution system×Sampling time		0.07	0.0002

[a]Means in the same column followed by the same letter are not significantly different according to the Tukey-Kramer test ($P < 0.05$). Mean comparison letters are applied to the main effects only in the absence of a significant interaction.

As given in Table 2.4, across all sampling times, approximately 90% of cistern water samples had free residual chlorine concentrations < 0.2 mg L^{-1} and 78% of these samples were positive for total coliforms. Of the remaining 10% of the cistern samples with free residual chlorine concentrations > 0.2 mg L^{-1}, about one-third tested positive for total coliforms, indicating that belowground cement cisterns are often contaminated regardless of the level of free residual chlorine concentrations. In contrast, out of the 52% of piped samples with free residual chlorine concentrations ≥ of 0.2 mg L^{-1}, only 8% of the samples were positive for total coliforms. Forty-eight percent of piped samples had free residual chlorine concentrations < 0.2 mg L^{-1} and 73% of these samples were positive for total coliforms (Table 2.4). Hence, for piped homes, the threshold level of 0.2 mg L^{-1} of free residual chlorine in the tap water is a reasonable indicator of whether or not drinking water is contaminated with total coliforms.

Community C: From January to December 2014, Community C screened a total of 577 water samples for total coliforms using the Colilert method (Figure 2.1c) and the percentage detection of total coliform was significantly greater in cistern water than in piped water. In total, 1% of the piped samples and 31% of the cistern samples collected in 2014 were positive for total coliforms, comparable to the averages reported for these water distribution systems across four band councils that form the Island Lake Tribal Council, including Community C (Figure 2.3). Total coliforms were consistently detected in cistern samples throughout the year but only in September and October in piped samples (Figure 2.1c). *E. coli* was only detected in cistern water samples both in May 2014 (notable snowmelt and flooding) and in August and September 2014 (notable heavy rainfall events) (Figure 2.2). Using APHA Method 9222 (standard membrane filter procedure), *E. coli* and total coliform counts in the belowground cisterns ranged from 1 to 1540 CFU 100 mL^{-1} in the one sampling event in Community C, whereas only up to 4 CFU 100 mL^{-1} of *E. coli* and total coliforms were detected in piped water samples. Also, cisterns had significantly higher DOC levels than piped water. All cistern samples collected in this community had free residual chlorine concentrations < 0.2 mg L^{-1} and 63% of the samples tested positive for total coliforms. In Community C, 70% of piped water samples also had free residual chlorine concentrations < 0.2 mg L^{-1} and 43% of these samples were positive for total coliforms (Table 2.4). None of the piped water samples that had free residual chlorine ≥ 0.2 mg L^{-1} were positive for total coliforms, again confirming that the threshold level of 0.2 mg L^{-1} is a reasonable indicator of whether total coliforms are present in piped tap water.

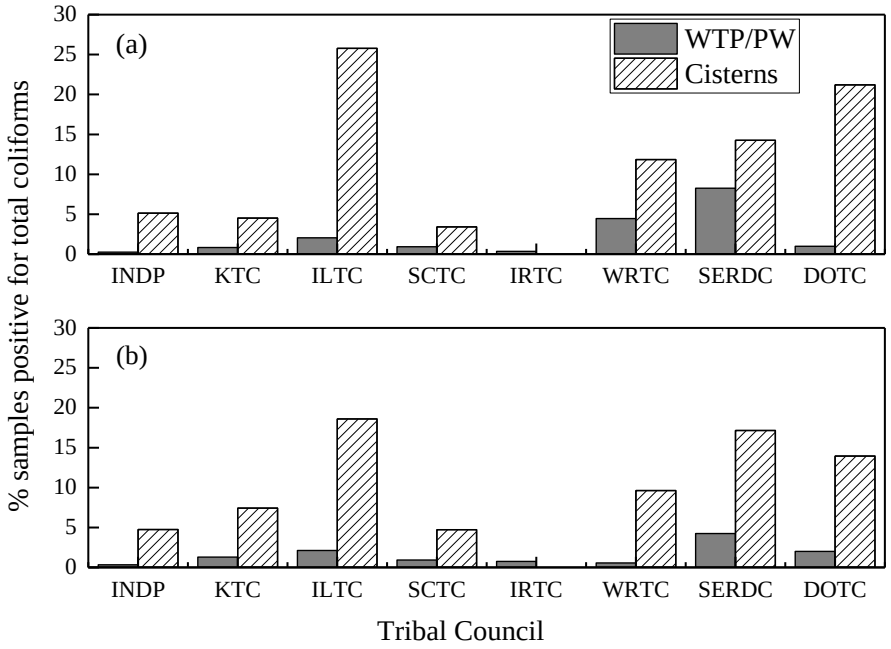

Figure 2.3. Percentage of water treatment plant or piped water (WTP/PW) and cistern water samples containing bacteria in: (a) 2014 and (b) 2018 in each First Nation Tribal Council in MB (n = 27,022) (INDP = No Tribal Council affiliated, KTC = Keewatin Tribal Council, ILTC = Island Lake Tribal Council, SCTC = Swampy Cree Tribal Council, IRTC = Interlake Reserves Tribal Council, WRTC = West Region Tribal Council, SERDC = Southeast Resource Development Council, DOTC = Dokota Ojibway Tribal Council).

Progress of bacterial water quality: Many First Nations communities in Canada continue to lack access to safe drinking water (Patrick 2011). For the First Nations communities studied, our results indicate a significant loss of chlorine during water distribution from the WTP to households. The low free residual chlorine concentrations in cistern water particularly contribute to positive detections of total coliforms in communities. Data obtained under the Access to Information Act (Figure 2.3) also indicate that the frequency of detecting coliform bacteria is substantially greater for cisterns than piped homes and in every Manitoba Tribal Council. There were no statistical differences in the percent of satisfactory samples (no detects) between 2014 and 2018 (Table 2.7),

suggesting that there is no clear indication of improved water quality in any of the three Tribal Councils, despite recent efforts by the Federal Government of Canada to strengthen on-reserve water infrastructure and water monitoring (Department of Finance Canada 2016). The exception to this was that for the Island Lake Tribal Council, there was a 7% increase in satisfactory cistern water samples in 2018 compared to 2014. However, this was mainly because one community in the Island Lake Tribal Council had only ten cistern samples tested in 2018 while they tested 41 samples in 2014, of which 30 had unsatisfactory results. By excluding this community from the Island Lake Tribal Council analysis, there were no significant differences in cistern water quality between 2014 and 2018.

Table 2.7. Percentage of water treatment plant or piped water (WTP/PW) and cistern water samples free from coliform bacteria in 2014 and 2018 in three First Nation tribal councils in Manitoba.

Tribal Council	WTP/PW (%)		Cisterns (%)	
	2014	2018	2014	2018
SCTC	99.1	99.1	96.6	95.3
WRTC	95.5	99.4	88.2	90.4
ILTC	97.9	97.88	74.2a	81.4b

[a] Means in the same row within one distribution system in a Tribal Council followed by the same letter are not significantly different according to the Tukey-Kramer test ($p < 0.05$).

Implications: Cistern water was always more likely to be contaminated with total coliforms than piped water, as previously reported for Community C (Farenhorst et al. 2017), and by including two other communities in our studies, we demonstrated here that this is particularly the case in

First Nations communities with belowground cisterns (Communities B and C). The levels of total coliforms and *E. coli* that were detected in the tap water of households with belowground cisterns are concerning as some strains of coliform bacteria can lead to respiratory, gastrointestinal and skin infection and make it more difficult for communities to protect themselves during epidemics and pandemics (Soller et al. 2010; Thomas et al. 2016). Given our findings, we recommend that households with belowground cisterns boil and hold water at a rolling boil for 1 min to inactivate bacteria as per Health Canada guidelines (Health Canada 2015). Notably, the monthly data obtained for the belowground cement cisterns in Community B indicated that the contamination was persistent throughout the year, suggesting that these cisterns need to be cleaned regularly or replaced with better options, i.e., connecting households to the WTP with pipes or using aboveground cisterns.

Collectively, the drinking water quality data collected in this study by various methods suggest that the current water distribution method and storage in cisterns is not working for all homes on First Nations reserves. The data also suggest that, within the same First Nations reserves, families living in homes served by cisterns are at a greater risk of contracting infections caused by bacterial pathogens than families living in homes with piped water. The addition of chlorine to the cisterns could ostensibly help ensure the safety of the water. However, given the high DOC in cistern water in Communities B and C (DOC in belowground cisterns in Community B and C were 8.86 and 10.32 mg L^{-1}, respectively), the added chlorine might be rapidly deactivated, including via reaction with DOC to produce trihalomethanes (Brown et al. 2011). Some of these trihalomethanes are potential carcinogens to animals and humans and cause reproductive abnormalities by acting as teratogens and to a lesser extent as mutagens (Mohamadshafiee and Taghavi 2012; Villanueva et al. 2015). The presence of biofilms on inner cistern walls may also increase trihalomethane

formation by increasing biomass-derived organic matter to react with available chlorine (Abokifa et al. 2016; Montoya-Pachongo et al. 2018).

Moreover, a study on the perceptions of drinking water quality showed that only 50% of households on First Nations reserves support chlorine addition as a disinfectant in water treatment (EKOS Research Associates Inc. 2011). Fifty-three percent of households opposed to the addition of chlorine in water treatment had concerns about the negative effects of chlorine on human health, whereas 17% did not like the taste. Therefore, the addition of more chlorine to the cistern will likely exacerbate these concerns. Additional in-house water treatment techniques such as filters and UV-light would help improve the drinking water quality; however, these methods would not be economical given the low household income of First Nations communities in Manitoba and would not be able to provide adequate water given the large number of residents in a household. However, additional filtration steps such as membrane filtration and advanced oxidation using ozone and ultraviolet radiation at the WTP would decrease microbial contaminants and organic and inorganic compounds that are nutrient sources for microorganisms, thereby reducing microbial regrowth in the distribution systems (Song et al. 2019; Zhou and Smith 2001). Also, it is important to identify the contaminant sources of belowground cisterns to minimize or eliminate drinking water contamination during storage.

2.5. Conclusion

Results of the study showed that cisterns are more contaminated with total coliforms compared to piped water. According to APHA Method 9222 (standard membrane filter procedure) results across all the communities and sampling times, 20% of the homes with piped water and 55% of the homes with cisterns showed positive detects for total coliforms and 14% of the piped homes

and 48% of the homes with cisterns were positive for *E. coli*. These data indicate that, within the same First Nations reserves, families living in homes with cisterns are at a greater risk of contracting water-borne infections than families living in homes with piped water. Contamination of water with *E. coli* and total coliforms was most prevalent and severe in homes whose tap water was supplied by belowground concrete cisterns, followed by belowground fibreglass cistern. In contrast, aboveground polyethylene cisterns that were being used in Community A had the potential to provide relatively safer drinking water. Therefore, as an immediate measure to ensure safe drinking water, damaged belowground cisterns should be replaced with aboveground polyethylene cisterns. The level of bacterial contamination in tap water derived from belowground concrete cisterns was significantly greater in May than in March and October, possibly because of the inflow of microbially contaminated snowmelt through cracks. Hence, cleaning all belowground concrete cisterns following snowmelt is another important practice to reduce the risk of human exposure to pathogenic bacteria.

2.6. Acknowledgments

The authors gratefully thank the Band Councils and the residents of the First Nations reserves who partnered with us in this research. This work was supported by the Natural Sciences and Engineering Research Council of Canada (NSERC) through its Collaborative Research and Training Experience Program (CREATE grant# 432009-2013) and Discovery Grant (DG) programs. The authors wish to acknowledge the NSERC Postgraduate Scholarships-Doctoral (PGS D) and the University of Manitoba Graduate Scholarship awarded to Amarawansha. In addition, we thank Ruidong Mi for the assistance in sampling and laboratory analysis.

2.7. References

Abokifa, A. A., Yang, Y. J., Lo, C. S., & Biswas, P. (2016). Investigating the role of biofilms in trihalomethane formation in water distribution systems with a multicomponent model. *Water Research*, *104*, 208–219. https://doi.org/10.1016/j.watres.2016.08.006

APHA. (2012). *Standard Methods for the Examination of Water and Wastewater*. (R. Baird, A. Eaton, & E. Rice, Eds.) (22nd ed.). American Public Health Association; American Water Works Association; Water Environment Federation.

Artiola, J., Rock, C., & Hix, G. (2012). Water storage tank disinfection, testing, and maintenance. The University of Arizona Cooperative Extension. College of Agriculture and Life Sciences, University of Arizona, Tucson, AZ, USA.

Baird, J. M., Summers, R., & Plummer, R. (2013). Cisterns and safe drinking water in Canada. *Canadian Water Resources Journal*, *38*, 121–134. https://doi.org/10.1080/07011784.2013.780790

Bradford, L., Waldner, C., McLaughlin, K., Zagozewski, R., & Bharadwaj, L. (2018). A mixed-method examination of risk factors in the truck-to-cistern drinking water system on the Beardy's and Okemasis First Nation Reserve, Saskatchewan. *Canadian Water Resources Journal*, *43*, 383–400. https://doi.org/10.1080/07011784.2018.1474139

Brown, D., Bridgeman, J., & West, J. R. (2011). Predicting chlorine decay and THM formation in water supply systems. *Reviews in Environmental Science and Bio-Technology*, *10*, 79–99. https://doi.org/10.1007/s11157-011-9229-8

Daley, K., Castleden, H., Jamieson, R., Furgal, C., & Ell, L. (2015). Water systems, sanitation, and public health risks in remote communities: Inuit resident perspectives from the

Canadian Arctic. *Social Science & Medicine*, *135*, 124–132. https://doi.org/10.1016/j.socscimed.2015.04.017

Department of Finance Canada. (2016). *A Better Future for Indigenous Peoples, Budget Plan 2016*. House of the Commons, Ottawa, Ontario.

EKOS Research Associates Inc. (2011). *Perceptions of Drinking Water Quality in First Nations Communities and General Population*. Ottawa, Ontario. https://www.ekospolitics.com/articles/015-11.pdf

Farenhorst, A., Li, R., Jahan, M., Tun, H. M., Mi, R., Amarakoon, I., et al. (2017). Bacteria in drinking water sources of a First Nation reserve in Canada. *Science of the Total Environment*, *575*, 813–819. https://doi.org/10.1016/j.scitotenv.2016.09.138

Fernando, D. M., Tun, H. M., Poole, J., Patidar, R., Li, R., Mi, R., et al. (2016). Detection of antibiotic resistance genes in source and drinking water samples from a first nations community in Canada. *Applied and Environmental Microbiology*, *82*, 4767–4775. https://doi.org/10.1128/AEM.00798-16

Fricker, C. R., Warden, P. S., & Eldred, B. J. (2010). Understanding the cause of false negative β-d-glucuronidase reactions in culture media containing fermentable carbohydrate. *Letters in Applied Microbiology*, *50*, 547–551. https://doi.org/10.1111/j.1472-765X.2010.02834.x

Graham, J. P., & Vanderslice, J. (2007). The effectiveness of large household water storage tanks for protecting the quality of drinking water. *Journal of Water and Health*, 5, 307–313. https://doi.org/10.2166/wh.2007.011b

Hach Company. (2013). Water Analysis Guide. Loveland, Colorado, USA.

Health Canada. (2009). *Guidelines for Canadian Drinking Water Quality: Guideline Technical Document- Chlorine*. Water, Air and Climate Change Bureau, Healthy Environments and

Consumer Safety Branch, Health Canada, Ottawa, Ontario. (Catalogue No. H128-1/09-588E).

Health Canada. (2015). *Guidance for Issuing and Rescinding Boil Water Advisories in Canadian Drinking Water Supplies*. Water and Air Quality Bureau, Healthy Environments and Consumer Safety Branch, Health Canada, Ottawa, Ontario (Catalogue No. H128-1/09-578-1E-PDF).

Health Canada. (2020). *Guidelines for Canadian Drinking Water Quality Summary Table*. Water and Air Quality Bureau, Healthy Environments and Consumer Safety Branch, Health Canada, Ottawa, Ontario.

Howard, G., & Bartram, J. (2003). *Domestic Water Quantity, Service, Level and Health (No. WHO/SDE/WSH/03.02). World Health Organization*. Geneva, Switzerland.

Lechevallier, M. W., Welch, N. J., & Smith, D. B. (1996). Full-scale studies of factors related to coliform regrowth in drinking water. *Applied and Environmental Microbiology*, *62*, 2201–2211. https://doi.org/10.1128/aem.00794-10

Mahler, B. J., Personne, J. C., Lods, G. F., Drogue, C., Personné, J.-C., Lods, G. F., & Drogue, C. (2000). Transport of free and particulate-associated bacteria in karst. *Journal of Hydrology*, 238, 179–193. https://doi.org/10.1016/s0022-1694(00)00324-3

Mohamadshafiee, M. R., & Taghavi, L. (2012). Health effects of trihalomethanes as chlorinated disinfection by products : A review article. *International Journal of Environmental and Ecological Engineering*, *6*, 545–551. http://www.epa.gov/iris/subst/0025.htm

Montoya-Pachongo, C., Douterelo, I., Noakes, C., Camargo-Valero, A., Sleigh, A., Escobar-Rivera, J.-C., & Torres-Lozada, P. (2018). Field assessment of bacterial communities and

total trihalomethanes: Implications for drinking water networks. *Science of the Total Environment, 616–617,* 345–354. https://doi.org/10.1016/j.scitotenv.2017.10.254

Neegan Burnside. (2011a). *National Assessment of First Nations Water and Wastewater Systems; National Roll-Up Report*: Department of Indian and Northern Affairs: Ottawa, ON, Canada.

Neegan Burnside. (2011b). *National Assessment of First Nations Water and Wastewater Systems;Manitoba Regional Roll-Up Report*: Department of Indian and Northern Affairs: Ottawa, ON, Canada.Patrick, R. J. (2011). Uneven access to safe drinking water for First Nations in Canada: Connecting health and place through source water protection. *Health & Place, 17,* 386–389. https://doi.org/10.1016/j.healthplace.2010.10.005

Rossman, L. A., Clark, R. M., & Grayman, W. M. (1994). Modeling chlorine residuals in drinking-water distribution systems. *Journal of environmental engineering, 120,* 803–820. https://doi.org/10.1061/(ASCE)0733-9372(1994)120:4(803)

Sadiq, R., Kleiner, Y., & Rajani, B. (2004). Aggregative risk analysis for water quality failure in distribution networks. *Journal of Water Supply Research and Technology-Aqua, 53,* 241–261. %3CGo

SAS Institute. 2013. SAS/STAT user's guide version 9.4. SAS Inst., Inc., Cary, NC.

Simões, L. C., & Simões, M. (2013). Biofilms in drinking water: problems and solutions. *Rsc Advances, 3,* 2520–2533. https://doi.org/10.1039/c2ra22243d

Soller, J., Embrey, M., Tuhela, L., Ichida, A., & Rosen, J. (2010). Risk-based evaluation of *Escherichia coli* monitoring data from undisinfected drinking water. *Journal of Environmental Management, 91,* 2329–2335. https://doi.org/10.1016/j.jenvman.2010.06.017

Song, K., Mohseni, M., & Taghipour, F. (2019). Mechanisms investigation on bacterial inactivation through combinations of UV wavelengths. *Water Research, 163*, 114875. https://doi.org/10.1016/j.watres.2019.114875

Stelmach, R. D., & Clasen, T. (2015). Household water quantity and health: A systematic review. *International Journal of Environmental Research and Public Health, 12*, 5954–5974. https://doi.org/10.3390/ijerph120605954

Thomas, T. K., Ritter, T., Bruden, D., Bruce, M., Byrd, K., Goldberger, R., et al. (2016). Impact of providing in-home water service on the rates of infectious diseases: Results from four communities in Western Alaska. *Journal of Water and Health, 14*, 132–141. https://doi.org/10.2166/wh.2015.110

Villanueva, C. M., Cordier, S., Font-Ribera, L., Salas, L. A., & Levallois, P. (2015). Overview of disinfection by-products and associated health effects. *Current environmental health reports, 2*, 107–115. https://doi.org/10.1007/s40572-014-0032-x

Winterbourn, J. B., Clements, K., Lowther, J. A., Malham, S. K., Mcdonald, J. E., & Jones, D. L. (2016). Use of Mytilus edulis biosentinels to investigate spatial patterns of norovirus and faecal indicator organism contamination around coastal sewage discharges. *Water Research*, 105, 241–250. https://doi.org/10.1016/j.watres.2016.09.002

Wohlsen, T. D. (2011). Comparative evaluation of chromogenic agar CM1046 and mFC agar for detection of *E. coli* and thermotolerant coliform bacteria from water samples. *Letters in Applied Microbiology, 53*, 155–160. https://doi.org/10.1111/j.1472-765X.2011.03086.x

Zhang, C., Li, C., Zheng, X., Zhao, J., He, G., & Zhang, T. (2017). Effect of pipe materials on chlorine decay, trihalomethanes formation, and bacterial communities in pilot-scale water

distribution systems. *International Journal of Environmental Science and Technology*, *14*, 85–94. https://doi.org/10.1007/s13762-016-1104-2

Zhou, H., & Smith, D. W. (2001). Advanced technologies in water and wastewater treatment. *Canadian Journal of Civil Engineering*, *28*, 49–66. https://doi.org/10.1139/S02-020

3. TRIHALOMETHANES IN DRINKING WATER FROM THREE FIRST NATION RESERVES IN MANITOBA, CANADA.

3.1. Abstract

Previous research indicates that the water distribution system used has a significant impact on the microbial quality of tap water sampled in First Nations reserves in Canada. This study collected tap water from homes in three First Nations reserves to compare the concentrations of four trihalomethanes and related water quality parameters between homes receiving piped water from a water treatment plant versus homes equipped with cisterns that are filled by a water truck. Health Canada sets the maximum acceptable concentration of total trihalomethanes (TTHMs) in treated water at 100 μgL^{-1} and of all the samples collected across time from household taps, 75% of piped samples and 70% of cistern samples had >100 $\mu g\ L^{-1}$ TTHMs. In all communities and across sampling times, trichloromethane ($CHCl_3$) was the dominant trihalomethane (42 – 96%) followed by bromodichloromethane ($CHBrCl_2$) (3 – 37%) and dibromochloromethane ($CHClBr_2$) (1 – 18%). Tribromomethane ($CHBr_3$) always contributed to less than 5% of TTHMs. Within each of the three First Nations reserves, the water distribution system had no significant effect on TTHM concentrations at the household level. Sampling month was a significant factor affecting TTHM concentration due to the differences in temporal changes in dissolved organic carbon of the source water. Results suggest that families in the studied First Nations reserves receive drinking water with high TTHM concentrations and that improvements to the water treatment plant might be the most effective way to minimize trihalomethane formation.

Key words: Total trihalomethanes, First Nations reserves, Tap water quality, Piped homes, Cistern-equipped homes, Trichloromethane ($CHCl_3$), Bromodichloromethane ($CHBrCl_2$) Dibromochloromethane ($CHClBr_2$), Tribromomethane ($CHBr_3$).

3.2. Introduction

Disinfection of source water in the water treatment process inactivates microorganisms to prevent the spread of waterborne illnesses through drinking water. The most widely used disinfection agent is chlorine introduced about a century ago in North America as a viable option in drinking water treatment. Chlorine has a high oxidizing potential accounting for its effectiveness in disinfecting (Gopal et al. 2007), but chlorine residuals' reaction with natural organic matter can also produce disinfection by-products, particularly trihalomethanes (Brown et al. 2011). Trihalomethanes are single carbon compounds with one to three halogens and include trichloromethane ($CHCl_3$), bromodichloromethane ($CHBrCl_2$) dibromochloromethane ($CHClBr_2$) and tribromomethane ($CHBr_3$). $CHCl_3$ and $CHBrCl_2$ have been identified as potential carcinogens to humans and other animals, and $CHBrCl_2$ can cause reproductive abnormalities by acting as a teratogen or possibly a mutagen (Mohamadshafiee and Taghavi 2012; Villanueva et al. 2015). The maximum acceptable concentration (MAC) of total trihalomethanes (TTHMs) in treated water varies across countries, ranging from 25 to 100 µg L^{-1} currently (Evlampidou et al. 2020). The MAC for TTHMs is derived from toxicological information in Canada and is set at 100 µg L^{-1} for treated drinking water (Health Canada 2019).

The level of trihalomethanes in treated water is influenced by the water treatment process used (e.g., filtration and disinfection techniques), as well as the type of source water used and its quality (Chhipi-Shrestha et al. 2018; Rodriguez and Serodes 2001; Rodrigues et al 2003; Wong et al.

2007). For example, Wei et al. (2010) found greater levels of TTHMs in the treated water when the source water was surface water rather than groundwater, predominantly because source water contains greater levels of natural organic matter that react with chorine during treatment. A short-term study (12 months) examined TTHMs' seasonal and special variation in municipal water distribution systems in Eastern Canada and found the highest TTHMs in the summer due to warmer temperatures and more organic matter in the source water (Scheili et al. 2015). TTHM concentrations tend to increase in the piped distribution system with increasing distance from the water treatment plant (WTP), increasing residence time, and increasing water temperature (Liang and Singer 2003; Rodriguez et al. 2003; Wei et al. 2010). In the Province of Manitoba, a 2018 survey showed that 38% of the public water systems out of 92 water systems tested exceeded 100 $\mu g\ L^{-1}$ TTHMs (Department of Conservation and Climate 2020).

The three groups of Indigenous peoples identified in the Canadian Constitution Act of 1982 are First Nations, Inuit and Metis. About one-third of the one million First Nations in Canada live on reserves, where many households lack access to safe drinking water (Dupont et al. 2014; EKOS Research Associates Inc. 2009). For example, several recent studies highlight that on First Nations reserves in the Canadian Prairies, including in Manitoba, total coliforms are frequently detected in tap water, particularly in First Nations households with cisterns (Bradford et al. 2018; Farenhorst et al. 2017; Fernando et al. 2016). In addition, TTHMs is among the most frequently exceeded parameter in treated water at WTPs on First Nations reserves (INAC 2003). In July 2019, two First Nations reserves in Eastern Canada declared a state of emergency after detecting high levels of TTHMs >100 $\mu g\ L^{-1}$ in treated water at WTP and tap water in piped homes (Abedi 2019).

In the Province of Manitoba, about one-half of households on First Nations reserves (51%) are connected to a WTP by pipes, whereas about one-third (31%) has drinking water delivered to

cisterns by tank trucks (Neegan Burnside 2011). (The remaining households rely on wells or do not have running water). To date, no study has compared the types and concentrations of trihalomethanes in the tap water of households with piped versus cistern water delivery. Biofilms have been detected on the inner walls of both pipes and cisterns, and biofilms have been shown to contribute to TTHMs formation (Brown et al. 2011; Xu et al. 2018). However, it is possible that the increased contact time during water storage in cisterns particularly facilitates the formation of trihalomethanes, especially when cistern water contains dissolved organic matter that readily reacts with chlorine (Brown et al. 2011).

This research focused on three First Nation reserves in Manitoba with relatively similar water treatment and distribution systems. Each community utilized surface water as source water for water treatment and sodium hypochlorite for disinfection. Each community had homes connected to the WTP by pipes and homes with cisterns filled by a tank truck. Community A is equipped with aboveground polyethylene cisterns stored in insulated shelters, while Communities B and C are equipped with belowground cisterns made out of concrete and fibreglass, respectively. The study's objective was to examine the effect of the water distribution system (piped vs. cistern) on the types and concentrations of trihalomethanes and related water quality parameters in the water distribution systems in these First Nations reserves.

3.3. Materials and Methods

3.3.1. Water Sample Collection and Analysis

Water samples were collected from three First Nations reserves herein referred to as Communities A (Swampy Creek Tribal Council), B (West Region Tribal Council) and C (Island Lake Tribal Council). Communities A and B are accessible by road, whereas Community C is a fly-in

community that is accessible by two flights per day during summer followed by a motorboat ride (no road access during summer) and ice roads during winter. Community A sources its water from a river whereas Communities B and C source their water from lakes. All three communities have conventional water treatment systems, including coagulation, flocculation, sedimentation, filtration, and disinfection using chlorine (sodium hypochlorite solution). Additionally, Community B is equipped with a reverse osmosis system and dissolved air floatation for clarification (sedimentation) and Community C is equipped with an ultraviolet disinfection system.

Communities A and B were sampled twice in 2015, and the fly-in Community C was sampled once in 2015. In February and October 2015 from Community A, in May and October 2015 from Community B and in June 2015 from Community C. The sampling times were decided on by the research partner in the communities. During each sampling event in a community, source water and treated water were sampled from taps inside the WTP, truck water samples were collected from the hose of the water delivery truck, and household samples were collected from kitchen taps in ten homes with piped water and ten homes with cisterns. Water samples for determination of trihalomethane concentrations and speciation were collected in 60 mL pre-cleaned amber glass bottles with polytetrafluoroethylene screw cap lids and containing 0.06 mL of 3% sodium thiosulfate solution. For pH and dissolved organic carbon (DOC) content analysis and determinations of humic and aromatic content indicators, water samples were collected in 250 mL polypropylene bottles.

Trihalomethanes were extracted and quantified following a similar method to Standard Methods 6232B (APHA 2012) in the Environmental Chemistry lab, Department of Chemistry, University of Manitoba and using an Agilent gas chromatograph 6890 (Agilent Technologies, CA, USA) coupled with an electron capture detector. $CHCl_3$, $CHBrCl_2$, $CHClBr_2$, and $CHBr_3$ were separated

on a fused silica DB-1MS capillary column (30 m × 0.25 mm × 0.25 mm film thickness) (Agilent Technologies, CA, USA). The gas chromatograph was calibrated using certified standards (Sigma-Aldrich, St. Louis, MO, USA) of individual trihalomethanes and the sum of concentrations determined in a sample was reported as TTHMs (Appendix 3.A). The pH of the samples was measured using a potable pH/DO meter (Extech DO700, Waltham, MA, USA). DOC was quantified using APHA Method 5310C with a Fusion Total Organic Carbon Analyzer (Teledyne Tekmar Inc., Mason, OH). Ultraviolet absorbance at 254 nm (UV254, m^{-1}), a measure of humic content in water, was measured using an Ultraspec 2100 pro UV/visible spectrophotometer (Biochrom, Cambridge, UK). Specific ultraviolet absorbance (SUVA, L mg^{-1} m^{-1}), a good indicator of humic content and aromatic carbon content in water, was calculated by multiplying the measured UV254 by 100 and dividing by DOC (Matilainen and Sillanpää 2010).

3.3.2. Statistical Analysis

Analysis of variance (ANOVA) using PROC GLIMMIX in SAS Version 9.4 (SAS Institute Inc. 2013) was performed separately for each community on data for $CHCl_3$, $CHBrCl_2$, $CHClBr_2$, $CHBr_3$, TTHMs, DOC, UV254, SUVA, and pH. For communities A and B, the water distribution system (piped and cisterns) and sampling time were used as a fixed effect, while for Community C, only the water distribution system was used as a fixed effect because samples were collected only once from this community. Homes within a community were considered as replicates. Statistical analyses were conducted for each community separately because cistern configurations (e.g., concrete vs. fibreglass vs. polyethylene) and setups (underground vs. aboveground) were not consistent among communities. The Tukey multiple comparison procedure was used to compare the least square means when three or more treatment means were compared. For all statistical analyses, significance was determined at $\alpha = 0.05$. For the source and treated water taps at the

WTP and the water truck, samples were collected in sequence from the same taps and truck, hence the water quality parameters determined for these samples were excluded from the analysis of variance.

3.4. Results

3.4.1. Trihalomethanes in Water from Water Treatment Plant Taps and Water Trucks

The pH ranged from 7.6 to 8.8 for source water and most often slightly decreased with the water treatment process (Figure 3.1). For both sampling times in Community B, relative to source water, the water treatment process reduced DOC in treated water by about 13%. In contrast, in both in Communities A and C and regardless of sampling time, DOC in treated water was numerically larger than in source water (Figure 3.1). Relative to source water treated water always showed between 32% to 78% (UV254) and between 35 % to 82% (SUVA) smaller absorbance values, indicating a reduced humic and aromatic carbon content in water (Figure 3.1). The source and treated water samples collected in the fall in Communities A and B (both in October) showed generally lower DOC, UV254 and SUVA values relative to the source and treated water samples collected from these communities moving into early- (February, Community A) and late spring (May, Community B).

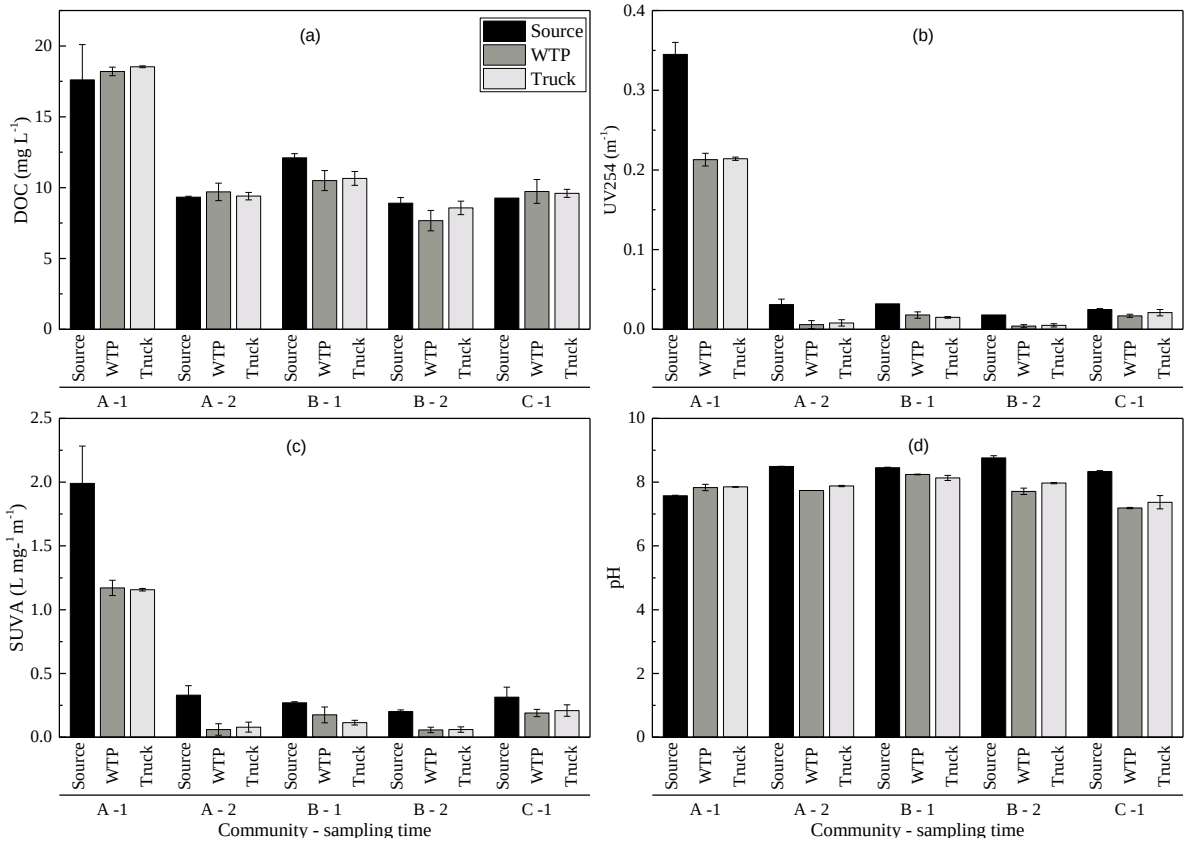

Figure 3.1. (a) Dissolved organic carbon (DOC), (b) UV absorbance at 254 nm (UV254), (c) specific UV absorbance (SUVA), and (d) pH of source water and treated water in the water treatment plant and of truck water in each community and for each sampling time. In Community A sampling time A - 1 = February 2015; A - 2 = October 2015; in Community B, sampling time B - 1 =May 2015; B - 2 = October 2015; in Community C the one sampling time C - 1 = June 2015. Error bars represent standard deviations.

Average TTHM concentrations in treated water at the WTP exceeded the Canadian Drinking Water Quality Guidelines in Community A in both February (187 µg L^{-1}) and October (109 µg L^{-1}), as well as in Community B in May (115 µg L^{-1}) (Figure 3.2, Appendix 3.B: Table 1). However, TTHM concentrations in treated water at the WTP were below the MAC when sampling Community B in October (72 µg L^{-1}) and Community C in June (81 µg L^{-1}). The largest TTHM concentrations in community A coincided with samples having the largest DOC, UV254 and

SUVA levels across all sampling times and communities (Figure 3.1). $CHCl_3$ was the dominant trihalomethane (42 – 96%) followed by $CHBrCl_2$ (3 – 37%) and $CHClBr_2$ (1 – 18%) (Figure 3.2). $CHBr_3$ always contributed to less than 5% of TTHMs and was not detected in the treated water at the WTP in Community C.

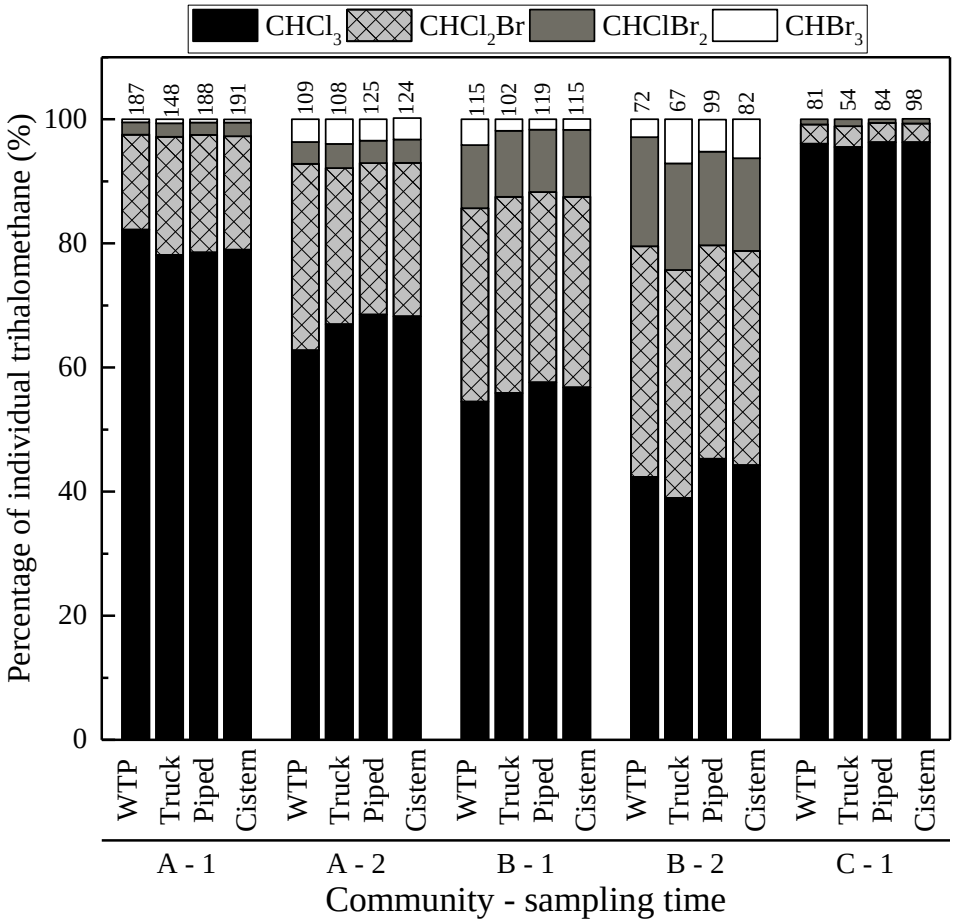

Figure 3.2. Percentage of individual trihalomethanes ($CHCl_3$, $CHBrCl_2$, $CHClBr_2$, and $CHBr_3$) in treated water samples from the water treatment plants (WTP), water delivery trucks, piped water and cistern water in the three communities for each sampling time. Numbers above the bars are mean TTHMs (µg L^{-1}). In Community A sampling time A - 1 = February 2015; A - 2 = October 2015; in Community B, sampling time B - 1 =May 2015; B - 2 = October 2015; in Community C the one sampling time C - 1 = June 2015.

Regardless of the community and sampling time, truck water had relatively similar DOC, UV254 and SUVA levels as the treated water at the WTP (Figure 3.1). However, relative to the treated water at the WTP, the TTHM concentrations in the truck water were substantially lower, for example by 33% in Community C and by 20% in Community A for samples taken in February. The reduction was numerically smaller in other cases, including for samples taken in Community A in October (1.2%) (Figure 3.2.). Overall, across communities and sampling times, loss of $CHCl_3$ accounted for 56 to 97% of the reduction in TTHMs, loss of $CHCl_2Br$ accounted for 2.6 to 35%, loss of $CHClBr_2$ accounted for 12% and loss of $CHBr_3$ accounted for 2%.

3.4.2. Trihalomethanes in Tap Water from Homes

In all three communities, piped and cistern water samples showed statistically similar concentrations of TTHMs (Table 3.1). TTHM concentrations in Community A homes ranged from 96 to 207 µg L^{-1}, in Community B from 45 to 160 µg L^{-1} and in Community C from 57 to 122 µg L^{-1} (Figure 3.2). Total THMs concentrations exceeded the Health Canada limit of 100 µg L^{-1} in 100% of piped and 83% of cistern water samples in Community A, in 61% of piped and 42% of cistern water samples in Community B and in 30% of piped and 50% of the cistern water samples in Community C. The largest contributor to TTHM presence, $CHCl_3$ concentration, was also not significantly different between piped and cistern water samples in any of the communities (Table 3.1). The $CHBrCl_2$ concentration was significantly greater in piped water samples compared to cistern water only in Community B (Table 3.1).

Table 3.1. Total trihalomethanes (TTHMs), $CHCl_3$, $CHBrCl_2$, $CHClBr_2$, and $CHBr_3$ concentrations in piped and cistern water in each community

		Community A					Community B					Community C			
Effect		$CHCl_3$	$CHBrCl_2$	$CHClBr_2$	$CHBr_3$	TTHMs	$CHCl_3$	$CHBrCl_2$	$CHClBr_2$	$CHBr_3$	TTHMs	$CHCl_3$	$CHBrCl_2$	$CHClBr_2$	TTHMs
		μg L^{-1}													
Distribution system	Piped	117a[a]	33.0a	4.12a	2.65a	157a	57.2a	35.2a	13.3	3.5a	109a	80.7a	2.52a	0.59a	84.1a
	Cistern	118a	32.7a	4.42a	2.58b	158a	51.5a	31.5b	12.1	3.5a	98.9a	94.6a	2.98a	0.56a	98.4a
Sampling time	1[b]	149a	35.2a	3.99b	0.98b	190a	67.9a	35.6a	11.9	1.94b	117a				
	2	85.5b	30.5b	4.56a	4.24a	125b	40.8b	31.2b	13.5	5.05a	90.7b				
		p-value													
Distribution system		0.99	0.74	0.06	0.13	0.83	0.12	0.03	0.03	0.84	0.15	0.22	0.15	0.22	0.16
Sampling time		<0.001	0.008	0.001	0.0001	<0.001	<0.001	0.01	0.006	0.001	0.0009				
Distribution system × sampling time		0.73	0.98	0.26	0.72	0.76	0.36	0.15	0.008	0.93	0.38				

[a] Means in the same column followed by the same letter are not significantly different according to the Tukey-Kramer test ($\alpha = 0.05$). Mean comparison for main effects is given when the interaction is not significant.

[b] In Community A sampling time 1 = Feb 2015; 2 = Oct 2015; In Community B, sampling time 1 = May 2015; 2 = Oct 2015; in Community C only one sampling time June 2015.

Tap water in homes has a pH ranging from 7.1 to 8.5 (Figure 3.3), with no significant differences between piped and cistern distribution systems, except that in Community C, the pH was significantly greater in cistern than in piped water (Figure 3.3, Appendix 3.B: Table 2). Piped and cistern water samples had statistically similar DOC and SUVA levels in Community A (Appendix 3.B: Table 2). DOC, UV254 and SUVA levels in Community B were significantly greater in cisterns than in piped water samples; however, numerical differences in DOC levels were greater between sampling times (May versus October) than between the type of water distribution systems (piped versus cistern) (Figure 3.3). DOC was also significantly greater in cistern than piped water in Community C (Figure 3.3; Appendix 3.B Table 2).

Total trihalomethanes concentrations in tap water from homes were significantly greater in February than in October in Community A and significantly greater in May than in October in Community B (Table 3.1). The $CHCl_3$ and $CHBrCl_2$ concentrations in tap water (averaged across piped and cistern water) were significantly greater in February in Community A and May in Community B compared to October samples in both communities (Table 3.1). However, $CHBr_3$ concentration was significantly lower in February in Community A and May in Community B compared to October samples in both communities (Table 3.1). The $CHClBr_2$ concentration in Community B piped water was the highest in May samples. These results suggest that time of the year is a dominant factor affecting the formations of types of trihalomethanes since source water quality changes with the time of the year. This corresponded to the results that the tap water in homes had significantly greater DOC, UV254 and SUVA levels in February than October in Community A and significantly greater DOC and SUVA levels in May than October in Community B (Figure 3.3, Appendix 3.B; Table 2). As noted above, the source and treated water

in the WTP, and the truck water, had numerically greater DOC, UV254 and SUVA levels in February than October in Community A and in May than October in Community B (Figure 3.1).

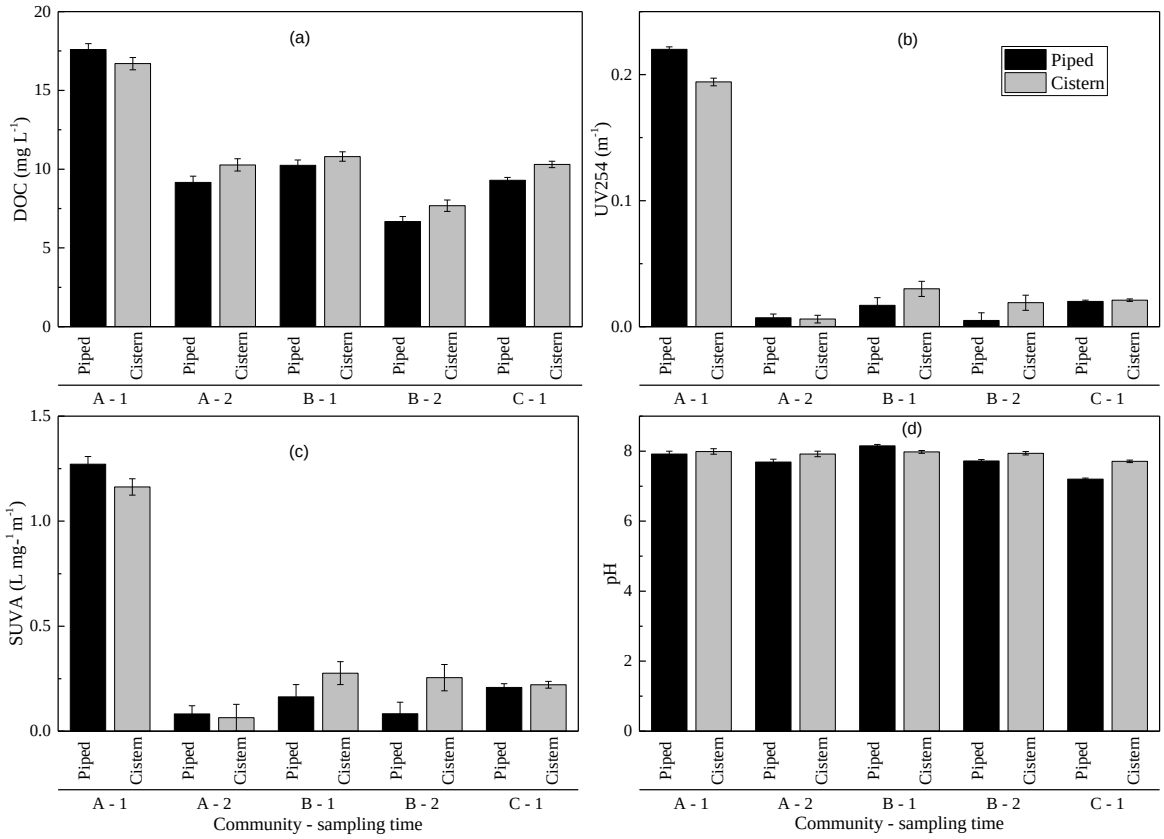

Figure 3.3. (a) Dissolved organic carbon (DOC), (b) UV absorbance at 254 nm (UV254), (c) specific UV absorbance (SUVA), and pH of piped and cistern water in each community. In Community A sampling time A - 1 = February 2015; A - 2 = October 2015; in Community C the one sampling time C - 1 = June 2015. Error bars represent standard error of means.

In each community, the relative proportions of the types of trihalomethanes detected in piped and

Figure 3.3. (a) Dissolved organic carbon (DOC), (b) UV absorbance at 254 nm (UV254), (c) specific UV absorbance (SUVA), and pH of piped and cistern water in each community. In Community A sampling time A - 1 = February 2015; A - 2 = October 2015; in Community C the one sampling time C - 1 = June 2015. Error bars represent standard error of means.

cistern water were reflective of the relative proportions of the types of trihalomethanes detected in the treated water WTP (Figure 3.2). However, for each sampling time in each community, TTHM concentrations in the majority of the samples were numerically greater in tap water from piped or cistern homes than in treated water at the WTP (Figure 3.2). This suggests a continued persistence and formation of trihalomethanes during water distribution. On average, TTHM concentrations in piped water were 1% (February) and 14% (October) greater than the TTHM concentrations in the treated water at the WTP in Community A, with corresponding percentages of 4% (May) and 38% (October) in Community B and 4% in Community C. Concentration of TTHMs in cistern water were 3% (February) and 14% (October) greater than the concentration of TTHMs in the treated water at the WTP in Community A, with corresponding percentages of 1% (May) and 15% (October) in Community B, and 21% in Community C. This increase in TTHMs during water distribution can be attributed to the presence of natural organic matter attached to the pipe and cistern walls which further enhance reactions with free chlorine producing TTHMs.

Relative to the treated water at the WTP, the substantial increase in concentrations of TTHMs in the tap water in piped and cistern homes in October in Communities A and B and for cisterns in Community C was due to the formation of $CHCl_3$ and $CHBrCl_2$ during water distribution. On average, in October, $CHCl_3$ concentrations in piped water were 17% and 54% greater than the $CHCl_3$ concentrations in the treated water at the WTPs in communities A and B, respectively, and such percentages were 17% and 25% for cisterns, respectively. Similarly, in October, $CHBrCl_2$ concentrations in piped water were 11% and 35% greater than the $CHBrCl_2$ concentrations in the treated water at the WTPs in Community A and B, respectively, and such percentages were 11% for cisterns in communities A and B. In Community C, the $CHCl_3$ concentration in piped water

was 4% greater and in cisterns was 21% greater than the $CHCl_3$ concentration in the treated water at the WTPs.

3.5. Discussion

The DOC, UV254 and SUVA were used to assess the level and nature of NOM in water samples. The communities were geographically located in regions (Canadian Prairies, Canadian Shield) known to have high DOC contents in surface water (Goss et al. 2017; Sadrnourmohamadi 2015), and source water DOC in all three communities was indeed relatively high, ranging from 8.9 to 17.6 mg L^{-1}. The low SUVA ($\leq$2 L mg^{-1} m^{-1}) values indicated that the source waters contained DOC that was relatively low in humic substances with high aromaticity but relatively rich in hydrophilic, non-humic and low molecular weight organic matter compounds (Edzwald and Van Benschoten 1990; Wong et al. 2007). DOC in source and treated water was greater for samples collected in February or May than for samples that were collected from communities in October. These greater DOC levels can be attributed to the DOC loading to surface water bodies by spring runoff due to snowmelt and saturated fall conditions. For example, in a study that involved a municipal WTP in northern England, Sharp et al. (2006) concluded that DOC in source water was greater in the spring than fall because warmer temperatures in the spring increased microbial driven organic matter decomposition and the subsequent release of DOC to surface water during rainfall-runoff. As well, studies conducted in England and Sweden reported high DOC values in source water during winter months, and this was attributed to increased availability of DOC in the watershed due to the physical disruption of the soil and mortality of fine roots due to freezing and thawing and lysis of freeze-damaged soil organisms (Haei et al. 2010; Sharp et al. 2006). In the

three communities in our study, UV254 and SUVA also changed with DOC concentration in both source waters, indicating that the DOC content and the organic matter composition also varied.

Several researchers (Goss et al. 2017; Goss 2011; Sadrnourmohamadi 2015) reported that surface waters within the Canadian Prairies and Canadian Shield have high concentrations of DOC and that its calcium hardness causes challenges in removing such precursor that can lead to the formation of trihalomethanes. If the water treatment process is inefficient in removing DOC in source water before chlorination, it can increase the trihalomethane concentrations in treated water after chlorination (Crittenden et al. 2012). Thus, the high concentrations of TTHMs observed in many of the samples can result from the inefficient removal of DOC by the water treatment process. Based on the low SUVA (<2 L mg^{-1} m^{-1}), it is possible that there were relatively high concentrations of the fulvic acid fraction in source water, and elevated fulvic acid concentrations have been shown to decrease DOC removal by conventional water treatment methods (Edzwald and Van Benschoten 1990). It is also possible that initial coagulation and clarification steps removed a portion of DOC in source water but that granular filtration re-introduced DOC into the treated water as observed for some municipal WTPs in Manitoba (Goss 2011). Across communities, the water treatment process in Community B was most effective in reducing DOC, UV254 and SUVA, likely because the WTP in Community B was equipped with a reverse osmosis system that promotes the removal of DOC during water treatment (Zazouli and Kalankesh 2017). However, for all communities more advanced water treatment processes might be required for the satisfactory removal of DOC during treatment. For example, Goss (2011) found that nanofiltration at a municipal WTP in Morris, Manitoba, successfully removed all DOC fractions controlling the formation of trihalomethanes.

Across the communities and sampling times, in three out of the five sampling times, the treated water samples at the WTPs had TTHM concentrations > 100 µg L^{-1}, indicating that the water treatment process needs to be augmented to decrease the formation of trihalomethanes effectively. $CHCl_3$ was the predominant compound contributing to this exceedance. Trihalomethane concentrations were lower in truck water relative to the WTP water. This reduction mainly came from reducing $CHCl_3$ concentration, probably via volatilization during handling and water delivery. Brominated trihalomethanes accounted for a relatively smaller portion of TTHMs, particularly in Community C, where < 5% of the TTHMs were brominated trihalomethanes. Previous studies that involved municipal WTPs in Manitoba also reported low concentrations of brominated trihalomethanes in treated water (Goss and Gorczyca 2013; Sadrnourmohamadi 2015). On average, TTHM concentrations increased during water distribution from the WTP to homes by 1 to 38% for households with piped water and 1 to 21% for households with cisterns. Across communities and sampling rounds, between 30 to 100% of piped and 42 to 83% of cistern water samples had TTHMs > 100 µg L^{-1}. This is concerning since the findings of epidemiological and animal based laboratory studies suggest that the long-term exposure to trihalomethanes increases risks for bladder cancer and reproductive defects such as lower birth weights and growth delay (Evlampidou et al. 2020; Mohamadshafiee and Taghavi 2012). Such concerns exist for other communities in Manitoba because a survey reported that almost half of the 100 public water systems surveyed had TTHMs > 100 µg L^{-1} in 2018 (Department of Conservation and Climate 2020).

Both Communities B and C have belowground cisterns and the DOC concentrations were significantly greater in cistern than piped water (Figure 3.3), probably from the seepage of

groundwater into cracked belowground cisterns during snowmelt and heavy rainfall (Kalbitz et al. 2000). Previous research has shown that specific TTHM production, which is the amount of TTHMs produced per unit of DOC, ranges from 11 to 1372 µg TTHMs/mg DOC in surface water from two Canadian rivers (Goss and Gorczyca 2013). However, there were no significant differences in the concentration of TTHMs between piped and cistern water samples in any of the communities. Although it is possible that trihalomethanes were more readily formed in cisterns than in piped water because of the greater DOC concentrations in cistern water, it is also likely that trihalomethanes were more readily lost from cistern water during storage. Specifically, trihalomethanes are relatively volatile compounds, with vapour pressure at 25 °C ranging from 0.80 kPa for $CHBr_3$ to 23.33 kPa for $CHCl_3$; therefore can be volatilized during water storage in cisterns (Brown et al. 2011). Sampling month appeared to be a significant factor affecting the concentration of TTHMs in piped and cistern water and this was likely due to temporal changes in DOC concentrations in the source water. For example, in communities A and B, the highest concentrations of TTHMs occurred when the DOC in source water and treated water were the highest. In order to better understand the temporal variation of TTHM concentration as a function of temporal variations in source water quality and environmental factors (e.g., temperature and rainfall), which will help implement measures to reduce TTHMs formation, more frequent sampling distributed over the year should be conducted. Many First Nations communities in Manitoba are remote and hence the building of human and infrastructural resources for communities to conduct monitoring studies is essential to accomplish more frequent sampling regimes and test innovative solutions to reducing TTHMs in drinking water.

3.6. Conclusion

In two of the three First Nations communities, water treatment plants had TTHM concentrations exceeding the 100 µg L^{-1} threshold set by Health Canada. The TTHM concentrations in the tap water of homes were sometimes larger than the in the treated water at the water treatment plant, suggesting that TTHMs formation occurs while the treated water is being distributed to homes. The water distribution method (piped or cistern) did not significantly affect TTHMs concentration with 75% of piped water samples and 70% of cistern water samples exceeding the 100 µg L^{-1} threshold. Exceeding the threshold means that tap water consumption in these homes poses a potential human health risk. $CHCl_3$ was always the most abundant TTHMs, followed by $CHBrCl_2$ and $CHClBr_2$. Currently, the Canadian regulatory process suggests taking quarterly samples at the point in the distribution system with the highest potential TTHMs levels. Given that the total and individual trihalomethane concentrations were relatively variable depending on which home in the community, the current data cannot explain where that point would be. A more intensive sampling density (number of homes within a community) and frequency (number of sampling rounds per year) could help determine the causes of these variations.

3.7. Acknowledgments

We gratefully thank the Chief and Band Councils of communities A, B and C, and other community members for the partnership established. Financial support for this research was supported by the Natural Sciences and Engineering Research Council of Canada (NSERC) through its Collaborative Research and Training Experience (grant# 432009-2013) and Discovery Grant (grant# RGPIN-2018-06030, grant# RGPIN-6417-2019) programs. We also gratefully

acknowledge the following awards to Amarawansha that assisted with student stipends: The NSERC Postgraduate Scholarship (Doctoral program) and The University of Manitoba Graduate Scholarship (Doctoral program). We highly acknowledge the technical assistants from Thor Halldorson, Department of Chemistry, University of Manitoba.

3.8. References

Abedi, M. (2019) 'Indigenous communities and water crises — is a real solution in the works?' *Global News*, July 27, 2019 (Online). https://globalnews.ca/news/5671392/attawapiskat-state-of-emergency-clean-water/ Accessed on May 13, 2020

Bradford, L., Waldner, C., McLaughlin, K., Zagozewski, R., & Bharadwaj, L. (2018). A mixed-method examination of risk factors in the truck-to-cistern drinking water system on the Beardy's and Okemasis First Nation Reserve, Saskatchewan. *Canadian Water Resources Journal*, *43*, 383–400. https://doi.org/10.1080/07011784.2018.1474139

Brown, D., Bridgeman, J., & West, J. R. (2011). Predicting chlorine decay and THM formation in water supply systems. *Reviews in Environmental Science and Bio-Technology*, *10*, 79–99. https://doi.org/10.1007/s11157-011-9229-8

Charisiadis, P., Andra, S. S., Makris, K. C., Christophi, C. A., Skarlatos, D., Vamvakousis, V., et al. (2015). Spatial and seasonal variability of tap water disinfection by-products within distribution pipe networks. *Science of the Total Environment*, *506*, 26–35. https://doi.org/10.1016/j.scitotenv.2014.10.071

Chhipi-Shrestha, G., Rodriguez, M., & Sadiq, R. (2018). Unregulated disinfection by-products in drinking water in Quebec: A meta analysis. *Journal of Environmental Management, 223,* 984–1000. https://doi.org/10.1016/j.jenvman.2018.06.082

Crittenden, J. C., Trussell, R. R., Hand, D. W., Howe, K. J., & Tchobanoglous, J. (2012). *MWH ' s Water Treatment Principles and Design* (3rd ed.). New Jersey: John Wiley & Sons, Inc.

Dominguez-Tello, A., Arias-Borrego, A., Garcia-Barrera, T., & Gomez-Ariza, J. L. (2015). Seasonal and spatial evolution of trihalomethanes in a drinking water distribution system according to the treatment process. *Environmental Monitoring and Assessment*, *187,* 1–19. https://doi.org/10.1007/s10661-015-4885-8

Dupont, D., Waldner, C., Bharadwaj, L., Plummer, R., Carter, B., Cave, K., & Zagozewski, R. (2014). Drinking water management: Health risk perceptions and choices in first nations and non-First Nations communities in Canada. *International Journal of Environmental Research and Public Health*, *11*, 5889–5903. https://doi.org/10.3390/ijerph110605889

Edzwald, J. K., & Van Benschoten, J. E. (1990). Aluminum coagulation of natural organic matter. In *Chemical Water and Wastewater Treatment* (pp. 341–359). Berlin, Heidelberg: Springer Berlin Heidelberg. https://doi.org/10.1007/978-3-642-76093-8_22

EKOS Research Associates Inc. (2009). *Water Quality On-Reserve Quantitative Research Submitted to*. Final Report submitted to Health Canada, Ottawa, Ontario. www.ekos.com. Accessed 20 December 2020

Evlampidou, I., Font-Ribera, L., Rojas-Rueda, D., Gracia-Lavedan, E., Costet, N., Pearce, N., et al. (2020). Trihalomethanes in drinking water and bladder cancer burden in the European Union. *Environmental Health Perspectives*, *128,* 1–14. https://doi.org/10.1289/EHP4495

Farenhorst, A., Li, R., Jahan, M., Tun, H. M., Mi, R., Amarakoon, I., et al. (2017). Bacteria in drinking water sources of a First Nation reserve in Canada. *Science of the Total Environment, 575*, 813–819. https://doi.org/10.1016/j.scitotenv.2016.09.138

Fernando, D. M., Tun, H. M., Poole, J., Patidar, R., Li, R., Mi, R., et al. (2016). Detection of antibiotic resistance genes in source and drinking water samples from a first nations community in Canada. *Applied and Environmental Microbiology*, *82*, 4767–4775. https://doi.org/10.1128/AEM.00798-16

Gopal, K., Tripathy, S. S., Bersillon, J. L., & Dubey, S. P. (2007). Chlorination byproducts, their toxicodynamics and removal from drinking water. *Journal of Hazardous Materials*, *140*, 1–6. https://doi.org/10.1016/j.jhazmat.2006.10.063

Goss, C. D. (2011). Characterization of natural organic matter and trihalomethane formation potential for NOM fractions isolated from two surface water sources in Manitoba. MSc thesis, University of Manitoba, Winnipeg, Manitoba, Canada.

Goss, C. D., & Gorczyca, B. (2013). Trihalomethane formation potential of DOC fractions isolated from two Canadian Prairie surface water sources. *Water Science and Technology: Water Supply*, *13*, 114–122. https://doi.org/10.2166/ws.2012.093

Goss, C. D., Wiens, R., Gorczyca, B., & Gough, K. M. (2017). Comparison of three solid phase extraction sorbents for the isolation of THM precursors from Manitoba surface waters. *Chemosphere*, *168*, 917–924. https://doi.org/10.1016/j.chemosphere.2016.10.118

Haei, M., Oquist, M., Buffam, I., Ågren, A., Blomkvist, P., Bishop, K., et al. (2010). Cold winter soils enhance dissolved organic carbon concentrations in soil and stream water. *Geophysical Research Letters*, *37*, 1-5.

INAC. (2003). *National Assessment of Water and Wastewater Systems in First Nations Communities : Summary Report*. Orangeville, ON: Indian and Northern Affairs Canada.

Kalbitz, K., Solinger, S., Park, J., Michalzik, B., & Matzner, E. (2000). Controls on the dynamics of dissolved organic matter in soils: a review. *Soil Science*, *165*, 277–304.

Liang, L., & Singer, P. C. (2003). Factors influencing the formation and relative distribution of haloacetic acids and trihalomethanes in drinking water. *Environmental Science & Technology*, *37*, 2920–2928. https://doi.org/10.1021/es026230q

Manitoba Water Stewardship. (2015). *Public Water System Compliance with Provincial Physical , Chemical , and Microbial Standards - Systems Open Year Round Public Water System Compliance with Provincial Physical , Chemical , and Microbial Standards - Systems Open Year Round*. https://www.gov.mb.ca/waterstewardship/odw/public-info/general-info/compliance_data/year_rnd_pws_may_29_2015.pdf

Matilainen, A., & Sillanpää, M. (2010). Removal of natural organic matter from drinking water by advanced oxidation processes. *Chemosphere*, *80*, 351–365. https://doi.org/10.1016/j.chemosphere.2010.04.067

Mohamadshafiee, M. R., & Taghavi, L. (2012). Health Effects of Trihalomethanes as Chlorinated Disinfection by Products : A Review Article. *International Journal of Environmental and Ecological Engineering*, *6*, 545–551. http://www.epa.gov/iris/subst/0025.htm

Neegan Burnside. (2011). *National Assessment of First Nations Water and Wastewater Systems;Manitoba Regional Roll-Up Report:* Department of Indian and Northern Affairs: Ottawa, ON, Canada.

Rodriguez, M. J., Vinette, Y., Sérodes, J.-B., & Bouchard, C. (2003). Trihalomethanes in drinking water of greater Québec region (Canada): Occurrence, variations and modelling.

Environmental Monitoring and Assessment, *89*, 69–93. https://doi.org/10.1023/A:1025811921502

Sadrnourmohamadi, M. (2015). Removal of trihalomethanes precursors from surface waters typical for Canadian Prairies and Shield. PhD thesis, University of Manitoba, Winnipeg, Manitoba, Canada.

SAS Institute Inc. (2013). SAS/STAT User's Guide version 9.4. *Statistical Analysis System (SAS Institute Inc.)*. Cary, NC.

Scheili, A., Rodriguez, M. J., & Sadiq, R. (2015). Seasonal and spatial variations of source and drinking water quality in small municipal systems of two Canadian regions. *Science of the Total Environment*, *508*, 514–524. https://doi.org/10.1016/j.scitotenv.2014.11.069

Sharp, E. L., Parsons, S. A., & Jefferson, B. (2006). Seasonal variations in natural organic matter and its impact on coagulation in water treatment. *Science of the Total Environment*, *363*, 183–194. https://doi.org/10.1016/j.scitotenv.2005.05.032

Villanueva, C. M., Cordier, S., Font-Ribera, L., Salas, L. A., & Levallois, P. (2015). Overview of Disinfection By-products and Associated Health Effects. *Current Environmental Health Reports*, *2*, 107–115. https://doi.org/10.1007/s40572-014-0032-x

Wei, J. R., Ye, B. X., Wang, W. Y., Yang, L. S., Tao, J., & Hang, Z. Y. (2010). Spatial and temporal evaluations of disinfection by-products in drinking water distribution systems in Beijing, China. *Science of the Total Environment*, *408*, 4600–4606. https://doi.org/10.1016/j.scitotenv.2010.06.053

World Health Organization. (2003). Trihalomethanes in Drinking-water. Background document for development of WHO Guidelines for Drinking-water Quality. *WHO*, *2*, 1–9. https://doi.org/WHO/SDE/WSH/03.04/74

Xu, J., Huang, C., Shi, X., Dong, S., Yuan, B., & Nguyen, T. H. (2018). Role of drinking water biofilms on residual chlorine decay and trihalomethane formation: An experimental and modeling study. *Science of the Total Environment*, *642*, 516–525. https://doi.org/10.1016/j.scitotenv.2018.05.363

4. EFFECTS OF BELOWGROUND CISTERN STORAGE OF DRINKING WATER ON CHLORINE DECAY KINETICS AND TRIHALOMETHANE FORMATION IN A MANITOBA FIRST NATIONS COMMUNITY

4.1. Abstract

Belowground cisterns are commonly used by households in First Nations reserves in Canada for storing water delivered by water delivery trucks. While storing water, the free residual chlorine concentration needs to be maintained at sufficient levels to prevent bacterial regrowth and the WHO recommends this level to be >0.2 mg L^{-1}. This study investigated the chlorine decay in belowground cisterns in homes on a First Nation reserve in the Province of Manitoba and conducted laboratory studies on cistern water to examine further chlorine decay and the formation of trihalomethanes formation in water during storage. Initial chlorine level impacted in-vitro bulk decay rates in laboratory flasks, with the free residual chlorine decay ranging from 0.01 to 0.15 mg L^{-1} day^{-1}. Free residual chlorine decay in belowground cisterns followed first-order kinetics with a field decay rate of 1.28 mg L^{-1} day^{-1}, suggesting substantial wall decay. In the laboratory study, chlorine decay was affected by initial chlorine concentration and dissolved organic carbon and ultraviolet (UV) absorbance at 254 nm. Even though TTHMs were readily formed in the cistern stored water in the laboratory study, the field study showed that concentrations of TTHMs decreased during storage, most likely due to volatilization. Our results underline the need for regular cleaning and proper maintenance of belowground cisterns to ensure safe drinking water for households on First Nations reserves.

4.2. Introduction

Chlorine compounds are the most commonly used disinfectants in drinking water treatment and assist in the inactivation of waterborne pathogens that can harm humans consuming drinking water (Crittenden et al. 2012). Although chlorine has a high oxidizing potential and hence high disinfecting effectiveness (Gopal et al. 2007), chlorine volatilization and reaction with various materials lead to chlorine decay and results in lesser overall effectiveness for suppressing microbial growth. Chlorine decay due to reactions within water is referred to as bulk chlorine decay. Chlorine can also react with distribution pipes or storage tank walls, a type of decay often described as wall decay. These two types of decay are collectively referred to as the chlorine demand of the water (Brown et al. 2011; Clark 2011). Bulk chlorine decay has been shown to be affected by the initial chlorine concentration of the water, the water temperature, as well as the types and amounts of organic and other inorganic compounds in the water (Devarakonda et al. 2010; Powell et al. 2000a; Vasconcelos et al. 2013). Understanding chlorine decay and chlorine demand are important to provide safe drinking water to consumers (Vasconcelos et al. 1997).

One of the main challenges in water chlorination is the production of disinfection by-products from the reaction of chlorine with natural organic matter in the water (Rodriguez et al. 2003; Sadiq and Rodriguez 2004). These reactions are mainly governed by initial chlorine dose, natural organic matter types and concentrations, temperature, pH, bromide concentration and contact time (Brown et al. 2011). More than 600 disinfection by-products have been identified with trihalomethanes frequently present in water systems (Kumari and Gupta 2015; Li and Mitch 2018). Trihalomethanes have been identified as potential carcinogens to animals and humans and can cause reproductive abnormalities (Krasner 2009; Villanueva et al. 2014). Health concerns related

to trihalomethanes prompted the setting of maximum acceptable concentrations of total trihalomethanes (TTHMs) in drinking water (Health Canada 2019; USEPA 2010; WHO 2008). In Canada, Health Canada has set the maximum concentration of TTHMs in treated water at 100 µg L^{-1} (Health Canada 2019).

Inadequate access to safe drinking water continues to be a major issue for First Nations reserves in Canada (Baijius and Patrick 2019; Neegan Burnside 2011a). For example, only one-half of the homes in First Nations reserves in the Province of Manitoba have direct piped water from WTPs (Neegan Burneside, 2011). About one-third of the homes have their drinking water delivered by truck and stored in cisterns. The remaining homes rely on well water or do not have running water. Available funding and site conditions (e.g., soil stoniness and drainage; groundwater table and slope; potential sources of contamination) influence the types of cisterns used (e.g., concrete, fibreglass, polyethylene; underground, aboveground) (Health Canada 2012). The presence of organic matter in cisterns can enhance chlorine decay reactions, thereby decreasing residual chlorine concentration and the disinfection efficiency (Baird et al. 2013; Graham and Vanderslice 2007). Chlorine can also react with biofilms and walls of the storage tanks (and pipe walls) and longer water storage and poor maintenance can increase biofilm formation in cisterns (Graham and Vanderslice 2007; Xu et al. 2018). Structural damage to belowground cement cisterns can often lead to seepage of outside water into the cisterns, leading to increased levels of particulates and dissolved material into the cistern, as well as to increased risks for microbial contamination (Artiola et al. 2012; Health Canada 2016; Mahler et al. 2000).

Several studies have focused on free residual chlorine decay in piped water in the distribution system (Brown et al. 2011; Vasconcelos et al. 1997; Zhang et al. 2017; Zhao et al. 2018). However,

we are not aware of any published study that has focused on free residual chlorine decay kinetics in cistern stored water at the consumer level, including in First Nations reserves. The objectives of this study were to determine the free residual chlorine decay kinetics, calculate the chlorine demand of the cistern stored water under natural environmental conditions, and evaluate the in-vitro bulk chlorine decay in cistern stored water under laboratory conditions. The formation during water storage of trichloromethane ($CHCl_3$), bromodichloromethane ($CHBrCl_2$) dibromochloromethane ($CHClBr_2$), and tribromomethane ($CHBr_3$) was also assessed during these field and laboratory experiments.

4.3. Materials and methods

4.3.1. Site description

The field study was conducted in a First Nation community in Northern Manitoba. The community has a conventional water treatment system that entails coagulation, flocculation, sedimentation, filtration, and disinfection using chlorine (sodium hypochlorite solution). Only 28.4% (60 homes) of the community's 211 homes receive piped water directly from the water treatment plant, whereas the remaining 71.6% (151 homes) have their water delivered by truck and stored in either belowground concrete cisterns or aboveground polyethylene cisterns. Structural damage evident in some of the belowground cisterns in the community has increased the risk of water contamination via seepage into the damaged cisterns, introducing pathogens and particulate (including organic) matter, which can react with chlorine in the water.

4.3.2. In-situ chlorine decay kinetics and trihalomethane formation in belowground cistern water

Tap water samples were collected from four homes equipped with belowground cisterns that had just been filled (within 1 h of sampling) by a water truck. Water samples were collected once every day until free chlorine concentration fell below detection. The samples were analyzed for free and total residual chlorine at the time of sampling using a Hach chlorine test kit and Hach chlorine colorimeter^{TM} II (VWR International, Mississauga, ON, Canada). Turbidity was measured using a portable AQ4500 turbidity meter (Thermo Scientific™, Beverly, MA, USA) and dissolved oxygen concentration and pH was measured using a pH/DO meter (Extech DO700, Waltham, MA, USA) at the time of sampling. Water samples for the analyses of trihalomethanes were collected into 60-mL amber bottles with PTFE screw cap lids containing 3% sodium thiosulfate solution (0.06 mL). Trihalomethanes were extracted using Standard Methods 6232B (APHA 2012) Liquid-Liquid Extraction and analyzed using an Agilent 7890A GC System (Agilent Technologies, Santa Clara, CA, USA) equipped with a CombiPAL (CTC Analytics AG, Zwingen, Switzerland) with a split/split-less injector. The separated species were detected with an electron capture detector and concentrations were calculated using an electron capture detector and calculated using external calibration and their sum was considered TTHMs.

4.3.3. In-vitro- bulk chlorine decay in cistern water

Treated water from three cisterns was collected in July 2016 and transported to the University of Manitoba in coolers for laboratory study. Water pH, turbidity and DOC were quantified using the methods described above at the beginning and end of the experiment. Ultraviolet absorbance at 254 nm (UV254, m^{-1}) was determined in all samples using an Ultraspec 2100 pro UV/visible spectrophotometer (Biochrom, Cambridge, UK). Specific ultraviolet absorbance (SUVA, L mg^{-1}

m^{-1}) was calculated as 100 × (UV254 / DOC). The experimental layout was a completely randomized design with three replicates. Aliquots (100 mL each) of the water samples were chlorinated with sodium hypochlorite (5% laboratory-grade) in 250 mL glass Erlenmeyer flasks to give initial chlorine concentrations of approximately 2.0, 4.0 and 8.0 mg L^{-1}. Deionized (DI), ultrapure water (Milli-Q® Direct 8/16, Millipore SAS, Mosheim, France) was dosed with the same initial chlorine concentrations to check the limitations of the methodology, including chlorine consumption by the glassware and volatilization losses. The flasks were then covered with parafilm and placed in a fume hood at room temperature. Free residual chlorine concentration was measured using a Hach chlorine test kit and Hach chorine colorimeter™ II at 10, 20, and 30 min and then at 1, 2, 4, 8, 24, 48, 72, 96, and 120 h after chlorination.

4.3.4. In-vitro trihalomethane formation in bulk cistern water

Because we were unable to acquire TTHMs information from our previous lab experiment due to instrument malfunction, we repeated the above chlorine decay experiment using three cistern water samples that were collected from the same First Nations community in July 2017. The pH, turbidity, DOC, UV254 and SUVA of the samples before chlorination and at the end of the experiment (after 120 h chlorination) were measured as described above. The experiment was laid out as a completely randomized block design with three initial chlorine levels (2.0, 4.0, and 8.0 mg L^{-1}) in a one-way treatment layout. Deionized water was included as a control and chlorinated with sodium hypochlorite (13% laboratory-grade) to the same initial chlorine levels. All treatments were replicated three times. Each experimental unit consisted of 1 L of water placed in a 2 L-Erlenmeyer flask. The flasks were then covered with parafilm and placed in a fume hood for the duration of the experiment. Water samples (10 ml) for TTHMs analysis were collected from each

unit at 1, 24, 48, 72, 96, and 120 h after chlorination and TTHMs were determined using the above methods.

4.3.5. Statistical analysis

Data were analyzed using SAS Version 9.4 (SAS Institute Inc. 2013). We tested three kinetic models commonly used in the literature to explain residual chlorine decay in drinking water. Generally, chlorine decay has been shown to follow first-order decay kinetics with respect to chlorine (Eq. 1) (Chambers et al. 1995; Powell et al. 2000a):

$$C = C_0 \ exp(-k_b t) \quad [1]$$

where C is the chlorine concentration (mg L^{-1}) at time t, C_0 is the initial chlorine concentration (mg L^{-1}), k_b is the bulk decay coefficient and t is the residence time in the pipe (d).

However, some studies demonstrated that other kinetic models better described chlorine bulk decay in a water distribution system (Brown et al. 2011; Powell et al. 2000b). One such model is the parallel first-order, which assumes two components to the reaction, each decaying according to a first-order law, but with different decay rate constants (Eq. 2):

$$C = C_0 z \ exp\left(-k_{bfast} t\right) + C_0(1 - z) exp(-k_{bslow} t) \quad [2]$$

where z is the ratio of fast to slow reactions (dimensionless), k_{bfast} is the bulk decay constant for fast reactions, and k_{bslow} is the bulk decay constant for slow reactions.

Another model is the limited first-order decay model, which assumes that a fraction of the initial chlorine concentration remains unchanged and only the remainder decays exponentially according to first-order kinetics (Eq. 3):

$$C = C^* + (C_0 + C^*)exp(-k_b t) \quad [3]$$

where C^* is the limiting chlorine concentration (mg L^{-1}).

The three kinetic models described above were compared for their fit to chlorine decay data using PROC NLIN in SAS. The best model for the data was selected based on the corrected Akaike information criterion. Model parameter estimates [free chlorine decay rate (k) and actual initial free chlorine concentration (C_0)] were considered significantly different between treatments if the 95% confidence intervals did not overlap. The time to reach a free chlorine concentration of 0.2 mg L^{-1} (Health Canada guidelines for safe drinking water) was calculated using the best-fitted model. Treatment effects on the time to 0.2 mg L^{-1} (which follows a gamma distribution) were then assessed using PROC GLIMMIX with DIST = gamma.

For the field study, analysis of variance (ANOVA) was conducted using PROC GLIMMIX to compare concentrations of individual THMs, TTHMs, and DOC, UV254, and SUVA on the day of cistern filling vs. measurements taken three days after cistern filling cisterns are refilled every four days by the water truck. Individual THMs and TTHMs followed a lognormal ($\log_e$) distribution, whereas DOC, UV254, and SUVA were normally distributed based on the Shapiro-Wilk test. Relationships between the bulk chlorine decay constant (k) and each of initial chlorine concentration, dissolved organic carbon, UV254, and SUVA were explored using simple linear regression analysis. For all statistical analyses, treatment effects were assessed at $\alpha = 0.05$.

4.4. Results and discussion

4.4.1. In-situ chlorine decay kinetics in belowground cistern water

Initial water quality attributes of the cistern water samples are presented in Table 4.1. Of the three kinetic models tested, the first-order kinetic model provided the best fit for free residual chlorine decay with a root mean square error of 0.13 mg L^{-1} and $p < 0.0001$:

$$C = 3.13e^{-1.28t} \quad [4]$$

where C is the free residual chlorine concentration (mg L^{-1}) at time t (d) and k is the first-order rate constant (d^{-1}).

Table 4.1. Selected initial chemical properties of belowground cistern water used in the field experiment.

	Initial free residual chlorine	Turbidity	DOC	UV254	SUVA	pH
	(mg L^{-1})	(NTU)	(mg L^{-1})	(m^{-1})	(L mg^{-1} m^{-1})	
Mean	0.75	0.77	3.4	0.016	0.005	7.85
SD [a]	0.34	0.15	0.2	0.010	0.003	0.14

[a] SD is the standard deviation of four replicates.

The first-order kinetic model is the most used kinetic model for in-situ chlorine decay in piped water distribution networks (Hua et al. 1999; Vasconcelos et al. 1997; Zhang et al. 2017). The three-parameter parallel first-order model is often fit well for chlorine decay data in piped water distribution systems as it considers both fast (k_{bfast}) and slower (k_{bslow}) reactions of chlorine (Al Heboos et al. 2017; Gang et al. 2003). However, in our in-situ chlorine decay study, all the fast reactions may have already been completed (just after chlorination at the water treatment plant) by

the time we collected samples from the cisterns.

In-situ chlorine decay rate in the cisterns, which included both bulk decay and wall decay, was 1.28 mg L^{-1} day^{-1}, indicating a high rate of chlorine decay in the cistern (Vasconcelos et al. 1997). Maintaining DOC below 1.8 mg L^{-1} is suggested to maintain an acceptable level of residual chlorine and reduce biofilm formation necessary for the biological stability of drinking water (Health Canada 2020b). The mean DOC concentration in the water samples was 3.4 mg L^{-1}. Thus, the high chlorine decay rate could be a result of DOC reacting with residual chlorine. Wall decay of chlorine could also play an important role because rough concrete surfaces are prone to develop biofilms (Characklis et al. 1990), and all cistern homes that were sampled had concrete belowground cisterns. Also, the microbial communities that have been previously observed in drinking water stored in concrete belowground cisterns indicate the presence of biofilms in cisterns (Farenhorst et al. 2017). Also, reaction of chlorine with sediments at the bottom of the cisterns and on the walls has been identified as a chlorine decay pathway (Artiola et al. 2012). Mass transfer of chlorine and other reactants between the bulk flow and the walls has been contributed to chlorine decay in pipe water distribution systems but not studied in cistern systems (Xu et al. 2018; Zhang et al. 2017). Studies have shown that the contribution of chlorine wall decay in piped distribution systems is greater than that of bulk decay reactions, especially in the presence of biofilms (Castro and Neves 2010; Xu et al. 2018).

There are limited water trucks in the First Nation reserve and households typically refill every four days. The half-life of free residual chlorine in the cisterns, calculated based on the first-order kinetic model, was 13 h. As such, based on the calculated model, in order to maintain a minimum of 0.2 mg L^{-1} free residual chlorine concentration in cistern water for at least 3 d (that is, until the

next re-filling), the minimum initial chlorine concentration in the cistern water should be 1.52 mg L^{-1}. This would be hard to achieve because we observed that free residual chlorine in water decreased immediately after filling the cisterns. This could result from free residual chlorine in truck-delivered water reacting with organic matter in the cisterns. Also, cisterns might have some leftover water with very low or zero free residual chlorine from the previous filling, which could dilute free residual chlorine in newly filled water. However, it is more common for First Nations households with cisterns to run out of water before a water truck is available to refill the cistern (Anderson 2020). Our study is the first to examine chlorine decay rates in cistern stored water at the household level. Further studies with a larger number of cisterns are needed to explore the processes behind the high chlorine decay rates observed in the present study and determine the prevalence of such high rates in other concrete cisterns and different types of cisterns such as polyethylene and fibreglass.

4.4.2. Trihalomethanes, DOC, UV254 and SUVA in belowground concrete cisterns

There was a 43% reduction in TTHMs during 3 d of water storage in the cisterns. However, the difference between 1 d and 3 d was not statistically significant due to the large variation in concentrations of TTHMs among cisterns, which resulted in low statistical power (Figure 4.1). Although not statistically significant, $CHCl_3$, $CHBrCl_2$, and $CHClBr_2$ concentrations numerically decreased on average by 50%, 37% and 36%, respectively, during the 3-d storage in cisterns. These results suggest that the free residual chlorine in the treated water delivered into cisterns by water trucks may readily react with organic matter in the cisterns as soon after refilling, resulting in the formation of TTHMs. The TTHMs may subsequently volatilize during storage as these compounds are moderately to extremely volatile, with vapour pressure at 25 °C ranging from 0.80 kPa for

$CHBr_3$ to 23.33 kPa for $CHCl_3$ (Brown et al. 2011; WHO, 2005). Levesque et al. (2006) reported a 43% reduction in TTHMs concentration after 2-day storage of chlorinated water in an uncovered pitcher in a refrigerator and attributed this to volatilization.

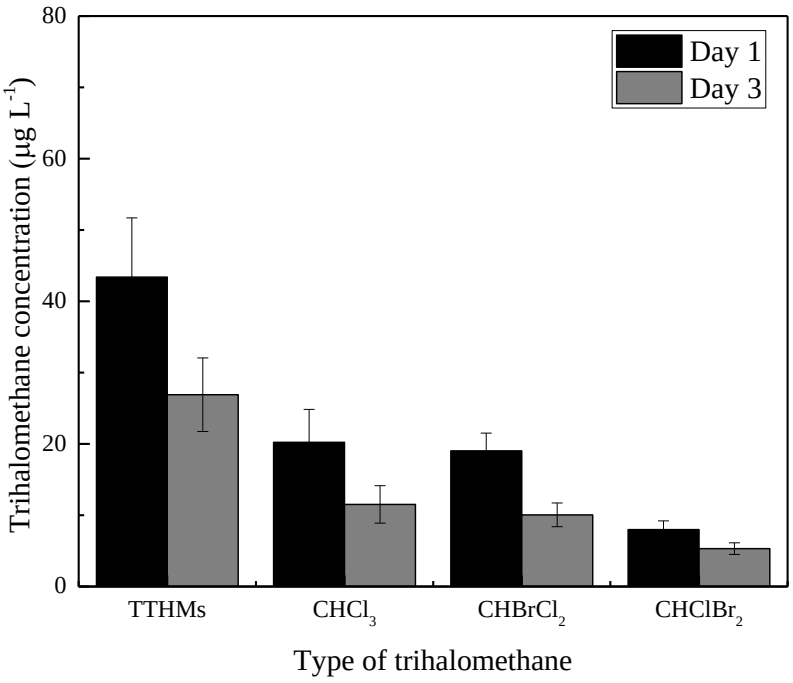

Figure 4.1. Changes in total (TTHMs) and individual trihalomethane concentrations in cistern stored water during three days of storage. Error bars represent standard errors of the mean.

The mean DOC concentration in the cistern water decreased by 25% over the 3-d storage period, although the reduction was not statistically significant due to the large variation and a small number of samples. This suggests that TTHMs were formed during at least the first two days of water storage in cisterns when there was enough residual chlorine to react with organic matter. However, mean UV254 and SUVA increased by 25% and 76%, respectively, indicating that the humic acid content of the water increased. This can be expected because the biofilms present in

the cisterns can continuously add organic matter, including humic material, into the water during storage (Rodriguez et al. 2003).

4.4.3. In-vitro bulk free residual chlorine decay kinetics in cistern water

Water quality attributes of cistern water collected from the First Nation community are presented in Table 4.2. Even though both the limited first-order model and the parallel first-order model also adequately described in-vitro bulk-free residual chlorine decay in the drinking water, based on the corrected Akaike information criterion, in-vitro bulk-free residual chlorine decay in re-chlorinated cistern water was best described by a first-order kinetic model (Table 4.3).

Table 4.2. Selected initial chemical properties of belowground cistern water used in the laboratory experiment

	DOC	UV254	SUVA	pH
	(mg L^{-1})	(m^{-1})	(L mg^{-1} m^{-1})	
Mean	2.3	0.012	0.005	7.79
SD [a]	0.4	0.005	0.002	0.06

[a] SD is the standard deviation of three replicates.

The first-order chlorine decay constant decreased significantly with increasing initial chlorine concentration (Table 4.3). There was a negative linear association between the initial chlorine concentration and k ($R^2 = 0.91$), with the relationship defined as:

$$k = \frac{0.366}{C_0} - 0.03 \quad [5]$$

Table 4.3. First-order kinetic model parameters for chlorine decay in chlorinated cistern water at different initial chlorine concentrations

Initial chlorine concentration	Model	*k* (95% *CI*)	RMSE (mg L^{-1})
2 mg L^{-1}	$C = 1.49e^{-0.15t}$	0.15 (0.13 – 0.16)a[a]	0.09
4 mg L^{-1}	$C = 3.11e^{-0.05t}$	0.05 (0.04 – 0.06)b	0.24
8 mg L^{-1}	$C = 5.95e^{-0.01t}$	0.01 (0.01 – 0.02)c	0.39

[a] Means in the same column followed by the same letter are not significantly different according to the Tukey multiple comparison procedure (α = 0.05).

Previous studies also found that the first-order decay constant was inversely related to the initial chlorine concentration (Brown et al. 2011; Powell et al. 2000a; Warton et al. 2006). When a low dose of chlorine is added to water, chlorine may react with easily reactable compounds resulting in rapid decay. However, when a higher dose of chlorine is added, chlorine may react both with fast and slow reactive organic compounds resulting in a slower overall decay (Hallam et al. 2003). A strong positive relationship was observed between initial DOC and *k* at the lowest initial chlorine level (2 mg L^{-1}) (R^2 = 0.99, *p* = 0.02) (Figure 4.2a). However, the significance of the relationship disappeared with increasing initial chlorine level, probably due to insufficient DOC to react with all the chlorine present at higher chlorine residual concentrations. Powell et al. (2000) also showed the dependence of *k* on DOC in a batch study using water from four different water treatment systems. In our study, the chlorine decay rate also increased with increasing initial UV254 of the water for 4 mg L^{-1} initial chlorine levels (R^2 = 0.99, *p* = 0.01) (Figure 4.2b). This is consistent with previous research, which showed a positive linear relationship between *k* and UV254 (Powell et al. 2000). The UV254 explains the humic fraction of the organic matter in water and it gives an

estimate of unsaturated bonds with which chlorine can react. However, the chlorine decay rate did not show a relationship with initial SUVA at any of the initial chlorine levels ($p > 0.05$). SUVA is considered a good indicator of humic content and aromatic carbon content of water (Matilainen and Sillanpää 2010). The SUVA has not yet been evaluated in-depth for its contribution to chlorine decay, even though its contribution to producing TTHMs has been widely studied (Hua et al. 2015; Matilainen and Sillanpää 2010; Rodriguez et al. 2003).

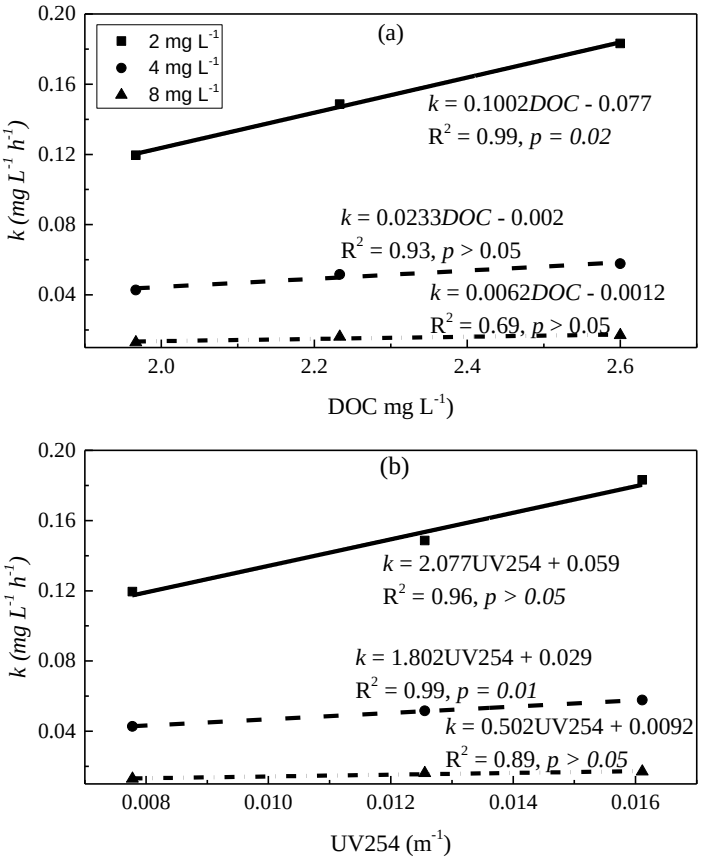

Figure 4.2. Relationships between in-vitro bulk chlorine decay constant (k) and (a) dissolved organic carbon (DOC) and (b) UV absorbance at 254 nm (UV254).

4.4.4. Trihalomethane formation in cistern water

After 30 min of chlorination, TTHMs were not significantly different among different initial chlorine concentrations (Figure 4.3). The highest concentrations for each trihalomethane was observed 24 h after chlorination. Concentrations of TTHMs at 4 and 8 mg L^{-1} initial chlorine levels were not significantly different and were steady until the end of the experiment. This could result from insufficient DOC to react with available chlorine in the water at high chlorination rates (4 and 8 mg L^{-1}). TTHMs and $CHCl_3$ concentrations decreased 2 d after chlorination at 2 mg L^{-1} initial chlorine level, indicating insufficient chlorine residual to react with available DOC in the water. At higher initial chlorine concentrations (4 and 8 mg L^{-1}), TTHMs were stable. However, it appears that most of the TTHMs production occurred during the first 20 h following chlorination (Figure 4.3). Concentrations of $CHBrCl_2$ and $CHClBr_2$ were not significantly different between any of the initial chlorine levels during the experiment (Figure 4.3). Previous research has shown that the formation of these compounds is mainly affected by bromide concentration in the water rather than the initial chlorine concentration or chlorine availability (Brown et al. 2011).

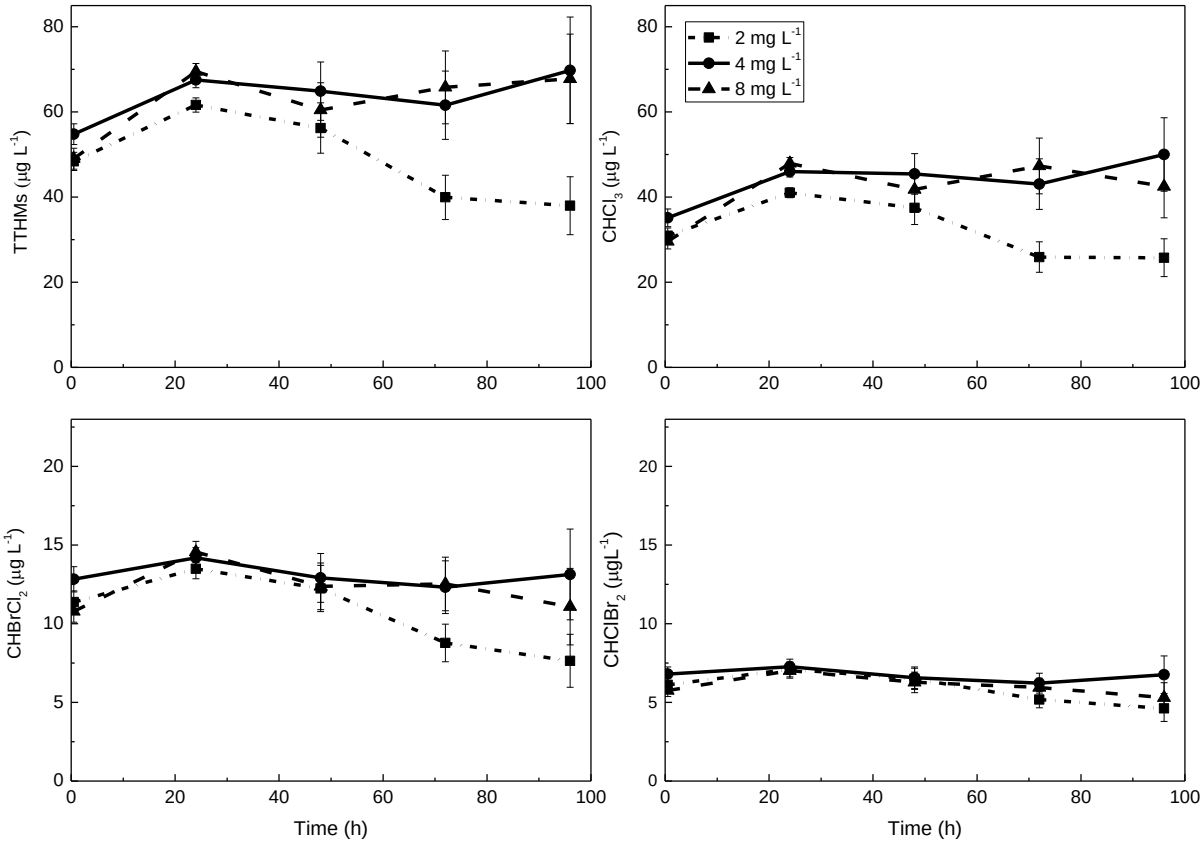

Figure 4.3. Temporal changes in total (TTHMs) and individual trihalomethane concentrations in cistern water at different initial chlorine levels.

4.4.5. Implications

This research has important implications on the use of belowground cisterns to store drinking water. The field study showed that in-situ chlorine decay rates are high, probably because of the contribution of biofilms in the cistern walls to chlorine wall decay. The lower bulk decay rate from the in-vitro lab study and the higher in-situ decay rate imply that the wall decay in belowground

cisterns plays a greater role in chlorine decay when compared to bulk chlorine decay. Faster chlorine decay may increase the risk of microbial re-growth and associated health risks such as gastrointestinal illnesses. Studies have shown large counts of *E. coli* and coliforms, which are indicators of fecal contamination of water, in cisterns in the presence of low free residual chlorine concentrations (Farenhorst et al. 2017). In a separate study of microbial quality variation in belowground cisterns in the community (Chapter 2), we also demonstrated that microbial quality of cistern water deteriorated with time due to chlorine disappearance. The addition of chlorine (booster chlorination) at critical points of the water distribution systems is usually practiced in large water distribution systems to maintain recommended free residual chlorine concentration and to prevent microbial growth (Islam et al. 2017). However, our field and laboratory studies demonstrated that added chlorine will react rapidly within the cistern. Our laboratory study demonstrated that the addition of chlorine can also increase TTHMs concentration in the water even though TTHMs decay over time. High concentrations of organic matter in belowground cisterns and biofilms on cistern walls can also contribute to TTHMs formation following chlorination (Montoya-Pachongo et al. 2018). Our results from a related study in the community showed TTHM concentrations exceeding Health Canada guidelines (100 µg L^{-1}) in cistern water samples (Chapter 3).

Removal of DOC at the water treatment plant and via regular cleaning of the cisterns are key to lowering DOC in the cisterns, maintaining adequate free residual chlorine levels in belowground cistern water. However, some belowground cisterns in the community appeared to be cracked and prone to contamination by dissolved and particulate organic matter from subsurface water. Therefore, it is critical to replace damaged belowground cisterns to ensure that sufficient levels of

free residual chlorine are maintained in the water.

Despite the limited sampling in our field study, our results provide important insights into the quality of water stored in belowground cisterns. There is a need for further studies employing larger number of cisterns and multiple communities comparing chlorine decay in both cistern and piped water. Also, evaluating chlorine decay in different cisterns such as aboveground polyethylene and fibreglass cisterns is important to understand the possibility of using different types of cisterns without compromising microbial quality water. Additionally, DOC in source water and, therefore in treated water varies over the year due to changes in temperature and rainfall and their effects on biological productivity (Hope et al. 1994). Therefore, evaluation of chlorine decay after onset of rainfall can also explain how external input of organic matter changes chlorine decay in belowground cisterns. However, the results suggest that belowground cisterns are unable to provide safe drinking water to the residents and therefore, they should be replaced by aboveground cisterns and homes need to be connected to the WTP by pipes when and wherever possible to ensure safe and equal water supply to people living in First Nations reserves.

4.5. Conclusion

Maintaining enough free residual chlorine in treated water is important to provide microbially safe drinking water to consumers. However, belowground cistern water tends to have considerably low free residual chlorine levels due to fast chlorine decay. In-situ decay of free chlorine in belowground cisterns in this study was best described by first-order kinetics. A high chlorine decay rate in belowground cistern water indicates a greater risk of microbial contamination during water storage in the cisterns. Regular cleaning and checking for structural damages of belowground

cisterns are required to ensure safe water for consumers. Replacing belowground cisterns with aboveground cisterns that can provide safe quality water or connecting to the WTP with pipes would be the best option to ensure a safe water supply to people living in First Nations communities.

4.6. Acknowledgements

This work was supported by the Natural Sciences and Engineering Research Council of Canada (NSERC) through its Collaborative Research and Training Experience Program (CREATE grant# 432009-2013) and Discovery Grant (DG) programs. Authors would like to acknowledge the cooperation and support of the Chief, Council, the Economic Development/Environmental Assistant Audrey Brass, Laura May and residence of the First Nation community. We thank Letie May-Lee and Mauli Gamhewage for their assistance in the laboratory and Ruidong Mi for help with sampling.

4.7. References

Al Heboos, S., Licskó, I., Heboos, S. Al, & Licskó, I. (2017). Application and comparison of two chlorine decay models for predicting bulk chlorine residuals. *Periodica Polytechnica Civil Engineering*, *61*, 7–13. https://doi.org/10.3311/PPci.9273

Anderson, K. 2021. Perceptions of household drinking water across a variety of water distribution systems in three First Nations in Manitoba. MSc thesis, University of Manitoba, Winnipeg, Manitoba, Canada.

APHA. (2012). *Standard Methods for the Examination of Water and Wastewater*. (R. Baird, A. Eaton, & E. Rice, Eds.) (22nd ed.). American Public Health Association; American Water

Works Association; Water Environment Federation.

Artiola, J., Rock, C., & Hix, G. (2012). Water storage tank disinfection, testing, and maintenance. The University of Arizona Cooperative Extension. College of Agriculture and Life Sciences, University of Arizona, Tucson, AZ, USA.

Baird, J. M., Summers, R., & Plummer, R. (2013). Cisterns and safe drinking water in Canada. *Canadian Water Resources Journal*, *38*, 121–134. https://doi.org/10.1080/07011784.2013.780790

Brown, D., Bridgeman, J., & West, J. R. (2011). Predicting chlorine decay and THM formation in water supply systems. *Reviews in Environmental Science and Bio-Technology*, *10*, 79–99. https://doi.org/10.1007/s11157-011-9229-8

Castro, P., & Neves, M. (2010). Chlorine decay in water distribution systems case study–Lousada network. *Electronic Journal of Environmental, Agricultural and Food Chemistry*, *2*, 261–266.

Chambers, V. K., Creasey, J. D., & Joy, J. S. (1995). Modeling free and total chlorine decay in potable water distribution systems. *Journal of Water Supply Research and Technology-Aqua*, *44*, 60–69.

Clark, R. M. (2011). Chlorine fate and transport in drinking water distribution systems: Results from experimental and modeling studies. *Frontiers of Earth Science*, *5*, 334–340. https://doi.org/10.1007/s11707-011-0194-x

Crittenden, J. C., Trussell, R. R., Hand, D. W., Howe, K. J., & Tchobanoglous, J. (2012). *MWH ' s Water Treatment Principles and Design* (3rd ed.). New Jersey: John Wiley & Sons, Inc.

Devarakonda, V., Moussa, N. A., VanBlaricum, V., Ginsberg, M., & Hock, V. (2010). Kinetics of Free Chlorine Decay in Water Distribution Networks. In *World Environmental and Water*

Resources Congress 2010 (pp. 4383–4392). Reston, VA: American Society of Civil Engineers. https://doi.org/10.1061/41114(371)446

EKOS Research Associates Inc. (2011). *Perceptions of Drinking Water Quality in First Nations Communities and General Population*. Ottawa, Ontario. https://www.ekospolitics.com/articles/015-11.pdf

Farenhorst, A., Li, R., Jahan, M., Tun, H. M., Mi, R., Amarakoon, I., et al. (2017). Bacteria in drinking water sources of a First Nation reserve in Canada. *Science of the Total Environment*, *575*, 813–819. https://doi.org/10.1016/j.scitotenv.2016.09.138

Gang, D., Clevenger, T. E., & Banerji, S. K. (2003). *Relationship of chlorine decay and THMs formation to NOM size*. *Journal of Hazardous Materials*, 96, 1–12.

Gopal, K., Tripathy, S. S., Bersillon, J. L., & Dubey, S. P. (2007). Chlorination byproducts, their toxicodynamics and removal from drinking water. *Journal of Hazardous Materials*, *140*, 1–6. https://doi.org/10.1016/j.jhazmat.2006.10.063

Graham, J. P., & Vanderslice, J. (2007). The effectiveness of large household water storage tanks for protecting the quality of drinking water. *Article in Journal of Water and Health*, *5*, 307–313. https://doi.org/10.2166/wh.2007.011b

Hallam, N. B., Hua, F., West, J. R., Forster, C. F., & Simms, J. (2003). Bulk Decay of Chlorine in Water Distribution Systems. *Journal of Water Resources Planning and Management*, *129*, 78–81. https://doi.org/10.1061/(asce)0733-9496(2003)129:1(78)

Health Canada. (2012). *Guidance for Designing, Installing, Maintaining and Decommissioning Drinking Water Cisterns in First Nations Communities South of 60°*. First Nations and Inuit Health Branch, Health Canada, Ottawa, Ontario.

Health Canada. (2019). *Guidelines for Canadian Drinking Water Quality Summary Table*. Water

and Air Quality Bureau, Healthy Environments and Consumer Safety Branch, Health Canada, Ottawa, Ontario.

Health Canada. (2020). *Guidance on Natural Organic Matter in Drinking Water*. Water and Air Quality Bureau, Healthy Environments and Consumer Safety Branch, Health Canada, Ottawa, Ontario. (Catalogue. www.canada.ca/en/health-canada/services/environmental-workplace-health/reports-. Accessed 5 December 2020

Hope, D., Billett, M. F., & Cresser, M. S. (1994). A review of the export of carbon in river water: fluxes and processes. *Environmental Pollution*, *84*, 301–324.

Hua, F., West, J. R., Barker, R. A., & Forster, C. F. (1999). Modelling of chlorine decay in municipal water supplies. *Water Research*, *33*, 2735–2746. https://doi.org/10.1016/S0043-1354(98)00519-3

Hua, G., Reckhow, D. A., & Abusallout, I. (2015). Correlation between SUVA and DBP formation during chlorination and chloramination of NOM fractions from different sources. *Chemosphere*, *130*, 82–89. https://doi.org/10.1016/j.chemosphere.2015.03.039

Islam, N., Rodriguez, M. J., Farahat, A., & Sadiq, R. (2017). Minimizing the impacts of contaminant intrusion in small water distribution networks through booster chlorination optimization. Stochastic Environmental Research and Risk Assessment, 31, 1759–1775. https://doi.org/10.1007/s00477-017-1440-x

Krasner, S. W. (2009). The formation and control of emerging disinfection by-products of health concern. *Philosophical Transactions of the Royal Society A: Mathematical, Physical and Engineering Sciences*, *367*, 4077–4095. https://doi.org/10.1098/rsta.2009.0108

Kumari, M., & Gupta, S. K. (2015). Modeling of trihalomethanes (THMs) in drinking water supplies: a case study of eastern part of India. *Environmental Science and Pollution Research*,

22, 12615–12623. https://doi.org/10.1007/s11356-015-4553-0

Levesque, S., Rodriguez, M. J., Serodes, J., Beaulieu, C., & Proulx, F. (2006). Effects of indoor drinking water handling on trihalomethanes and haloacetic acids. *Water Research*, *40*, 2921–2930. https://doi.org/10.1016/j.watres.2006.06.004

Li, X. F., & Mitch, W. A. (2018). Drinking water disinfection byproducts (DBPs) and human health effects: multidisciplinary challenges and opportunities. *Environmental Science and Technology*, *52*, 1681–1689. https://doi.org/10.1021/acs.est.7b05440

Mahler, B. J., Personne, J. C., Lods, G. F., & Drogue, C. (2000). Transport of free and particulate-associated bacteria in karst. *Journal of Hydrology*, *238*, 179–193. https://doi.org/10.1016/s0022-1694(00)00324-3

Matilainen, A., & Sillanpää, M. (2010). Removal of natural organic matter from drinking water by advanced oxidation processes. *Chemosphere*, *80*, 351–365. https://doi.org/10.1016/j.chemosphere.2010.04.067

Montoya-Pachongo, C., Douterelo, I., Noakes, C., Camargo-Valero, A., Sleigh, A., Escobar-Rivera, J.-C., & Torres-Lozada, P. (2018). Field assessment of bacterial communities and total trihalomethanes: Implications for drinking water networks. *Science of the Total Environment*, *616–617*, 345–354. https://doi.org/10.1016/j.scitotenv.2017.10.254

Neegan Burnside. (2011). *National Assessment of First Nations Water and Wastewater Systems;Manitoba Regional Roll-Up Report*: Department of Indian and Northern Affairs: Ottawa, ON, Canada.

Powell, J. C., Hallam, N. B., West, J. R., Forster, C. F., & Simms, J. (2000). Factors which control bulk chlorine decay rates. *Water Research*, *34*, 117–126. https://doi.org/10.1016/S0043-1354(99)00097-4

Powell, J. C., West, J. R., Hallam, N. B., Forster, C. F., & Simms, J. (2000). Performance of various kinetic models for chlorine decay. *Journal of Water Resources Planning and Management, 126*, 13–20.

Rodriguez, M. J., Vinette, Y., Sérodes, J.-B., & Bouchard, C. (2003). Trihalomethanes in drinking water of greater Québec region (Canada): Occurrence, variations and modelling. *Environmental Monitoring and Assessment, 89*, 69–93. https://doi.org/10.1023/A:1025811921502

Sadiq, R., & Rodriguez, M. J. (2004). Disinfection by-products (DBPs) in drinking water and predictive models for their occurrence: a review. *Science of the Total Environment, 321*, 21–46. https://doi.org/10.1016/j.scitotenv.2003.05.001

SAS Institute Inc. (2013). SAS/STAT User's Guide version 9.4. *Statistical Analysis System (SAS Institute Inc.)*. Cary, NC.

Vasconcelos, J., Rossman, L. A., Grayman, W. M., Boulos, P. F., & Clark, R. M. (1997). Kinetics of chlorine decay. *American Water Works Association*, *89*, 54–65.

Villanueva, C. M., Kogevinas, M., Cordier, S., Templeton, M. R., Vermeulen, R., Nuckols, J. R., et al. (2014). Assessing exposure and health consequences of chemicals in drinking water: Current state of knowledge and research needs. *Environmental Health Perspectives, 122*, 213–221. https://doi.org/10.1289/ehp.1206229

Warton, B., Heitz, A., Joll, C., & Kagi, R. (2006). A new method for calculation of the chlorine demand of natural and treated waters. *Water Research, 40*, 2877-2884. https://doi.org/10.1016/j.watres.2006.05.020

Xu, J., Huang, C., Shi, X., Dong, S., Yuan, B., & Nguyen, T. H. (2018). Role of drinking water biofilms on residual chlorine decay and trihalomethane formation: An experimental and

modeling study. *Science of the Total Environment*, *642*, 516–525. https://doi.org/10.1016/j.scitotenv.2018.05.363

Zhang, C., Li, C., Zheng, X., Zhao, J., He, G., & Zhang, T. (2017). Effect of pipe materials on chlorine decay, trihalomethanes formation, and bacterial communities in pilot-scale water distribution systems. *International Journal of Environmental Science and Technology*, *14*, 85–94. https://doi.org/10.1007/s13762-016-1104-2

Zhao, Y., Yang, Y. J., Shao, Y., Neal, J., & Zhang, T. (2018). The dependence of chlorine decay and DBP formation kinetics on pipe flow properties in drinking water distribution. *Water Research*, *141*, 32–45. https://doi.org/10.1016/j.watres.2018.04.048

5. OVERALL SYNTHESIS

5.1. Summary of research

The United Nations Human Rights Council specifies that the human right to water "entitles everyone to be sufficient, safe, acceptable, physically accessible and affordable water for personal and domestic uses". However, First Nations living on reserves in Canada are 90 times more likely to have no access to running water than other Canadians (Morrison et al. 2015). First Nations on reserves have a 26 times greater risk of contracting water-borne infections than the Canadian national average (Bradford et al. 2016).

The Province of Manitoba accounts for 13% of First Nations in Canada (Statistics Canada 2017), and approximately 59% of First Nations in Manitoba live on reserves (INAC 2014). Only 51% of homes on First Nations reserves in Manitoba have water piped directly to their households from a water treatment plant. About one-third (31%) of households have their drinking water delivered by trucks and stored in cisterns, with the remaining homes (18%) relying on well water or having no running water (Neegan Burnside 2011). First Nations communities use different types of cisterns, namely aboveground polyethylene cisterns located in insulated sheds or belowground concrete, polyethylene, or fibreglass cisterns. The type of cistern selected for a home is influenced by the amount of funding available for water infrastructure in the community and geographical factors such as surface topography, soil conditions including texture and stoniness, ground water levels, and risk to flooding (Health Canada 2012).

The overall objective of this research was to evaluate the quality of piped versus cistern water distribution systems in three First Nations communities. The three communities (A, B and C) have

comparable water treatment systems but are equipped with different types of cisterns: Community A has aboveground polyethylene cisterns, Community B has belowground concrete cisterns and Community C has belowground fibreglass cisterns. The research included monitoring the entire water distribution system, including samples obtained from source water (lake or river), collected from a tap in the WTP, as well as samples from treated water in the WTP, from the water delivery truck, and from homes served with piped and cistern water. In Communities A and C, samples from water leaving the WTP and the truck water were free from *E. coli* and total coliforms, indicating that the water treatment systems in these two communities effectively removed potentially pathogenic microbial contaminants from source water. In contrast, the treated water samples from the WTP and water truck in Community B were occasionally positive for *E. coli* and total coliforms, indicating that improvements are needed to ensure effective and safe water treatment and delivery processes.

Across all sampling times, none of the piped water samples but 30% of the cistern samples in Community A, 39% of piped water samples but 73% of cistern water samples in Community B, and 30% of piped water samples but 62% of cistern water samples in Community C tested positive for total coliforms. Monthly microbial water quality data collected from the communities and Health Canada also showed that averaged across the sampling period, the percentage detection of total coliforms was significantly greater in cistern water than in piped water in Communities B and C (Community A did not have enough cistern samples tested to do statistical analysis). Further, data obtained from Indigenous Services Canada for every Manitoba Tribal Council under the Access to Information Act indicated that the frequency of detection of coliform bacteria was substantially greater for cisterns than piped homes. Total coliforms were detected in cistern

samples throughout the year, except in winter months. In addition, *E. coli* was detected from April to May in Community B and in May, August, and September in Community C, coinciding with snowmelt flooding and heavy rainfall events observed. These results suggest that in each First Nations community, families living in homes with cisterns are at a greater risk of contracting water-borne infections than families in homes with piped water. Detections of *E. coli* and total coliforms were highest in Communities B and C, where belowground cisterns were used, whereas aboveground polyethylene cisterns that were being used in Community A had the potential to provide safer drinking water.

Free and total chlorine concentrations in cistern water were significantly lower than in piped water in all three communities. Across all sampling times, 38% of the piped water samples but 75% of the cistern samples in Community A, 48% of piped water samples but 88% of cistern water samples in Community B, and 70% of piped water samples but 100% of cistern water samples in Community C had free residual chlorine concentrations < 0.2 mg L^{-1}. This indicates a considerable chlorine loss during water distribution both in piped and cistern water in each community. Out of these low free residual chlorine (<0.2 mg L^{-1}) samples, none of the piped water samples and 33% of the cistern water samples in Community A, 73% of piped water samples and 78% of cistern water samples in Community B, and 43% of piped water samples and 63% of cistern water samples in Community C tested positive for total coliforms. This suggests that microbial regrowth was lower in the piped distribution system than cisterns, even under low free residual chlorine levels. Overall, Chapter 2 demonstrates that cistern storage reduces the microbial quality of stored water consistently across communities due to the increased presence of *E. coli* and total coliforms. The detection of *E. coli* and total coliforms was most frequent in homes that received water through

belowground concrete cisterns, followed by belowground fibreglass cisterns, whereas aboveground polyethylene cisterns were being used in Community A provided safer drinking water.

As discussed in Chapter 3, the TTHM concentrations in treated water at the WTP exceeded the Canadian Drinking Water Quality Guidelines (100 µg L^{-1}) in both sampling times in Community A and in May in Community B. Across all sampling times and communities, $CHCl_3$ was the dominant trihalomethane, ranging from 42 to 96% of TTHMs. For each sampling time in each community, concentrations of TTHMs in most samples were numerically greater (1 – 38%) in tap water from piped or cistern homes than in treated water at the WTP, suggesting a continued persistence and formation of TTHMs during water distribution. In all three communities, concentrations of TTHM were not significantly different between piped and cistern water samples. However, most of the times, TTHMs concentrations exceeded Health Canada guidelines. The TTHMs ranged from 96 to 207 µg L^{-1} in Community A homes, 45 to 160 µg L^{-1} in Community B and 57 to 122 µg L^{-1} in Community C. Across sampling times, concentrations of TTHMs exceeded the Health Canada limit of 100 µg L^{-1} in 100% of piped and 83% of cistern water samples in Community A, 61% of piped and 42% of cistern water samples in Community B and 30% of piped and 50% of cistern water samples in Community C. Results also showed a variation of TTHMs with sampling time due to changes in DOC, UV254 and SUVA in the source water where an increase in these attributes was associated with an increase in TTHM concentrations .

The research presented in Chapter 4 of this thesis investigated chlorine decay and TTHMs formation in belowground cisterns in Community B. This community was selected for further study because its cisterns indicated a poorer microbial water quality and higher DOC

concentrations than Communities A and C. In-situ chlorine decay of the belowground concrete cisterns was evaluated by measuring chlorine concentrations in the cistern water until below the detection limit (0.02 mg L^{-1}) and free residual chlorine concentration dropped 0.02 mg L^{-1} within three days. In-situ chlorine decay followed first-order kinetics with a decay constant (*k*) of 1.28 mg L^{-1} d^{-1}. Based on this decay rate, the half-life of free residual chlorine in the cisterns was 13 h, i.e., free chlorine levels would decrease by 50% every 13 h. In order to maintain 0.2 mg L^{-1} free chlorine, which Health Canada recommends to prevent microbial regrowth in the water distribution systems for at least 3 d (that is, until the next re-filling according to communications with the community members), a minimum of 1.52 mg L^{-1} free chlorine concentration would be required. This is not realistic to achieve, as well as that this chlorine level would be about twice as large as the average chlorine levels in the water distribution system that provides tap water for residents in the City of Winnipeg (City of Winnipeg 2020).

The in-vitro laboratory study showed that bulk chlorine decay in cistern stored water also followed first-order kinetics, but at a lower decay rate (0.01 to 0.15 mg L^{-1} h^{-1}). The lower in-vitro bulk decay rate and the higher in-situ decay rate imply that the wall decay in belowground cisterns plays a greater role in chlorine decay when compared to bulk chlorine decay. The rate constant of in-vitro bulk chlorine decay decreased with increasing initial chlorine level and increased with DOC, UV254 and SUVA. This emphasizes the need to maintain the lowest possible levels of DOC and related parameters to help maintain free residual chlorine in cistern water.

There was a reduction in TTHMs and individual trihalomethane concentrations in the belowground cisterns during the three days of the field study. This was likely due to volatilization during the storage. The laboratory study also showed that the highest TTHM concentrations occurred after

24 h of chlorination and remained stable for higher initial chlorine levels while decreasing for the lowest initial chlorine concentration (2 mg L^{-1}). This suggests that, at high chlorine levels, DOC can act as a limiting factor for TTHMs production, whereas chlorine limits TTHMs production at low initial chlorine levels.

5.2. Practical implications of the research

This research demonstrates that tap water quality is different in piped versus cistern homes in First Nations reserves, with the main differences being that tap water in piped homes has greater free residual chlorine concentrations and is less likely to be contaminated with coliform bacteria. These results point to the need to improve water distribution systems in the study locations, and likely in other First Nations reserves and rural municipalities that use cisterns to store drinking water. Results indicate that cistern storage, especially belowground concrete cisterns, reduces the quality of stored water consistently due to *E. coli* and total coliforms. The relatively fast residual chlorine decay observed (i.e., a decay half-life of 13 h) increases the risk of microbial regrowth. Chapters 2 and 3 showed that DOC in belowground cisterns is higher compared to piped water in Communities B and C. This could lead to faster decay of free residual chlorine and increases in TTHMs. Replacing damaged belowground cisterns with aboveground polyethylene cisterns would be a better alternative when there is no access to direct piped water. However, connecting homes to the mainline would be a more appropriate solution for ensuring consistent access to safe and sufficient drinking water. However, Indigenous Services Canada approves funding for connecting homes to the mainline versus cistern systems based on 20-yr life cycle cost and as given in *the Water and Wastewater Policy and Level of Services Standards* (AANDC 2011) “the most

economically feasible, physically appropriate system to meet the water and wastewater needs of the community in question shall be chosen". Also, the recommended lot frontage to be considered for a conventional high-pressure piped water system should be on average 30 m (AANDC 2011), however, most of the First Nations homes do not meet this requirement due to their traditional and cultural preferences (APTN National News 2021). For households using belowground cisterns, frequent cleaning and repairing damaged cisterns are important necessities to ensure safe drinking water for First Nations families. However, shortages in funding and human resources (with confined environment certification) are realistic challenges that interfere with First Nations communities' ability to improve on safer drinking water for households relying on cisterns. Resources to train personnel and enable community-led strategies to clean belowground cisterns through adequate funding are essential to ensure the safe use of cisterns to provide clean drinking water in homes. Additional in-house water treatment techniques such as filters and UV lights would help improve drinking water quality. However, these methods would not be economical given the low household income of First Nations communities and would not be able to provide adequate water given a large number of residents in a household.

Some of the water samples collected from WTP in Community B were positive for *E. coli* and total coliforms, emphasizing the need for additional water treatment processes. Higher chlorination levels according to the chlorine demand of the source water and advanced filtration techniques such as membrane filtration steps are some of the interventions that can be implemented to improve treated water quality.

High concentrations of TTHMs in water samples in all three communities also point to the need for the implementation of advanced water treatment steps. For example, membrane filtration

techniques such as nanofiltration have been proven to reduce TTHMs formation by removing most of the organic compounds responsible for TTHMs production (Goss 2011). Advanced oxidation using ozone or UV effectively reduce TTHMs concentration in treated water (Chin and Bérubé 2005; Méité et al. 2015). The use of other alternative disinfection agents, such as chloramines and chlorine dioxide, are other approaches to minimize TTHMs formation (Richardson et al. 2000). Notwithstanding the considerable improvement in operator certification, ongoing and specific training is required to keep pace with the advancement of the water treatment techniques, such as membrane filtration and advanced oxidation. Standardized wages, increased discussion between the operator and the community leadership, and sufficient technical and financial support from Indigenous Services Canada will enhance the retention and productivity of certified operators in producing and distributing safe drinking water. Indigenous Services Canada's 2017 – 2018 Departmental Results Report (Indigenous Services Canada 2018) mentioned that the percentage of primary operators certified to the level of the system they operated was 71% due to continuous investment in the Circuit Rider Training Program. However, 29% of operators are still in need of specific training, emphasising the need for continuous support with financial and human resources towards operator training to increase First Nations' capacity to effectively operate the water systems.

Monthly microbial data collected from the three communities were used to compare piped and cisterns and variation of microbial contamination during the year. Community A had sampled cistern water only over a few months and less frequently from April 2014 to March 2015 due to inadequate resources to analyze the recommended number of samples (personal communication with the Community Based Water Quality Monitor). Health Canada (2017) reports that only 80%

of public water distribution systems met recommended sampling frequency in 2016-2017, indicating that the less frequent sampling observed in Community A is not an isolated problem. This also highlights the need for further interventions, such as increased funding to meet recommended sampling frequency for bacteriological parameters (Health Canada 2017). Also, Health Canada recommends only quarterly sampling for cisterns, as well as that the water analysis of damaged or poor quality cisterns should not be conducted until the repairs are done, or the cisterns are cleaned and disinfected (Health Canada 2007). However, First Nations residents continue to use such cisterns mainly because they do not have other options. Therefore, such guidelines should be modified to capture the actual situation in the communities because it undermines the issues associated with the use of cisterns in First Nations communities.

5.3. Limitations of the study and recommendations for future studies

One limitation of the study was the limited number of piped and cistern samples collected during the sampling round in each community, thereby making it difficult to capture the variation in water quality as influenced by factors such as distance from the water treatment plant. We collected only 10 samples from each water distribution system per sampling time. We had to limit the number of samples due to the remote location of the communities, whereas the analysis of microbial quality is time-sensitive. Two drive-in communities were located 410 and 600 km from Winnipeg and one community was accessible only via a one-and-half hour-long flight and a boat ride. In the fly-in community, samples were transported by motorboat and air and consequently had a longer time window between sampling and laboratory analysis. The APHA/Standard Methods protocol requires that drinking water samples be analysed for *E. coli* and total coliforms within 30 h of

sampling. For practical reasons and to reduce bias, samples from all communities were analysed within 24-30 h after sampling. If logistics had been permitted, samples would have been analysed within 24 h (recommended for routine monitoring). It is also important for First Nations communities to have increased access to laboratories in communities, as this will empower communities to detect concerns more readily and in a more cost-effective manner.

Coliform bacteria, TTHMs and other operational parameters varied with sampling time. However, samples were collected only up to three times from two communities, with the timing dependent on the community being able to accommodate the university researchers. More frequent sampling within a short period of time and representing different seasons of the year would have provided a better understanding of the temporal variation of microbial quality, concentrations of TTHMs and chlorine decay observed for tap water in homes, and how changes influence such temporal variations in source water quality or flooding events due to snowmelt runoff. However, to address these limitations with a low number of samples and less frequent sampling, the communities agreed to share *E. coli* and total coliforms data obtained through other monitoring programs and captured a full year. These data are also presented in the thesis and microbial quality data (microbially satisfactory and unsatisfactory water samples) obtained under the Access to Information Act from Indigenous Services Canada. These additional data support the conclusion obtained through this experimental work that cistern water is more often contaminated with coliform bacteria than piped water. There appear to be temporal variations in microbial quality during a year, with contamination more likely in spring and summer months.

In this study, we analysed only indicator bacteria, *E. coli* and total coliforms because the First Nations communities regularly analyze indicator bacteria as per Health Canada's recommendation

(INAC 2010). Even though monitoring *E. coli* and total coliforms provide important information on potential contamination of water, this indicator does not provide detailed information on the microbial composition of the water (Harwood et al. 2005; USEPA 2016). This was the first research by our research group that involved multiple communities and examined different cisterns. Following up on the findings of this study, other members of the research group have since formulated studies to evaluate further pathogens and the molecular composition of piped and cistern water distribution systems in First Nations communities.

For this study, three types of cisterns were examined (aboveground polyethylene cisterns, belowground concrete cisterns and belowground fibreglass cisterns), but from each community, we could collect samples from only one type of cistern. If this study could have sampled different types of cisterns within the same community, the effect of cistern type on water quality could have been better evaluated. However, a given First Nations reserve likely has only one or two types of cisterns, so it might be challenging to find opportunities to provide for an experimental design that involves multiple cisterns comparisons within the same community. Another option would be to extend the study to more communities to increase the sample size, which was provided by the Tribal Council data in Chapter 2.

Even though this thesis provides substantial important information about drinking water quality issues, especially unequal access to safe drinking water within a community and between communities, there are still many gaps that need to be addressed to realize safe drinking water for households on First Nations reserves. The further development of additional or alternative water treatment technologies that work well for small communities is an important research area to ensure safe drinking water for First Nations reserves, and the protection and advancement of the

incorporation of traditional knowledge in leading this further development are paramount.

This study showed higher concentrations of TTHMs in treated water, which resulted from reactions between chlorine residuals and organic matter (Chapter 3). A previous study (Goss 2011) found that coagulation and clarification removed all the DOC; however, the hydrophobic acid and all hydrophilic compounds increased after the granular activated carbon filtration. Therefore, each step in the water treatment process should be evaluated to investigate the effectiveness or problems associated with each step of the water treatment process in DOC removal, which is necessary to decrease the formation of TTHMs. New technologies such as membrane filtration techniques can be introduced to improve organic matter removal depending on the results. Future studies should also examine alternative water treatment methods such as biological filtration, which is more effective in removing microbial and chemical contaminants by biological removal of compounds that can either be energy or nutrient sources for bacteria (Peterson et al. 2007).

One of the major limitations of the chlorine decay study in Community B was the limited number of cisterns sampled and tested. It is imperative to increase the sample number to perform sound statistical analyses. Sampling was started with 20 cisterns, but the sampling regime is intrusive by needing to repeatedly visit the same household at specific times during the day when family members have other commitments as required by their professional or personal responsibilities. No compensation was provided to obtain samples in homes, and this should be considered in future studies given the time commitment that is expected of families in homes that are to be sampled. Due to the unavailability of residents over the sampling period, a complete sampling regime was only accomplished for four homes with cisterns. Field-scale evaluation of chlorine decay and changes in the microbial community and chlorine decay in both cisterns and piped water might be

important to continue to alert the Federal Government of Canada that the current system of providing drinking water via belowground cisterns is inadequate.

Chlorine decay rates in cistern water were evaluated in a lab lab-scale batch study using four cistern water samples collected from Community B. Increasing the number of cisterns and comparing chlorine decay in both cistern and cisterns piped water are recommended. Also, evaluating chlorine decay in different cisterns such as aboveground polyethylene and fibreglass cisterns is important to understand the possibility of using different types of cisterns without compromising microbial quality water. For example, polyethylene cisterns that might result in a slower chlorine decay would benefit store water for a longer period compared to concrete cisterns with a higher chlorine decay. DOC in source water and in treated water varies over the year due to seasonal changes in temperature and rainfall and their effects on biological productivity (Hope et al. 1994). Therefore, seasonal changes of chlorine decay in belowground cisterns should also be evaluated. Event-based (after onset of rainfall or snowmelt event) evaluation of chlorine decay is also important. It can provide insights on how a potential sudden input of water (containing organic matter) seeping into damaged underground, can influence the rate of chlorine decay in cisterns.

Personal communications with members of the participating First Nations communities improved the understanding of potential contaminant sources impacting source and drinking water quality. This information will be helpful in further studies on source tracking for indicator and pathogenic microorganisms and DOC. Also, the contribution of seepage of subsurface contaminants and related factors such as environmental conditions (rainfall, temperature) and soil factors (soil type, stoniness, infiltration capacity) need to be carefully evaluated in future studies for a better understanding of the contribution of each factor on belowground cistern contamination with DOC

and microorganisms. This type of study will also help select the appropriate cistern type for the specific community depending on the environmental and soil factors. It is estimated that most free bacterial cells in bulk water are released from biofilms attached to the solid surfaces of the distribution system (Chan et al. 2019; Lechevallier et al. 1987), and 95% of bacteria in drinking water networks are in biofilms (Wingender and Flemming 2011). Therefore, biofilm development in cisterns and their effect on drinking water quality should be studied in detail, as is the need to identify the frequency of clearing required to prevent such biofilms from forming. Many First Nations communities in Manitoba are remote and hence the building of human and infrastructural resources within communities to conduct monitoring studies is essential to accomplish more frequent sampling regimes and test innovative solutions for reducing microbial contaminants and TTHMs in drinking water. The current study and proposed future studies will be beneficial to efforts aimed at providing safe drinking water to First Nations communities in Manitoba and Canada.

5.4. References

Aboriginal Affairs and Northern Development Canada (AANDC) (2011). *Water and Wastewater Policy and Level of Services Standards (Corporate Manual System), Protocols and guidelines for water systems.* https://www.sac-isc.gc.ca/eng/1312228309105/1533729544122. Accessed 01 November 5, 2021

Abokifa, A. A., Yang, Y. J., Lo, C. S., & Biswas, P. (2016). Investigating the role of biofilms in trihalomethane formation in water distribution systems with a multicomponent model. *Water Research*, *104*, 208–219. https://doi.org/10.1016/j.watres.2016.08.006

APTN National News. (2021). ‘It doesn’t make sense’: Feds pledged spend billions on First Nations water plants, but not on the pipes to carry fresh water to homes. APTNNews.ca. February 23, 2021 (online) https://www.aptnnews.ca/national-news/water-plants-first-nations-water-pipes/?fbclid=IwAR2gGYGsNVfBg4vk3vLslcBE1uxkSIYq0Er0kwro09xtscWfPV0QtXtkocY. Accessed October 28, 2021

Artiola, J., Rock, C., & Hix, G. (2012). Water storage tank disinfection, testing, and maintenance. The University of Arizona Cooperative Extension. College of Agriculture and Life Sciences, University of Arizona, Tucson, AZ, USA.

Baird, J. M., Summers, R., & Plummer, R. (2013). Cisterns and safe drinking water in Canada. *Canadian Water Resources Journal*, *38*, 121–134. https://doi.org/10.1080/07011784.2013.780790

Bradford, L. E. A., Bharadwaj, L. A., Okpalauwaekwe, U., & Waldner, C. L. (2016). Drinking water quality in indigenous communities in Canada and health outcomes: A scoping review. *International Journal of Circumpolar Health 75,* 32336. https://doi.org/10.3402/ijch.v75.32336

Brown, D., Bridgeman, J., & West, J. R. (2011). Predicting chlorine decay and THM formation in water supply systems. *Reviews in Environmental Science and Bio-Technology*, *10*, 79–99. https://doi.org/10.1007/s11157-011-9229-8

Chan, S., Pullerits, K., Keucken, A., Persson, K. M., Paul, C. J., & Rådström, P. (2019). Bacterial release from pipe biofilm in a full-scale drinking water distribution system. *npj Biofilms and Microbiomes*, *5*, 1–8. https://doi.org/10.1038/s41522-019-0082-9

Chin, A., & Bérubé, P. R. (2005). Removal of disinfection by-product precursors with ozone-UV advanced oxidation process. *Water Research*, *39*, 2136–2144. https://doi.org/10.1016/j.watres.2005.03.021

Farenhorst, A., Li, R., Jahan, M., Min, H., Mi, R., Amarakoon, I., et al. (2017). Bacteria in drinking water sources of a First Nation reserve in Canada. *Science of the Total Environment*, *575*, 813–819.

Goss, C, D. (2011). Characterization of natural organic matter and trihalomethane formation potential for NOM fractions isolated from two surface water sources in Manitoba. MSc thesis, University of Manitoba, Winnipeg, Manitoba, Canada.

Hallam, N. B., West, J. R., Forster, C. F., & Simms, J. (2001). The potential for biofilm growth in water distribution systems. *Water Research*, *35*, 4063–4071. https://doi.org/10.1016/S0043-1354(01)00248-2

Harwood, V. J., Levine, A. D., Scott, T. M., Chivukula, V., Lukasik, J., Farrah, S. R., & Rose, J. B. (2005). Validity of the Indicator Organism Paradigm for Pathogen Reduction in Reclaimed Water and Public Health Protection. *Applied and Environmental Microbiology*, *71*, 3163–3170. https://doi.org/10.1128/AEM.71.6.3163-3170.2005

Health Canada. (2007). *Procedure Manual for Safe Drinking Water in First Nations Communities South of 60 °*.

Health Canada. (2012). *Guidance for Designing, Installing, Maintaining and Decommissioning Drinking Water Cisterns in First Nations Communities South of 60°*. First Nations and Inuit Health Branch, Health Canada, Ottawa, Ontario.

Health Canada. (2017). Health Canada Departmental Resutls Report: Supplementary Information Tables 2016-17. https://www.canada.ca/en/health-

canada/corporate/transparency/corporate-management-reporting/departmental-performance-reports/2016-17-supplementary-information-tables.html. Accessed 22 September 2020

Hope, D., Billett, M. F., & Cresser, M. S. (1994). A review of the export of carbon in river water: fluxes and processes. *Environmental Pollution*, *84*, 301–324.

INAC. (2010). *Protocol for Centralised Drinking Water Systems in First Nations Communities Standards for Design, Construction, Operation, Maintenance, and Monitoring of Centralised Drinking Water Systems*. Gatineau, Quebec. http://www.ainc-inac.gc.ca/enr/wtr/index-eng.asp. Accessed 25 August 2020

INAC. (2014). First Nations in Manitoba. https://www.aadnc-aandc.gc.ca/eng/1100100020400/1100100020404. Accessed 29 September 2020

Indigenous Services Canada. (2018). *Indigenous Services Canada 2017 – 18 Departmental Results Report*. Ottawa, Ontario.

Lechevallier, M. W., Babcock, T. M., & Lee, R. G. (1987). *Examination and Characterization of Distribution System Biofilms*. *Applied and Environmental Microbiology* (Vol. 53).

Lechevallier, M. W., Welch, N. J., & Smith, D. B. (1996). Full-scale studies of factors related to coliform regrowth in drinking water. *Applied and Environmental Microbiology*, *62*, 2201–2211. https://doi.org/10.1128/aem.00794-10

Méité, L., Fotsing, M., & Barbeau, B. (2015). Efficacy of Ozone to Reduce Chlorinated Disinfection By-Products in Quebec (Canada) Drinking Water Facilities. *Ozone: Science & Engineering*, *37*, 294–305. https://doi.org/10.1080/01919512.2014.946592

Mi, R., Patidar, R., Farenhorst, A., Cai, Z., Sepehri, S., Khafipour, E., & Kumar, A. (2019). Detection of fecal bacteria and antibiotic resistance genes in drinking water collected from

three First Nations communities in Manitoba, Canada. *FEMS Microbiology Letters*, *366*, 67. https://doi.org/10.1093/femsle/fnz067

Morrison, A., Bradford, L., & Bharadwaj, L. (2015a). Quantifiable progress of the First Nations Water Management Strategy, 2001–2013: Ready for regulation? *Canadian Water Resources Journal*, *40*, 352–372. https://doi.org/10.1080/07011784.2015.1080124

Morrison, A., Bradford, L., & Bharadwaj, L. (2015b). Quantifiable progress of the First Nations Water Management Strategy, 2001–2013: Ready for regulation? *Canadian Water Resources Journal*, *40*, 352–372. https://doi.org/10.1080/07011784.2015.1080124

Murphy, H. M., Corston-Pine, E., Post, Y., & McBean, E. A. (2015). Insights and opportunities: Challenges of Canadian First Nations drinking water operators. *International Indigenous Policy Journal*, *6*. https://doi.org/10.18584/iipj.2015.6.3.7

Patrick, R. J. (2011). Uneven access to safe drinking water for First Nations in Canada: Connecting health and place through source water protection. *Health & Place*, *17*, 386–389. https://doi.org/10.1016/j.healthplace.2010.10.005

Richardson, S. D., Thruston, A. D., Caughran, T. V., Chen, P. H., Collette, T. W., Schenck, K. M., et al. (2000). Identification of new drinking water disinfection by-products from ozone, chlorine dioxide, chloramine, and chlorine. *Water, Air, and Soil Pollution*, *123*, 95–102.

Statistics Canada. (2017). *Aboriginal peoples in Canada: Key results from the 2016 Census*. *2016 Census*. https://doi.org/Catalogue no. 11-001-X

USEPA. (2016). *Six-Year Review 3 Technical Support Document for Microbial Contaminant Regulations*. Office of Water, Washington, DC.

Wingender, J., & Flemming, H. C. (2011). Biofilms in drinking water and their role as reservoir for pathogens. *International Journal of Hygiene and Environmental Health*, *214*, 417–423. https://doi.org/10.1016/j.ijheh.2011.05.009

6. APPENDICES

Supplementary Information: Chapter 3

Appendix 3.A. Reagents used for trihalomethane extraction and analysis

Pentane (99.9%), which was purchased from Fisher Scientific (Millipore, Bedford, MA, USA) was used as the extraction solvent. Analytical grade anhydrous sodium sulfate (Na_2SO_4) was purchased from Fisher Sc. (St. Louis, MO, USA). Chloroform (CF) (98.6%), Bromoform (BF) (97.9%), Dibromochloromethane (DBCM) (98.7%), and Bromodichloromethane (BDCM) (99%) were purchased from Sigma-Aldrich (St. Louis, MO, USA) as neat standards. Dibromomethane which was used as internal standards in the analysis of THMs, was obtained from Sigma Aldrich (St. Louis, MO, USA). HPLC grade water which was purchased from Fisher Scientific, was used for standard preparation (Millipore, Bedford, MA, USA).

Appendix 3.B: Table 1. Total trihalomethanes (TTHMs), $CHCl_3$, $CHBrCl_2$, $CHClBr_2$, and $CHBr_3$ concentrations of treated water in water treatment plant (WTP) and Truck water in each community. Standard deviations are given in the parentheses.

Community	**Location**	Sampling Date	THMs	$CHCl_3$	$CHBrCl_2$	$CHClBr_2$	$CHBr_3$
			$\mu g\ L^{-1}$				
Community A	WTP	02/15	187 (7.27)	147 (6.39)	35.1 (0.85)	3.5 (0.02)	0.87 (0.01)
		10/15	109 (3.73)	73.1 (10.4)	27.4 (2.09)	4.1 (0.24)	4.27 (0.12)
	Truck	02/15	149 (7.27)	116 (0.66)	28.3 (0.39)	3.2 (0.02)	0.97 (0.03)
		10/15	108 (12.8)	72.1 (2.35)	27.1 (1.35)	4.2 (0.24)	4.26 (0.01)
Community B	WTP	05/15	115 (7.12)	64.0 (5.32)	36.6 (1.54)	12.0 (0.43)	1.98 (0.03)
		10/15	71.8 (2.94)	29.2 (1.02)	25.6 (1.09)	12.1 (0.77)	4.86 (0.15)
	Truck	05/15	102 (4.02)	57.3 (2.71)	32.4 (1.21)	10.9 (0.39)	1.90 (0.12)
		10/15	67.0 (0.86)	26.1 (0.53)	24.6 (0.43)	11.5 (0.21)	4.76 (0.11)
Community C	WTP	06/15	81.1 (2.32)	77.9 (2.36)	2.52 (0.03)	0.7 (0.08)	0.00 (0.00)
	Truck	06/15	54.2 (8.90)	51.8 (8.69)	1.82 (0.22)	0.6 (0.00)	0.00 (0.00)

Appendix 3.B; Table 2. Dissolved organic carbon (DOC), UV absorbance at 254 nm (UV254) and specific UV absorbance (SUVA) of piped and cistern water in each community.

		Community A				Community B				Community C			
Effect		DOC	UV254	SUVA	pH	DOC	UV254	SUVA	pH	DOC	UV254	SUVA	pH
		mg L^{-1}	m^{-1}	Lmg^{-1} m^{-1}		mg L^{-1}	m^{-1}	L mg^{-1} m^{-1}		mg L^{-1}	m^{-1}	L mg^{-1} m^{-1}	
Distribution system	Piped	13.4	0.114	0.613a	7.81a	8.13b [a]	0.011b	0.123b	7.93	9.29b	0.020a	0.208a	7.20b
	Cistern	13.5	0.100	0.676a	7.95a	8.86a	0.024a	0.266a	7.96	10.3a	0.021a	0.221a	7.71a
Sampling time	1	17.1	0.207	1.217a	7.96a	10.5a	0.023a	0.220a	8.07				
	2	9.7	0.007	0.073b	7.80a	6.48b	0.012a	0.169a	7.83				
Distribution system x Sampling time	Piped x 1	17.6a	0.220a	1.271	7.92	10.2	0.017	0.163	8.15a				
	Piped x 2	9.16b	0.007c	0.082	7.69	6.67	0.005	0.083	7.72c				
	Cistern x 1	16.7a	0.194b	1.163	7.99	10.8	0.030	0.277	7.98ab				
	Cistern x 2	10.3b	0.006c	0.064	7.92	7.68	0.019	0.255	7.94b				
		p-value											
Distribution system		0.76	<0.001	0.11	0.08	0.04	0.03	0.02	0.53	0.002	0.73	0.59	<0.001
Sampling time		<0.001	<0.001	<0.001	0.06	<0.001	0.07	0.39	<0.001				
Distribution system x Sampling time		0.01	<0.001	0.25	0.35	0.58	0.96	0.62	0.0003				

a Means in the same column followed by the same letter are not significantly different according to the Tukey-Kramer test ($\alpha = 0.05$). Mean separation for main effects is given when the interaction is not significant.

b In community A sampling time 1 = Feb 2015; 2 = Oct 2015; In community B, sampling time 1 =May 2015; 2 = Oct 2015; in community C only one sampling time June 2015.