2023

Integrating Consumer Risk Perception and Awareness With Simulation-Based Quantitative Microbial Risk Assessment Using a Coupled Systems Framework: A case study of private groundwater users in Ontario

Tessa Latchmore
School of Environmental Studies, Queen’s University, Kingston, Ontario, Canada

Sarah Lavallee
School of Environmental Studies, Queen's University, Kingston, Ontario, Canada

Paul Hynds
Technological University Dublin, Paul.Hynds@tudublin.ie

See next page for additional authors

Funder: Queens University Ontario

Authors

Tessa Latchmore, Sarah Lavallee, Paul Hynds, R. Stephen Brown R. Stephen Brown, and Anna Majury

Journal of Environmental Management 331 (2023) 117112

Contents lists available at ScienceDirect

Journal of Environmental Management

journal homepage: www.elsevier.com/locate/jenvman

Research article

Integrating consumer risk perception and awareness with simulation-based quantitative microbial risk assessment using a coupled systems framework: A case study of private groundwater users in Ontario

Tessa Latchmore [a], Sarah Lavallee [a], Paul D. Hynds [b,*], R. Stephen Brown [a], Anna Majury [a,c,**]

[a] School of Environmental Studies, Queen's University, Kingston, Ontario, Canada
[b] Environmental Sustainability and Health Institute, Technological University Dublin, Dublin, Ireland
[c] Public Health Ontario, Kingston, Ontario, Canada

ARTICLE INFO

Keywords:
Private water wells
Quantitative microbial risk assessment
Waterborne illness
Coupled systems framework
Risk management
E. coli

ABSTRACT

Private well users in Ontario are responsible for ensuring the potability of their own private drinking water source through protective actions (i.e., water treatment, well maintenance, and regular water quality testing). In the absence of regulation and limited surveillance, quantitative microbial risk assessment (QMRA) represents the most practical and robust approach to estimating the human health burden attributable to private wells. For an increasingly accurate estimation, QMRA of private well water should be represented by a coupled model, which includes both the socio-cognitive and physical aspects of private well water contamination and microbial exposure. The objective of the current study was to determine levels of waterborne exposure via well water consumption among three sub-groups (i.e., clusters) of private well users in Ontario and quantify the risk of waterborne acute gastrointestinal illness (AGI) attributed to *Giardia*, shiga-toxin producing *E. coli* (STEC) and norovirus from private drinking water sources in Ontario. Baseline simulations were utilized to explore the effect of varying socio-cognitive scenarios on model inputs (i.e., increased awareness, protective actions, aging population). The current study uses a large spatio-temporal groundwater quality dataset and cross-sectional province-wide survey to create socio-cognitive-specific QMRA simulations to estimate the risk of waterborne AGI attributed to three enteric pathogens in private drinking waters source in Ontario. Findings suggest significant differences in the level of exposure among sub-groups of private well users. Private well users within Cluster 3 are characterised by higher levels of exposure and annual illness attributable to STEC, *Giardia* and norovirus than Clusters 1 and 2. Provincial incidence rates of 520.9 (1522 illness per year), 532.1 (2211 illness per year) and 605.5 (5345 illness per year) cases/100,000 private well users per year were predicted for private well users associated with Clusters 1 through 3. Established models will enable development of necessary tools tailored to specific groups of at-risk well users, allowing for preventative public health management of private groundwater sources.

1. Introduction

Human populations and their behaviours represent a critical component of the global hydrogeological cycle through extraction, government policies and social (i.e., individual and/or community) behaviours which drive water usage, quality, quantity, and regulation (i.e., socio-hydrogeology) (Re, 2015; Di Pelino et al., 2019). As such, communities and individuals have the capacity to both contribute to and solve groundwater issues. This is particularly true of private groundwater supplies in Ontario, where an estimated 1.6 million individuals (Statistics Canada, 2019) rely on private wells which are exempt from regulations, including the Ontario Safe Drinking Water Anthonj et al. (2022) and the Ontario Clean Water Act (2006). Thus, private well water users are responsible for ensuring the potability of their own private drinking water source. Protective actions; namely, water treatment, well maintenance, and regular water quality testing (i.e., well water stewardship) are critical in preventing human exposure (via consumption) to waterborne pathogens (Hynds et al., 2013; Fox

* Corresponding author. Environmental Sustainability and Health Institute, Technological University Dublin, Park House, 191 N Circular Rd, Dublin, Ireland.
** Corresponding author. Public Health Ontario, Kingston, Ontario, Canada.
E-mail addresses: hyndsp@tcd.ie (P.D. Hynds), anna.majury@oahpp.ca (A. Majury).

https://doi.org/10.1016/j.jenvman.2022.117112
Received 21 October 2022; Received in revised form 23 November 2022; Accepted 20 December 2022
Available online 19 January 2023
0301-4797/

et al., 2016). However, lack of protective actions has been identified among private well users in Canada and elsewhere. For example, while bacteriological testing is available free of charge to private well users in Ontario, testing rates are considered low relative to the number of wells in the province (Kreutzwiser et al., 2011; Roche et al., 2013; Maier et al., 2014). Limited engagement in protective actions, coupled with a poor understanding of the heath burden attributable to private well water, make it difficult to assess overall well water quality and subsequently develop evidence-based public health interventions for private drinking water systems (Colley et al., 2019).

In the absence of regulation and with limited surveillance, quantitative microbial risk assessment (QMRA) represents the most practical and robust approach to estimating the human health burden attributable to private wells and subsequently inform public health efforts (Owens et al., 2020). A recently developed QMRA of private well water in Ontario, delineated by aquifer type (i.e., consolidated and unconsolidated), indicates that consumption of contaminated private well water may be responsible for 4823 AGI cases annually, highlighting the significant risk contaminated private well water represents in rural Ontario (Latchmore et al., 2022a). While QMRA has been employed across the globe to assist with water safety planning (Smeets et al., 2010; Petterson and Ashbolt, 2016), to date, no QMRA has sought to include the socio-cognitive aspects of microbial exposure actively and empirically (e.g., well owner awareness, perceptions, protective actions). The social barriers to protective actions (e.g., gaps in knowledge related to testing importance) and/or exposure (e.g., private well water consumption patterns) must be considered to improve current outreach strategies and risk communication interventions (Chappells et al., 2015; Re, 2015). Thus, an accurate exposure framework, and by extension QMRA, for private water wells should be represented by a coupled systems framework, which includes both the socio-cognitive and physical (e.g., contamination frequency and duration) aspects of private well water contamination and microbial exposure (Di Pelino et al., 2019).

Protective actions represent the primary measures available to well owners for assessing and/or mitigating their exposure to contaminated private well water; however, the decision to undertake these actions depends on multiple socio-cognitive factors, including perceived risk associated with consuming well water (i.e., based on quality and subsequent consequences) and awareness of contamination sources and pathways (Munene and Hall, 2019; Seliga et al., 2022). Recent studies have identified associations between awareness, risk perception and protective behaviour among private well users (Hynds et al., 2014a; Munene and Hall, 2019; Lavallee et al., 2021a). Lavallee et al. (2022) identified higher mean levels of awareness and groundwater risk perception among users reporting previous well water testing ($p = 0.015$ and $p = 0.023$). However, a significant knowledge gap exists regarding the quantitative link and potential causative effects of cognition on (waterborne) exposure, and the quantitative effect of risk perception and human behaviour on private well water quality (Aerts et al., 2018). Likewise, establishing public health strategies based solely on physical systems may result in inappropriate and/or unsustainable strategies for well users that do not account for behavioural, experiential, or cognitive differences between individuals and communities. Thus, risk assessments (e.g., QMRA) that classify well user exposure based on socio-demographic (e.g., income, education) and socio-cognitive variables (e.g., perception, knowledge, experience), will provide an increasingly accurate depiction of the human health risks associated with private well water, resulting in efficacious public health interventions that are tailored to "risk-specific" sub-groups, as opposed to a "one-size-fits-all" approach.

Accordingly, the current study aimed to build upon methodologies and findings recently presented by Latchmore et al. (2022a) and Lavallee et al. (2022) via development of a "coupled" (i.e., socio-cognitive, and physical systems) probabilistic QMRA for Ontario private well users. More specifically, the current study sought to: (1) use three statistically distinct population clusters identified by Lavallee et al. (2022) to assess and quantify levels of waterborne exposure via well water consumption, (2) utilize the framework proposed by Latchmore et al. (2022a) to quantify the risk of waterborne AGI attributed to *Giardia*, shiga-toxin producing *E. coli* (STEC) and norovirus from private drinking water sources in Ontario, as these represent the most frequently reported protozoan, bacterial and viral water-borne pathogens in Canada, and contribute to significant morbidity and mortality, both regionally and internationally (Hynds et al., 2014a; Wallender et al., 2014; Murphy et al., 2016; Health Canada, 2019; Owens et al., 2020; Sorensen et al., 2021); and (3) explore the effect of varying socio-cognitive scenarios (i.e., increased awareness, protective actions, aging population) on baseline simulations to guide development of future public health interventions and management strategies.

2. Methods

2.1. Integration of private well users risk perception and awareness

A province-wide cross-sectional survey was developed to assess awareness, attitudes, risk perception, health beliefs and behaviours among private well owners across Ontario with respect to their personal well water supply and local/regional sources of groundwater contamination (Lavallee et al., 2021a). Lavallee et al. (2022) characterised private well users based on socio-cognitive risk domains (defined as cognitive factors potentially affecting personal behaviours [i.e., protective actions]). To quantify and compare results, a scoring protocol for four socio-cognitive risk domains (i.e., awareness, risk perception, attitude, and belief) and associated sub-domains (e.g., (personal) source risk perception, regional risk perception) was developed and validated (Lavallee et al., 2021a), followed by two-step cluster analysis to identify sub-groups within the survey cohort based on risk domain scores. Three statistically distinct respondent clusters were identified, as follows: Cluster 1 "low awareness (A) and low source risk perception (SRP)" (Low A/SRP), Cluster 2 "low A and moderate SRP" (Low A/Mod SRP) and Cluster 3 "High A and SRP" (High A/SRP) (Table 1). Hierarchical logistic regression was employed to identify key demographic and experiential variables associated with cluster membership, with further statistical analyses revealing significant associations between cluster membership and protective actions (Table 1). The methodology and results of risk domain scoring protocols, cluster identification and profiling are described in detail by Lavallee et al. (2022) and are presented in Supplementary Materials (Table S1-S12).

Associations between daily well water consumption volume, bottled water use (captured by the province-wide survey) and cluster membership were examined via linear-by-linear association (i.e., ordinal $\chi2$). Specifically, associations between daily well water consumption ranges (i.e., number of cups consumed per day), and cluster membership were examined. Chi-square tests of independence were used to identify associations between categorical (dichotomous) variable pairs (e.g., use of bottled water and cluster membership). To develop a discrete mean value and continuous distribution for use in waterborne QMRA,

Table 1
Summary of socio-cognitive clusters among private well users in Ontario ($n = 1140$) (Lavallee et al., 2022).

Cluster 1 *Low awareness & source risk perception (n = 200; 17.5%)*	Cluster 2 *Low awareness & moderate source risk perception (n = 301; 26.4%)*	Cluster 3 *High awareness & source risk perception (n=639; 56.1%)*
• Northern, Central, and Southeastern Ontario • Gender: Male • Age: 18-24 • Property: Seasonal • Property ownership: Rented	• Education: High School • Gender: Female • Inherited well with property • Bottled water users	• Education: University • Property Type: Permanent • Drilled (i.e., borehole) well users • More likely to test

deterministic midpoint values were assigned to each consumption range based on assumed within-range normality as previously described by Hynds et al. (2012) and Lavallee et al. (2021b).

2.1.1. Cluster-specific QMRA model framework

Distinct cluster-specific (i.e., well user type) risk assessments were developed to produce risk estimates directly reflective of identified private well user socio-cognitive profiles in Ontario (Table 1). Accordingly, the overarching risk assessment framework comprised nine models (i.e., three pathogens x three private well user clusters) to estimate the daily exposure and subsequent human health risk associated with consumption of contaminated private well water in Ontario. The current study utilized the QMRA framework and model inputs developed by Latchmore et al. (2022a), as follows:

Step 1. Exposure Assessment

$$ED = C_{ecoli} \text{ x } 1/R \text{ x } PI \text{ x } V_{DW} \quad (1)$$

where ED = daily pathogen exposure per person (CFU/day; cyst/day; pfu/day), C_{ecoli} = *E. coli* loading (cfu/mL), R = detection sensitivity (%), PI = pathogen contribution (%), Vdw = Daily water consumption (mL/day).

Step 2. Hazard Characterization

$$PDinf\_STEC = [1\text{-}(1 + ED/\beta)]^{\alpha} \quad (2a)$$

$$PDinf\_giardia = 1\text{- } \exp(\text{-}ED^{*}r) \quad (2b)$$

$$PDinf\_norovirus = P^{*}(1\text{-}\exp(\text{-}ED)) \quad (2c)$$

where: PDinf = daily probability of infection (%), ED = daily pathogen exposure per person, β = Beta Poisson parameter value, α = Beta Poisson parameter value, r = Exponential parameter value, P = Fractional Poisson parameter

$$PAinf = [1\text{-}(1\text{-}PDinf)^{DUR}] \text{ x } Fc \quad (2d)$$

where PAinf = annual probability of infection (%), DUR = contamination duration (days/annum), Fc = contamination frequency (%)

Step 3. Risk Characterization

$$\text{Health burden per annum} = PAinf \text{ x } Pop\ (1\text{-}NCon) \text{ x } MR \quad (3)$$

where PAinf = probability of infection per annum (%), Pop = affected population/population at risk (number of consumers on wells, with well water as primary drinking source), NCon = rate of non-consumption (%), MR = morbidity ratio (%).

Statistical distributions were developed and tested using R (version 3.2.1) and RStudio (version 1.1.447) with the add-on packages ggplot2 (version 3.2.1), fitdistrplus (1.0–14) and mc2d (version 0.1–21). Kolmogorov-Smirnov statistics were used to examine goodness-of-fit between fitted parametric distributions and observed data (Delignetter-Muller and Dutang, 2015). All QMRA stochastic iterations were run in R using 2-dimensional Monte Carlo (2DMC) simulations with the package mc2d (Pouillet and Delignetter-Muller, 2010). This approach was selected based on previous risk assessment studies of private groundwater wells (Burch et al., 2021; Latchmore et al., 2022a) as it permits individual and concurrent measurement of the effects of parameter variability and uncertainty on risk estimates and distributions. Each 2DMC simulation was based on 10,000 iterations. A summary of model inputs is provided in Table 2. Sensitivity analysis was carried out using Spearman's rank order correlation coefficient (Rho) to determine the significant input variables contributing to uncertainty in the risk calculations (Cummins et al., 2010; Hunter et al., 2011; Pouillot and Delignette-Muller, 2010).

Table 2
Summary of Cluster-Specific input variables, descriptions, and distributions.

Input	Variable	Description	Distribution	Source
E. coli Loading	C_{ecoli}	Stochastic	Gamma (*shape* = 0.3579, *rate* = 0.0516*)*	Current Study
Contamination frequency	Fc	Stochastic	Beta (α = 0.0169, β = 0.5464)	Current Study
Contamination Duration (days/annum)	DUR	Stochastic	Lognormal (μ = 2.930, σ = 0.5577)	Current Study
Pathogen Contribution *STEC* *Giardia* *Norovirus*	PI	Stochastic	Uniform (min = 0.011, max = 0.099) (Point Estimate) $1.30x10^{-4}$ Uniform (min =$1.64x10^{-5}$, max = $1.60x10^{-1}$)	Murphy et al. (2016) Aerts et al. (2018) Burch et al. (2021)
Detection Sensitivity	R	Stochastic	Uniform (min = 0.93, max = 1)	Hynds et al. (2014b) Unpublished Data
Consumption Volume (mL/day)	V_{dw}	Stochastic [Cluster 1] [Cluster 2] [Cluster 3]	Lognormal (μ = 6.7211, σ = 0.7344; mL = 1084 SD = 737) Lognormal (μ = 6.8859, σ = 0.6623; mL = 1191 SD = 697) Lognormal (μ = 6.9734, σ = 0.6513; mL = 1283 SD = 606)	Current Study
Dose Response: *STEC* *Giardia* *Norovirus*	β:α	Stochastic	Beta-Poisson (α = 398.9 β = 3.96 × 10^4) Exponential (r = 0.0199) Beta-Poisson (α = 0.04 β = 0.05)	Teunis et al. (2008a) Teunis and Havelaar (2002) Rose and Slifko (1999) Teunis et al. (2008a)
Morbidity Ratio: *STEC* *Giardia* *Norovirus*	MR	Stochastic	See Table 4	Meta-analysis (n = 50)
Contamination Rate	CR	Deterministic	0.034	Current Study
Affected Population	Pop	Deterministic	11% of the Ontario population on PW Ontario Pop = 14.07 mill x .11 = **1.57 million private well users**	Statistics Health Canada (2019)
Rate of non-consumption	NCon	Deterministic [Cluster 1] [Cluster 2] [Cluster 3]	0.22 0.23 0.14	Current Study

2.2. Model inputs

2.2.1. Private well water data sources

Model inputs (e.g., *E. coli* loading, contamination frequency, contamination duration) were derived from the Well Water Information System (WWIS) dataset and the Well Water Testing Dataset (WWTD) from 2010 to 2017, inclusive. A full description of dataset development and components has previously been described in detail by Latchmore et al. (2020). Briefly, the WWIS dataset contains all well records and is maintained by the Ontario Ministry of the Environment, Conservation and Parks. Well records include details pertaining to well construction, location, pump test results, geological formation in which the well is constructed, and general information regarding water quality. The WWTD dataset contains all bacteriological testing performed free of charge at one of the eleven provincial laboratories and is maintained by the province of Ontario. All submitted samples are processed and analyzed for Total Coliform and *E. coli* via direct membrane filtration and culture, in compliance with MECP Method #E3407 (Membrane filtration method using DC agar for the simultaneous detection and enumeration of Total Coliforms and *Escherichia coli* in drinking water) (Ministry of the Environment, 2010). The combined WWIS-WWTD dataset contained 702,861 samples from 253,136 wells.

2.2.2. *Escherichia coli concentration and contamination frequency*

To simulate *E. coli* concentration in Ontario private well water, a probabilistic distribution was developed using all contaminated private well samples (≥1 CFU/100 mL) in the WWIS-WWTD data. The gamma distribution provided the most appropriate fit for *E. coli* loading (shape = 0.3579, rate = 0.0516; Table 3; Figure S1).

Contamination frequency was estimated by examining the proportion of wells contaminated at least once per year using the WWTD-WWIS dataset (Latchmore et al., 2020, 2022b). Findings indicate that private wells had *E. coli* present on approximately 3% of sampling occasions i.e., approximately 3 contamination events per 100 samples (Table 3). Sampling results from the entire WWIS-WWTD database were fit to the most appropriate distribution. Goodness of fit tests indicate the beta distribution provided the most appropriate fit (α = 0.0169, β = 0.5464; Figure S2).

2.2.3. *Contamination duration*

An understanding of *E. coli* survival and subsequent contamination sequence duration [defined as the number of days (per annum) with *E. coli* > 0 CFU/100 mL for individual private well water] in private well water is central to developing an accurate QMRA. Accordingly, the WWTD-WWIS dataset was employed to evaluate in-situ *E. coli* die-off rates and subsequent contamination duration (*E. coli*) in Ontario, and the methodology is explained in depth by: Latchmore et al. (2022b). Further, the probability of contamination event overlap [defined as >1 contamination event occurring sequentially, resulting in an event characterised by increased duration and a point increase in concentration] in Ontario private well water was calculated (Latchmore et al., 2022b).

Contamination frequency (number of contamination events per year; Section 1.2.3.2), employed in concurrence with calculated mean concentration and mean time to zero (Table 2) result in an estimated mean duration of 19 (CI 97.5%: 2.92–64.7; min 0 max 314) "annual contamination days" among private well water in Ontario, based on one contamination event. The Kolmogorov-Smirnov statistic indicate the lognormal distribution (μ = 2.930, σ = 0.5577) provides a statistically appropriate fit for contamination duration (Figure S3).

Table 3
Escherichia coli occurrence in the WWIS-WWTD database stratified by year.

Year	*E. coli* positive samples (Annual detection rate)	Mean *E. coli* concentration for positive samples (CFU/100 mL)
2010	3863/96,088 **(4.0%)**	7.30
2011	3669/94,859 **(3.9%)**	7.48
2012	2711/90,790 **(2.9%)**	6.89
2013	3590/87,322 **(4.1%)**	7.30
2014	3074/84,226 **(3.6%)**	7.21
2015	2512/79,437 **(3.2%)**	7.36
2016	1694/79,046 **(2.4%)**	5.53
2017	2265/79,605 **(2.8%)**	4.99
Mean	23,371/691,373 **(3.4%)**	**6.96**

2.2.4. *Pathogen contribution*

Regionally-specific pathogen occurrence data are typically costly and complex to obtain, and as a result, groundwater QMRAs frequently rely on fecal indicator bacteria (FIB) data and/or previously published pathogen estimates (e.g., pathogen to FIB ratios). This is particularly true of unregulated private drinking water systems located in rural/remote areas, making accurate and consistent data collection and analysis challenging (Haas et al., 2000; Howard et al., 2006). Further, the large sample volumes (e.g., up to 100 L) required for detection make it difficult to monitor drinking water for all enteric pathogens (Cabral, 2010; Felleiter et al., 2020). Previous studies have identified positive correlations between *E. coli* in private groundwater systems and the presence and concentration of bacterial pathogens ($r = 0.636$, $p = 0.02$; Hynds et al., 2014b) and viruses ($r = 0.33$, $p < 0.001$; Fox et al., 2016). Therefore, in the current study, the presence and concentration of enteric pathogens were extrapolated from total *E. coli* CFU per 100 mL. In the context of the current risk assessment, pathogen contribution is defined as the likely ratio of enteric pathogens to laboratory confirmed *E. coli* in private well water in Ontario. As described by Latchmore et al., 2023, a uniform distribution of distribution of $1.1x10^{-2}$ to $9.9x10^{-2}$ was employed for STEC. A point estimate ratio of $1.34x10^{-4}$ was used for *Giardia* (Robertson et al., 2006). Finally, a uniform distribution of $1.64x10^{-5}$ to $1.6x10^{-1}$ was employed for norovirus (Silverman et al., 2013).

2.2.5. *Private well water consumption volume and bottled water usage*

Daily self-reported consumption volume was significantly different between the three identified population clusters, therefore daily well water consumption was input independently via three distinct distributions (Table 2). Results indicate a mean daily well water consumption rate of 1084 mL/day (SD = 737 mL/day), 1191 mL/day (SD = 697 mL/day) and 1283 mL/day (SD = 606 mL/day) for Clusters 1, 2 and 3, respectively (Table 2). As previously described by Lavallee et al. (2021b) and Latchmore et al. (2022a), deterministic midpoint ranges were assigned to each consumption range to develop continuous distributions for QMRA. Goodness of fit tests indicate that the log-normal distribution provided an appropriate fit for private well water consumption among all three clusters (Cluster 1: μ = 6.7211, σ = 0.7344; Cluster 2: μ = 6.8859, σ = 0.6623; Cluster 3: μ = 6.9734, σ = 0.6513; Figure S4a to S4c). In addition to consumption volume, the three Clusters differed significantly with respect to bottled water usage. Bottled water usage was deterministically applied to the Cluster-Specific QMRAs (Table 2).

2.2.6. *Dose-response parameters*

The dose-response parameters employed in the current study are based on previously published estimates for shiga-toxin producing *E. coli* (STEC), *Giardia* and norovirus and are equivalent to those used by Latchmore et al. (2022a). Briefly, a Beta-Poisson (BP) model has previously been applied in waterborne QMRA of shiga-toxin producing *E. coli* (Haas et al., 2000; Hynds et al., 2014b; Murphy et al., 2016). More specifically, the hierarchical BP dose-response model developed by Teunis et al. (2008a) was employed in the current study using 95th percentile parameters ($\alpha = 398.9$, $\beta = 3.96 \times 10^4$; Equation (2a)). For *Giardia*, the exponential dose-response curve (Equation (2b)) was used with $r = 0.0199$ (Rose and Slifko, 1999; Teunis et al., 2008b; Balderrama-Carmona et al., 2014). Finally, the modified BP model (Equation (2a)) is most frequently used for simulating norovirus exposure in waterborne QMRA (Teunis et al., 2008a; Murphy et al., 2016). The BP model was used with the parameters α = 0.04 and β = 0.05 (Teunis et al., 2008a).

2.2.7. *Population at risk*

The total at-risk population was estimated from Census data (Statistics Canada, 2019) and the province-wide survey (Lavallee et al., 2021b). The use of a treatment system was not significantly different among identified clusters (Lavallee et al., 2022). More specifically, approximately 43.5% (83/191), 45.6% (135/296) and 44.7% (285/637) of individuals in Clusters 1 through 3, who treat their well water, use a treatment system effective against microbial contamination (e.g., reverse osmosis, membrane filtration, shock chlorination) (Lavallee et al., 2022). Thus, all individuals who rely on private well water were considered at-risk regardless of the presence of well water treatment. Private well owners may not provide the required maintenance of their treatment systems and/or some well owners incorrectly believe their system is effective against microbial contamination (e.g., water softener, iron removal) (Kreutzwiser et al., 2011; Malecki et al., 2017; Lavallee et al., 2021b). Therefore, the total at-risk population was estimated to be 1,573,775 (Statistics Canada, 2019). Further delineation was undertaken based on the composition of each cluster, whereby Clusters 1 through 3 represent 17.5%, 26.4% and 56.1% of the total at-risk population and are assumed to be uniform across the entire population (Lavallee et al., 2022). Cluster profiles were unable to be ascertained for children as the age distribution of the province wide survey was predominately individuals >25 years old (99.3%; 1161/1169).

2.2.8. Scenario analysis

Scenario analyses simulate potential future outcomes by considering the effect of alternative and/or changing events on model inputs (Schweizer and Kurniawan, 2016). Developed 'Cluster-Specific' QMRA simulations were used to assess the likely impact of a successful public health intervention and/or policy addressing the major socio-cognitive gaps presented in Clusters 1 through 3 i.e., 'Cluster 4'. More specifically, the hypothetical Cluster 4 represents the 'ideal' well owner in terms of public health protection through which well and groundwater stewardship is undertaken appropriately and consistently via well water quality testing, treatment, and maintenance. Additionally, scenarios that consider the transitioning of private well owners between clusters over time was examined. For example, well users in Cluster 2 are characterised by "short-term residency" (i.e., recently inheriting the well with the property; Table 1). However, over time, survey results indicate shifting behaviours as well users gain more experience with and/or confidence in their well, thus transitioning to Cluster 3. Scenario analysis was undertaken using multivariable nodes as a third dimension in the mc2d package (Delignetter-Muller and Dutang, 2015).

2.2.9. Pooled-analyses

To provide risk estimates relevant to both scientific and public health authorities, several pathogen-specific clinical variables are necessary including attack rate, hemolytic uremic syndrome rates (specific to STEC) and hospitalization rate. Thus, fifty of first most relevant published academic peer-reviewed articles relating to outbreaks of waterborne *Giardia*, norovirus and STEC were analyzed (Table S13). Kolmogorov-Smirnov tests were employed to assess goodness-of-fit between parametric distributions and empirical metadata for development of statistical (input) distributions (Delignetter-Muller and Dutang, 2015) (Table 4).

3. Results

3.1. Exposure assessment

Members of Cluster 3 consumed significantly more well water than Clusters 1 and 2 ($p = 0.004$, $p = 0.022$). Conversely, Clusters 1 and 2 consumed significantly more bottled water than Cluster 3 ($p = 0.002$; $p = 0.027$). Thus, Cluster-specific exposure assessments and consequent QMRAs were developed to account for varying levels of consumption and bottled water usage between clusters. Cluster-specific QMRA simulations predict that 97.5% of private well water users who drink from an *E. coli* contaminated source, during a contamination event, consume 0–17 *E. coli* CFU/day, with a predicted mean daily exposure of 5 CFU/day, 6 CFU/day and 12 CFU/day for Clusters 1 through 3, respectively. Maximum likely rates of ingestion found at high *E. coli* concentrations (e.g., >70 CFU/day) in concurrence with high daily well water consumption (>3500 mL/day) equated to 82 CFU/day, 89 CFU/day and 131 CFU/day for Clusters 1 through 3, respectively.

3.2. Hazard characterization

Equations 2 a-d were used to calculate the daily and annual probabilities of infection, during a contamination event, from pathogen contaminated private well water for all three Clusters. Overall, highest daily and annual probabilities of infection within all clusters are attributable to norovirus followed by *Giardia* and STEC (Table 5). For example, the predicted mean daily likelihood of symptomatic infection for individuals in Cluster 1 during a contamination event were 2.3% SD $= 1.1\text{x}10^{-1}$), 0.95% (SD $= 6.0\text{x}10^{-2}$) and 0.017% (SD $= 2.3\text{x}10^{-3}$), for norovirus, STEC and *Giardia*, respectively. The predicted mean annual probability of infection ranged from 4.2% (SD $= 2.5\text{x}10^{-2}$) for STEC to 9.1% (SD $= 5.6\text{x}10^{-2}$) for norovirus. Likewise, for private well users in Cluster 2 (Low A & Mod SRP), predicted daily probabilities of infection ranged from 0.020% (SD $2.3\text{x}10^{-3}$) to 1.9% (SD $9.5\text{x}10^{-2}$), with the predicted annual probability of infection ranging from 4.5% (SD $8.4\text{x}10^{-2}$) for STEC to 9.5% (SD $5.7\text{x}10^{-2}$) for norovirus. Highest predicted mean daily probabilities of infection were attributed to Cluster 3 (High A & SRP), equating to 2.5% (SD $= 1.1\text{x}10^{-1}$), 1.0% (SD $= 6.4\ \text{x}10^{-2}$) and 0.027% (SD $= 2.6\text{x}10^{-3}$) for norovirus, STEC and *Giardia*, respectively. The mean annual probability of infection for private well users in Cluster 3 ranged from 0.039% (SD $= 6.8\text{x}10^{-4}$) for *Giardia* to 12% (SD $= 6.9\text{x}10^{-}$

Table 4
Results of pooled – analyses ($n = 50$) and distribution fitting for clinical variables. Based on previous literature the beta-general, PERT, lognormal, logistic, normal and pareto distributions were all tested for fit.

Variable	n	Range	Distribution	Parameters	Goodness of fit *p*
Morbidity Ratio	14[1]	Min – 6.7%	Logistic	$\mu = 0.224$ $s = 0.095$	0.87
	21[2]	Max – 64%	Normal	$\mu = 0.243$	0.85
	15[3]	Min – 3.1%	Normal	$\sigma = 0.169$	0.78
		Max – 92%		$\mu = 0.243$	
		Min – 5%		$\sigma = 0.169$	
		Max – 63%			
HUS rate[1]	7	Min – 0.3%	Normal	$\mu = 0.077$	0.948
		Max – 27.2%		$\sigma = 0.027$	
Hospitalization rate	9	Min – 7.5%	Normal	0.013	0.999
	2[2]	Max – 35.7%		0.030	
	2[3]	Deterministic			
		Deterministic			
Mortality rate[1]	5[1]	Min – 0.3%	Logistic	$\mu = 0.023$ $s = 0.011$	0.95
		Max – 6.3%			

1 STEC data
2 Norovirus data
3 Giardia data

Table 5
Daily and annual probability of infection of STEC, *Giardia* and norovirus attributable to private well water in Ontario delineated by cluster membership.

	Daily and Annual Probability of Infection (PD*inf*)		
	Cluster 1	Cluster 2	Cluster 3
STEC	$9.5x10^{-3}$ (SD $6.0x10^{-2}$) [a] $4.2x10^{-2}$ (SD $8.1x10^{-2}$) [b]	$9.9x10^{-3}$ (SD $6.3x10^{-2}$) [a] $4.5x10^{-2}$ (SD $8.4x10^{-2}$) [b]	$1.0x10^{-2}$ (SD $6.4x10^{-2}$) [a] $5.0x10^{-2}$ (SD $8.9x10^{-2}$) [b]
Norovirus	$2.3x10^{-2}$ (SD $1.1x10^{-1}$) [a] $9.1x10^{-1}$ (SD $6.7x10^{-2}$) [b]	$1.9x10^{-2}$ (SD $9.5x10^{-2}$) [a] $9.5x10^{-2}$ (SD $5.7x10^{-2}$) [b]	$2.5x10^{-2}$ (SD $1.1x10^{-1}$) [a] $1.2x10^{-1}$ (SD 6.9 x 10^{-2}) [b]
Giardia	$1.7x10^{-4}$(SD $2.3x10^{-3}$) [a] $2.9x10^{-4}$ (SD $6.9x10^{-4}$) [b]	$2.0x10^{-4}$ (SD $2.3x10^{-3}$) [a] $3.1x10^{-4}$ (SD $6.1x10^{-4}$) [b]	$2.7x10^{-4}$ (SD $2.6x10^{-3}$) [a] $3.9x10^{-4}$ (SD $6.8x10^{-4}$) [b]

a = daily probability of infection, b = annual probability of infection

^{2}) for norovirus.

3.3. Risk characterization

Cluster-specific QMRA simulations predict a total of 9,078 cases of illness in Ontario annually across all three pathogens due to consumption of contaminated private well water, equating to a crude incidence rate (CIR) of 576.8/100,000 private well owners (1 infection per 200 well users). Private well users within Cluster 3 are predicted to be at a higher risk of illness compared to Clusters 1 and 2; simulations predict a total of 5,345 (CIR = 605.5/100,000) total illnesses per year attributable to contaminated private well water in Cluster 3, while Cluster 1 and Cluster 2 are associated with 1,522 [CIR = 520.9/100,000] and 2,211 [CIR = 532.1/100,000] cases of illness per year, respectively (Table 6). A majority of cases within all three clusters are associated with norovirus (75.4%; 6,845/9,078), followed by STEC (24.1%; 2,189/9,078) and *Giardia* (0.48%; 44/ 9,078). Overall, 286 hospitalizations (CI 97.5%: 89 – 334) per year are predicted to occur due to primary infections (i.e., does not include secondary infections) from untreated, contaminated private well water in Ontario (Table 7). Private well users within Cluster 3 account for approximately 53% (152/286) of hospitalizations, with infection attributable to STEC accounting for 69.1% (105/152) of hospitalizations, followed by norovirus (30.3%; 46/152) and *Giardia* (0.66%;1/152). Based on a pooled-analyses of hemolytic uremic syndrome (HUS) contraction rates, Cluster-Specific QMRA simulations predict a total of 35 cases per year, with 21 (59%), 9 (31.8%) and 5 (9%) cases occurring among private well owners within Clusters 3, 2 and 1, respectively. Consequently, approximately one to five HUS-related deaths are predicted to occur in Ontario due to contaminated private well water consumption.

Overall, 286 hospitalizations (CI 97.5%: 89 – 334) per year are predicted to occur due to primary infections (i.e., does not include secondary infections) from untreated, contaminated private well water in Ontario (Table 7). Private well users within Cluster 3 account for approximately 53% (152/286) of hospitalizations, with infection attributable to STEC accounting for 69.1% (105/152) of hospitalizations, followed by norovirus (30.3%; 46/152) and *Giardia* (0.66%;1/152). Based on a pooled-analyses of hemolytic uremic syndrome (HUS) contraction rates, Cluster-Specific QMRA simulations predict a total of 35 cases per year, with 21 (59%), 9 (31.8%) and 5 (9%) cases occurring among private well owners within Clusters 3, 2 and 1, respectively. Consequently, approximately one to five HUS-related deaths are predicted to occur in Ontario due to contaminated private well water consumption.

Table 6
Predicted illnesses per annum attributable to STEC, *Giardia* and norovirus from untreated private well water in Ontario for Cluster 1, 2 and 3.

	Annual Illnesses [Crude Incidence Rates]		
	Cluster 1	Cluster 2	Cluster 3
STEC	308 (147 – 410) [111.8/100,000]	541 (262-717) [130.2/100,000]	1,340 (662 – 1,850) [152.7/100,000]
Giardia	6 (0 – 83) [2.2/ 100,000]	11 (0 – 120) [2.6/ 100,000]	27 (9 – 75) [3.2/ 100,000]
Norovirus	1,208 (753-1,349) [399.3/100,00]	1,659 (1,063 – 1,837) [438.6/ 100,000]	3,978 (2,586 – 4,391) [450.6/ 100,000]
Total Illness	1,522 [520.9/ 100,000]	2,211 [532.1/ 100,000]	5,345 [605.5/ 100,000]

3.4. Scenario and sensitivity analysis

QMRA simulations were used to examine a scenario whereby an idealised “Cluster 4” exists and was characterised by implementation of a public health intervention that aims to increase and maintain well water stewardship behaviours (i.e., comprehensive testing programs, well and treatment system maintenance) among private well users from Cluster 3 (high A and SRP). Results indicate that the burden of infection within Cluster 3 would decrease by approximately 91% from 5,345 to 536 cases for all three pathogens in the case of overarching behavioural change (i.e., a successful intervention reaching 100% of the Cluster 3 population) (Fig. 1). More specifically, behaviour change may include and be measured by an increase in well water testing by private well users, coupled with treatment system installation in high-risk areas and/or sources (e.g., shallow wells, areas with thin overburden (0 – 2 m)). Alternatively, for an effective intervention reaching 50% of the Cluster 3 population and resulting in behavioural change (i.e., increased well water stewardship), total annual illness would decrease from 5,345 to 2,677 cases. An additional scenario considering the transitioning of private well users from Cluster 2 to Cluster 3 was also examined (i.e., aging population profile). Results indicate that total annual illness may increase from 5,345 to 7,098 cases per year for all three pathogens if 50% of the Cluster 2 population were to transition to Cluster 3, without implementation of a public health intervention. Alternatively, with just 10% of Cluster 2 members transitioning, total annual illness would increase by approximately 12.5% from 5,345 to 6,021 case per year.

Sensitivity analysis identified contamination frequency (Fc) as the model parameter with the greatest overall impact on all twelve models (correlation coefficient ranging from 0.751 to 0.844), followed by pathogen contribution (P1) and *E. coli* loading (C_{ecoli}) (Table S12).

4. Discussion

The differing socio-demographic profiles, individual awareness, perceptions, and behaviours among private well users represent a significant limitation for developing effective, sustainable, and focused public health interventions using a “one-size-fits-all” approach. Demographically and experientially specific (i.e., well user clusters) exposure assessments and subsequent risk estimates are therefore critical in quantifying realistic and accurate levels of exposure to waterborne pathogens in the community (Brouwer et al., 2018). Categorising levels of risk among private well users represents a significant step forward in identifying the extent to which well user behaviours influence exposure to waterborne pathogens, thus informing development of evidence-based, bespoke tools and interventions. To the authors knowledge, the current study is the first to employ socio-cognitively-specific QMRA simulations to estimate the risk of AGI attributable to enteric pathogens across private well user sub-groups. Developed models provide an effective and transferable template for public health authorities and governments to preserve and protect private well water for current and future well users.

Table 7
Predicted HUS rates, hospitalization rates and mortality per annum attributable to STEC, *Giardia* and norovirus from untreated private well water in Ontario delineated by cluster membership.

	HUS cases			Hospitalizations			Mortality		
	Cluster 1	Cluster 2	Cluster 3	Cluster 1	Cluster 2	Cluster 3	Cluster 1	Cluster 2	Cluster 3
STEC	5 (0–8)	9 (014)	21 (6–34)	35 (0–98)	62 (22–110)	105 (59–198)	1 (0–4)	2 (0–4)	5 (0–10)
Giardia	NA			0 (0–11)	0 (0–15)	1 (0–22)	NA		
Norovirus	NA			13 (0–29)	24 (4–55)	46 (8–109)	NA		
Total	35			48	86	152	8		

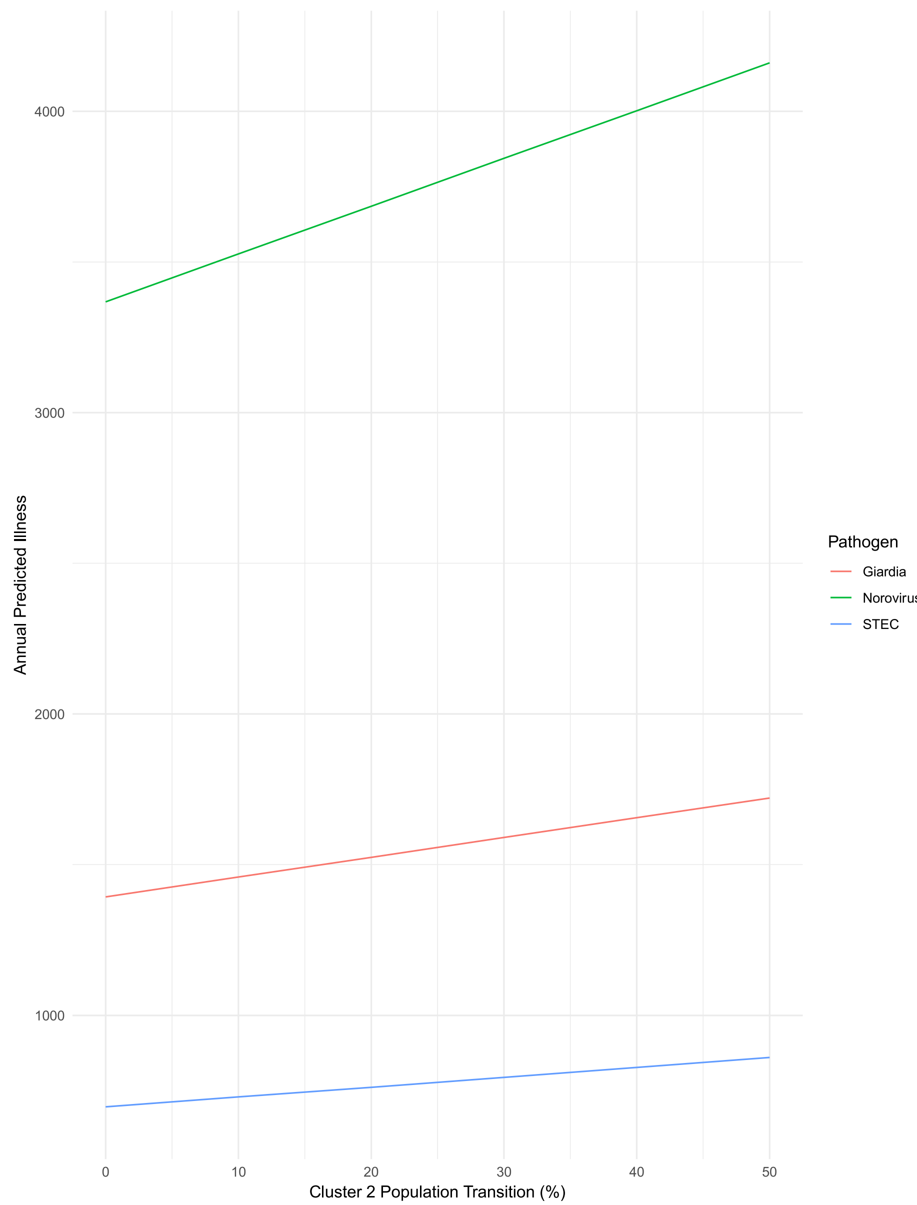

Fig. 1. Results of the scenario analysis of private well users within Cluster 2 transitioning to Cluster 3 without implementation of a successful public health intervention on number of predicted illnesses per year attributable to norovirus, *Giardia* and Shiga-toxin producing *E. coli*.

4.1. Cluster – specific waterborne AGI rates

QMRA simulations predict that private well users within Cluster 3 are characterised by higher levels of exposure (via contaminated private well water consumption) and annual illness attributable to STEC, *Giardia* and norovirus than Clusters 1 and 2. More specifically, provincial incidence rates of 520.9 (1,522 illness per year), 532.1 (2,211 illness per year) and 605.4 (5,345 illness per year) cases/100,000 private well users per year were predicted for private well users associated with Clusters 1 to 3, respectively. The Guidelines for Drinking Water Quality established by Health Canada use an annual target risk of 1×10^{-6} DALYs per person per year (Health Canada, 2019), thus the risk estimates simulated as part of the current study clearly and significantly exceed the Health Canada tolerable risk level by several orders of magnitude. A previous QMRA of Ontario private well water conducted by the authors estimates that consumption of contaminated private well water in Ontario is responsible for approximately 4,648 AGI cases annually. Thus, current study estimates of 9,078 AGI cases annually are nearly double that of previous Ontario estimates. The inclusion of well user socio-cognitive attributes via cluster profiles allows for inclusion of increasingly and user-specific simulation inputs (e.g., well and bottled water consumption), likely leading to a more accurate depiction of the health burden attributable to private well water. Thus, the authors consider that previous QMRAs that do not account for well user behaviour likely under-estimate the risk associated with private well water. It is important to note that the previous QMRA by Latchmore et al. (2022a) accounted for individuals who reported effective microbial treatment, while the current cluster-specific QMRA assumes all individuals who rely on private well water are at-risk regardless of the presence of well water treatment due to lack of significance associated with treatment system among clusters (section 2.2.7). Nonetheless, when treatment failure (e.g., lack of effective microbial treatment) was examined (via scenario analysis) in Latchmore et al. (2022a), total annual illness increased to 6,244 cases per year, approximately 30% less than the total annual illness estimated in the current study.

Private well owners within Cluster 3 (i.e., "high awareness and source risk perception") typically consume significantly more well water and less bottled water than private well owners from Clusters 1 and 2. Moreover, while private well owners from Cluster 3 were more likely to report previous supply testing, approximately 41% (235/579) of these users indicated that they had only had their well water tested once (Lavallee et al., 2022). Recent studies have reported that well owners who test their groundwater supply with insufficient frequency (≤ once a year) tend to exhibit a high degree of (unfounded) confidence in their well water quality, resulting in a belief that subsequent testing is unnecessary (Chappells et al., 2015; Musacchio et al., 2021). Further, Colley et al. (2019) report that the longer an individual has resided in the household, the less likely they are to have tested their well water in the last five years. Accordingly, private well owners within Cluster 3 exhibit complacency toward the potability of their well water due to previous "acceptable" test results, with length of residency likely playing a role. "One-off" well water testing is insufficient to determine the long-term potability of a private groundwater supply given *E. coli* detection varies significantly based on dynamic spatial (e.g., local agricultural cycles) and temporal (e.g., seasonal rainfall) factors (Atherholt et al., 2015; Latchmore et al., 2020; Qayyum et al., 2020). Complacency regarding one's private well water quality results in an increased risk of exposure to waterborne pathogens via consumption of untested and/or untreated private well water. Future work should seek to examine if clusters are directly associated with groundwater quality, thus compounding higher rates of exposure via increased consumption. It is critical that private well users are made aware of the comprehensive testing program specific to their local groundwater environment and the appropriate tools to identify and manage water quality fluctuations at the household and community level (Morris et al., 2016; Anthonj et al., 2022). For example, a tool that provides private well owners with source-specific remediation recommendations following a contamination event (e.g., well maintenance, treatment, re-testing), specific to season (e.g., time of year) and well location may decrease health risk via increased well water stewardship.

Private well users within Cluster 1 (Low A & SRP) are characterised by the lowest predicted levels of exposure. More specifically, Cluster 1 members were characterised by a significantly higher proportion of young male (18–24 years old) renters (i.e., not homeowners; Table 1). Low awareness and source risk perception may be associated with lower levels of private well water consumption as individuals are unaware of how to ensure potability of their private well water supply and thus, perceive bottled water as safer and easier (i.e., lower level of effort to ensure safety) (Jones et al., 2006; Hu et al., 2011; Roche et al., 2013; Gholson et al., 2018). Similarly, private well users within Cluster 2 (Low A & Mod SRP) were significantly more likely to have inherited the well with their property in concurrence with consuming bottled water (Lavallee et al., 2022). A lack of experience with well construction (i.e., well siting and drilling) may play a fundamental role in groundwater exposure and waterborne illness insofar as private well users who were not present during well construction (e.g., Cluster 1 [property renters] and 2 [inherited the well]) may perceive bottled water as being safer than their private water supply, thus reducing exposure to waterborne pathogens (Lavallee et al., 2021b). However, engaging in protective actions is considerably less expensive than single-use plastics, with frequent and routine participation in these practises (e.g., well water testing, treatment) providing a similar level of human health protection. Well owner awareness is significantly related to confidence in maintaining the well and previous experience with well contractors during the construction process has been correlated with an increased awareness of possible contamination sources and the importance of appropriate well location (Roche et al., 2012; Hynds et al., 2013; Lavallee et al., 2021a; Mooney et al., 2021). Future public health interventions should aim to increase awareness among young adults and new homeowners regarding how best to protect and maintain their private well water through source maintenance, treatment, and frequent testing as opposed to source avoidance and subsequent overuse of other less environmentally sustainable resources (e.g., bottled water).

Overall, 152 hospitalizations, 21 cases of HUS and five HUS-related deaths per year due to infections attributable to contaminated private well water are predicted to occur within Cluster 3 (High A & SRP). As previously mentioned, higher levels of awareness exhibited among Cluster 3 members may be misconstrued as overconfidence in their long-term well water potability/stability, even in the absence of appropriate well stewardship behaviours (Lavallee et al., 2021a). Overconfidence in the quality of their water supply may lead to further undiagnosed or misdiagnosed waterborne illnesses as the well user may fail to attribute acquired illnesses to their private drinking water source, leading to clinicians not being consulted, or if consulted, not given all the pertinent information to make a correct diagnosis for provincial notification (Hrudey and Hrudey, 2007). Previous research has indicated that well owners are aware of contamination risks in their region (e.g., proximity to livestock and septic systems, flooding), but may not appropriately "downscale" this risk to their own water supply unless there is a notable change in organoleptic properties (e.g., taste, smell, colour) (Hynds et al., 2014b; Malecki et al., 2017; Munene and Hall, 2019; Lavallee et al., 2021b; Mooney et al., 2021). Further, results from the current survey cohort show that while private well user awareness was generally high (71%), findings indicate a low level of awareness related to waterborne pathogens, with 50.8% of respondents unable to identify any pathogens. Thus, well owners may fail to appreciate the link between water and health due to normally good health and/or misattribute infection to foodborne or secondary transmission, ultimately leading to continued use of unsafe drinking water sources (Ochoo et al., 2017; Reynolds et al., 2020; Seliga et al., 2022). It is important to note that providing education related to waterborne disease and pathogens does not always directly result in behaviour change (i.e., increase in well water stewardship) if well users are not provided with the appropriate means to personally recognize risks and subsequently implement proactive behaviours (Rundblad, 2008; Gupta et al., 2012; Anthonj et al., 2022). For example, an adverse test result (>0 *E. coli* CFU/100 mL) may result in little or no action to reduce exposure if an individual is unable to interpret the results in relation to waterborne pathogens and human health risk, hasn't the financial means to install a treatment system, or remains unsure as to whether (or when) they should re-test their water supply (Chappells et al., 2015; Flanagan et al., 2018).

4.2. Scenario analysis: public health interventions and cluster transition

It may be anticipated that socio-cognitive and socio-demographic patterns will change over-time via governance (e.g., programs etc.) and/or population aging or movement. The Ontario population is projected to increase by 35.6% (from 14.6 to 19.8 million individuals) from

2019 to 2046, with the number of elderly individuals (>65 years old) projected to effectively double (2.5–4.9 million individuals) over the same period (Government of Ontario, 2020). Further, the rural population (i.e., those dependent on private well water) of Ontario grew by approximately 2% between 2016 and 2021, with rural growth projected to double by 2025 (Statistics Canada, 2022a,b). Public health interventions and educational campaigns surrounding private well water stewardship will be fundamental in ensuring the health and safety of Ontario's current and future private well users and its vulnerable sub-populations (e.g., the elderly, pregnant women, infants, immunocompromised) (Gerba et al., 1996).

Results of scenario analyses indicate the critical and time-sensitive need for tailored public health interventions and tools for private well users that can address water quality and well maintenance requirements in real-time. The potential outcome of an efficacious public health intervention indicates that the burden of infection within Cluster 3 would decrease by approximately 91% from 5,345 to 536 cases across all three pathogens if 100% of the Cluster 3 population were successfully targeted (Fig. 2). Conversely, total annual illnesses were projected to increase from 5,345 to 7,098 cases cases per year across all three pathogens if 50% of the Cluster 2 sub-population transition to Cluster 3, in the absence of any public health intervention (Fig. 1). The estimates and the future projections presented in the current study may serve as a baseline and/or cost benefit analyses for public health authorities and government officials to develop the necessary tools and educational supports that are tailored to specific groups (i.e., Clusters) of at-risk well users, allowing for preventative public health management of private groundwater sources, which are often overlooked at the provincial and federal level (Hrudey et al., 2006).

4.3. Recommendations

Results from the current study highlight the complexity of private well water exposure and the limitations in developing "one-size-fits-all" tools and interventions for the private well user. To date, no tools have been developed for well-users that consider both the physical and social aspects of private well water exposure. Notably, bottom-up approaches, particularly those facilitated by "pocket technologies" (i.e., smartphone

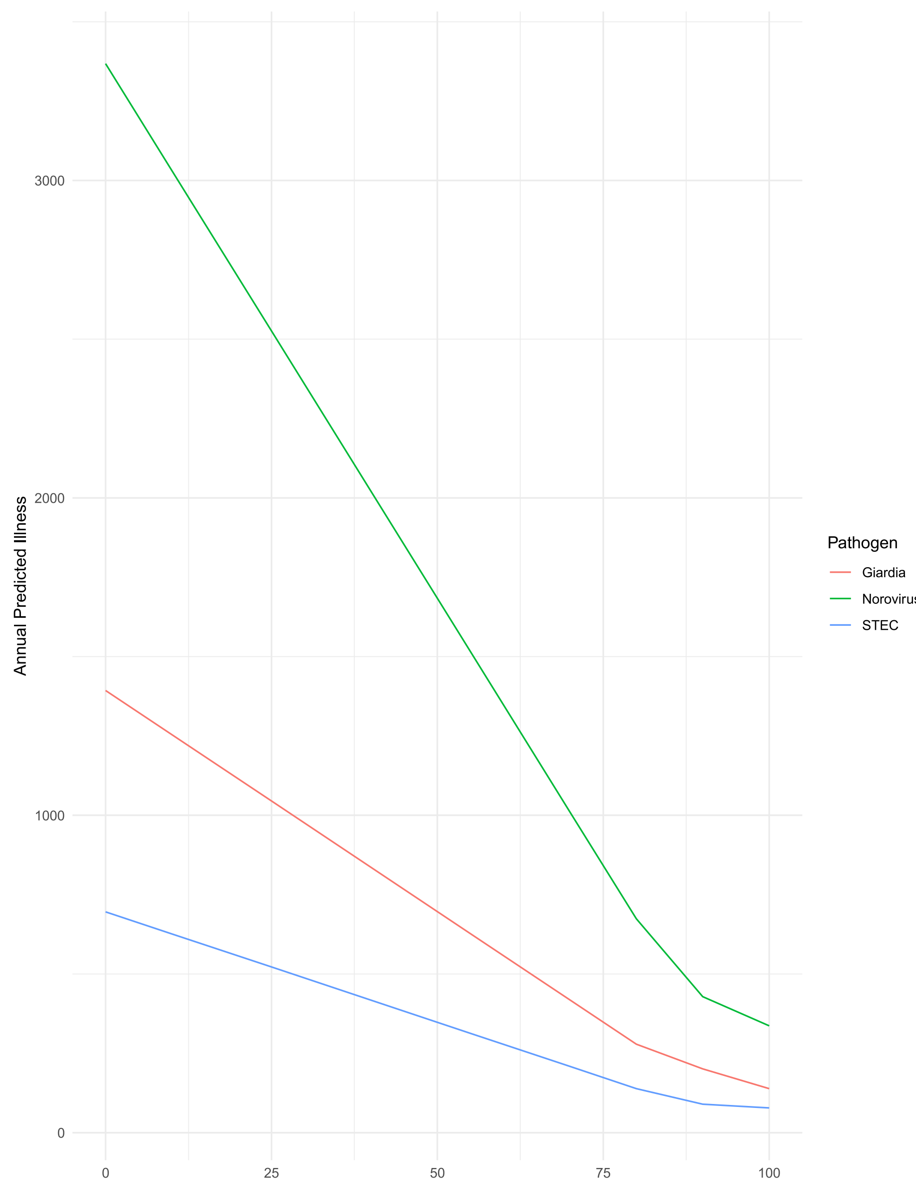

Fig. 2. Results of the scenario analysis of the implementation of a successful public health intervention and the creation of a 'Cluster 4' (i.e., ideal well user) on number of predicted illnesses per year attributable to norovirus, *Giardia* and Shiga-toxin producing *E. coli*.

applications), have been identified as a potential path forward for risk communication that not only engage well users but facilitate private well stewardship at the household level (Hynds et al., 2018; Hoffman et al., 2019). A well user-specific tool or web-based application could be used to identify and mitigate the risk associated with private well water, communicating risks, interpretations, and solutions in real-time. A smartphone or web-based application would have the potential for widespread impact, with reminders for well water testing (and re-testing), logging and interpretation of test results, well maintenance and source protection customized to the well user's physical location, socio-demographic, and socio-cognitive attributes. For example, Latchmore et al. (2020 & 2022b) indicated the risk associated with private well water contamination significantly differs based on hydrogeological setting, season and well depth, and found the mean duration and overlap probability of a contamination event in private well water in Ontario to be "aquifer-specific", with calculated mean contamination durations of 18 days (consolidated aquifers) and 11 days (unconsolidated aquifers), respectively. Thus, results from studies examining the risk factors for private well water contamination in Ontario, such as the current study, could be used as a foundation for tool development.

A tool of this kind would require significant partnerships across disciplines, including but not limited to public health authorities, well inspectors, government officials, (hydro)geologists, software engineers, and application programmers, in addition to resources for development, implementation across diverse demographics, and updating (Hoffman et al., 2019). However, the risk of waterborne AGI attributable to private well water in Ontario identified in this study, as well as Latchmore et al. (2022a) highlights the need for a significant investment in tools that will safeguard private drinking water sources for current and future generations. For example, it has been estimated that the cost per case of AGI in Ontario is $1089 when over-the counter medications, lost patient or parental work time and costs to the health care system are included (Majowicz et al., 2006). The current study estimate of 9,078 AGI cases per year attributable to private well water would incur costs to the province of approximately $9.8 million per year, not including the additional cost to the well owner for possible well maintenance and/or treatment installation. Accounting for inflation, this is equivalent to $13.9 million per year in the current economic climate (Statistics Canada, 2022). Thus, the cost of implementing a province-wide tool or application would be of significant benefit to private well users and greatly outweigh the cost of retroactively managing and remediating one or more waterborne outbreaks which cost both private and public systems millions of dollars (Vicente and Christoffersen, 2006; Chyzheuskaya et al., 2017; Gilpin et al., 2020; Collier et al., 2021).

4.4. Limitations

While QMRA frameworks represent the most practical and robust approach for estimating the human health burden attributable to private well water, they encompass inherent limitations. In the current study, and the majority of groundwater-related QMRAs, pathogen co-occurrence and contribution represent a notable data limitation, as the authors were unable to directly analyze the *E. coli*-to-pathogen ratio across the study area and were thus compelled to rely on previously published estimates, potentially misrepresenting the overall human health risk. Further, while the current study findings leave little doubt as to the effect that cluster membership has on waterborne exposure, the authors were unable to determine if cluster membership directly influences raw groundwater quality. While Hynds et al. (2013) did not find a correlation between consumer awareness and *E. coli* concentration in the Republic of Ireland, future research should seek to assess cluster membership (which also includes individual risk perception) and private well water quality concurrently via sampling in the province of Ontario. Additionally, given the varying survival times and transport mechanisms associated with viruses and protozoa in untreated groundwater due to differences in microbial characteristics, mechanisms, and resistance to treatment, findings should be interpreted and employed cautiously. Finally, given the age demographic of the province-wide survey, children could not be accurately or definitively represented in the current Cluster-Specific QMRA framework. However, it may be presumed that adults in the household assume primary responsibility for engaging in protective actions. Further, given the overall novelty of the presented socio-cognitive QMRA, it is important to establish baseline risk estimates before applying the framework to diverse populations and age-groups not currently captured by Clusters 1 through 3.

5. Conclusions

Overall, to the authors knowledge, the current study is the first to employ socio-cognitively-specific QMRA simulations to estimate the risk of AGI attributable to enteric pathogens across private well user subgroups. Significant differences in the level of exposure among subgroups of private well users were identified, thus, future risk assessments should aim to classify well user exposure based on socio-demographic (e.g., income, education) and socio-cognitive variables (e.g., perception, knowledge, experience). Private well users within Cluster 3 are characterised by higher levels of exposure and annual illness attributable to STEC, *Giardia* and norovirus than Clusters 1 and 2, highlighting the need for tools that are tailored to specific groups of at-risk well users, allowing for preventative public health management of private groundwater sources.

Credit author statement

Tessa Latchmore: Conceptualization, Data curation, Formal analysis, Investigation, Methodology, Writing- original draft, Writing-review & editing. **Sarah Lavallee**: Conceptualization, Data curation, Formal analysis, Investigation, Methodology, Writing- original draft, Writing-review & editing **Paul Hynds**: Conceptualization, Investigation, Methodology, Supervision, Writing-review & editing. **R. Stephen Brown**: Conceptualization, Funding acquisition, Supervision, Writing-review & editing. **Anna Majury**: Conceptualization, Funding acquisition, Supervision, Writing-review & editing.

Declaration of competing interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

Data availability

The data that has been used is confidential.

Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.jenvman.2022.117112.

References

Aerts, J.C., Botzen, W.J., Clarke, K.C., Cutter, S.L., Hall, J.W., Merz, B., Michel-Kerjan, E., Mysiak, J., Surminski, S., Kunreuther, H., 2018. Integrating human behaviour dynamics into flood disaster risk assessment. Nat. Clim. Change 8 (3), 193–199. https://doi.org/10.1038/s41558-018-0085-1.

Anthonj, C., Setty, K., Ferrero, G., Poague, K., Yaya, A.M.A., Augustijn, E.W., Marsh, A.J., 2022. Do health risk perceptions motivate water-and health-related behaviour? A systematic literature review. Sci. Total Environ. 819, 152902 https://doi.org/10.1016/j.scitotenv.2021.152902.

Atherholt, T.B., Korn, L.R., Louis, J.B., Procopio, N.A., 2015. Repeat sampling and Coliform bacteria detection rates in New Jersey domestic wells. Groundw. Monit. Remediat. https://doi.org/10.1111/gwmr.12094.

Balderrama-Carmona, A.P., Gortáres-Moroyoqui, P., Álvarez-Valencia, L.H., Castro-Espinoza, L., Mondaca-Fernández, I., de Jesús Balderas-Cortés, J., et al., 2014.

Occurrence and quantitative microbial risk assessment of Cryptosporidium and *Giardia* in soil and air samples. Int. J. Infect. Dis. 26, 123–127.

Brouwer, Andrew F., Masters, Nina B., Joseph, NS Eisenberg, 2018. Quantitative microbial risk assessment and infectious disease transmission modeling of waterborne enteric pathogens. Current environmental health reports 5 (2), 293–304. https://doi.org/10.1007/s40572-018-0196-x.

Burch, T.R., Stokdyk, J.P., Spencer, S.K., KiekeJr, B.A., Firnstahl, A.D., Muldoon, M.A., Borchardt, M.A., 2021. Quantitative Microbial Risk Assessment for Contaminated Private Wells in the Fractured Dolomite Aquifer of Kewaunee County, Wisconsin. Environmental Health Perspectives. https://doi.org/10.1289/EHP7815.

Cabral, J.P., 2010. Water microbiology. Bacterial pathogens and water. Int. J. Environ. Res. Publ. Health 7 (10), 3657–3703. https://doi.org/10.3390/ijerph7103657.

Chappells, H., Campbell, N., Drage, J., Fernandez, C.V., Parker, L., Dummer, T.J., 2015. Understanding the translation of scientific knowledge about arsenic risk exposure among private well water users in Nova Scotia. Sci. Total Environ. 505, 1259–1273. https://doi.org/10.1016/j.scitotenv.2013.12.108.

Chyzheuskaya, A., Cormican, M., Srivinas, R., O'Donovan, D., Prendergast, M., O'Donoghue, C., Morris, D., 2017. Economic assessment of waterborne outbreak of cryptosporidiosis. Emerg. Infect. Dis. 23 (10), 1650.

Cummins, E., Kennedy, R., Cormican, M., 2010. Quantitative risk assessment of Cryptosporidium in tap water in Ireland. Sci. Total Environ. 408 (4), 740–753. https://doi.org/10.1016/j.scitotenv.2009.11.008.

Colley, S.K., Kane, P.K., MacDonald Gibson, J., 2019. Risk communication and factors influencing private well testing behavior: a systematic scoping review. Int. J. Environ. Res. Publ. Health 16 (22), 4333. https://doi.org/10.3390/ijerph16224333.

Collier, S.A., Deng, L., Adam, E.A., Benedict, K.M., Beshearse, E.M., Blackstock, A.J., et al., 2021. Estimate of burden and direct healthcare cost of infectious waterborne disease in the United States. Emerg. Infect. Dis. 27 (1), 140. https://doi.org/10.3201/eid2701.190676.

Delignette-Muller, M.L., Dutang, C., 2015. fitdistrplus: an R package for fitting distributions. J. Stat. Software 64 (4), 1–34.

Di Pelino, Schuster-Wallace, Corinne, Hynds, Paul D., Dickson-Anderson, Sarah E., Anna, Majury, 2019. A coupled-systems framework for reducing health risks associated with private drinking water wells. Can. Water Resour. J./Revue canadienne des ressources hydriques 44 (3), 280.

Felleiter, S., McDermott, K., Hall, G., Sheth, P., Majury, A., 2020. Exploring private water wells for fecal sources and evidence of pathogen presence in the context of current testing practices for potability in Ontario. Water Quality Research Journal 55 (1), 93–105. https://doi.org/10.2166/wqrj.2019.035.

Flanagan, S.V., Gleason, J.A., Spayd, S.E., Procopio, N.A., Rockafellow-Baldoni, M., Braman, S., Chillrud, S.N., Zheng, Y., 2018. Health protective behavior following required arsenic testing under the New Jersey Private Well Testing Act. Int. J. Hyg Environ. Health 221 (6), 929–940. https://doi.org/10.1016/j.ijheh.2018.05.008.

Fox, M.A., Nachman, K.E., Anderson, B., Lam, J., Resnick, B., 2016. Meeting the public health challenge of protecting private wells: proceedings and recommendations from an expert panel workshop. Sci. Total Environ. 554, 113–118. https://doi.org/10.1016/j.scitotenv.2016.02.128.

Gerba, C.P., Rose, J.B., Haas, C.N., 1996. Sensitive populations: who is at the greatest risk? Int. J. Food Microbiol. 30 (1–2), 113–123. https://doi.org/10.1016/0168-1605(96)00996-8.

Gholson, D.M., Boellstorff, D.E., Cummings, S.R., Wagner, K.L., Dozier, M.C., 2018. Consumer water quality evaluation of private and public drinking water sources. J. Water Health 16 (3), 369–379. https://doi.org/10.2166/wh.2018.206.

Gilpin, B.J., Walker, T., Paine, S., Sherwood, J., Mackereth, G., Wood, T., et al., 2020. A large scale waterborne Campylobacteriosis outbreak, Havelock North, New Zealand. J. Infect. 81 (3), 390–395.

Gupta, N., Mutukkanu, T., Nadimuthu, A., Thiyagaran, I., Sullivan-Marx, E., 2012. Preventing waterborne diseases: analysis of a community health worker program in Rural Tamil Nadu, India. J. Community Health 37 (2), 513–519.

Haas, C.N., Thayyar-Madabusi, A., Rose, J.B., Gerba, C.P., 2000. Development of a dose-response relationship for *Escherichia coli* O157: H7. Int. J. Food Microbiol. 56 (2–3), 153–159. https://doi.org/10.1016/S0168-1605(99)00197-X.

Health Canada, 2019. Guidance on the Use of Quantitative Microbial Risk Assessment in Drinking Water. Water and Air Quality Bureau, Healthy Environments and Consumer Safety Branch. Health Canada, Ottawa, Ontario (Catalogue No. H144-59/2019E-PDF).

Hoffman, T., Hynds, P., Schuster-Wallace, C., Dickson-Anderson, S., Majury, A., 2019. Harnessing smart technology for private well risk assessment and communication. Water Security 6, 100026. https://doi.org/10.1016/j.wasec.2019.100026.

Howard, G., Pedley, S., Tibatemwa, S., 2006. Quantitative microbial risk assessment to estimate health risks attributable to water supply: can the technique be applied in developing countries with limited data? J. Water Health 4 (1), 49–65. https://doi.org/10.2166/wh.2006.0004.

Hrudey, S.E., Hrudey, E.J., Pollard, S.J., 2006. Risk management for assuring safe drinking water. Environ. Int. 32 (8), 948–957. https://doi.org/10.1016/j.envint.2006.06.004.

Hrudey, S.E., Hrudey, E.J., 2007. Published case studies of waterborne disease outbreaks—evidence of a recurrent threat. Water Environ. Res. 79 (3), 233–245. https://doi.org/10.2175/106143006X95483.

Hu, Z., Morton, L.W., Mahler, R.L., 2011. Bottled water: United States consumers and their perceptions of water quality. Int. J. Environ. Res. Publ. Health 8 (2), 565–578. https://doi.org/10.3390/ijerph8020565.

Hunter, P.R., De Sylor, M.A., Risebro, H.L., Nichols, G.L., Kay, D., Hartemann, P., 2011. Quantitative microbial risk assessment of cryptosporidiosis and *giardia*sis from very small private water supplies. Risk Anal.: Int. J. 31 (2), 228–236. https://doi.org/10.1111/j.1539-6924.2010.01499.x.

Hynds, P.D., Misstear, B.D., Gill, L.W., 2012. Investigation of water consumption patterns among Irish adults for waterborne quantitative microbial risk assessment (QMRA). J. Water Health 10 (3), 453–464. https://doi.org/10.2166/wh.2012.025.

Hynds, P.D., Misstear, B.D., Gill, L.W., 2013. Unregulated private wells in the Republic of Ireland: consumer awareness, source susceptibility and protective actions. J. Environ. Manag. 127, 278–288. https://doi.org/10.1016/j.jenvman.2013.05.025.

Hynds, P.D., Thomas, M.K., Pintar, K.D.M., 2014a. Contamination of groundwater systems in the US and Canada by enteric pathogens, 1990–2013: a review and pooled-analysis. PLoS One 9 (5), e93301. https://doi.org/10.1371/journal.pone.0093301.

Hynds, P.D., Gill, L.W., Misstear, B.D., 2014b. A quantitative risk assessment of verotoxigenic E. coli (VTEC) in private groundwater sources in the Republic of Ireland. Hum. Ecol. Risk Assess. 20 (6), 1446–1468. https://doi.org/10.1080/10807039.2013.862065.

Hynds, P., Regan, S., Mooney, E., O'Dwyer, J., 2018. Development of a pocket technology Groundwater Risk Application for Local Evaluation (GRAppLE) in highly groundwater reliant populations. In: EGU General Assembly Conference Abstracts, p. 7685.

Jones, A.Q., Dewey, C.E., Doré, K., Majowicz, S.E., McEwen, S.A., David, W.T., Eric, M., Carr, D.J., Henson, S.J., 2006. Public perceptions of drinking water: a postal survey of residents with private water supplies. BMC Publ. Health. https://doi.org/10.1186/1471-2458-6-94.

Kreutzwiser, R., de Loe, R., Imgrund, K., Conboy, M.J., Simpson, H., Plummer, R., 2011. Understanding stewardship behaviour: factors facilitating and constraining private water well stewardship. J. Environ. Manag. 92 (4), 1104–1114. https://doi.org/10.1016/j.jenvman.2010.11.017.

Latchmore, T., Hynds, P., Brown, R.S., Schuster-Wallace, C., Dickson-Anderson, S., McDermott, K., Majury, A., 2020. Analysis of a large spatiotemporal groundwater quality dataset, Ontario 2010–2017: informing human health risk assessment and testing guidance for private drinking water wells. Sci. Total Environ. 738, 140382 https://doi.org/10.1016/j.scitotenv.2020.140382.

Latchmore, T., Hynds, P.D., Brown, R.S., McDermott, K., Majury, A., 2022a. Assessing the risk of acute gastrointestinal illness attributable to three enteric pathogens from contaminated private water wells in Ontario. Int. J. Hyg Environ. Health.

Latchmore, T., Hynds, P.D., Brown, R.S., McDermott, K., Majury, A., 2022b. Estimating the duration and overlap of *Escherichia coli* contamination events in private groundwater supplies for quantitative risk assessment using a multiannual (2010–2017) provincial dataset. Environ. Pollut. 309, 119784 https://doi.org/10.1016/j.envpol.2022.119784.

Lavallee, S., Hynds, P.D., Brown, R.S., Schuster-Wallace, C., Dickson-Anderson, S., Di Pelino, S., et al., 2021a. Examining influential drivers of private well users' perceptions in Ontario: a cross-sectional population study. Sci. Total Environ. 763, 142952 https://doi.org/10.1016/j.scitotenv.2020.142952.

Lavallee, S., Latchmore, T., Hynds, P.D., Brown, R.S., Schuster-Wallace, C., Anderson, S. D., Majury, A., 2021b. Drinking water consumption patterns among private well users in Ontario: implications for exposure assessment of waterborne infection. Risk Anal. https://doi.org/10.1111/risa.13676.

Lavallee, S., Hynds, P., Brown, R.S., Majury, A., 2022. Classification of Sub-populations for Quantitative Risk Assessment Based on Awareness and Perception: A Cross-Sectional Population Study of Private Well Users in Ontario. Science of the Total Environment, 159677. https://doi.org/10.1016/j.scitotenv.2022.159677.

Maier, A., Krolik, J., Randhawa, K., Majury, A., 2014. Bacteriological testing of private well water: a trends and guidelines assessment using five years of submissions data from southeastern Ontario. Can. J. Public Health 105 (3), 203–208. https://doi.org/10.17269/cjph.105.4282.

Majowicz, S.E., McNab, W.B., Sockett, P., Henson, S., Doré, K., Edge, V.L., Buffett, M.C., Fazil, A., Read, S., McEwen, S., Stacey, D., 2006. Burden and cost of gastroenteritis in a Canadian community. J. Food Protect. 69 (3), 651–659. https://doi.org/10.4315/0362-028X-69.3.651.

Malecki, K.M., Schultz, A.A., Severtson, D.J., Anderson, H.A., VanDerslice, J.A., 2017. Private-well stewardship among a general population based sample of private well-owners. Sci. Total Environ. 601, 1533–1543. https://doi.org/10.1016/j.scitotenv.2017.05.284.

Ministry of the Environment, Conservation and Parks, 2010. Protocol of Accepted Drinking Water Testing Methods Version 2.0. E3388–The Determination of N-Nitrosamines in Water by Gas Chromatography-High Resolution Mass Spectrometry (GC/HRMS), Laboratory Services Branch. Ontario Ministry of the Environment Ontario.

Mooney, S., O'Dwyer, J., Lavallee, S., Hynds, P.D., 2021. Private groundwater contamination and extreme weather events: the role of demographics, experience and cognitive factors on risk perceptions of Irish private well users. Sci. Total Environ. 784, 147118 https://doi.org/10.1016/j.scitotenv.2021.147118.

Morris, L., Wilson, S., Kelly, W., 2016. Methods of conducting effective outreach to private well owners - a literature review and model approach. J. Water Health 14, 167–182. https://doi.org/10.2166/wh.2015.081.

Munene, A., Hall, D.C., 2019. Factors influencing perceptions of private water quality in North America: a systematic review. Syst. Rev. 8 (1), 1–15. https://doi.org/10.1186/s13643-019-1013-9.

Murphy, H., Thomas, M., Schmidt, P., Medeiros, D., McFadyen, S., Pintar, K., 2016. Estimating the burden of acute gastrointestinal illness due to *Giardia*, Cryptosporidium, Campylobacter, *E. coli* O157 and norovirus associated with private wells and small water systems in Canada. Epidemiol. Infect. 144 (7), 1355–1370. https://doi.org/10.1017/S0950268815002071.

Musacchio, A., Andrade, L., O'Neill, E., Re, V., O'Dwyer, J., Hynds, P.D., 2021. Planning for the health impacts of climate change: flooding, private groundwater contamination and waterborne infection–A cross-sectional study of risk perception,

experience and behaviours in the Republic of Ireland. Environ. Res. 194, 110707 https://doi.org/10.1016/j.envres.2021.110707.
Ochoo, B., Valcour, J., Sarkar, A., 2017. Association between perceptions of public drinking water quality and actual drinking water quality: a community-based exploratory study in Newfoundland (Canada). Environ. Res. 159, 435–443. https://doi.org/10.1016/j.envres.2017.08.019.
Owens, C.E., Angles, M.L., Cox, P.T., Byleveld, P.M., Osborne, N.J., Rahman, M.B., 2020. Implementation of quantitative microbial risk assessment (QMRA) for public drinking water supplies: systematic review. Water Res. 174, 115614 https://doi.org/10.1016/j.watres.2020.115614.
Petterson, S.R., Ashbolt, N.J., 2016. QMRA and water safety management: review of application in drinking water systems. J. Water Health 14 (4), 571–589. https://doi.org/10.2166/wh.2016.262.
Pouillot, R., Delignette-Muller, M.L., 2010. Evaluating variability and uncertainty separately in microbial quantitative risk assessment using two R packages. Int. J. Food Microbiol. 142 (3), 330–340. https://doi.org/10.1016/j.ijfoodmicro.2010.07.011.
Qayyum, S., Hynds, P., Richardson, H., McDermott, K., Majury, A., 2020. A geostatistical study of socioeconomic status (SES), rurality, seasonality and index test results as drivers of free private groundwater testing in southern Ontario, 2012–2016. Sci. Total Environ. https://doi.org/10.1016/j.scitotenv.2020.137188.
Re, V., 2015. Incorporating the social dimension into hydrogeochemical investigations for rural development: the Bir Al-Nas approach for socio-hydrogeology. Hydrogeol. J. 23 (7), 1293–1304. https://doi.org/10.1007/s10040-015-1284-8.
Reynolds, C., Checkley, S., Chui, L., Otto, S., Neumann, N.F., 2020. Evaluating the risks associated with Shiga-toxin-producing *Escherichia coli* (STEC) in private well waters in Canada. Can. J. Microbiol. 66 (5), 337–350. https://doi.org/10.1139/cjm-2019-0329.
Robertson, L.J., Hermansen, L., Gjerde, B.K., 2006. Occurrence of Cryptosporidium oocysts and Giardia cysts in sewage in Norway. Applied and Environmental Microbiology 72 (8), 5297–5303.
Roche, S.M., Jones, A.Q., Majowicz, S.E., McEwen, S.A., Pintar, K.D.M., 2012. Drinking water consumption patterns in Canadian communities (2001–2007). J. Water Health 10 (1), 69–86. https://doi.org/10.2166/wh.2011.051.
Roche, S.M., Jones-Bitton, A., Majowicz, S.E., Pintar, K.D., Allison, D., 2013. Investigating public perceptions and knowledge translation priorities to improve water safety for residents with private water supplies: a cross-sectional study in Newfoundland and Labrador. BMC Publ. Health 13 (1), 1–13. https://doi.org/10.1186/1471-2458-13-1225.
Rose, J.B., Slifko, T.R., 1999. *Giardia*, Cryptosporidium, and Cyclospora and their impact on foods: a review. J. Food Protect. 62 (9), 1059–1070. https://doi-org.proxy.queensu.ca/10.4315/0362-028X-62.9.1059.
Rundblad, G., 2008. The semantics and pragmatics of water notices and the impact on public health. J. Water Health 6 (S1), 77–86. https://doi.org/10.2166/wh.2008.130.
Schweizer, V.J., Kurniawan, J.H., 2016. Systematically linking qualitative elements of scenarios across levels, scales, and sectors. Environ. Model. Software 79, 322–333. https://doi.org/10.1016/j.envsoft.2015.12.014.
Seliga, A., Spayd, S.E., Procopio, N.A., Flanagan, S.V., Gleason, J.A., 2022. Evaluating the impact of free private well testing outreach on participants' private well stewardship in New Jersey. J. Water Health 20 (1), 1–11. https://doi.org/10.2166/wh.2021.018.
Silverman, A.I., Akrong, M.O., Amoah, P., Drechsel, P., Nelson, K.L., 2013. Quantification of human norovirus GII, human adenovirus, and fecal indicator organisms in wastewater used for irrigation in Accra, Ghana. Journal of water and health 11 (3), 473–488.
Smeets, P.W.M.H., Rietveld, L.C., Van Dijk, J.C., Medema, G.J., 2010. Practical applications of quantitative microbial risk assessment (QMRA) for water safety plans. Water Sci. Technol. 61 (6), 1561–1568. https://doi.org/10.2166/wst.2010.839.
Sorensen, J.P., Aldous, P., Bunting, S.Y., McNally, S., Townsend, B.R., Barnett, M.J., et al., 2021. Seasonality of enteric viruses in groundwater-derived public water sources. Water Res. 207, 117813.
Statistics Canada, 2019. Table 38-10-0274-01 Households and the Environment Survey, Dwelling's Main Source of Water. https://doi.org/10.25318/3810027401-eng.
Statistics Canada, 2022a. Table 18-10-0004-01 Consumer Price Index, Monthly, Not Seasonally Adjusted. https://doi.org/10.25318/1810000401-eng.
Statistics Canada, 2022b. Population Projections for Canada (2018 to 2068), Provinces and Territories (2018 to 2043): Technical Report on Methodology and Assumptions. Statistics Canada catalogue number 91-620. https://www150.statcan.gc.ca/n1/pub/91-620-x/91-620-x2019001-eng.htm.
Teunis, P.F., Havelaar, A.H., 2002. Risk assessment for protozoan parasites. Int. Biodeterior. Biodegrad. 50 (3–4), 185–193. https://doi.org/10.1016/S0964-8305(02)00085-9.
Teunis, P.F.M., Ogden, I.D., Strachan, N.J.C., 2008a. Hierarchical dose response of *E. coli* O157: H7 from human outbreaks incorporating heterogeneity in exposure. Epidemiol. Infect. 136 (6), 761–770. https://doi.org/10.1017/S0950268807008771.
Teunis, P.F., Moe, C.L., Liu, P., Miller, E., Lindesmith, L., Baric, R.S., Calderon, R.L., 2008b. Norwalk virus: how infectious is it? J. Med. Virol. 80 (8), 1468–1476.
Vicente, K.J., Christoffersen, K., 2006. The Walkerton *E. coli* outbreak: a test of Rasmussen's framework for risk management in a dynamic society. Theor. Issues Ergon. Sci. 7 (2), 93–112.
Wallender, E.K., Ailes, E.C., Yoder, J.S., Roberts, V.A., Brunkard, J.M., 2014. Contributing factors to disease outbreaks associated with untreated groundwater. Ground Water 52 (6), 886–897.
Government of Ontario, 2020. Ontario Population Projections; Available at: https://www.ontario.ca/page/ontario-population-projections#foot-1 (Accessed 17th January, 2023).