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Microbial risk assessment and mitigation options for wastewater treatment in Arctic Canada

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ABSTRACT

Populations in Arctic Canada are strongly connected to, and draw sustenance from, the physical environment. Recreation and food harvesting locations, however, may be impacted by the basic wastewater treatment and disposal processes used in the region. Within these mixed socio-ecological systems, people may unknowingly be exposed to wastewater pathogens, either by direct contact or indirectly through activities resulting in exposure to contaminated locally harvested food. The objectives of this research are to estimate microbial health risks attributable to wastewater effluent exposure in Arctic Canada and evaluate potential mitigation options. A participatory quantitative microbial risk assessment (QMRA) approach was used. Specifically, community knowledge and information describing human activity patterns in wastewater-impacted environments was used with microbial water quality data to model a range of exposure scenarios and risk mitigation options. In several exposure scenario results, estimated individual annual risk of acute gastrointestinal illness exceeds a proposed tolerable target of 10^{-3}. These scenarios include shore recreation and consumption of shellfish harvested near primary mechanical treatment plants at low tide, as well as travel in wetland portions of passive treatment sites during spring freshet. These results suggest that wastewater effluent exposures may be contributing to gastrointestinal illness in some Arctic communities. Mitigation strategies, including improved treatment and interventions aimed at deterring access to disposal areas reduce risk estimates across scenarios to varying degrees. Overall, well-designed passive systems appear to be the most effective wastewater treatment option for Arctic Canada in terms of limiting and managing associated microbial health risks. This research demonstrates a novel application of QMRA and provides science-based evidence to support public health, water, and sanitation decisions and investment in Arctic regions.

1. Introduction

Across Arctic Canada, traditionally semi-nomadic Indigenous populations balance food harvesting and recreational customs with the requisite sanitation and disease prevention measures of life in permanent settlements. Given the extreme temperatures and high infrastructure costs in the region, many conventional wastewater treatment options are not feasible (Johnson et al., 2014). Most Arctic communities utilize decentralized passive systems consisting of wastewater stabilization ponds (WSP) and adjoining wetlands, with a few operating mechanical treatment facilities that discharge directly to marine or fresh waters. A limitation of both types of systems, as they are currently designed and operated, is their minimal pathogen removal capabilities (Huang et al., 2018). Consequently, partially treated effluent containing microbial pathogens of risk to human health is released into the receiving environment (Huang et al., 2018; Krumhansl et al., 2015). Many of these pathogens can lead to acute gastrointestinal illness (AGI), including diarrhea and vomiting, as well as other diseases following

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exposure to even very low doses (Leclerc et al., 2002). Indigenous and non-Indigenous residents of Arctic communities maintain strong ties to their natural surroundings as a source of food, identity, and livelihood (Bjerregaard et al., 2004; Cunsolo Willox et al. 2012). Given the proximity of wastewater treatment sites to communities, effluent may be released to areas used and valued by the local population inadvertently causing adverse health impacts. For instance, people may unknowingly be exposed to wastewater pathogens while fishing, hunting, harvesting food or while engaged in other recreational and occupational activities (Donaldson et al., 2010; Nilsson et al., 2013).

Estimates of AGI incidence in Arctic Canada range up to six times greater than the national average (Harper et al., 2015a; Thomas et al., 2013) and above rates in many less industrialized countries (Harper et al., 2015a; Mathers et al., 2002; WHO 2006a). The specific role water plays in AGI transmission is unclear. Numerous environmental and behavioural risk factors have been explored (Harper et al., 2015a; Masina et al., 2019; Mosites et al., 2018; Wright et al., 2018); however, as of yet, there is limited evidence of any specific associations with AGI (Goldfarb et al., 2013; Iqbal et al., 2015). As research on AGI in the Arctic continues (Hastings et al., 2014; Thivierge et al., 2016), the potential link with wastewater contamination (Daley et al., 2015) remains a concern among regional health authorities and communities, which are often limited in terms of financial, technical, and infrastructural resources (Hennessy and Bressler 2016; Pardhan-Ali et al., 2013).

The remoteness of the Arctic region often constrains extensive epidemiological, microbiological, and field-based studies of environmental health risks; thus, comprehensive datasets on local pollution sources are limited. Furthermore, the potential for quantifying exposures in this context are difficult as human behaviours leading to contact with contaminants and risk of disease are also shaped by cultural, economic, and social factors (Brown et al., 2011). Therefore, standard literature-based values pertaining to exposure frequencies and magnitudes may not be directly generalizable to Indigenous populations (Barber and Jackson 2015; Knibbs and Sly 2014) and Arctic communities (Suk et al., 2004).

Quantitative microbial risk assessment (QMRA) is an approach employed to characterize health risks attributable to a microbial hazard. The disease burden can be estimated based on stochastic models and the concentration and distribution of indicator organisms when direct measurements of pathogens at points of exposure are not available or possible (WHO 2016). QMRA designs are flexible and have been adapted for use in data-limited settings within less industrialized global regions (Ferrer et al., 2012; Howard et al., 2006; Hunter et al., 2009). Additionally, this type of risk assessment has been previously applied in situations where inadvertent exposure to wastewater effluent may have occurred through food harvesting and recreation (Fuhrimann et al., 2017; Fuhrimann et al., 2016; Henao-Herreño et al., 2017; Yapo et al., 2014). Innovatively combining participatory research methods with traditional risk assessment frameworks is also increasing as a means of improving understanding of human interactions with contaminated areas (Ramirez-Andreotta et al., 2014). Engaging with the communities affected can lead to exposure models and risk management strategies that are more reflective of the population's social and cultural practices (Nguyen-Viet et al., 2009).

The results of a QMRA can be compared to protective health-based targets. Currently in Arctic Canada, pollutant-based effluent quality standards are the predominant measure used to determine and manage the risk posed to human health by wastewater discharges (CCME 2009). Health-based targets offer a more directly comparable measure by establishing a tolerable level of additional disease burden attributable to a given exposure (Rose and Gerba 1991). Currently, however, no such targets have been established for Arctic Canadian regions. For context, the World Health Organization (WHO) tolerable risk level for water related infectious disease for drinking water as well as wastewater use in agriculture is 10^{-4} (WHO 2006a; refer to Table 2 in Mara 2008 for conversion between disability adjusted life years and tolerable risk). Governments can choose to adopt, or adapt-and-adopt, this guideline based on the state of knowledge concerning waterborne disease in their jurisdiction as well as social and economic conditions. Dependent on the local situation, a less stringent tolerable risk of illness target of 10^{-3} or 10^{-2} may be more appropriate in combination with regular monitoring and incremental improvement efforts (Mara 2008; WHO 2006a). QMRA can also be used to evaluate the potential impact of such efforts on risk reduction (WHO 2016). Types of mitigation include engineering controls and designs to improve treated water quality (Machdar et al., 2013; Weir et al., 2011) or behavioural interventions intended to limit human contact to contaminated environments (Katukiza et al., 2014; Labite et al., 2010).

Using a QMRA approach, the objectives of this research were to: 1) characterize the exposure pathways and risk of illness (AGI, specifically) associated with wastewater effluent in Arctic Canada and; 2) to identify and evaluate interventions that may be effective in reducing health risk. The guiding purposes of the research were to provide findings that serve as an initial evidence base on this issue and to offer an adaptable model that can be used further as a decision-making tool by stakeholders in the region.

2. Materials and methods

2.1. Research approach

This research was guided by an ecosystems approach to health (ecohealth). Ecohealth research attempts to address complex issues occurring at the intersection of environment, society, and human health, emphasizing core principles such as systems thinking, stakeholder participation, and knowledge-to-action (Charron 2012; Forget and Lebel 2001).

This research was based in the Territory of Nunavut, a region of Inuit Nunangat (the Inuit home land, water, and ice of Canada) and builds on an existing wastewater research relationship between the academic-based authors and the Government of Nunavut. In accordance with Inuit research priorities (ITK 2018; Tri-Council 2018), territorial government organizations and community-level stakeholders were engaged and included throughout the research process, with an end goal of producing results that translate into practical health improvements.

The research design was a form of participatory risk assessment, wherein a QMRA model was applied in an Arctic community setting. The conventional assessment framework and data sourcing methods were tailored to include local perspectives and experiences in effort to link Inuit knowledge (Inuit Qaujimajatuqangit) and scientific understanding of water, sanitation, and human health.

2.2. Model overview

The model builds upon a conceptual framework (Daley et al., 2018a) and an initial screening-level, point-estimate assessment of risk in case study sites (Daley et al., 2019). Specifically, an inferential QMRA model – rather than community-specific – was designed to reflect hypothetical Arctic wastewater treatment systems, receiving environment conditions, and exposure pathways. Exposure scenarios were parameterized with probability distributions whenever possible. In instances where there was insufficient data to generate a distribution, point estimates were used. The input parameter values used in the model were sourced from water quality data, community knowledge, and peer-reviewed literature. The results represent probability distributions of annual AGI risk to individuals who partake in each activity. Base cases, which simulate current conditions, were assessed first. Risk mitigating interventions were then formulated and evaluated. In the absence of an established health-based target for wastewater discharges in Arctic Canadian communities, results are benchmarked against a tolerable waterborne risk level derived from WHO guidelines for safe wastewater management (Mara et al. 2008; WHO 2006a). The inputs and equations involved in each of the four stages of QMRA (hazard identification, exposure assessment, dose-response, and risk characterization) are described in the following subsections.

2.3. Hazard identification

The hazard source is partially-treated domestic wastewater effluent. The passive treatment systems in use in Arctic Canada vary greatly from site to site in terms of initial design, current condition, and operational management (Ragush et al., 2015; Schmidt et al., 2016). Subject to natural conditions, the effluent largely remains frozen within a WSP during the subzero (°C) period of the year, which is from approximately October to May in most of the region. During the warmer months effluent either continuously seeps from the WSP into a wetland, or alternatively if the holding cell is structurally sound, is detained within the WSP until being manually decanted using a pump. Upon release, the effluent flows through the wetland and into a receiving water body. In arctic conditions, these passive systems have typically been shown to provide a primary level of treatment (Balch et al., 2018; Hayward et al., 2014; Yates et al., 2012) and do not reliably remove human pathogens (Huang et al., 2018).

A few Arctic communities use mechanical wastewater treatment processes such as filters or aerobic treatment units. The systems are capable of providing secondary treatment under optimal conditions, though most achieve only preliminary or primary levels of treatment (Johnson et al., 2014). These systems continuously discharge effluent directly from an enclosed facility into receiving water environments. Mechanical treatment systems are less subjective to natural environmental processes than passive systems (Bitton 2005); however, the application of mechanical wastewater treatment in the Arctic has proven challenging in other regards. Mechanical systems require significantly more financial investment, energy, daily operation, maintenance, and technical expertise. These factors in combination with the extreme temperatures and remoteness of the region have resulted in extended periods of compromised treatment in some communities (Johnson et al., 2014). None of the current mechanical wastewater treatment systems being used in Arctic Canada have a disinfection process.

Six pathogenic agents that are routinely present in nondisinfected effluent and transmissible via incidental ingestion of contaminated water or food were included in the assessment. These included three bacteria (pathogenic *Escherichia coli, Salmonella* spp., and *Campylobacter* spp.), one virus (rotavirus), and two protozoa (*Giardia* spp. and *Cryptosporidium* spp.). The selections were based on microorganisms detected in Arctic wastewater treatment systems by Huang et al. (2018), as well as a review of important pathogenic infections in the region. Huang et al. (2018) demonstrated that pathogenic *E. coli* and *Salmonella* spp. were present in treated wastewater discharged into the receiving environment. These authors did not detect *Campylobacter* spp. within either of the two sites they studied. Nevertheless, *Campylobacter* spp., along with *Salmonella* spp. and *Giardia* spp., was included in the QMRA due to their significance as sources of AGI in the region (Pardhan-Ali et al., 2012b; Goldfarb et al., 2013). Manore et al. (2020) also detected accumulated *Giardia* in some samples of shellfish tissue in Iqaluit, Nunavut. *Cryptosporidium* spp. was also included based on a recent emergence of infections (Thivierge et al., 2016). Finally, rotavirus was included in the model based on its global significance as a pathogen affecting children and as a reported source of AGI in Arctic Canada (Desai et al., 2017; Gurwith et al., 1983). Worth noting, norovirus was also considered for inclusion given that it is similarly a leading viral cause of AGI worldwide (Ahmed et al., 2014). In comparison to rotavirus, however, limited evidence of norovirus relating to AGI in Arctic Canada could be located. For simplification purposes within the assessment, the pathogenic strains of each agent that are associated with AGI are implied.

2.4. Exposure assessment

The concentrations of specific pathogenic agents within effluent-impacted environment at points of human exposure were estimated using an indirect method. The process is described in the subsequent paragraphs and a list of the corresponding QMRA model distributions, parameters, and references is presented in Table 1. (Additional detail is provided in the Supplementary Materials.)

To begin, a dataset of indicator *E. coli* concentrations (a common fecal indicator organism) in raw influent, treated effluent, and water from the immediate receiving environments in five Arctic sites was sourced. Additional details on the data that was used, and the associated references, are provided in Table 1 of the Supplementary Material. In brief, the dataset includes two sites operating mechanical systems (Iqaluit and Pangnirtung, Nunavut), and three using passive systems (Naujaat, Pond Inlet, and Sanikiluaq, Nunavut). In sites operating mechanical systems, where effluent is continuously discharged directly to marine waters, sampling took place during both high and low tide cycles to account for the noted impact of water exchange on contaminant concentration in tidal receiving environments (Gunnarsdóttir et al., 2013). At sites where effluent was discharged from a stabilization pond to a wetland, sampling was scheduled during spring freshet (i.e. spring thaw) in June and late summer in September to capture the high variability that occurs over the span of the passive treatment season (Hayward et al., 2014; Yates et al., 2012).

Indicator *E. coli* analysis was conducted on the samples either using the Colilert-18 method and Quanti-Tray/2000 system in accordance with manufacturer's instructions (IDEXX Laboratories Inc. 2013) or via standard methods at the Maxxam Analytics commercial laboratory in Montréal, Quebec (APHA 2012). Concentration results were in the form of the most probable number of *E. coli* in 100 mL (MPN/100 mL). For full descriptions of the sampling and analysis methods, refer to Greenwood (2016), Hayward et al. (2018), and Neudorf et al. (2017). Field data were supplemented with literature values (Westrell et al. 2004). Using this information base, probability distributions were fitted to parameter ranges to characterize the indicator *E. coli* concentrations at initial release (C_0). Additionally, input from municipal employees and review of operational records was used to explain periods of high concentrations and estimate the frequency and duration of reduced or failed treatment periods (City of Iqaluit 2015; Johnson et al., 2014).

Most human exposures to wastewater effluent are likely to occur at locations beyond the initial release points and immediate mixing zones where sampling occurred, as these areas are commonly recognized among community members as being heavily contaminated (Daley et al., 2015). Therefore, indicator *E. coli* concentrations beyond that range, at distances where exposures are more likely to occur, were estimated using a first-order kinetic model. This model is widely applied to characterize microbial inactivation or decay within environmental media (Haas et al., 2014; Stetler et al., 1992). The natural logarithms of

Table 1
Quantitative microbial risk assessment model parameters, distributions, and assumptions used to estimate pathogen concentrations in wastewater effluent-impacted environments in Arctic Canada.

Description	Units	Distribution and values
Concentration of indicator *E. coli* at effluent release (C_0)[a]		
Mechanical	MPN/100 mL	Pareto (1×10^4; 0.48)[b]
Passive	MPN/100 mL	Uniform (1×10^5; 1×10^6)[c]
Reduction rate coefficient (k)[a]		
Mechanical: low tide	1/m	Point estimate (−0.0048)
Mechanical: high tide	1/m	Point estimate (−0.0357)
Passive: spring	1/m	Point estimate (−0.0090)
Passive: summer	1/m	Point estimate (−0.0198)

[a] Refer to Supplementary Materials for more information.
[b] Pareto distribution (location; shape).
[c] Uniform distribution (minimum; maximum).

the observed *E. coli* concentrations in the dataset were first plotted and linearly regressed against distance from the effluent release points for each of the five sites under varying tidal or seasonal conditions. Next, first-order concentration reduction constants (k, m^{-1}) were derived from each slope line. From among the calculated reduction constants, the modelling coefficients that were most representative of typical systems and conditions found across the Arctic were chosen. The coefficients were then used as reduction rate constants (k) in a first-order model (Eq. (1)) to predict *E. coli* concentrations (C_{dist}) as a function of initial concentration at effluent release points (C_0) and distance ($dist$), under similar base case conditions.

$$C_{dist} = C_0 * e^{-k(dist)} \tag{1}$$

All behavioural elements of the exposure scenarios included in the QMRA model were grounded in community-based information. Localized knowledge and descriptions of human-environment interactions formed the primary data source. These data were supplemented with literature based exposure values. Corrective factors were assumed in some instances to adapt standard exposure magnitude and frequency values to the local context and population, as has been practiced in other QMRA models (Barker et al., 2014; Fuhrimann et al., 2016). The local data were collected using participatory epidemiology techniques (Barber and Jackson, 2015; Leung et al., 2004; O'Fallon and Dearry 2002) in the five aforementioned Nunavut communities. Between 2013 and 2016, a total of 42 interviews were held with key informants, which included wastewater operators, public health staff, wildlife conservation officers, and subsistence hunters, fishers, and harvesters. The interviews included site-mapping exercises and questionnaires designed to gather information regarding activity patterns, food harvest amounts, and awareness of potential hazards in and near wastewater treatment areas. Community forums were also held, during which approximately 100 additional members of the public provided feedback and validation of preliminary exposure scenarios. Site assessments of the treatment and potential exposure areas, led by engineers and local partners, were also

Table 2
Quantitative microbial risk assessment model parameters, distributions, and assumptions used to develop exposure scenarios in wastewater effluent-impacted environments in Arctic Canada.

Treatment system	Exposure pathway and parameters	Conditions	Units	Distribution and values	References
Mechanical	**Shoreline recreation**				
	Distance (*dist*)	Low tide / high tide	m	Uniform (1000; 1500)[a]	Supplementary Materials
	Frequency (*freq*)	Low tide / high tide	m	Point estimate (105)	Supplementary Materials
	Ingestion volume (*V*)	Low tide / high tide	mL	Triangular (3.8; 7.6; 22.8)[b]	Dorevitch et al., 2011; McBride et al., 2013
Mechanical	**Small craft boating**				
	Distance (*dist*)	Low tide	m	Uniform (2000; 3500)[a]	Supplementary Materials
	Distance (*dist*)	High tide	m	Uniform (1000; 1500)[a]	Supplementary Materials
	Frequency (*freq*)	Low tide / high tide	m	Point estimate (105)	Supplementary Materials
	Ingestion volume (*V*)	Low tide / high tide	mL	Triangular (5.8; 11.6; 34.8)[b]	Dorevitch et al., 2011; McBride et al., 2013
Mechanical	**Netfishing**				
	Distance (*dist*)	Low tide	m	Uniform (2000; 3500)[a]	Supplementary Materials
	Distance (*dist*)	High tide	m	Uniform (1500; 2500)[a]	Supplementary Materials
	Frequency (*freq*)	Low tide / high tide	m	Point estimate (85)	Supplementary Materials
	Ingestion volume (*V*)	Low tide	mL	Triangular (3.8; 7.6; 58.0)[b]	Dorevitch et al., 2011; McBride et al., 2013
Mechanical	**Shellfish harvesting**				
	Distance (*dist*)	Low tide	m	Uniform (1000; 2500)[a]	Supplementary Materials
	Frequency (*freq*)	Low tide	m	Point estimate (40)	Supplementary Materials
	Ingestion volume (*V*)	Low tide	mL	Triangular (10.0; 35.0; 50.0)[b]	Fuhrimann et al., 2017, Fuhrimann et al., 2016; WHO 2006b
Mechanical	**Shellfish consumption**				
	Distance (*dist*)	Low tide	m	Uniform (1000; 2500)[a]	Supplementary Materials
	Frequency (*freq*)	Low tide	m	Point estimate (40)	Supplementary Materials
	Ingestion volume (*V*)	Low tide	g	Triangular (15.0; 60.0; 75.0)[b]	Health Canada 2007; Moya 2004
	Accumulation factor	Low tide	–	Point estimate (10)	CEFAS 2014
	Cooking reduction factor	Low tide	–	Point estimate (0.5)	Supplementary Materials
Passive	**Shoreline recreation**				
	Distance (*dist*)	Spring / summer	m	Uniform (1500; 2000)[a]	Supplementary Materials
	Frequency (*freq*)	Spring	m	Point estimate (25)	Supplementary Materials
	Frequency (*freq*)	Summer	m	Point estimate (40)	Supplementary Materials
	Ingestion volume (*v*)	Spring / summer	mL	Triangular (3.8; 7.6; 22.8)[b]	Dorevitch et al., 2011; McBride et al., 2013
Passive	**Small craft boating**				
	Distance (*dist*)	Spring / summer	m	Uniform (1500; 2000)[a]	Supplementary Materials
	Frequency (*freq*)	Spring	m	Point estimate (25)	Supplementary Materials
	Frequency (*freq*)	Summer	m	Point estimate (40)	Supplementary Materials
	Ingestion volume (*V*)	Spring / summer	mL	Triangular (3.8; 11.6; 34.8)[b]	Dorevitch et al., 2011; McBride et al., 2013
Passive	**Netfishing**				
	Distance (*dist*)	Spring / summer	m	Uniform (1500; 2000)[a]	Supplementary Materials
	Frequency (*freq*)	Spring	m	Point estimate (35)	Supplementary Materials
	Frequency (*freq*)	Summer	m	Point estimate (50)	Supplementary Materials
	Ingestion volume (*V*)	Spring / summer	mL	Triangular (5.8, 11.6, 58.0)[b]	Dorevitch et al., 2011; McBride et al., 2013
Passive	**Wetland travel**				
	Distance (*dist*)	Spring / summer	m	Uniform (250; 1000)[a]	Supplementary Materials
	Frequency (*freq*)	Spring	m	Point estimate (35)	Supplementary Materials
	Frequency (*freq*)	Summer	m	Point estimate (45)	Supplementary Materials
	Ingestion volume (*V*)	Spring /summer	mL	Triangular (10; 35; 50)[b]	Fuhrimann et al., 2017; Fuhrimann et al., 2016; WHO 2006b

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

carried out in each community. It was assumed that a suite of exposures based on conditions in these five communities, which span a range of treatment systems, population sizes, and receiving environments, provides a reasonably representative range of base case model scenarios for Arctic Canada.

Six activities were included as exposure pathways in the base case model: shoreline recreation; small craft boating; netfishing; shellfish harvesting; shellfish consumption; and wetland travel. Each pathway is described in the following paragraphs and a summary of all the corresponding distributions, parameters, and literature references is provided in Table 2. Input variables include distance (*dist*), which is the location where the exposure event occurs as measured in metres from the effluent release point, and exposure frequency (*freq*), the number of exposure events per person per year. Values for both variables were estimated based on localized data (see Supplementary Materials for more information). Ingestion volumes (*V*), the amount of media ingested per person per exposure event, are literature based assumptions. The transmission route in five of the six exposures is incidental ingestion of contaminated water (i.e. droplets or hand-to-mouth contact). The exception is the shellfish consumption scenario wherein the route is ingestion of contaminated tissue. The parameters of incedental water ingestion volume for shoreline recreation, small craft boating, and netfishing are based on water recreation exposure values (Dorevitch et al., 2011; McBride et al., 2013). Dorevitch et al. (2011) group activities as either low, mid, or high contact exposures with average ingestion rates per hour of 3.8, 5.8, and 10.0 mL, respectively, and advise using three times the average hourly rate as a conservative maximum estimate. Values associated with wetland travel and shellfish harvesting exposures were sourced from assessments of agricultural and aquacultural labor in wastewater-irrigated settings that estimated an incidental water ingestion maximum of 50.0 mL per day (Fuhrimann et al., 2017; Fuhrimann et al., 2016; WHO 2006b). Triangular distributions (minimum; most likely; maximum) were assumed and fitted to this maximum value. In absence of reliable estimates of shellfish harvest yields in Arctic Canada (Priest and Usher 2004), the shellfish consumption value per exposure event was established upon a standard seafood portion for North American Indigenous populations (Health Canada 2007; Moya 2004). Separate exposure scenarios were constructed for each set of physical environment conditions (low tide / high tide or spring / summer, as applicable) as human activity parameters varied in some instances. The model assumed no human exposures of any kind during the non-open water months (approximately October through May).

Shoreline recreation: Shorelines are hubs of recreational and work-related activity in Arctic communities. Serving multiple purposes, shorelines provide access points to fresh and marine waters as well as storage space for boats and equipment. They also function as walking paths, children's play areas, and rod fishing locations. Shallow wading and splashing as well as handling of wet fish and equipment are expected; however, swimming or full submersion is infrequent. Therefore, a low-contact exposure rate (Dorevitch et al., 2011; McBride et al., 2013) was applied and a two-hour event duration.

Small craft boating: Small water craft are widely used across Arctic Canada for recreation, transportation, work, and food harvesting in aquatic environments. Small open-top crafts with outboard motors are most common in addition to larger motorized boats as well as canoes and kayaks. While boating, incidental ingestion of water could occur via launching from shore, fishing, spray, or splash from motors or paddles, or a fall into the water. A mid-contact exposure rate classification (Dorevitch et al., 2011; McBride et al., 2013) and two-hour event duration were designated.

Netfishing: Netfishing involves the setting and retrieving of large weighted nets, ropes, and buoys, typically by hand, from aboard a boat. Incidental water ingestion is plausible during all stages of the process. Similar to small craft boating, this scenario was also valued as a mid-contact exposure (Dorevitch et al., 2011; McBride et al., 2013). A corrective factor of five times the average, rather than three, was applied as a maximum parameter however, due to the intensified actions and submerged equipment. Non-commercial netfishing, hence no use of specialized clothing or decontamination measures, was assumed. The assumed event duration was two hours.

Wetland travel: The wetland travel exposure pathway is only applicable in locations operating passive wastewater treatment systems. While it is commonly known within communities that the WSP is a hazardous area to be avoided, the potential health risk posed in the adjoining, effluent-impacted wetland is less apparent. Fencing and signage are often erected around the perimeter of the stabilization pond but they usually do not extend to the wetland portions of the treatment areas. People may enter these areas while hunting small game, picking berries, collecting geese eggs, or on route elsewhere. Means of travel include walking, all-terrain vehicle, or snowmobile during the spring when snow is still present within the wetland. Incidental water ingestion could occur following contact with soil, vegetation, clothing, or equipment that has been contaminated with effluent. Additionally, all-terrain vehicles and snowmobiles will, as they traverse the wetland, spray soil particles and create droplets of water, which may be inadvertently ingested by the vehicle riders.

Shellfish harvesting: The shellfish harvesting exposure scenario was only included in the mechanical QMRA assessment, and only during low tide conditions. This modeling decision was based on local descriptions of the locations where this activity is commonly practiced. Shellfish, predominantly clams, are harvested by digging them from the exposed sea bed in coastal areas during low tide, either by hand or with a shovel. Fecal coliforms can become concentrated within the bottom sediment of the sea bed in effluent-impacted waters (Ford 2005; Heaney et al., 2012). Exposure may occur following the handling of shellfish and contact with contaminated water, soil, or tools.

Shellfish consumption: Shellfish consumption, also only applicable in mechanical system sites and during low tide, was assessed independently of harvesting. Shellfish filter large quantities of seawater and pathogens can become concentrated within their digestive tissue (Bitton 2005). Infective agents are then communicable to humans via ingestion (Ford 2005). To account for the accumulation of pathogens within the raw tissue, a factor of 10 times the *E. coli* concentration in the water at the harvest location was assumed based on a critical review of published data (CEFAS 2014). Most infectious pathogens can be killed or inactivated through cooking; however, shellfish is commonly consumed raw or partially cooked (Butt et al., 2004). Community data did in fact reveal a predilection for raw or lightly cooked shellfish among some residents in the region. To reflect this local practice, a reduction factor of 0.5 was then assumed and applied to the *E. coli* concentration within the tissue.

2.5. Dose response

The dose-response stage of a QMRA describes the relationship between levels of exposure a person experiences and the probability of a health outcome. The health outcome modelled in this research was AGI. The steps and equations involved are described in the ensuing paragraphs and the corresponding parameters, distributions, and assumptions are listed in Table 3.

The dose of *E. coli* ($d_{E.\ coli}$) a person ingests at exposure (MPN) was calculated by multiplying, C_{dist}, the concentration of indicator *E. coli* in the environmental media at the exposure distance (MPN/mL) by the volume (*V*) of water or tissue (mL or g) accidently ingested per event (Eq. (2)).

$$d_{E.\ coli} = C_{dist} \cdot V \tag{2}$$

Indicator *E. coli* was the only obtainable organism data. It was assumed that the reduction in *E. coli*, obtained using the first-order model (Eq. (1)), can be used to conservatively predict the inactivation, dilution, or sedimentation of specific enteric pathogens within the effluent-receiving environment (Nevers and Boehm 2011; Schoen and Ashbolt 2010). Based on WHO (2016) guidance documents and

Table 3
Dose-response model parameters, distributions, and assumptions used in quantitative microbial risk assessment of acute gastrointestinal illness associated with wastewater effluent-impacted environments in Arctic Canada.

Description	Distribution and values	References
Ratio of pathogenic organism per indicator *E. coli* (*E. coli: Path*)		
Pathogenic *E. coli*	Point estimate (0.08)	Haas et al. (1999); Howard et al. (2006)
Salmonella spp.	Triangular (1×10^{-4}; 1×10^{-3}; 1×10^{-2})[a]	Craig et al. (2003)
Campylobacter spp.	PERT (1×10^{-6}; 5.5×10^{-6}; 1×10^{-5})[b]	Fuhrimann et al. (2017); Mara et al. (2007); WHO (2006b)
Rotavirus	PERT (1×10^{-6}; 5.5×10^{-6}; 1×10^{-5})[b]	Fuhrimann et al. (2017); Mara et al. (2007)
Giardia spp.	Uniform (1×10^{-7}; 1×10^{-5})[c]	Machdar et al. (2013)[f]
Cryptosporidium spp.	PERT (1×10^{-7}; 5.5×10^{-7}; 1×10^{-6})[d]	Fuhrimann et al. (2017); Mara et al. (2007); WHO (2006b)
Dose-response models [*P(d)*]		
Pathogenic *E. coli* (EIEC)	Beta-Poisson (0.16; 2.11×10^{6})[d]	CAMRA (2015); Dupont et al. (1971)
Salmonella spp.	Beta-Poisson (0.389; 1.68×10^{4})[d]	CAMRA (2015); McCullough and Eisele (1951)
Campylobacter spp.	Beta-Poisson (0.14; 890.38)[d]	Black et al. (1988); CAMRA (2015)
Rotavirus	Beta-Poisson (0.253; 6.17)[d]	CAMRA (2015); Ward (1986)
Giardia spp.	Exponential (0.020)[e]	CAMRA (2015); Rendtorff (1954)
Cryptosporidium spp.	Exponential (0.057)[e]	CAMRA (2015); Messner et al. (2001)
Morbidity ratios ($P_{ill \mid inf}$)		
Pathogenic *E. coli*	0.35	Fuhrimann et al. (2017); Machdar et al. (2013); Westrell (2004)
Salmonella spp.	0.80	Westrell (2004); WHO (2006b)
Campylobacter spp.	0.30	Fuhrimann et al. (2017); Machdar et al. (2013); Westrell (2004)
Rotavirus	0.50	Barker et al. (2014); Westrell (2004); WHO (2006b)
Giardia spp.	0.90	Schoen and Ashbolt (2010)
Cryptosporidium spp.	0.79	Fuhrimann et al. (2017)

[a] Triangular distribution (minimum, most likely; maximum).
[b] Project evaluation and review techniques distribution (PERT) (minimum; most likely; maximum).
[c] Uniform distribution (minimum; maximum).
[d] Beta-Poisson distribution (α; N_{50}).
[e] Exponential distribution (r).
[f] General protozoa ratio. Machdar et al. (2013) provide values only, so uniform distribution is assumed.

microbial risk assessment approaches used in other data-limited contexts (Howard et al., 2006; Fuhrimann et al., 2016), ratios were then used to infer the level of relationship between concentrations of indicator *E. coli* and each pathogen included in the model. The ratios were sourced from wastewater literature when possible. When ratios derived from wastewater were not available, it was necessary to source ratios from recreational and drinking water literature. Specifically, wastewater-derived ratios were used for *Campylobacter*, rotavirus, and *Cryptosporidium* (Mara et al., 2007; Fuhrimann et al., 2016; 2017; WHO 2006b), drinking water ratios for pathogenic *E. coli* and *Giardia* (Haas et al. 1999; Howard et al., 2006; Machdar et al., 2013), and a recreational water ratio for *Salmonella* (Craig et al., 2003). The pathogen-specific doses, d_{path} (MPN) are then obtained by multiplying $d_{E.\ coli}$ by corresponding inference ratios, (*E. coli: Path*) (Eq. (3)).

$$d_{path} = d_{E.\ coli} \cdot (E.\ coli:\ Path) \tag{3}$$

The probability of infection [*P(d)*] at a single dose (*d*) for each pathogen was estimated using either the exponential (Eq. (4)) or beta-Poisson model (Eq. (5)), which are established as applicable to most microorganisms and exposures (Haas et al., 2014). With the exponential function (Eq. (4)), the natural logarithm base (*e*) and the probability that one organism survives to cause an infection within the human host (*r*) are pathogen-specific constants. The beta-Poisson model (Eq. (5)) is a two-parameter function with slope parameter α and median infectious dose N_{50}.

$$P(d) = 1 - e^{-rd} \tag{4}$$

$$P(d) = 1 - \left[1 + \left(\frac{d}{N_{50}}\right) \cdot \left(2^{1/\alpha} - 1\right)\right]^{-\alpha} \tag{5}$$

Morbidity ratios ($P_{ill \mid inf}$) sourced from literature were then applied to these probabilities to estimate the number of infections that resulted in symptomatic cases, which represents the probability of illness following a single exposure event ($P_{ill,path}$) (Eq. (6)).

$$P_{ill,\ path} = P(d) \cdot P_{ill \mid inf} \tag{6}$$

2.6. Risk characterization

Monte Carlo simulations were used in the risk characterization stage of the QMRA. Samples from the pre-specified data distributions were repeatedly drawn (10 000 iterations) to model the probability of the health outcome (Haas et al., 2014). The probability of illness from a single exposure event ($P_{ill,path}$), as calculated with Eq. (6), was combined with the frequency of exposure events per person per year (*freq*) to arrive at the individual annual probability of AGI ($P_{ill,\ annual}$) associated with each exposure scenario (Eq. (7)).

$$P_{ill,\ annual} = 1 - \left(1 - P_{ill,\ path}\right)^{freq} \tag{7}$$

The risk results only apply to individuals in the specified exposure group (e.g. shellfishers harvesting near the mechanical treatment plant during low tide), and not an entire community population. It is assumed that individuals can simultaneously belong to more than one exposure group (e.g. an individual may be a shellfisher and a netfisher). The model was developed using Crystal Ball software (Oracle 2017).

Sensitivity analysis was conducted to prioritize potential control points in the system where risk reducing mitigations may be effective. Specifically, rank order correlation was used to evaluate the impact of the variability and uncertainty within the model inputs on the base case risk results. Rank order correlation is a nonparametric approach, which is based on less stringent distributional assumptions and provides relatively conservative estimates. This feature is beneficial in risk assessment research when the actual distributions of input variables are typically unknown (Vose 2008). Based on this analysis, potential mitigations were configured and assessed.

3. Results and discussion

3.1. Base case scenarios

Given that this study represents the first assessment in this context, the 75th percentile risk levels were conservatively chosen as the result values to be compared to the proposed tolerable risk benchmark of 10^{-3}. Figs. 1, 2, and 3 present box-and-whisker plots of the three exposure scenarios with the highest individual annual risk estimates. Of the three scenarios, two are activities associated with the mechanical treatment and low tide conditions: shore recreation and shellfish consumption. The third scenario, wetland travel during spring freshet, is from the passive

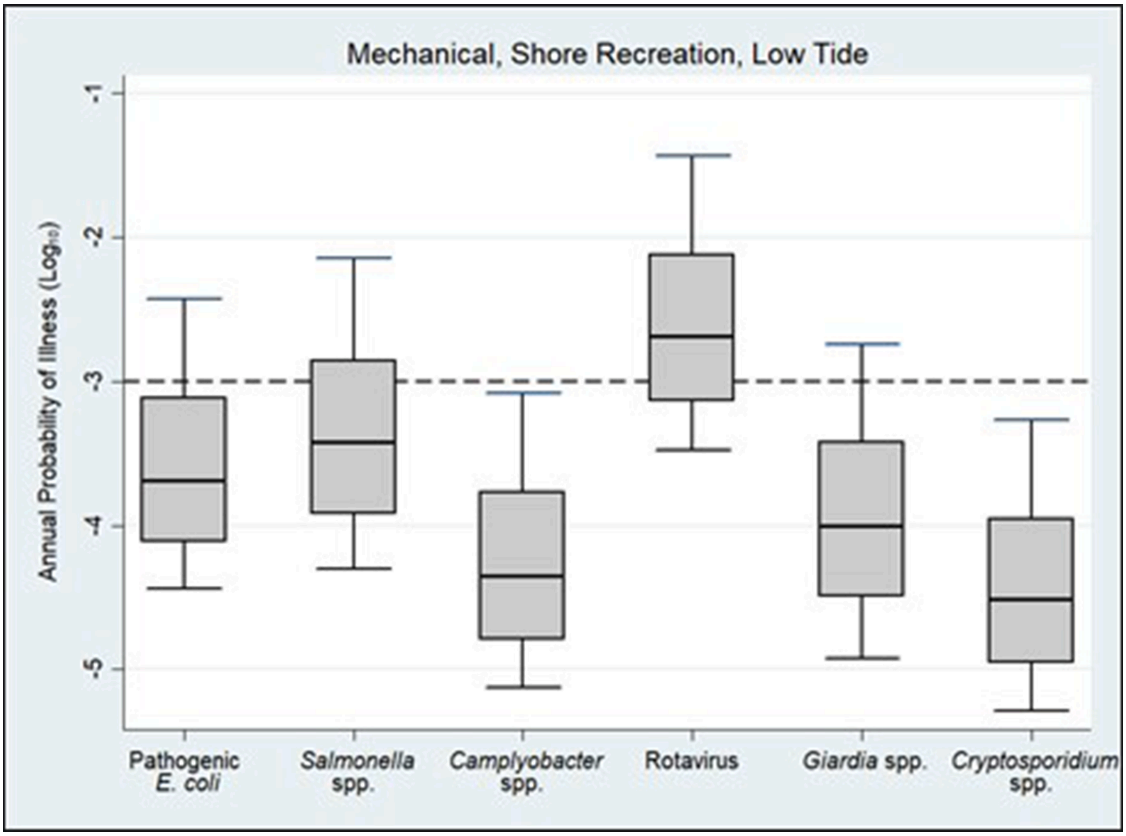

Fig. 1. Box-and-whisker graph of individual annual probabilities of acute gastrointestinal illness caused by enteric pathogens associated with 'mechanical, shore recreation, low tide' wastewater effluent exposure scenario in Arctic Canada under baseline conditions. The probabilities were estimated using a quantitative microbial risk assessment. Boxes represent 25th and 75th percentiles, solid lines within boxes are medians, and whiskers are 10th and 90th percentiles. Large dashed line denotes a potential tolerable risk guideline (10^{-3}).

treatment model. Of the six pathogens modelled, rotavirus and *Salmonella* spp. were projected to pose the highest risk, followed by pathogenic *E. coli, Giardia* spp., *Campylobacter* spp., and *Cryptosporidium* spp. In each of the three presented exposures scenarios, the 75th percentile risk level for at least two pathogens exceeded the benchmark. Although not included in the figure, it should also be noted that the 75th percentile risk level for rotavirus, singly, was near 10^{-3} in the mechanical-shellfish harvest-low tide and passive-wetland travel-summer scenarios. Most of the annual risk probabilities were log-normally distributed. Exceptions were some pathogens in very low risk scenarios ($\leq 10^{-12}$). These lower probabilities followed Weibull or Gamma distributions, which are similar to log-normal (Vose 2008).

Of the remaining passive system scenarios, the majority of annual risk estimates were much lower than the wetland travel-spring exposure, with 75th percentiles $\leq 10^{-6}$. Engineering assessments of arctic wetland treatment systems have also emphasized the spring freshet as a period of higher risk if the adjoining WSP is undersized or has a breached berm (Hayward et al., 2018). Under such circumstances, wastewater that has been accumulating and remained frozen within the WSP throughout the winter thaws quickly and is discharged into the wetland at a high rate (Hayward et al., 2014; Yates et al., 2012). The consequence is an influx of untreated contaminants in the wetland treatment area and receiving water body (Huang et al., 2018). Community input shows that spring is also a potential time for increased human activity within treatment wetlands. As sea and lake ice begin to thin and melt, people travelling by all-terrain vehicles begin to alter their inland routes toward these areas, consequently increasing exposure frequencies.

The mechanical treatment estimates exhibited a pronounced difference in risk between low and high tidal conditions. All exposures modelled during high tide produced 75th percentile risk estimates less than or equal to 10^{-16}. Aside from the highest risk pathways noted above, the remaining low tide exposures, small craft boating and netfishing, had 75th percentile risk levels between 10^{-5} and 10^{-7}. Despite the marked difference in risk estimates between tidal conditions, it is unlikely that an operational change whereby effluent is only released from the plant during high tide would be possible. The current mechanical systems operating in Arctic Canada are not designed with the holding capacity to detain large volumes of wastewater, as would be necessary between tidal cycles. One such system, in Iqaluit, is semi-centralized so raw influent is continuously being piped into the plant; therefore, it must be processed and discharged in a timely manner. The community of Pangnirtung has a decentralized system with all homes and buildings serviced by wastewater pump trucks, which then discharge to the treatment plant. The restrictions that would be necessary to align pump truck service with tidal schedules would be severely disruptive to community life. Such practices may simply create additional sanitation issues at the household level through backups and overflows as home wastewater holding tanks require emptying via pump truck multiple times per week (Daley et al., 2014). Previous engineering and ecological assessments have suggested that continuously discharging mechanical treatment systems are not well-suited for Arctic conditions. Specifically, Greenwood (2016) and Krumhansl et al. (2015) demonstrated that the continuous discharge of nondisinfected effluent can have a negative environmental impact on the receiving water habitat over 500 m from the effluent source. These QMRA results show that such wastewater management practices also have potential to elevate human health risks in the region. Risks are more pronounced when effluent is discharged during low tide conditions; a period when the sea bed is exposed and minimal dilution occurs (Gunnarsdóttir et al., 2013).

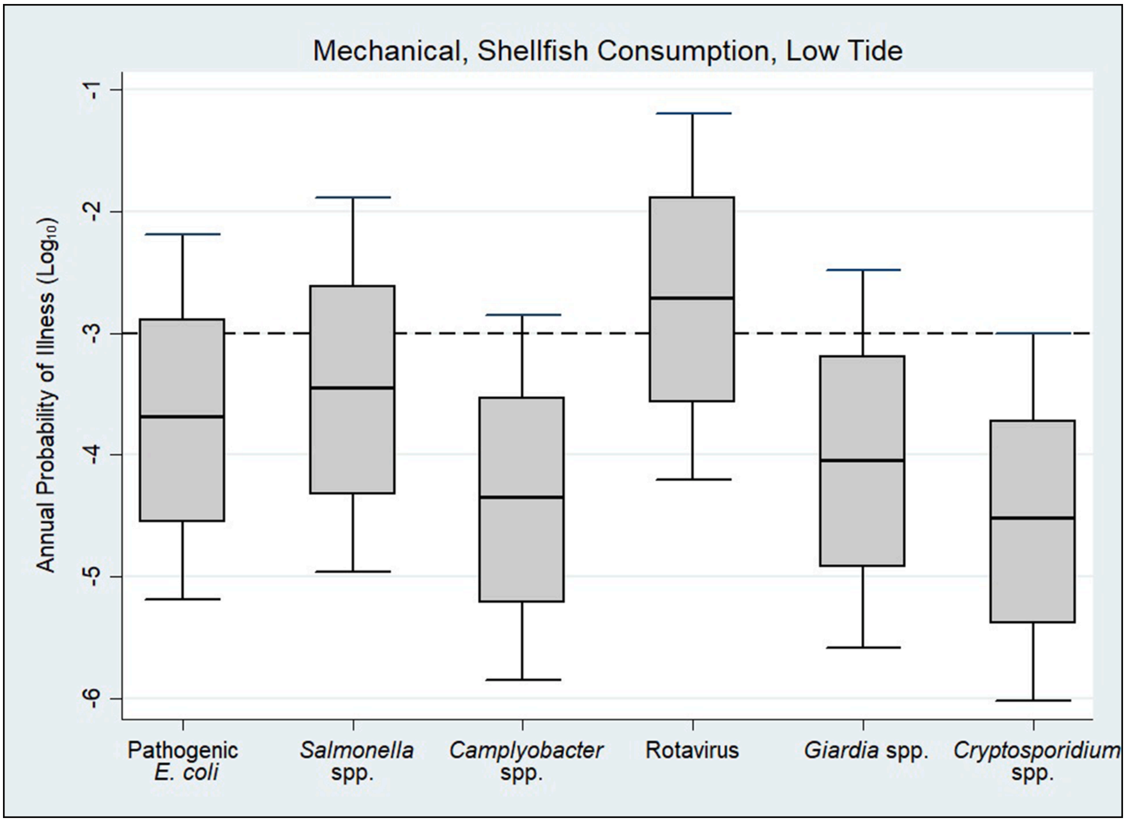

Fig. 2. Box-and-whisker graph of individual annual probabilities of acute gastrointestinal illness caused by enteric pathogens associated with 'mechanical, shellfish consumption, low tide' wastewater effluent exposure scenario in Arctic Canada under baseline conditions. The probabilities were estimated using a quantitative microbial risk assessment. Boxes represent 25th and 75th percentiles, solid lines within boxes are medians, and whiskers are 10th and 90th percentiles. Large dashed line denotes a potential tolerable risk guideline (10^{-3}).

In Arctic Canada, territorial health departments are authorized to inspect and respond to wastewater-related issues (Government of Nunavut 1990), although there are no specific health-based targets applied to wastewater discharges. Also, the Canadian Shellfish Sanitation Program – an intended nationwide food safety program – has never been established in northern Canada (Canadian Food Inspection Agency 2019). In the absence of documented health-based targets for the region, the WHO safe wastewater reuse adapt-and-adopt value of 10^{-3} annual risk of illness, one order of magnitude higher than the WHO standard global guideline, was selected as a comparative benchmark for these QMRA results. This choice was based on limited epidemiological data on waterborne and shellfish-related illness in the Arctic as well as the nature of the exposure pathways. Most established waterborne illness guidelines are drawn from recreational water settings or wastewater reuse for agriculture and aquaculture. Recreational water criteria suggest a tolerable per event risk of gastrointestinal illness of approximately 3.0×10^{-2} for exposures such as swimming at a beach (USEPA 2012b). In agriculture and aquaculture settings where wastewater is intentionally used for irrigation purposes, an annual tolerable risk of illness of either 10^{-4} or 10^{-3} is applied for both fieldworkers and consumers (Mara 2008; WHO 2006a). The exposure pathways in the wastewater-impacted environment in Arctic Canada, however, differ from those in the reviewed guidelines. Some, such as shore recreation and small craft boating, classify as recreational but others are unique to this setting. Foraging activities such as netfishing and shellfish harvesting compare somewhat to agriculture and aquaculture, but with the distinction that the food being harvested is wild and not farmed. This distinction is important given the central role of subsistence activities in Indigenous communities (Suk et al., 2004), view of the immediate environment as a vital source of nourishment (Cunsolo Willox et al. 2012), and risk of contaminant bioaccumulation in the diets of Arctic Indigenous populations (Donaldson et al., 2010). While 10^{-3} was chosen as a benchmark for this analysis, policy makers may want to consider the use of multiple guidelines to account for the various types of exposure pathways.

3.2. Sensitivity analysis and risk mitigation options

The results of the sensitivity analysis conducted on the three base case exposure scenarios that exceeded the risk benchmark are presented in Table 4. Distance from the effluent release to exposure location (*dist*) was identified as the parameter with the highest mean correlation coefficient across the three scenarios (−0.71), followed by concentration of indicator *E. coli* at effluent release (C_o) (0.53). The ratio of pathogenic organism per indicator *E. coli* (*E. coli: Path*) and ingestion volume (*V*) correlation coefficients values were lower with means of 0.22 and 0.16, respectively.

The sensitivity analysis was used to identify leverage points where risk reducing mitigations may be most effective. Two specific mitigations were theorized and modelled: one targeted at decreasing the concentration of indicator *E. coli* within effluent at initial release points (C_0) and the second at increasing the distance between effluent release points and locations of human activity where exposure is likely to occur (*dist*). The mitigation designs, including the corresponding model parameter adjustments are described in the following paragraphs. Figs. 4, 5, and 6 present the impact of the mitigations on estimated individual annual risk for the three exposure scenarios that exceeded the benchmark, per pathogen, as compared to base case results.

Mitigation 1–Improved treatment

Mitigation 1 is an engineering control aimed at improving wastewater treatment and thus reducing the initial concentration of pathogen in the effluent being discharged into the receiving environment. For

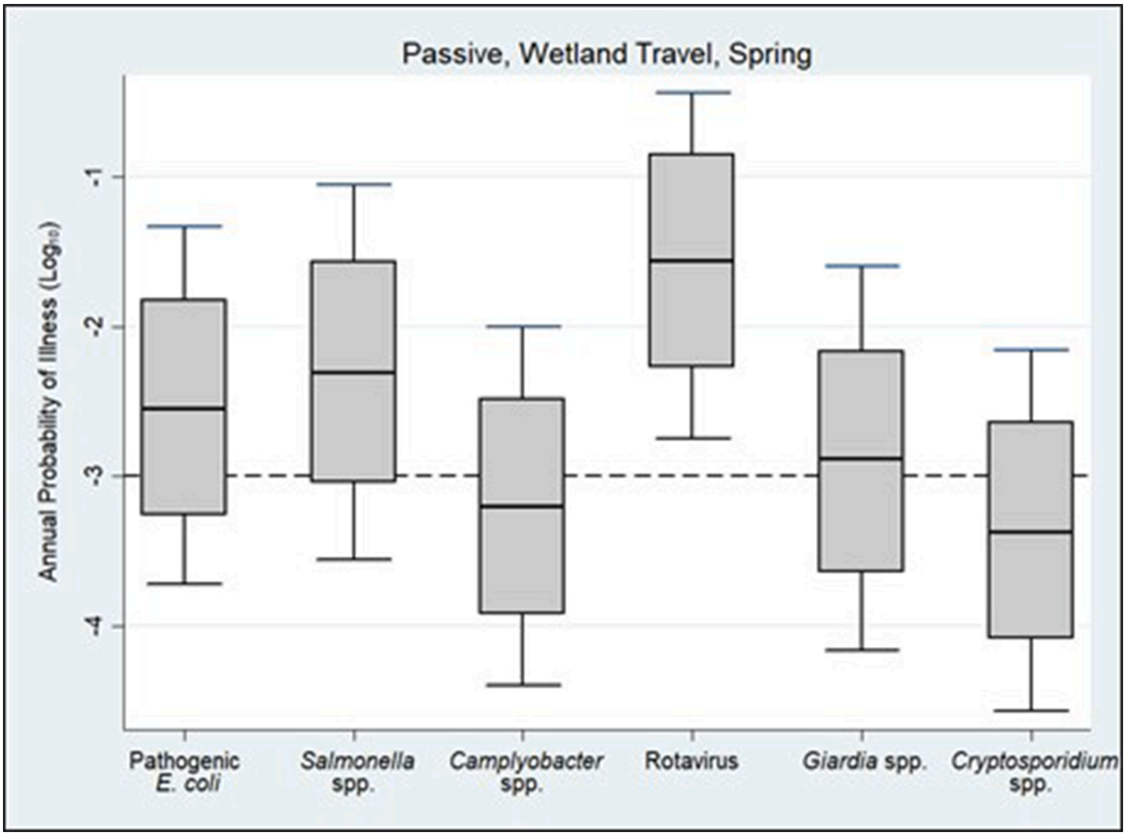

Fig. 3. Box-and-whisker graph of individual annual probabilities of acute gastrointestinal illness caused by enteric pathogens associated with 'passive, wetland travel, spring' wastewater effluent exposure scenario in Arctic Canada under baseline conditions. The probabilities were estimated using a quantitative microbial risk assessment. Boxes represent 25th and 75th percentiles, solid lines within boxes are medians, and whiskers are 10th and 90th percentiles. Large dashed line denotes a potential tolerable risk guideline (10^{-3}).

Table 4
Sensitivity analysis of base case scenarios that exceeded a tolerable risk benchmark (10^{-3}) of individual annual probability of acute gastrointestinal illness caused by enteric pathogens in an Arctic Canada wastewater exposure risk assessment model.

Parameters[a]	Correlation coefficients		
Treatment system Exposure pathway Conditions	Mechanical Shore recreation Low tide	Mechanical Shellfish consumption Low tide	Passive Wetland travel Spring
Distance (*dist*)[b]	−0.43 – −0.38	−0.80 – −0.76	−0.95 – −0.88
E. coli at effluent release (C_o)	0.74 – 0.84	0.51 – 0.54	0.24 – 0.27
Inference ratio (*E. coli: Path*)	0.18 – 0.40	0.10 – 0.26	0.13 – 0.32
Ingestion volume (*V*)	0.21 – 0.24	0.12 – 0.12	0.14 – 0.15

Values represent the range (min to max) of the rank order correlation coefficients across modelled pathogens for input variables in relation to individual annual probability of illness ($P_{ill,\ annual}$).

[a] Full definition of parameters available in Tables 1, 2, and 3.

[b] Negative values indicate inverse relationship between variable and $P_{ill,\ annual}$.

mechanical systems, this reduction could be accomplished by adding additional treatment units (UV, chlorination, and filtration) to the treatment process to remove pathogens. Within the model, initial concentration of *E. coli* (C_o) is characterized by a Pareto distribution. The improved treatment was parameterized by first adjusting the location parameter, which determines the minimum possible value, from 10^4 to 10^2, which is the achievable treatment level by chlorination (Bitton 2005). Some pathogens, such as *Cryptosporidium* and *Giardia* are resistant to chlorination; however for the purposes of this analysis we are assuming that the mechanical treatment system would have additional units (e.g. UV) to ensure removal of the full suite of pathogens. In turn, the shape parameter was adjusted from 0.48 to 0.15 to maintain a fit that represents the documented 5 – 10% failure rate of mechanical systems in Arctic Canada (City of Iqaluit 2015). Upon reassessing shore recreation and shellfish consumption at low tide conditions, an approximate 3 to 5 fold reduction was seen at the 75th risk level across pathogens for both scenarios; dropping them all below the 10^{-3} benchmark. Note that in both scenarios the 90th percentile risk level was similar with or without mitigation, remaining above the benchmark for several pathogens. This result is due to the incorporated failure rate in the design, currently a reality of these systems in arctic conditions (Johnson et al., 2014).

In passive systems, improved treatment requires designing and constructing an adequately sized WSP capable of eliminating overflow and leakage. The effect is that wastewater would be detained within the WSP, undergoing a full passive treatment season, rather than continuously seeping from the onset of spring freshet. Effluent would then be manually decanted from the WSP in a controlled discharge exclusively during a one-month period in late summer, just prior to freeze-up. The adjoining wetland could also be engineered to slow and direct the flow of effluent to increase retention times. The improved stabilization pond would produce a 1-log reduction in *E. coli* concentration at the point of discharge to the wetland (Bitton 2005). In modelling terms, the parameters of the uniformly distributed initial indicator *E. coli* concentration were adjusted to a minimum of 10^4 and a maximum of 10^5. Additionally, changing to a controlled decant at the conclusion of the passive treatment season dictates using the summer pathogen reduction coefficient for the wetland treatment component (Table 1). Exposure

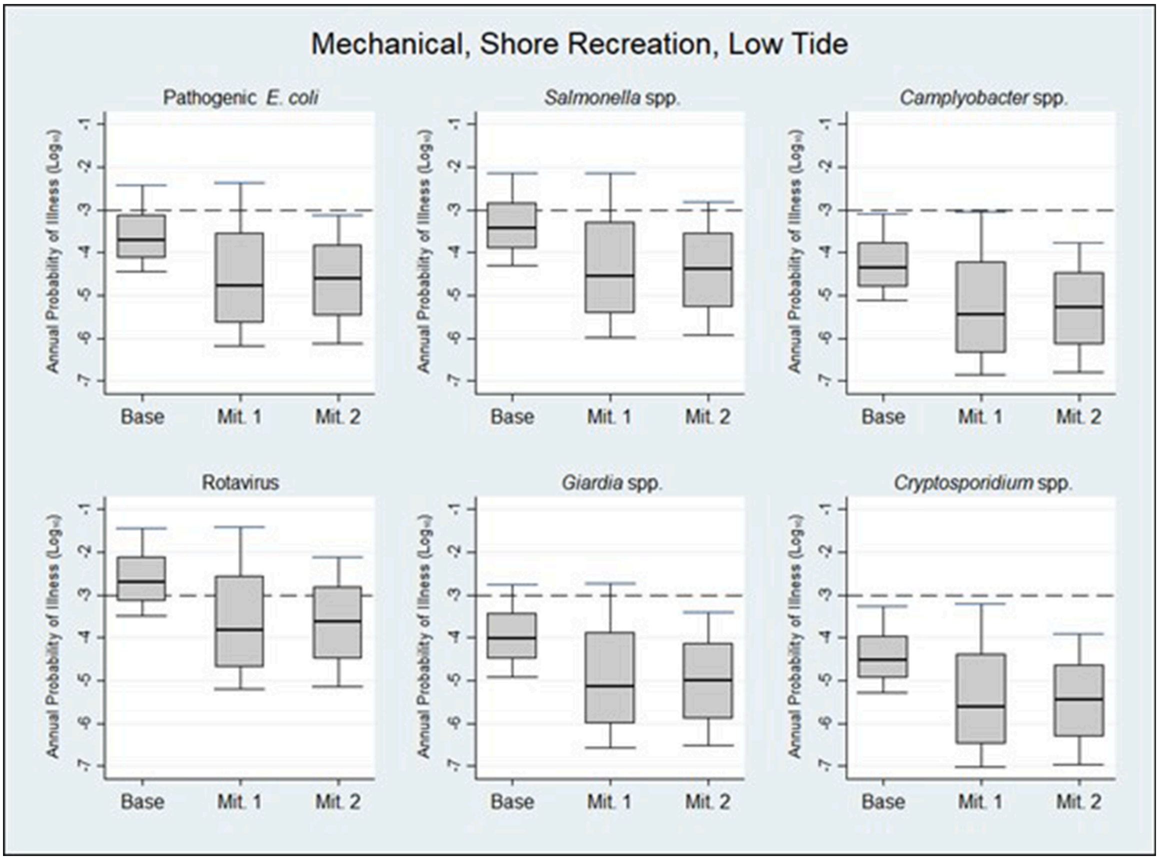

Fig. 4. Box-and-whisker graphs of individual annual probability of acute gastrointestinal illness caused by enteric pathogens associated with 'mechanical, shore recreation, low tide' wastewater effluent exposure scenario in Arctic Canada, under baseline conditions (Base) and mitigations (Mit. 1–improved treatment, Mit. 2–behavioural change). The probabilities were estimated using a quantitative microbial risk assessment. Boxes represent 25th and 75th percentiles, solid lines within boxes are medians, and whiskers are 10th and 90th percentiles. Large dashed lines denote a potential tolerable risk guideline (10^{-3}).

event frequency was also decreased to 20, to reflect the shorter time period during which human contact with pathogens could occur in the wetland. The risk reduction to wetland travel as a result of this mitigation is substantial, as 75th percentile values for all pathogens drop to 10^{-4} or lower, which is approximately 2000 fold lower than base case risk.

Mitigation 2–Behavioural change

Mitigation 2 involves interventions intended to inform people of wastewater hazards and change the patterns of human activity occurring in the treatment areas and receiving environments. Behavioural change mitigations should ultimately be chosen based on what is acceptable, appropriate, and culturally relevant to the local population (Nguyen-Viet et al., 2009). Options in this setting may include public health messaging or signage and fencing at the initial points of effluent discharge. It is assumed that these interventions are preventative initiatives, as opposed to enforced by-laws. As such, some people may still choose to enter these spaces to gain access to established travel routes and food harvesting locations. A portion of the exposed population, however, will likely alter their behaviour patterns and shift activity to locations further away from the effluent release source.

In the passive system model, the minimum parameter of the uniformly distributed distance (*dist*) variable was increased from 250 to 500 m. All other values remained the same. The result was an approximate 5 fold reduction in risk at the 75th percentile level for spring-wetland travel exposure across pathogens. Even so, pathogenic *E. coli*, rotavirus, *Giardia*, and *Salmonella* 75th percentile values remain at or above 10^{-3}. Within the mechanical treatment model, the minimum distance (*dist*) parameter was unaltered from the base case setting of 1000 m as this original value was based on the existing level of public awareness concerning hazards in the area directly surrounding mechanical treatment facilities. Instead, the maximum parameter was increased by 1000 m for both scenarios to simulate the shoreline recreation and shellfish consumption exposure populations moving further away from the treatment facility in response to the mitigation. The result was an approximate 3 to 5 fold decrease in 75th percentile risk values across pathogens for both scenarios; dropping all but rotavirus below the 10-3 benchmark.

Overall, both types of mitigation reduced the estimated AGI risk attributable to wastewater exposures. With respect to mechanical treatment specifically, the impact was similar across the two options. There is greater inherent uncertainty in the mitigation 2 results, however, as the effectiveness of improved treatment processes is more predictable than actions intended to change human behaviour. Regarding passive systems, the improved treatment mitigation was more effective, strengthening the case for well-designed stabilization pond and wetland systems in Arctic conditions (Balch et al., 2018). Infrastructure costs in the Arctic are exorbitant and decisions related to upgrading wastewater treatment should be made based on whether the investments will result in significantly improved health or environmental outcomes. Appropriate technology choices and rational allocation of resources should be part of setting priorities within an overall public health strategy and water safety plan (Murphy et al., 2009; WHO 2016). For comparative purposes, and keeping in mind that costs are highly variable, a mechanical treatment facility with disinfection capability in a medium-sized Arctic Canada community (pop. 1500) would likely cost upwards of $5 – 10 M in Canadian dollars (CAD). Additionally, annual operational and maintenance costs could range from CAD$300–800 thousand, a large portion of which get allocated to energy expenses. The

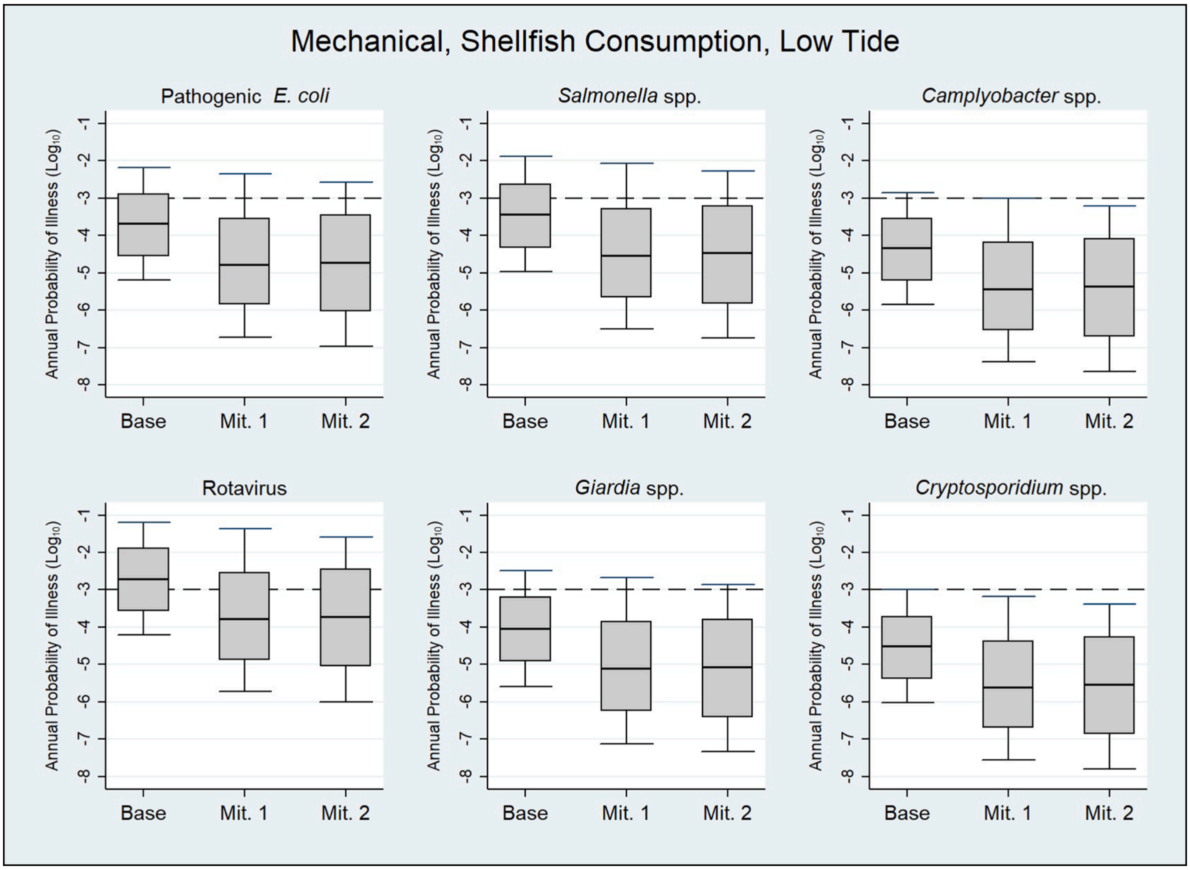

Fig. 5. Box-and-whisker graphs of individual annual probability of acute gastrointestinal illness caused by enteric pathogens associated with 'mechanical, shellfish consumption, low tide' wastewater effluent exposure scenario in Arctic Canada, under baseline conditions (Base) and mitigations (Mit. 1–improved treatment, Mit. 2–behavioural change). The probabilities were estimated using a quantitative microbial risk assessment. Boxes represent 25th and 75th percentiles, solid lines within boxes are medians, and whiskers are 10th and 90th percentiles. Large dashed lines denote a potential tolerable risk guideline (10^{-3}).

initial cost of building a properly engineered passive WSP and wetland treatment system is estimated at CAD$5M, but with far less operational costs required (Johnson et al., 2014).

3.3. Limitations

The sensitivity analysis also provided insight into which health risk assessment variables would benefit from more site data. The model used relies exclusively on *E. coli* as a fecal indicator organism. Although this approach was necessary due to lack of data availability, *E. coli* can be a relatively weak indicator of pathogen concentrations (Harwood et al., 2005). Similarly, enterococci are considered a preferred fecal indicator in marine waters, if available (Health Canada 2012). In order to reduce some of the uncertainty inherent in this approach, additional pathogen-specific datasets are highly desired for future risk assessments. A related source of uncertainty within this study is the likely differences in environmental decay between pathogens, which were not accounted for. This study relied on a first-order kinetics approach to estimate *E. coli* decay and then the use of ratios to translate *E. coli* values to pathogens for the QMRA. Although there is precedent for this indirect approach in sanitation research within data limited settings (Fuhrimann et al., 2016; WHO 2016; 2006a), it necessitates sourcing some of the *E. coli*-to-pathogen ratios from surface or drinking water literature if a comparable wastewater study is not available. Assuming that identical ratios apply across water sources does constrain interpretation of the study. Pathogen fate and transport models developed specifically for arctic conditions would be of great benefit to future wastewater research (Cho et al., 2016). However, all Arctic microbiology research–water, medical, or otherwise–is currently limited by a lack of laboratory facilities in the remote region. Therefore, for the time being, *E. coli* analysis remains the practical indicator organism given the low cost and ease of processing. More research is also recommended specifically on the human health risks associated with shellfish consumption in the Arctic as the QMRA results presented here provide only a starting estimate. Shellfish are an easily accessible, and therefore important, food source in the region (Harrison and Loring 2016), yet caution is warranted as worldwide they are commonly associated with wastewater contamination and cases of AGI (Ford 2005).

The exposure pathways assessed in this QMRA were developed using local knowledge from predominantly Inuit communities in Nunavut. As such, the findings may or may not be directly transferable to other communities and Indigenous populations in the Arctic. The model was deliberately designed to be inferential and is easily adaptable to other communities and exposure scenarios given the necessary input to define and parameterize the human-environment interactions. This type of data can be collected and inserted into the model by community members and stakeholders without the need for extensive training.

4. Conclusion

Building on an initial screening-level model (Daley et al., 2019), this research provides the first in-depth risk assessment of AGI attributable to wastewater treatment systems in Arctic Canada. Three exposure scenarios included in the assessment exceeded a proposed tolerable annual 75th percentile risk target of 10^{-3}. These scenarios included: shore recreation near mechanical treatment sites during low tide; consumption of shellfish harvested near mechanical treatment sites during low tide; and wetland travel near passive treatment sites during spring

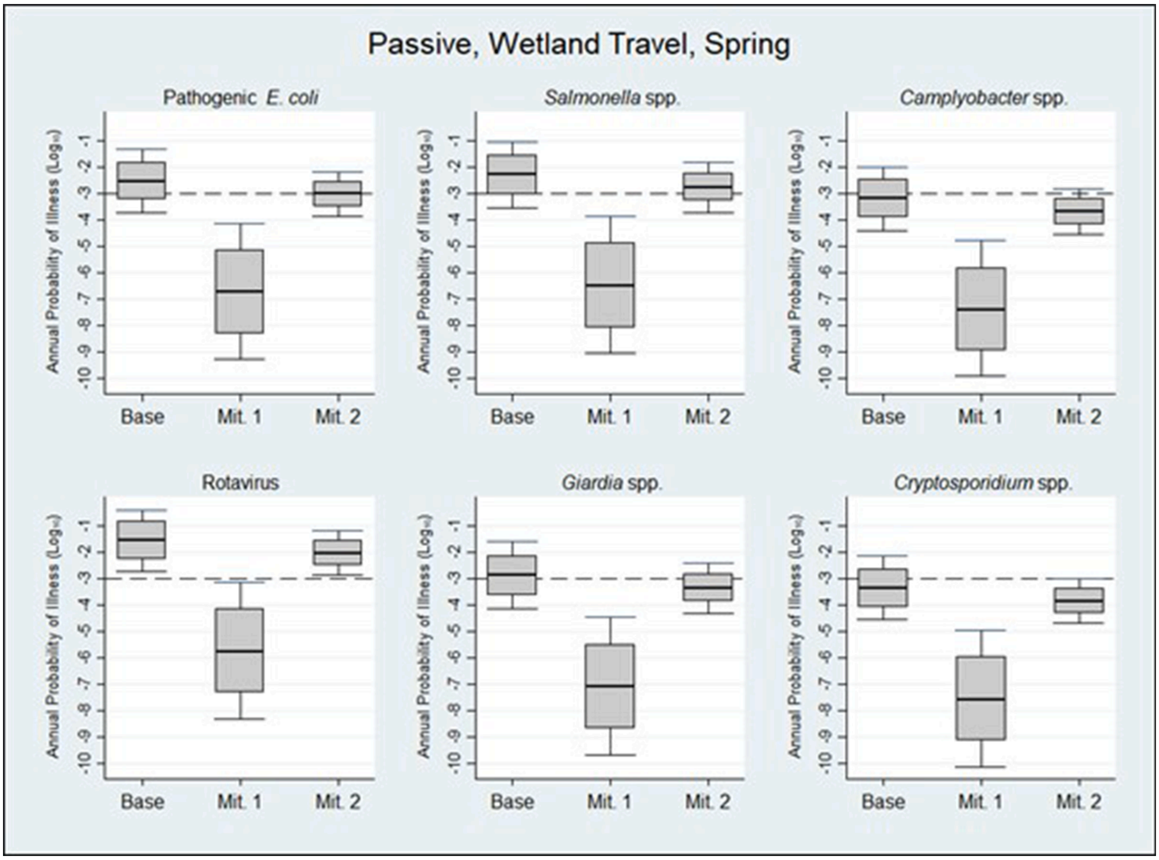

Fig. 6. Box-and-whisker graphs of individual annual probability of acute gastrointestinal illness caused by enteric pathogens associated with 'passive, wetland travel, spring' wastewater effluent exposure scenario in Arctic Canada, under baseline conditions (Base) and mitigations (Mit. 1–improved treatment, Mit. 2–behavioural change). The probabilities were estimated using a quantitative microbial risk assessment. Boxes represent 25th and 75th percentiles, solid lines within boxes are medians, and whiskers are 10th and 90th percentiles. Large dashed lines denote a potential tolerable risk guideline (10^{-3}).

freshet. Lower risk probabilities were estimated in all other scenarios. These base case results suggest that human exposure to wastewater effluent via food harvesting and recreational activities may be above benchmark risk levels selected for this study. Mitigation in the form of engineering controls and behavioural interventions were shown to have potential to reduce risk to varying degrees. On the whole, engineered passive systems, incorporating controlled summer discharge schedules and risk communication messaging, appear the most appropriate wastewater treatment option for Arctic communities.

This research was conducted using a modified participatory QMRA approach. Participatory epidemiology-based data collection methods including interviews, site-mapping, and public forums were used with the conventional risk assessment framework. Thereby, local knowledge of activity patterns in wastewater-impacted environments centered the exposure scenario development process. As such, the results offer an evidence base for water, sanitation, and public health policy and actions in Arctic Canada that is grounded in community knowledge. This study also lends perspective to the greater body of emerging epidemiology and microbiology research investigating various aspects of waterborne pathogens and enteric disease in the Arctic. More broadly, elements of this research may also be relevant to other locations where basic wastewater treatment practices are utilized.

CRediT authorship contribution statement

Kiley Daley: Writing – original draft, Methodology, Formal analysis, Investigation, Data curation, Visualization. **Rob Jamieson:** Writing – review & editing, Supervision, Conceptualization, Funding acquisition, Project administration. **Daniel Rainham:** Writing – review & editing, Conceptualization, Supervision. **Lisbeth Truelstrup Hansen:** Writing – review & editing, Supervision, Conceptualization. **Sherilee L Harper:** Writing – review & editing, Conceptualization.

Declaration of Competing Interest

None.

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Supplementary materials

Supplementary material associated with this article can be found, in the online version, at doi:10.1016/j.mran.2021.100186.

References

Ahmed, A.M., Hall, A.J., Robinson, A.E., Verhoef, L., Premkumar, P., Parashar, U.D., Koopmans, M., Lopman, B.A., 2014. Global prevalenence of norovirus in cases of gastroenteritis: a systematic review and meta-analysis. Lancet Infecti. Dis. 14, 725–730.

American Public Health Association [APHA], 2012. Standard Methods for the Examination of Water and Wastewater, 22nd ed. Water Environment Federation, Washington, DC.

Balch, G., Hayward, J., Jamieson, R., Wootton, B., Yates, C., 2018. Recommendations for the use of tundra wetlands for treatment of municipal wastewater in Canada's Far North. In: Nagabhatla, N., Metcalfe, C. (Eds.), Multifunctional wetlands: Pollution abatement and Other Ecological Services from Natural and Constructed Wetlands. Spring International Publishing, New York, pp. 83–120.

Barber, M., Jackson, S., 2015. Knowledge making': issues in modelling local and indigenous ecological knowledge. Hum. Ecol. 43, 119–130.

Barker, S.F., Amoah, P., Dreschsel, P., 2014. A probabilistic model of gastroenteritis risk associated with consumption of street food salads in Kumasi, Ghana: evaluation of methods to estimate pathogen dose from water, produce or food quality. Sci. Total Environ. 487, 130–1142.

Bitton, G., 2005. Wastewater Microbiology, 3rd ed. Wiley, Hoboken, NJ.

Bjerregaard, P., Young, T.K., Dewailly, E., Ebbesson, S.O.E., 2004. Indigenous health in the Arctic: an overview of the circumpolar Inuit population. Scand. J. Public Health 32 (5), 390–395.

Black, R.E., Levine, M.M., Clements, M.L., Hughes, T.P., Blaser, M.J., 1988. Experimental *Campylobacter jejuni* infection in humans. J. Infect. Dis. 157 (3), 472–479.

Brown, P.J., Armelagos, G.J., Maes, K.C., 2011. Humans in a world of microbes: the anthropology of infectious disease. In: Singer, M., Erickson, P.I. (Eds.), A Companion to Medical Anthropology. Blackwell Publishing Ltd, Malden, MA, pp. 251–270.

Butt, A.A., Aldridge, K.E., Sanders, C.V., 2004. Infections related to ingestion of seafood. Part I: viral and bacterial infections. Lancet Infecti. Dis. 4, 201–212.

Canadian Council of the Ministers of the Environment [CCME]. (2009). *Canada-wide strategy for the management of municipal wastewater effluent*. http://www.ccme.ca/assets/pdf/cda_wide_strategy_mwwe_final_e.pdf. Accessed 15 June 2016.

Canadian Food Inspection Agency. (2019). *Canadian shellfish sanitation program*. Retrieved from http://www.inspection.gc.ca/food/requirements-and-guidance/preventive-controls-food-businesses/fish/cssp/eng/1563470078092/1563470123546.

Center for Advancing Microbial Risk Assessment [CAMRA]. (2015). *Table of recommended best-fit parameters*. http://qmrawiki.org/content/recommended-best-fit-parameters. Accessed 26 April 2020.

Centre for Environment Fisheries & Aquaculture Science [CEFAS], 2014. A Critical Review of the Current Evidence For the Potential Use of Indicator Species to Classify UK Shellfish Production Sreas (FS512006). Food Standards Agency, London.

Charron, D.F., 2012. Ecohealth: origins and Approach. In: Charron, D.F. (Ed.), Ecohealth Research in Practice. Springer, New York, pp. 1–30.

Cho, K.H., Pachepsky, Y.A., Oliver, D.M., Muirhead, R.W., Park, Y., Quilliam, R.S., Shelton, D.R., 2016. Modeling fate and transport of fecally-derived microorganisms at the watershed scale: state of the science and future opportunities. Water Res. 100, 38–56.

City of Iqaluit. (2015). *2015 Annual water license report*. https://www.nwb-oen.ca/. Accessed 3 December 2018.

Craig, D.L., Fallowfield, H.J., Cromar, N.J., 2003. Effectiveness of guideline faecal indicator organism values in estimation of exposure risk at recreational coastal sites. Water Sci. Technol. 47 (3), 191–198.

Cunsolo Willox, A., Harper, S.L., Ford, J., Landman, K., Houle, K., Edge, V.L., Rigolet Inuit Community Government, 2012. From this place and of this place:" climate change, sense of place, and health in Nunatsiavut, Canada. Soc. Sci. Med. 75 (3), 538–547.

Daley, K., Castleden, H., Jamieson, R., Furgal, C., Ell, L., 2014. Municipal water quantities and health in Nunavut households: an exploratory case study in Coral Harbour, Nunavut, Canada. Int. J. Circumpolar Health 73 (1), 23843.

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. Soc. Sci. Med. 135, 124–132.

Daley, K., Jamieson, R., Rainham, D., Truelstrup Hansen, L., 2018a. Wastewater treatment and public health in Nunavut: a microbial risk assessment framework for the Canadian Arctic. Environ. Sci. Pollut. Res. 25 (33), 32860–32872.

Daley, K., Jamieson, R., Rainham, D., Truelstrup Hansen, L., Harper, S.L., 2019. Screening-level microbial risk assessment of acute gastrointestinal illness attributable to wastewater treatment systems in Nunavut, Canada. Sci. Total Environ. 657, 1253–1264.

Desai, S., Miners, A., Smieja, M., Serra, E., Amagoalik, L., Barker, B., Nishi, W., Yansouni, C.P., Pernica, J.M., Barker, K., Goldfarb, D.M., 2017. Results of the multisite Nunavut acute childhood gastrointestinal surveillance project. In: Paper presented at the ArcticNet International Arctic Change Conference. Quebec, Canada. http://www.arcticnetmeetings.ca/ac2017/.

Donaldson, S., Van Oostdam, J., Tikhonov, C., Feeley, M., Armstrong, B., Ayotte, P., Boucher, O., Bowers, W., Chan, L., Dallaaire, F., Dallaire, R., Dewailly, E., Edwards, J., Egeland, G.M., Fontaine, J., Furgal, C., Leech, T., Loring, E., Muckle, G., Nancarrow, T., Pereg, D., Plusquellec, P., Potyrala, M., Receveur, O., Shearer, R.G., 2010. Environmental contaminants and human health in the Canadian Arctic. Sci. Total Environ. 408 (22), 5165–5234.

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

Dupont, H.L., Formal, S.B., Hornick, R.B., Snyder, M.J., Libonati, J.P., Gheahan, D.G., Labrec, E.H., Kalas, J.P., 1971. Pathogenesis of *Escherichia coli* diarrhea. New Engl. J. Med. 285, 1–9.

Ferrer, A., Nguyen-Viet, H., Zinsstag, J., 2012. Quantification of diarrhea risk related to wastewater contact in Thailand. Ecohealth 9, 49–59.

Ford, T., 2005. Water and health. In: Frumkin, H (Ed.), Environmental health: From global to Local. Jossey-Bass, San Francisco, CA, pp. 454–518.

Forget, G., Lebel, J., 2001. An ecosystem approach to human health. Int. J. Occup. Environ. Health 7, 3–38.

Fuhrimann, S., Nauta, M., Pham-Duc, P., Thuy Tram, N., Nguyen-Viet, H., Utziner, J., Cissé, G., Winkler, M.S., 2017. Disease burden due to gastrointestinal infections among people living along the major wastewater system in Hanoi, Vietnam. Adv. Water Resour. 108, 439–449.

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

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

Government of Nunavut (1990). *Public sewerage systems regulations (RRNWT Nu 1990 c P-22)*. http://canlii.ca/t/kgx4. Accessed date: 12 December 2019.

Greenwood, M. (2016). Assessment of Water Quality Impacts in Marine Environments Receiving Municipal Wastewater Effluent Discharges in Nunavut (M.ASc. Thesis). Dalhousie University, Halifax, Nova Scotia. Available from http://dalspace.library.dal.ca.

Gunnarsdóttir, R., Jenssen, P., Jensen, P.E., Villumsen, A., Kallenborn, R., 2013. A review of wastewater handling in the Arctic with special reference to pharmaceuticals and personal care products (PPCPs) and microbial pollution. Ecol. Eng. 50, 76–85.

Gurwith, M., Wenman, W., Gurwith, D., Brunton, J., Feltham, S., Greenberg, H., 1983. Diarrhea among infants and young children in Canada: a longitudinal study in three northern communities. J. Infect. Dis. 147 (4), 685–692.

Haas, C.N., Rose, J.B., Gerba, C.P., 2014. Quantitative Microbial Risk Assessment, 2nd ed. Wiley Publishing, Hoboken, NJ.

Harper, S.L., Edge, V.L., Ford, J., Thomas, M.K., Pearl, D.L., Shirley, J., Indigenous Health Adaptation to Climate Change Research Group, Rigolet Inuit Community Government, McEwen, S.A., 2015a. Acute gastrointestinal illness in two Inuit communities: burden of gastrointestinal illness in Rigolet and Iqaluit, Canada. Epidemiol. Infect. 143 (14), 3048–3063.

Harrison, H.L., Loring, P.A., 2016. Urban harvests: food security and local fish and shellfish in Southcentral Alaska. Agric. Food Secur. 5, 16.

Hastings, E., Yasui, Y., Hanington, P., Goodman, K., CANHelp Working Group, 2014. Community-driven research on environmental sources of *H. pylori* infection in arctic Canada. Gut. Microbes 5 (5), 606–617.

Hayward, J., Jackson, A., Yost, C.K., Truelstrup Hansen, L., Jamieson, R., 2018. Fate of antibiotic resistance genes in two Arctic tundra wetlands impacted by municipal wastewater. Sci. Total Environ. 642, 1415–1428.

Hayward, J., Jamieson, R., Boutilier, L., Goulden, T., Lam, B., 2014. Treatment performance assessment and hydrological characterization of an arctic tundra wetland receiving primary treated municipal wastewater. Ecol. Eng. 73, 786–797.

Health Canada. (2007). *Eating well with Canada's food guide–First Nations, Inuit and Métis* (No. 3426 Cat. H34-159/2007E-PDF). http://publications.gc.ca/site/eng/9.650446/publication.html. Accessed 5 September 2018.

Health Canada, 2012. Guidelines for Canadian Recreational Water Quality, 3rd ed. (Cat. H129-15/2012E). https://www.canada.ca/en/health-canada/services/publications/healthy-living/guidelines-canadian-recreational-water-quality-third-edition.html Accessed 22 May 2020.

Heaney, C.D., Sams, E., Dufour, A.P., Brenner, K.P., Haugland, R.A., Chern, E., Wing, S., Marshall, S., Love, D.C., Serre, M., Noble, R., Water, T.J., 2012. Fecal indicators in sand, sand contact, and risk of enteric illness among beachgoers. Epidemiology 23, 95–106.

Henao-Herreño, L.X., López-Tamayo, A.M., Ramos-Bonilla, J.P., Haas, C.N., Husserl, J., 2017. Risk of illness with Salmonella due to consumption of raw unwashed vegetables irrigated with water from the Bogotá River. Risk Anal. 37 (4), 733–743.

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. Appl. Environ. Microbiol. 71 (6), 3163–3170.

Hennessy, T., Bressler, J., 2016. Improving health in the Arctic region through safe and affordable access to household running water and sewer services: an Arctic Council initiative. Int. J. Circumpolar Health 75, 31149.

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

Huang, Y., Truelstrup Hansen, L., Ragush, C., Jamieson, R., 2018. Disinfection and removal of human pathogenic bacteria in arctic waste stabilization ponds. Environ. Sci. Pollut. Res. 25 (33), 32881–32893.

Hunter, P.R., Zmirou-Navier, D., Hartemann, P., 2009. Estimating the impacts of health of poor reliability of drinking water interventions in developing countries. Sci. Total Environ. 407, 2621–2624.

IDEXX Laboratories Inc. (2013). Colilert-18 test kit procedure. http://www.idexx.com/en/water/water-products-services/colilert-18/. Accessed 22 May 2020.

Inuit Tapiriit Kanatami [ITK]. (2018). *National Inuit Strategy on Research* (ISBN: 978-0-9699774-2-1). https://www.itk.ca/wp-content/uploads/2018/03/National-Inuit-Strategy-on-Research.pdf. Accessed 10 February 2020.

Iqbal, A., Goldfarb, D., Slinger, R., Dixon, B., 2015. Prevalence and molecular characterization of *Cryptosporidium* spp. and *Giardia duodenalis* in diarrhoeic patients in the Qikitani Region, Nunavut, Canada. Int. J. Circumpolar Health 74, 27713.

Johnson, K., Prosko, G., Lycon, D. (2014). The challenges with mechanical wastewater systems in the Far North. Paper presented at the *2014 Annual Conference of Western Canada Water and Waste Association*. Regina, Saskatchewan, Canada. https://issuu.com/cryofront/docs/2016_rev_bookmarked_northern_wastew. Accessed 7 October 2019.

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

Knibbs, L., Sly, P., 2014. Indigenous health and environmental risk factors: an Australian problem with global analogues? Glob Health Action 7, 23766.

Krumhansl, K., Krkosek, W., Greenwood, M., Ragush, C., Schmidt, J., Grant, J., Barrell, J., Lu, L., Lam, B., Gagnon, G., Jamieson, R., 2015. Assessment of Arctic community wastewater impacts on marine benthic invertebrates. Environ. Sci. Technol. 49 (2), 760–766.

Labite, H., Lunani, I., van der Steen, P., Vairavamoorthy, K., Drechsel, P., Lens, P., 2010. Quantitative microbial risk analysis to evaluate health effects of interventions in the urban water system of Accra, Ghana. J. Water Health 8 (3), 417–430.

Leclerc, H., Schwartzbrod, L., Dei-Cas, E., 2002. Microbial agents associated with waterborne disease. Crit. Rev. Microbiol. 28 (4), 371–409.

Leung, M.W., Yen, I.H., Minkler, M., 2004. Community-based participatory research: a promising approach for increasing epidemiology's relevance in the 21st century. Int J Epidemiol 33 (3), 499–506.

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

Mara, D., Sleigh, P., Blumenthal, U., Carr, M., 2007. Health risks in wastewater irrigation: comparing estimates from quantitative microbial risk analysis and epidemiological studies. J. Water Health 5 (1), 39–50.

Mara, D., 2008. A Numerical Guide to Volume 2 of the Guidelines and Practical Advice On How to Transpose Them Into National Standards. World Health Organization, Geneva, Switzerland.

Masina, S., Shirley, J., Allen, J., Sargeant, J.M., Guy, R.A., Wallis, P.M., Weese, J.S., Cunsolo, A., Bunce, A., Harper, S.L., 2019. Weather, environmental conditions, and waterborne *Giardia* and *Cryptosporidium* in Iqaluit, Nunavut. J. Water Health 17 (1), 84–97.

Mathers, C.D., Stein, C., Ma Fat, D., Rao, C., Inoue, M., Tomijima, N., Bernard, C., Lopez, A.D., Murray, C.J.L., 2002. Global Burden of Disease 2000, Version 2: Methods and Results. World Health Organization, Geneva, Switzerland.

McBride, G.B., Stott, R., Miller, W., Bambic, D., Wuertz, S., 2013. Discharge-based QMRA for estimation of public health risks from exposure to stormwater-borne pathogens in recreational waters in the United States. Water Res. 47, 5282–5297.

McCullough, N.B., Eisele, C.W., 1951. Experimental human salmonellosis. II. Immunity studies following experimental illness with *Salmonella meleagridis* and *Salmonella anatum*. J. Immunol. 66 (5), 595–608.

Messner, M.J., Chappell, C.L., Okhuysen, P.C., 2001. Risk assessment for *Cryptosporidium*: a hierarchical Bayesian analysis of human dose response data. Water Res. 35 (16), 3934–3940.

Mosites, E., Miernyk, K., Priest, J.W., Bruden, D., Hurlburt, D., Parkinson, A., Klejka, J., Hennessy, T., Bruce, M.G., 2018. *Giardia* and *Cryptosporidium* antibody prevalence and correlates of exposure among Alaska residents, 2007–2008. Epidemiol. Infect. 146, 888–894.

Moya, J., 2004. Overview of fish consumption rates in the United States. Human Ecol. Risk Assess. 10, 1195–1211.

Murphy, H.M., McBean, E.A., Farahbakhsh, K., 2009. Appropriate technology–A comprehensive approach for water and sanitation in the developing world. Technol. Soc. 31, 158–167.

Neudorf, K., Huang, Y., Ragush, C., Yost, C., Jamieson, R., Truelstrup Hansen, L., 2017. Antibiotic resistance genes in municipal wastewater treatment systems and receiving waters in Arctic Canada. Sci. Total Environ. 598, 1085–1094.

Nevers, M.B., Boehm, A.B., 2011. Modeling fate and transport of fecal bacteria in surface water. In: Sadowsky, M.J., Whitman, R.L. (Eds.), The Fecal Bacteria. American Society for Microbiology, Washington, DC, pp. 165–187.

Nguyen-Viet, H., Zinsstag, J., Schertenleib, R., Zurbrügg, C., Obrist, B., Montangero, A., Surkinkul, N., Koné, D., Morel, A., Cissé, C., Koottatep, T., Bonfoh, B., Tanner, M., 2009. Improving environmental sanitation, health, and well-being: a conceptual framework for integral interventions. Ecohealth 6 (2), 180–191.

Nilsson, L.M., Destouni, G., Berner, J., Dudarev, A.A., Mulvad, G., Oland, J.Ø., Parkinson, A., Tikhonov, C., Rautio, A., Evengård, B., 2013. A call for urgent monitoring of food and water security based on relevant indicators for the Arctic. Ambio 42, 816–822.

O'Fallon, L.R., Dearry, A., 2002. Community-based participatory research as a tool to advance environmental health sciences. Environ. Health Perspect. 110 (Supplement 2), 155–159.

Oracle. (2017). *Oracle Crystal Ball (11.1.2.4.850)*. https://www.oracle.com/applications/crystalball/.

Pardhan-Ali, A., Wilson, J., Edge, V.L., Furgal, C., Reid-Smith, R., Santos, M., McEwen, S. A., 2012b. A descriptive analysis of notifiable gastrointestinal illness in the Northwest Territories, Canada, 1991-2008. BMJ Open 2, e000732.

Pardhan-Ali, A., Wilson, J., Edge, V.L., Furgal, C., Reid-Smith, R., Santos, M., McEwen, S. A., 2013. Community-level risk factors for notifiable gastrointestinal illness in the Northwest Territories, Canada, 1991-2008. BMC Public Health 13, 63.

Priest, H., Usher, P., 2004. The Nunavut wildlife Harvest study: Nunavumi umayukhiukhimayut Ihivgiugutait. Nunavut Wildlife Management Board, Iqaluit, NU.

Ragush, C., Schmidt, J., Krkosek, W., Gagnon, G., Truelstrup Hansen, L., Jamieson, R., 2015. Performance of municipal waste stabilization ponds in the Canadian Arctic. Ecol. Eng. 83, 413–421.

Ramirez-Andreotta, M.D., Brusseau, M.L., Artiola, J.F., Maier, R.M., Gandolfi, A.J., 2014. Environmental research translation: enhancing interactions with communities at contaminated sites. Sci. Total Environ. 497-498, 651–664.

Rendtorff, R.C., 1954. The experimental transmission of human intestinal protozoan parasites: II. *Giardia lamblia* cysts given in capsules. Am. J. Epidemiol. 59 (2), 209–220.

Rose, J.B., Gerba, C.P., 1991. Use of risk assessment for development of microbial standards. Water Sci. Technol. 24 (2), 29–34.

Schmidt, J., Ragush, C., Krkosek, W., Gagnon, G., Jamieson, R., 2016. Characterizing phosphorus removal in passive waste stabilization ponds in Arctic communities. *Arctic* Sci. 2, 1–14.

Schoen, M., Ashbolt, N., 2010. Assessing pathogen risk to swimmers at non-sewage impacted recreational beaches. Environ. Sci. Technol. 44, 2286–2291.

Stetler, R.E., Waltrip, S.C., Hurst, C.J., 1992. Virus removal and recovery in the drinking water treatment train. Water Res. 26, 727–731.

Suk, W., Avakian, M., Carpenter, D., Groopman, J., Scammell, M., Wild, C., 2004. Human exposure monitoring and evaluation in the Arctic: the importance of understanding exposures to the development of public health policy. Environ. Health Perspect. 112 (2), 113–120.

Thivierge, K., Iqbal, A., Dixon, B., Dion, R., Lévesque, B., Cantin, P., Cédilotte, L., Ndao, M., Proulx, J., Yansouni, C., 2016. *Cryptosporidium hominis* is a newly recognized pathogen in the Arctic Region of Nunavik, Canada: molecular characterization of an outbreak. PLoS Negl. Trop. Dis. 10 (4), e0004534.

Thomas, M.K., Murray, R., Flockhart, L., Pintar, K., Pollari, F., Fazil, A., Nesbitt, A., Marshall, B., 2013. Estimates of the burden of food-borne illness in Canada for 30 specified pathogens and unspecified agents, circa 2006. Foodborne Pathog. Dis. 10, 639–648.

Tri-Council (Canadian Institutes of Health Research, Natural Sciences and Engineering Research Council of Canada, and Social Sciences and Humanities Research Council), 2018. Research involving the first nations, Inuit, and Métis Peoples of Canada. In Tri-Council Policy Statement: Ethical Conduct for Research Involving Humans. https://www.pre.ethics.gc.ca. Accessed 10 February 2020.

United States Environmental Protection Agency [USEPA]. (2012b). *Recreational Water Quality Criteria*. Office of Water (830-F-12-058). http://water.epa.gov/scitech/swguidance/standards/criteria/health/recreation/. Accessed 5 September 2018.

Vose, D., 2008. Risk Analysis: A quantitative Guide, 3rd ed. Wiley, Hoboken, NJ.

Weir, M.H., Razzolini, M.R.P., Rose, J.B., Masago, Y., 2011. Water reclamation redesign for reducing *Cryptosporidium* risks at a recreational spray park using stochastic models. Water Res. 45, 6505–6514.

Westrell, T., 2004. *Microbial Risk Assessment and Its Implications For Risk Management in Urban Water Systems* (Ph.D Thesis). Linköping University, Linköping, Sweden. Available from. http://liu.divaportal.org.

Westrell, T., Schonning, C., Stenstrom, T.A., Ashbolt, N., 2004. QMRA and HACCP for management of pathogens in wastewater and sewage sludge treatment and reuse. Water Sci. Technol. 50 (2), 23–30.

World Health Organization [WHO], 2006a. WHO Guidelines For the Safe Use of wastewater, Excreta and greywater. Volume 1. Policy and Regulatory Aspects. WHO Press, Geneva, Switzerland.

World Health Organization [WHO], 2006b. Guidelines For the Safe Use of wastewater, Excreta and greywater. Volume 2. Wastewater use in Agriculture. WHO Press, Geneva, Switzerland.

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

Wright, C.J., Sargeant, J.M., Edge, V.L., Ford, J.D., Farahbakhsh, K., Shiwak, I., Flowers, C., Harper, S.L., Rigolet Inuit Community Government, Indigenous Health Adaptation to Climate Change Research Team, 2018. Water quality and health in northern Canada: stored drinking water and acute gastrointestinal illness in Labrador Inuit. Environ. Sci. Pollut. Res. 25 (33), 32975–32987.

Yapo, R.I., Koné, B., Bonfoh, B., Cissé, G., Zinsstag, J., Nguyen-Viet, H., 2014. Quantitative microbial risk assessment related to urban wastewater and lagoon water reuse in Abidjan, Côte d'Ivoire. J. Water Health 12 (2), 301–309.

Yates, C., Wootton, B., Murphy, S., 2012. Performance assessment of arctic tundra municipal wastewater treatment wetlands through an arctic summer. Ecol. Eng. 44, 160–161.