Titre: Title:	Vulnerability assessment of drinking water intakes to microbial contamination during combined sewer overflows under global change: A bottom-up approach
Auteurs: Authors:	Baptiste Leveque, Albert Irakiza Shyaka, Mouhamed Ndong, Raja Kammoun, Jean-Baptiste Burnet, Sarah Dorner, & Françoise Bichai
Date:	2025
Type:	Article de revue / Article
Référence: Citation:	Leveque, B., Shyaka, A. I., Ndong, M., Kammoun, R., Burnet, J.-B., Dorner, S., & Bichai, F. (2025). Vulnerability assessment of drinking water intakes to microbial contamination during combined sewer overflows under global change: A bottom-up approach. PLOS Water, 21 pages. https://doi.org/10.1371/journal.pwat.0000352

PLOS Water

RESEARCH ARTICLE

Vulnerability assessment of drinking water intakes to microbial contamination during combined sewer overflows under global change: A bottom-up approach

Baptiste Leveque[1], Albert Irakiza Shyaka[1], Mouhamed Ndong[2], Raja Kammoun[1]*, Jean-Baptiste Burnet[1], Sarah Dorner[1], Françoise Bichai[1]

1 Department of Civil, Geological and Mining engineering, Polytechnique Montréal, Montréal, Quebec, Canada, **2** Department of Chemical Engineering and Applied biology, Cheikh Anta Diop University, Dakar, Senegal

* raja.kammoun@polymtl.ca

Citation: Leveque B, Shyaka AI, Ndong M, Kammoun R, Burnet J-B, Dorner S, et al. (2025) Vulnerability assessment of drinking water intakes to microbial contamination during combined sewer overflows under global change: A bottom-up approach. PLOS Water 4(5): e0000352. https://doi.org/10.1371/journal.pwat.0000352

Editor: Daniel Reddythota, Faculty of Water Supply & Environmental Engineering, ArbaMinch Water Technology Institute (AWTI), ETHIOPIA

Received: November 15, 2024

Accepted: March 17, 2025

Published: May 28, 2025

Data availability statement: The data utilized in this study were obtained from municipalities and are subject to confidentiality agreements,

Abstract

Combined sewer overflows (CSOs) are a major cause of microbial contamination in urban rivers, especially during summer low flow periods. This study analyzes the vulnerability of drinking water intakes (DWIs) to CSO-derived microbial contamination in an urban river located in Quebec, Canada, under global change. The vulnerability assessment of DWIs was based on the *Escherichia coli* (*E. coli)* concentrations and conducted using a novel bottom-up approach. Unlike the traditional top-down approach, the bottom-up approach incorporates a wide range of climate information sources, including historical data, stochastic climate simulations, and outputs from General Circulation Models, without the need for extensive recalibration or reliance on downscaled models. It also allows local capacities and system-specific factors to be taken into account, providing a more adaptable framework for regions with limited data. *E. coli* concentrations from CSOs were generated stochastically, while hydrographs were generated by a deterministic method. A hydrodynamic and water quality models were used to investigate the impact of simultaneous overflows, their duration, *E. coli* concentrations, and peak overflow and river flow. The study revealed a significant impact of simultaneous overflows on the mean and maximum simulated *E. coli* concentrations at DWIs, particularly during extended CSO durations and with higher discharged *E. coli* concentrations. The extreme-low river flow rates significantly increased mean and peak *E. coli* concentrations, altering the pollutograph shape at DWIs. Future climate projections indicate a decrease in summer low flows, potentially exacerbating the vulnerability of water sources to CSO contamination. Source water protection plans need to consider vulnerable periods, characterized by reduced contaminant dilution alongside high numbers of simultaneous overflows, high contaminants concentration, and prolonged durations. The bottom-up approach

which prevent their public sharing. However, all relevant data necessary to replicate the findings are fully presented within the main text and supplemental materials of this paper. Researchers requiring additional information may contact the corresponding author, who can facilitate requests with the data providers in accordance with the applicable legal and ethical frameworks.

Funding: This work was partially funded through the start-up fund of F. Bichai (Polytechnique Montreal) (FB) and the NSERC CREATE project PURE (BL) (grant number: 528220-2019), and received financial support from the Ministère de l'Environnement et de la Lutte contre les Changements Climatiques (MELCC) of Québec, Canada (SD) (grant number: 04 5856). The funders had no role in study design, data collection and analysis, decision to publish, or preparation of the manuscript. Funds from NSEC CREATE project PURE and MELCC provided scholarships to Baptiste Leveque (BL).

Competing interests: The authors have declared that no competing interests exist.

proposed in this study can be applied in jurisdictions with limited data and covers a range of potential risks using probabilistic scenarios, including extreme scenarios, without using a hydroclimatic model.

1. Introduction

Globally, combined sewer overflows (CSOs) remain one of the major sources of microbial contamination of surface waters in urban watersheds [1–7]. CSOs frequently cause critical microbial contamination events in receiving watercourses during summer low flows, which are often preceded by high-intensity and/or long-lasting precipitation episodes [3,8].

Climate change, the greatest challenge of the 21st century [9], poses major challenges due to the vulnerability of natural and human ecosystems to meteorological events and anthropogenic pressures such as urbanization and land use change [10]. Global change threatens both the quantity and quality of surface waters [11–14] which serve as major sources of drinking water supplies to a large part of the global urban population [15]. The predicted impact of climate change in various countries and regions around the world is expected to result in an increase the untreated water volume, frequency, and/or duration of overflows from combined sewer structures into rivers or lakes, as is the case in Canada [16,17], the United States [18], France [19], Italy [20], England [21], Germany [22], and Norway [23]. Other recent studies that assessed the impact of global change on the microbial quality of surface waters in highly urbanized territories pointed out that microbial surface water quality depends mainly on anthropogenic changes (population growth), but less on climate change [24–26]. Despite the disparity in watersheds across such studies (pedology and geology, land use and cover, etc.), human impacts generally appeared to be the predominant risk factor.

Surface water microbial quality raises public health concerns, particularly due to the proximity of many drinking water intakes (DWIs) from overflow structures. Depending on the number and location of these structures relative to DWIs, contamination levels at DWIs can be high [3,27]. However, drinking water production systems are vulnerable to peak contamination events at DWIs, as well as to the highly variable characteristics of CSOs, including duration, microbial concentration, and load [3,27–29]. This vulnerability arises because drinking water system operation relies on more stable conditions [30].

While an increase in *E. coli* concentrations may occur, it does not necessarily trigger an adjustment in treatment levels. Treatment processes often operate based on average *E. coli* concentrations, which may not directly respond to fluctuations in peak concentrations. Thus, peak contamination events are considered as major risk factors for public health, especially in the absence of adequate treatment and operation [31]. Therefore, a comprehensive understanding of how climate change influences both average and peak *E. coli* concentrations from CSOs at DWIs is necessary to support decision-making on the implementation of effective risk control measures at the source.

Numerous studies have investigated microbial risks and hydraulic vulnerabilities associated with CSOs under climate change, employing both quantitative and qualitative methodologies. For instance, Gooré Bi et al. [17] applied the PCSWMM model to assess the sensitivity of urban drainage systems under current and future climate conditions. Their analysis focused on quantitative variables such as peak flow (Q_{CSO}) and discharge volume (V_D) during extreme precipitation events, complemented by ecotoxicological risk indices derived from event fluxes. While this approach offered valuable insights into ecotoxicological risks for the St. Lawrence River, it did not consider changes in surface water quality criteria, infrastructure vulnerability, or risks to DWIs.

Other studies have employed climate model-driven assessments to project CSO impacts. For example, Fortier et al. [16] utilized simulated rainfall data from the Canadian Regional Climate Model (CRCM) to estimate future changes in CSO frequency and duration. Although this approach highlighted hydraulic impacts on urban drainage systems, it relied on a single pair of climate models and emission scenarios, which limited its robustness. Additionally, microbial risks and DWI impacts were not explicitly addressed. Similarly, Rodriguez et al. [32] explored diverse future rainfall scenarios using EURO-CORDEX regional climate models for 2085–2099, aligned with three Representative Concentration Pathways (RCPs): 2.6, 4.5, and 8.5. This study assessed changes in yearly CSO discharge volumes and the adaptive capacity of green infrastructure under future rainfall conditions. Roseboro et al. [18] conducted a similar investigation for Buffalo (New York), integrating small time-step future rainfall data with long-term continuous sewer network modeling. Using the worst-case scenario (RCP 8.5), their study demonstrated the efficacy of urban stormwater management in enhancing system resilience. However, like the aforementioned studies, it did not evaluate microbial risks or DWI impacts.

While these methodologies have advanced the understanding of hydraulic and infrastructural vulnerabilities, they have largely focused on surface water quality or reactive measures. Few studies have integrated microbial risk assessments, particularly in the context of DWIs, underscoring a significant gap in addressing these risks under climate change scenarios.

Although these studies have enhanced knowledge of urban drainage system performance, they are predominantly rooted in the top-down risk assessment approach. This widely used approach starts with future emission projections to develop climate scenarios, which then inform biophysical impact studies and adaptation strategies [33–36]. This approach relies on numerical evaluations of climate variations, using General Circulation Models (GCMs) to project future precipitation and temperature, which then inform hydrologic and impact models [33,34]. Regardless of its potential usefulness, the top-down approach has several limitations, including high data requirements, methodological complexity, and challenges in simulating regional repercussions [35,37,38]. GCMs, generated at coarse spatial scales, require bias correction and downscaling, which introduce cascading uncertainties [34,39,40], complicating the communication of climate projection uncertainties [35]. Additionally, future contaminants loads add another layer of uncertainty, as they vary by site and scenario [41]. For instance, the risk space may be defined as the range of possible variation in the concentration of fecal coliforms depending on hydroclimatic parameters (precipitation, temperature, river flows, etc.).

Given the limitations related to uncertainties in the top-down approach, a bottom-up vulnerability assessment approach has been suggested to further explore the full possible risk space [37,42], relying more on possibilities than probabilities [43]. This approach is crucial for climate adaptation policies, as it prioritizes specific inherent system attributes, including exposure, sensitivity, and adaptive capacity over downscaled GCMs projections [35,39]. The strength of the bottom-up approach lies in its ability to incorporate a wide range of climate information, including historical data, stochastic climate simulations, and GCM output [39]. In contrast to the top-down approach, which utilizes climate data differently in the final processing phases [34], the bottom-up strategy begins by identifying key climatic factors and then link them to reliable and readily available climate data. This allows the incorporation of different sources of climate information without the need to rerun the modeling study [34,39]. Conway et al. [35] describe the bottom-up approach as a people-centred approach because it provides knowledge of current and changing conditions, risks, and responses based on understanding of experts, notably scientists and stakeholders.

To the best of our knowledge, no prior study has applied a bottom-up approach to assess the vulnerability of DWIs to microbial contamination from CSOs under climate change in Quebec (Canada). Southern Quebec was chosen as the study site in this research due its high reliance on surface waters for drinking water supply—serving approximately 70% of its population [44]—as well as its numerous urban watersheds with CSO infrastructure..Urbanized areas, where many of these drinking water supplies are located, are particularly vulnerable to microbial and chemical contamination [2,3,29,30], with CSOs being a significant contributor to microbial contamination [41]. This vulnerability is expected to worsen as climate change drives an increase in both the intensity and frequency of extreme precipitation events [45], resulting in more frequent and prolonged CSO discharges [17]. Notably, Petrucci et al. [46] predict an exponential increase in the number of CSO events in southern Quebec due to the impact of climate change, highlighting the urgent need for strategic and proactive measures to mitigate these risks. Additionally, the Quebec government has mandated vulnerability assessment for DWIs [47], highlighting the urgent need for practical, locally adaptable assessment method. The insights gained from this study not only address a critical gap in Quebec's water resource management but also provide valuable knowledge applicable to other urban regions worldwide that face similar challenges. As highlighted in review of Leveque et al. [48], regions with comparable climates, based on the Köppen-Geiger classification established for the period 1980–2016, could benefit from these findings to inform strategic water resource management and mitigation measures.

To address the following research questions: (1) How does climate change impact the frequency and intensity of microbial contamination events at DWIs? (2) What are the primary environmental and anthropogenic factors driving vulnerability to CSO-related contamination? (3) How can a bottom-up approach support decision-making on climate-resilient water management compared to the traditional top-down approach? This study proposes a bottom-up approach to quantify the vulnerability of DWIs to microbial contamination in a highly urbanized river under climate change, using *E. coli* as an indicator of microbial contamination. To evaluate the usefulness of this novel methodological approach in guiding source protection strategies, the method was applied to a densely urbanized river in southern Quebec. A short-term future horizon (< 10 years) was considered, in line with the Quebec government requirements for vulnerability assessment of drinking water sources. Specific objectives were to (1) develop scenarios of river flow changes and scenarios of *E. coli* contamination events associated with CSOs (concentration, duration, volume, frequency of CSOs); (2) quantify the impacts of global change on the vulnerability of a DWI to microbial contamination by applying a hydrodynamic model of transport dispersal of fecal contaminants originating from CSOs in the river and (3) identify the key factors influencing the vulnerability of drinking water intakes for realistic global change mitigation strategies. The approach used in this work is intended to be accessible and quick to set up. It aims to be broadly used by municipalities, water managers, governments, and environmental organizations responsible for the vulnerability assessment of drinking water intakes. This is particularly relevant as the top-down approach is deemed unrealistic for most municipalities given the large human and financial resources required, while being also subject to important uncertainty, especially at a local scale.

2. Materials and methods

2.1. Study site

The case study river, located in southern Quebec, is almost exclusively fed by a river draining one of the largest watersheds in Eastern Canada (146,000 km^2), known to be one of the most hydraulically regulated with over 100 dams [49]. In 2011, a rock sill was lowered at the river entrance to increase incoming flow rates, thereby affecting the river flow regime, especially during the low-flow period from mid-August to mid-September [50]. Over a length of 42 km, the river delimitates two highly urbanized territories, exceeding an urbanization rate of 75% in 2010 [51]. The flow regime is severely disrupted by the presence of about 100 islands, which are generally small (the largest having a length of about 2.5 km), and three flow regulation structures, including the dam located at the entrance to the river, which limits the flow to 780 m^3/s. The other two dams are located in the second half of the river. The water level is measured by four hydrometric stations operated by the Centre d'Expertise Hydrique du Québec (CEHQ) and Environment and Climate Change Canada. Only the station operated by Environment and Climate Change Canada measures the river flow. The hydrometric station downstream of the dam at the entrance to the river has been recording daily water levels since 1999.

Five (5) DWIs are located on the river, supplying drinking water to more than 458,000 people in 2016. Over the 2013–2018 period studied, the river's microbial quality was mainly affected by the discharge of treated effluents [52] and raw sewage bypasses from 8 wastewater treatment plants and 85 CSOs structures located throughout the length of the river [53]. Forty percent of the overflow structures are located on the south shore, and overflows often occur during snowmelt and summer low flows, resulting from intense and/or lengthy precipitation events. The river's microbial contamination, which tends to increase downstream, is significantly attributed to these overflows [50].

Although agricultural activities are limited to the northern part of the watershed [54], these have little influence on the risk of microbial contamination, as most farms use chemical fertilizers or mineral fertilizers rather than manure [51]. Wastewater treatment plants effluents (treated and untreated) and agriculture activities, which are also potential sources of microbial contamination, were not considered in this study, as the objective was to specifically quantify the impact of global change on microbial contamination from CSO structures using the bottom-up approach because these urban overflows hamper full use of the river as water supply or recreation [50]. To achieve this objective, the study focused on a particular section of the river from its entrance up to 50 m downstream from DWI-3 (Fig 1). This site was chosen because it accurately represents the river watershed, as it includes 3 DWIs and 33 CSO structures, which are roughly evenly distributed along both banks of this section (approximately 15 km).

2.2. Bottom-up approach

Future microbial contamination levels at DWI-3 under global change were assessed based on a bottom-up approach, using *E. coli* as an indicator of recent fecal contamination [15]. In Quebec, the design of the drinking water treatment plants and their performance requirements are based on *E. coli* measurements in raw water [55] and the vulnerability of DWI to microbial contaminants is assessed based on these measurements [47]. In this study, fecal coliform data were also available at DWI-3. Several studies have shown significant variations in the *E. coli*/fecal coliform ratio [56,57], even exceeding the theoretical threshold of 1 due to counting errors. To be conservative, *E. coli* concentrations were estimated from fecal coliform concentrations assuming a ratio of 1:1. This assumption aligns with values commonly used in similar studies and provides a reasonable approximation in the absence of site-specific *E. coli* measurements.

The water quality model used in this study was previously calibrated and validated under current climate conditions by Shyaka [58] for the same study site. In that study, a sensitivity analysis was conducted to assess the influence of key hydrodynamic and water quality parameters, ensuring the robustness of the model. Statistical analysis, including Pearson correlation, mean squared error (MSE), and Spearman's rho, were performed to verify the reliability of the selected calibration parameters by comparing simulated outputs with observed data at monitoring stations [58].

As the present study builds upon Shyaka's validated model [58], the same parameterization was adopted as a baseline for future climate scenario simulations. Given that the previous sensitivity analysis confirmed the stability of the model outputs with respect to key parameters, additional sensitivity testing for the *E. coli*/fecal coliform ratio was not performed. However, for drinking water, it is the order of magnitude of *E. coli* concentrations that drive differences in treatment requirement classes. As such, minor variations in the *E. coli* to fecal coliform ratio on an arithmetic scale are expected to have a negligible impact.

This study used a bottom-up approach to examine various scenarios related to *E. coli* contamination. The first step involved generating a semi-probabilistic model of *E. coli* loads by using a stochastic method to determine *E. coli* concentrations (section 2.3). The model was developed based on different peak factors (FP_c = 1, 10, 100) that reflect the variation of *E. coli* concentrations in overflows. Additionally, it incorporates different reduction factors at maximum flow

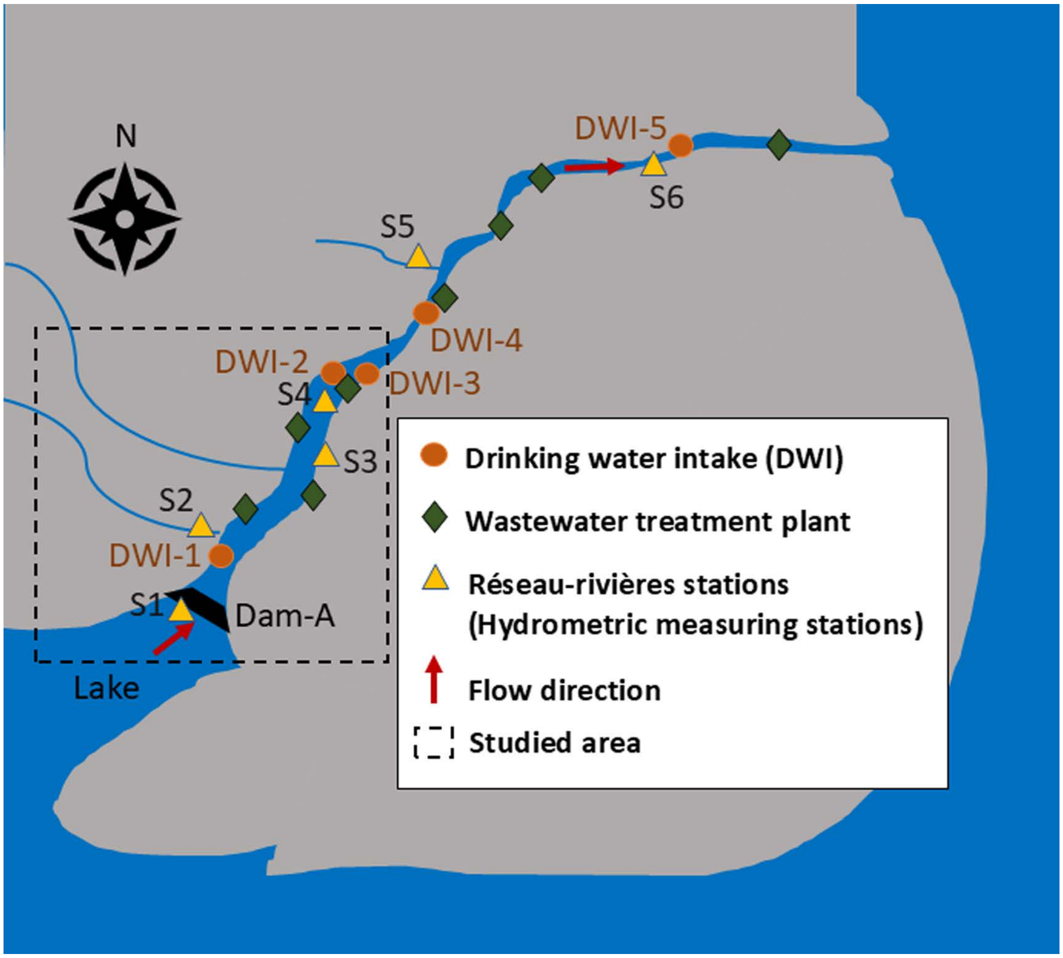

Fig 1. Schematic representation of the case study river: locations of drinking water intakes, wastewater treatment plant, and existing hydrometric measuring stations.

https://doi.org/10.1371/journal.pwat.0000352.g001

of the overflow structure ($FP_q = 1, 0.8, 0.5, 0.3$), which indicate the extent to which the maximum flow rate of the structure is reduced under specific conditions or interventions. The scenarios were also categorized according to different overflow durations ($T = 90, 300, 690, 990$ min), specifically chosen to reflect the observed duration of overflow events at the study site's specific overflow structures. Next, the impact of overflows on *E. coli* contamination at DWI-3 was studied using this semi-probabilistic model and hydrodynamic model coupled with 3D water quality modeling (Section 2.4). The simulation analysis was conducted under different scenarios, involving 33, 20, and 10 simultaneously active overflow structures at different river flows ($Q = 11, 110, 207$ m^3/s), facilitating a comprehensive examination of their potential impacts on microbial contamination levels in the river. By incorporating scenarios with different numbers of active overflow structures, the analysis evaluates the respective impacts on contamination levels. Similarly, by considering diverse river flow rates, the study examines how fluctuations in natural flow patterns affect contaminant dispersion and dilution within the river.

Fig 2 summarizes the methodology described above to generate *E. coli* concentrations at the studied DWI following overflows. Explanations regarding the choices behind all the parameters and scenarios used for the semi-probabilistic model and the hydrodynamic model coupled with water quality are provided in sections 2.3 and 2.4, respectively.

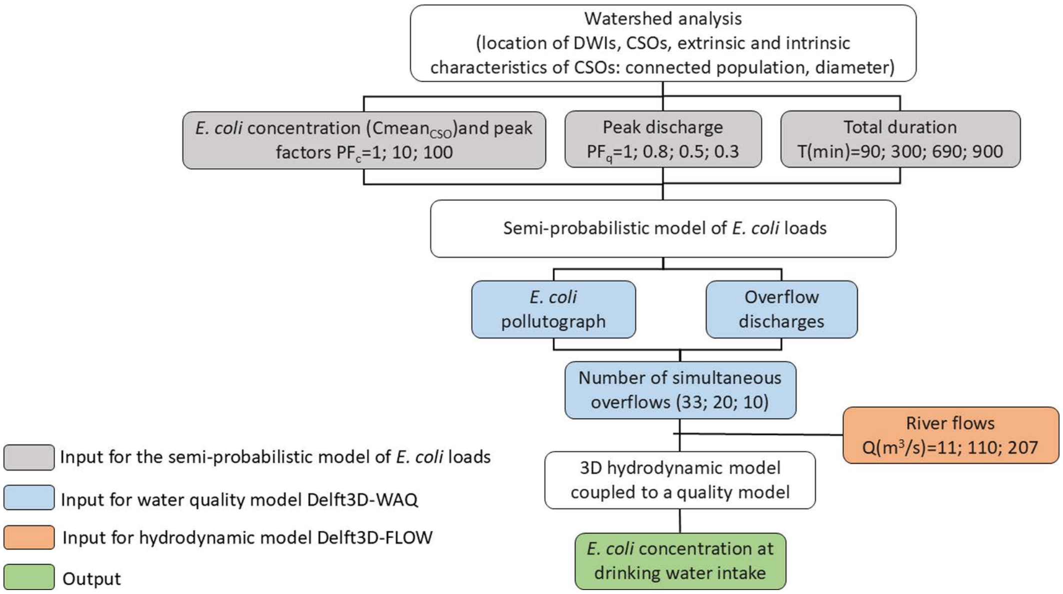

Fig 2. Methodological approach to assess the vulnerability of riverine drinking water intakes.

https://doi.org/10.1371/journal.pwat.0000352.g002

2.3. Semi-probabilistic model of *E. coli* loads

The CSO scenarios in this study were constructed using a semi-probabilistic model of CSO loads recently proposed by Taghipour et al. [59] to assess the impact of CSOs on surface water quality. The model consisted of the three steps: (1) generating discharge duration, (2) generating discharge flows, and (3) generating *E. coli* concentrations. In contrast to deterministic load models, which assume a constant mean value in *E. coli* concentration [41], this semi-probabilistic model accommodated for the variability of *E. coli* loading values during overflow events since contaminant concentrations may fluctuate [59].

Table 1 presents the rationale behind each scenario choice that can be applied in the Quebec study site. The scenarios were chosen based on the available data on CSOs in Quebec, which indicate high variability in discharge duration ranges. Four representative overflow duration were selected for this study, ranging from 90 minutes to 990 minutes. Missing *E. coli* data were addressed by using estimation methods derived from the population served by each combined sewer structure (as described in Table 1). While some uncertainties in the input data may arise due to incomplete records or variations in sewer system performance, we ensured the inclusion of all plausible scenarios in the model. As a result, any potential uncertainties in the data are negligible in the final outputs, allowing for robust conclusions about the impact of CSOs on water quality. Overall, the scenarios were designed to reflect the complexity of CSO dynamics in Quebec and to provide a robust framework for studying their impacts on water quality.

2.4. Hydrodynamic and water quality models

2.4.1. River flow scenarios. The low-flow period during summer is especially critical for source protection due to the low dilution capacity of some waterways [10,60]. Therefore, the impacts of climate change are evaluated in the summer period (June 1 – October 31) on two distinct spatial scales: (1) impacts on the water regime are evaluated at the regional level; (2) impacts on point sources (CSOs) of microbial contamination are evaluated at the local level [41].

In southern Quebec, at regional level, future climate projections suggest that increased temperatures and evapotranspiration could further decrease summer and fall flows, as well as summer low flows of the river. Changes in precipitation

Table 1. Selection of CSO scenarios for overflow duration, overflow discharge flows, and *E. coli* concentration.

CSO scenario parameters	Scenarios	Justification of scenarios choice
Overflow duration	90 min 300 min 690 min 990 min	The choice of overflow duration scenarios (90 min, 300 min, 690 min, and 990 min) is justified based on previous research conducted by [28] on a near-by river in Quebec. This study showed that the duration of overflows can vary depending on the season and type of precipitation. Autumn overflows, which occur at the end of summer low flows, tend to be longer than summer overflows. However, summer precipitation can also cause long-lasting overflows that last for more than 10 hours. Due to the limited knowledge on CSO duration resulting from current CSO monitoring practices in Quebec [8], it is assumed that the duration of overflow at each CSO is identical, which is simplistic assumption.
Overflow discharge flows	$PF_q = 1$ $PF_q = 0.8$ $PF_q = 0.5$ $PF_q = 0.3$	To determine the hydrographs of overflows, a deterministic model based on normalized parameters was used as explained in S1 Text. The model is formulated based on the total duration of overflow (T) and the peaks flow (Q_{PCSO}), which was calculated from the Manning-Strickler formula. This formula involves various parameters related to the overflow structure and the hydraulic regime: Manning coefficient (*n*), wetted surface (*S*), hydraulic radius (R_h), and slopes of the hydraulic gradient (*S*). To study the impact of the overflow rate on DWI contamination, reduction factors were applied to the Q_{PCSO}. In addition to the baseline situation with $PF_q = 1$, three reduction factors were used: $PF_q = 0.8$, $PF_q = 0.5$, and $PF_q = 0.3$. The S1 Text reports the equations used to calculate the overflow discharge flows.
E. coli concentration in overflows	$PF_c = 1$ $PF_c = 10$ $PF_c = 100$	The concentration of *E. coli* in overflows varies over time [29], so this study aimed to determine these concentrations using a stochastic model. The timing of peak concentration also varied, with extreme events showing later peaks [59]. Three probability distributions were identified for normalized *E. coli* concentration in CSOs based on the overflow duration as demonstrated by Taghipour et al. [59] and as detailed in the S2 Text. The mean concentration of *E. coli* per capita in CSO discharge was estimated to be 5.5 CFU/100mL/pers, which was used to determine the mean concentration of *E. coli* at overflow structure during an overflow event. Population growth and high intra-event variability were taken into consideration, and different factors (PF_C) of 1, 10, and 100 were applied to estimate the maximum *E. coli* concentration at each CSO. These factors were based on the observed differences of up to three orders of magnitude in the event scale in the study of Madoux-Humery et al. [29].

https://doi.org/10.1371/journal.pwat.0000352.t001

patterns and temperature will impact the characteristics of CSOs (duration, frequency, concentration, and flow) at the local level [10]. The inactivation of *E. coli* increases with raising temperatures, but the intensity and frequency of extreme precipitation are likely to increase regardless of greenhouse gas emissions [61]. This will lead to an increase in the number and characteristics of CSOs in a non-linear way. Dray weather periods are expected to increase, promoting the accumulation of *E. coli* in sewer systems due to sorption and sedimentation process [62]. These impacts of climate change at the local and regional levels are described in further detail in the S3 Text.

The strong regulation of the case study river and of the Ottawa River on which it depends complicates the study of its water quantity in the context of global change. To our knowledge, no hydroclimatic projection for the case study river has been made accessible to date. On a nearby river, Jalliffier-Verne et al. [41] showed that under equivalent CSO scenarios, changes in summer low flows due to climate change did not cause a significant variation in *E. coli* concentration at DWIs. During the period of low flows, the concentration of *E. coli* at DWIs was much more sensitive to the high concentrations released by the overflow structures. Thus, on some rivers threatened by CSOs, variations in river regimes do not appear to be a major risk factor for the vulnerability of DWIs to microbial contamination events. In addition, the excavation work at the entrance to the river in 2011 aimed at increasing the river flow to a minimum of 11 m^3/s during the low-water period. Thus, critical low flows, as experienced before 2011, and which threatened drinking water supply security, are unlikely to be reached again, even under the worst-case climate change scenario. Based on the two previous points, the study of the water quality of the river in the context of climate change was made from three flowrate scenarios in the summer period. The first is conservative and could represent a worst-case scenario. It corresponds to the worst historical summer low flows series measured over the historical period 1961–2020. The other two scenarios were selected from flow rates measured after excavation from 2012 to 2018. These scenarios, which correspond to mean flow rates of 110 m^3/s and 207 m^3/s, are adapted to the current hydroclimatic context, in addition to assessing the influence of high-water velocity on the vulnerability of DWIs to high-contamination events.

2.4.2. Model description. The impact of CSOs on *E. coli* contamination of the river was studied using 3D hydrodynamic modeling. To limit computational time, it was decided to use as few layers as possible, the aim being to understand the phenomena of the propagation of *E. coli* in rivers from the CSOs mainly in the horizontal plane, assuming a mean *E. coli* concentration across the water column. The river was modeled with five vertical layers. This is a reasonable compromise between calculation time and model accuracy for future uses.

The open-source software Deflt3D was used to develop the model, utilizing only the FLOW [63] and WAQ [64,65] modules. The FLOW module allows the multidimensional hydrodynamic simulation of the river, while the WAQ module is used to study the water quality. For further information on the equations and numerical diagrams of the hydrodynamic-quality coupled model, Deltares [64] can be consulted.

The hydrodynamic simulations were conducted with a 15-second time step to ensure the stability of the numerical scheme. For the water quality part of the model, the simulations were performed with a 5-minute time step, which is the optimal time step ensuring stability of the numerical scheme and speed of the calculation time, but the results were exported with a 15-minute time step, deemed sufficient to determine the peak events.

For boundary conditions in the water quality model, *E. coli* concentration at the river entrance was assumed constant and was set to 43 CFU/100ml, which represents the mean concentration of fecal coliforms measured during summer at Réseau-Rivières sampling station S1 (Fig 1) from 2015 to 2017 [66]. These values are very low compared to the values measured at downstream sampling stations S3, S4 and S6 (Fig 1).

From *E. coli* discharge (via overflows) to DWI, *E. coli* concentrations in the river follow a first-order decay rate degradation reaction according to the Chick-Watson law (S3 Text). In the water quality module, *E. coli* inactivation depends on salinity, water temperature and solar radiation [64,65]. The dissolved salts concentration was assumed constant and equal to 0.5 g/kg, which is representative of freshwater [64]. A constant water temperature of 15°C was used. This value is close to the average summer temperature at the Réseau-rivières Station S1 (Fig 1), over the 2015–2017 period. Similarly, solar radiation was set to a constant value of 230 W/m^2, which falls within the recommended range of 190 and 250 W/m^2 for the summer months [64]. Other studies have explored the relative importance of modeling parameters. Processes governing flow and mixing dominate the fate and transport of pathogens at the time scales relevant for modeling CSO discharges to a river [62,67]. Over longer time scales, biotic factors such as predation by zooplankton are strongly influenced by temperature and drive seasonal inactivation rates [68,69].

Finally, simulations were made assuming uniform dispersion in all three directions. The model was calibrated with the value of 1 m^2/s for dispersal coefficients in the 1st and 2nd direction (horizontal plane), and 1.10^{-7} m^2/s for the vertical dispersion coefficient, as recommended in the literature [64,65]. The model does not account for the settling and resuspension of *E. coli* from sediments, nor for the possible growth of *E. coli* in the river, which should be negligible given the short transport times in the studied section of the river.

In total, over 100 simulations were carried out to study the impact of simultaneous overflows, their duration, *E. coli* concentration, and peak flow at DWI-3.

3. Results and discussion

Simulations were conducted to assess the impact of overflows on *E. coli* concentration at DWI-3 based on various scenarios (concentration, duration, discharge flow rate, number of simultaneous overflows) even in the absence of observed data, for different river flow conditions as outlined in Section 2. The results of these simulations and their discussion are presented in sub-sections bellow, covering the impact of simultaneous overflows, overflow duration and maximum *E. coli* concentration, peak CSO flow, and the impact of the river and overflow duration on *E. coli* concentration at DWI-3. All simulations showed negligible differences in *E. coli* concentrations across the water column, supporting the use of a mean concentration in the analysis of the results.

3.1. Impact of simultaneous overflows

The impact of the number of simultaneous overflows on the *E. coli* concentration at DWI-3 was investigated by considering 33, 20 and 10 CSOs closest to the DWI-3. Two pairs of input data ($PF_c = 1$; $T = 300$ min) and ($PF_c = 10$; $T = 690$ min), were used for the simulations, which were performed using a flow series with an average river flow rate of 207 m^3/s. The results of these simulations are presented in Fig 3.

Both datasets show that CSOs located closest to the DWI (see 10 CSOs results in Fig 3) pose the greatest risk to the vulnerability of the source water to microbial hazards. This is primarily due to these CSOs contributing the most to the first and highest peak in *E. coli* concentration at DWI-3. Therefore, targeting efforts towards mitigation CSOs located near DWIs could prove to be efficient strategy in reducing the risk of microbial contamination, as demonstrated in previous studies [10,27,41].

However, it is important to acknowledge that CSOs situated further upstream of DWI-3 also have a significant impact on the source water quality. In case of long duration of overflows and high *E. coli* concentrations ($PF_c = 10$; $T = 690$ min), the second peak at DWI-3 can be significant and can exceed the extreme concentration threshold of 1,500 MPN (Most Probable Number)/100ml, which prevents the use of the source of water for drinking water production [41]. Moreover, an increased number of simultaneous overflows increase the mean *E. coli* concentration at the DWI, with high levels being maintained over a longer period. In general, CSOs situated further upstream from the DWI could potentially impact *E. coli* concentrations, especially in combination with high summer flows, high overflow duration and high *E. coli* concentrations. Although the hypothesis that all 33 CSOs will be simultaneously active is conservative, better CSO monitoring is required to disprove this assumption and refine the analysis. A possible extension of this study could focus on the CSO structures that were identified as being the most critical in terms of overflow frequency [52] to pinpoint the structures that pose the

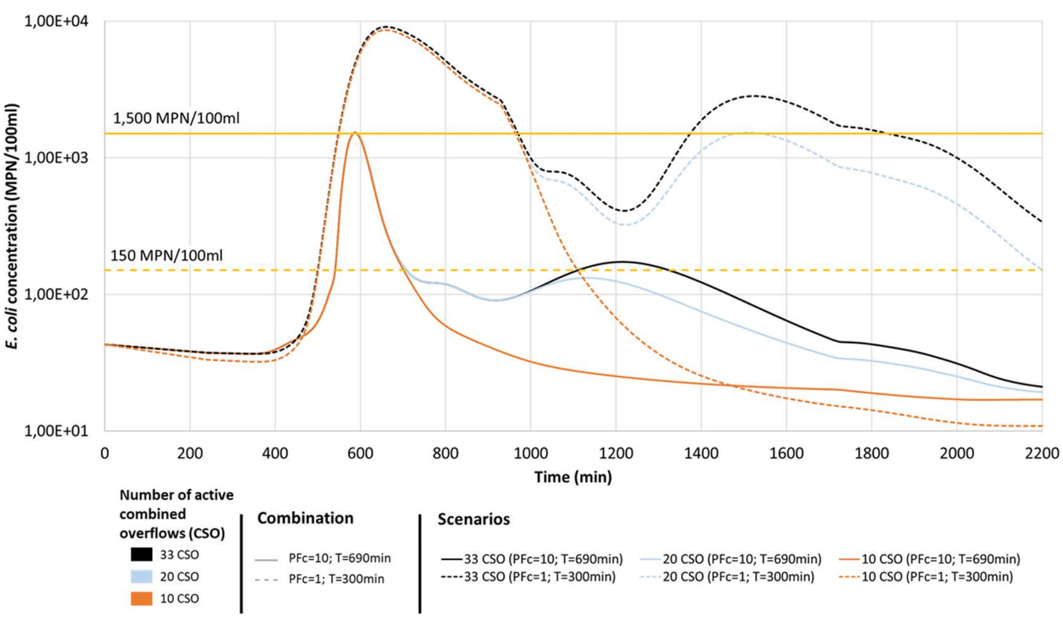

Fig 3. Impact of simultaneous overflows on E. coli concentration at DWI-3 for two scenarios: (PFc = 1; T = 300 min) and (PFc = 10; T = 690 min) when Qmean = 207 m3/s.

https://doi.org/10.1371/journal.pwat.0000352.g003

greatest problematic to water quality. Furthermore, these results (Fig 3) demonstrate the importance of assessing the risks posed by CSOs based on cumulative overflows, rather than evaluating the risks of each individual overflow structure separately, as required by the Quebec regulations for DWI vulnerability assessment [47].

3.2. Impact of overflow duration and maximum *E. coli* concentration

This section examines the impact of overflow duration (T = 90, 300, 690, 990 min) and maximum *E. coli* concentration (PF_c = 1, 10, 100) on water quality at the DWI-3. For these simulations, all 33 CSOs were assumed to be simultaneously active, and an average river flow rate of 207 m^3/s was used. The selection of the 33 CSOs was based on their significant impact on water quality at the DWI, as shown in section 3.1. Fig 4 presents the results of these simulations.

These results indicate a significant increase in *E. coli* concentration peaks at the studied DWI during longer duration of overflows (T > 690 min), as compared to shorter duration (T = 90, 300 min). Maximum *E. coli* concentrations show nearly two orders of magnitude (2 log) variation between 90-min and 690 min overflow duration. However, beyond a certain duration (e.g., between 690 and 990 min), the peak concentration remains invariant. Increased duration of overflows (e.g., over 300 min) are shown to generate high concentrations of *E. coli* at the DWI for a longer time, especially above the threshold of 150 CFU/100ml. This threshold also determines the classification of raw water quality and the necessary drinking water treatment in Quebec [55].

Indeed, the lag between the peaks of *E. coli* concentration at the DWI for different overflow duration is due to the semi-probabilistic model, rather than the transport of *E. coli* in the river. The semi-probabilistic model assumptions for *E. coli* loading indicate that longer overflow duration results in a delayed peak of *E. coli* at the DWI.

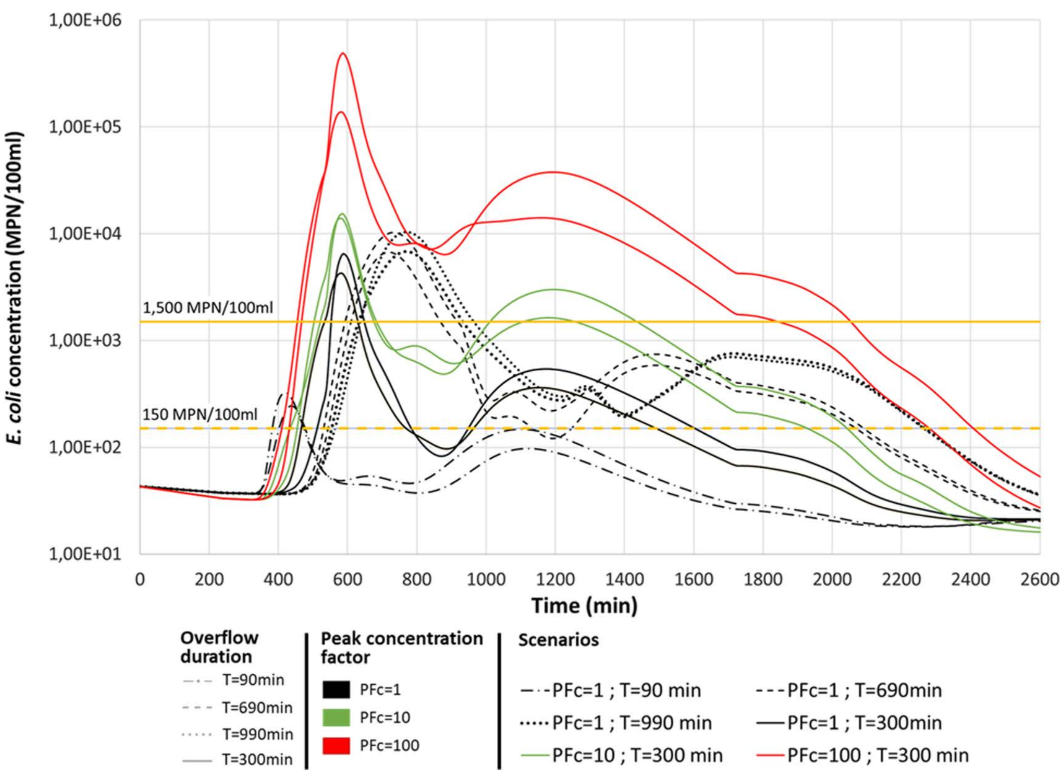

Fig 4. Impact of overflow duration (T = 90, 300, 690 et 990 min) and maximum E. coli concentration discharged (PFc = 1, 10, 100) on the concentration of *E. coli* at DWI-3 when Qmean = 207 m3/s.

https://doi.org/10.1371/journal.pwat.0000352.g004

The increase in peak *E. coli* concentrations from CSOs has a significant impact on peak concentrations at DWI-3, especially for the second peak, as shown in Fig 4 for a 300-min CSO duration. The effect of more concentrated *E. coli* overflows on peak concentrations at the DWI in non-linear. For instance, a 10-fold increase in the PF_c peak factor increases the first peak *E. coli* concentrations at the DWI by about 0.5 log. However, a 100-fold increase in PF_c increases this first *E. coli* peak by 1 to 1.5 log and increases the second peak by about 2 log, which is a significant impact on the drinking water treatment plant.

For peak concentration factors $PF_c = 1$ or $PF_c = 10$, simulated *E. coli* concentrations are more realistic than those obtained for the peak factor $PF_c = 100$, which leads to *E. coli* concentrations at the DWI well beyond historical values.

The bottom-up approach, as employed in this study, focuses on evaluating vulnerabilities by utilizing existing data and larger-scale hydrological processes, without incorporating local precipitation patterns. While this approach offers the advantage of flexibility and reduces the need for high-resolution climate data, it also has inherent limitations. For instance, it may oversimplify certain regional dynamics, especially in areas where localized precipitation and extreme weather events significantly influence vulnerability, such as in the case of overflows (Section 1). In contrast, the top-down approach provides a more detailed projection of future climate conditions, particularly precipitation patterns, using GCMs and downscaling techniques (Section 1). However, this approach comes with its own set of limitations, such as high data requirements, uncertainties in downscaling methods, and challenges in simulating regional effects (Section 2). The reliance on global models and their biases can complicate the assessment of localized impacts, especially in areas with complex hydrological processes. Despite these limitations, the bottom-up and top-down approaches are complementary. The bottom-up approach, by focusing on the present and utilizing available data, can provide insights into current system vulnerabilities and identify immediate areas for adaptation. On the other hand, the top-down helps frame potential future risks and projections, offering a broader view of long-term challenges. Therefore, combining both the bottom-up and top-down approaches can provide a more comprehensive vulnerability assessment, as also explained in other studies [33,35,39].

3.3. Impact of peak discharge flow from CSOs

As discussed in the preceding section (Section 3.2), it was shown that an increase in *E. coli* concentration peaks at DWI due to the influence of long periods of overflows and a high peak concentration factor ($PF_c = 100$). In this section, simulations were carried out for a scenario where $PF_c = 100$ and $T = 690$ min, assuming that all the CSOs are simultaneously active, and using an average river flow rate of 207 m^3/s. However, the peak discharge flow from CSOs ($PF_q = 1$, 0.8, 0.5, 0.3) was varied to explore its impact on the overall *E. coli* concentration at the DWI-3. The results presented in Fig 5 demonstrate that a decrease in peak discharge flow (PF_q) has the potential to reduce peak *E. coli* concentrations, as well as the average *E. coli* concentration at the DWI-3. This reduction in CSO discharge flow leads to a downward vertical shift on the *E. coli* pollutograph established at DWI-3. It should be noted, however, that while CSO discharge flow rate has a significant influence on the risk of high *E. coli* contamination at DWI-3, its impact is less pronounced than the impact of maximum *E. coli* concentrations at CSOs. In fact, a reduction of 50% to 70% in peak discharge flow may result in a reduction of approximately 1 log in the concentration of *E. coli* at the DWI. However, the reduction is less significant for lower reductions of around 20% in peak overflow discharge.

In view of the results, it is recommended that PF_q values ranging from 0.8 to 0.5 be used instead of an overly conservative value of 1 ($PF_q = 1$). It is unlikely that the pipes will reach full capacity during overflow events, and it is more probable that peak flows will be between 50% and 80% of the maximum capacity of the pipes. Therefore, adopting a more realistic PF_q value within this range would provide a more accurate representation of the *E. coli* concentration levels during overflow events.

3.4. Impact of the river flow rate

To gain a deeper understanding of the factors affecting *E. coli* contamination in the studied area, additional simulations were conducted under varying low river flows conditions, including a high flow rate ($Q = 207$ m^3/s) during the studied

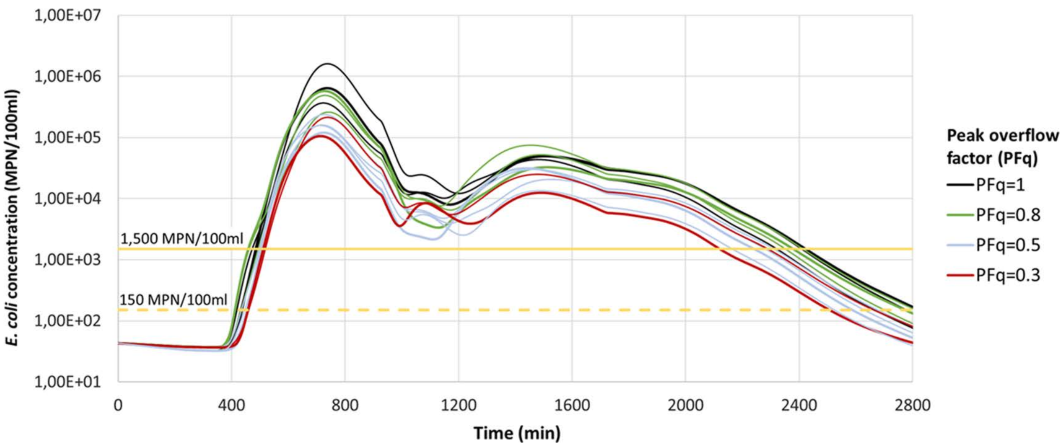

Fig 5. Impact of peak discharge flow (PFq = 1, 0.8, 0.5, 0.3) on *E. coli* concentration at DWI-3 with PFc = 100, T = 690 min and a mean river flow Qmean = 207 m3/s.

https://doi.org/10.1371/journal.pwat.0000352.g005

period, a normal flow rate (Q = 110 m^3/s), and an extreme low flow rate (Q = 11 m^3/s) that is unlikely to occur in future climate scenarios. These simulations further explore the impact of river flow rates on *E. coli* concentrations under the condition of all 33 CSOs being active. The simulations considered the peak factor on *E. coli* concentration of $PF_c = 1$ and different overflow durations of T = 90; 300; 690 min. The results are presented in Fig 6.

The shape of the *E. coli* pollutograph at the studied DWI is significantly affected by the decrease in river flow rate during summer low flows. Under normal (Q = 110 m^3/s) and high (Q = 207 m^3/s) flows, the pollutograph typically exhibits a double-peaked structure due the temporal dispersion of *E. coli* plumes originating from different CSOs along the river. However, when the flow is extremely low (*Q* = 11 m^3/s), the second peak of *E. coli* concentration no longer appears, even for long-duration overflows (T = 300, 690 min). This phenomenon is primarily attributed to the reduced flow velocities, which slow the downstream transport of the *E. coli*, particularly from the more distant CSOs. As a result, increased residence time enhances the likelihood of *E. coli* degradation before reaching DWI-3. Additionally, the onset of the *E. coli* peak induced by the ten closest CSOs to DWI-3 is delayed by approximately 12 hours due to the slower velocity of transport under low-flow conditions. These findings highlight the strong influence of hydrodynamics on the timing and shape of the *E. coli* concentration profiles, particularly in systems with significant CSOs contributions.

While the prolonged residence time under extreme low flow increases *E. coli* inactivation, it is partially offset by the river's reduced dilution capacity. Consequently, the peak *E. coli* concentration remains relatively similar for higher low flows (Q > 110 m^3/s). However, at very low flows (Q = 11 m^3/s), long-duration overflows lead to a lower maximum *E. coli* concentration compared to high flows (Table 2). This reduction can be explained by the cumulative effect of inactivation over the extended residence time. In contrast, the mean *E. coli* concentration at DWI-3 increases substantially under extreme low flows (Table 3), suggesting that chronic exposure to microbial contamination could be a more pressing issue than peak events. This finding is particularly relevant for public health, as annual microbial risk assessments are more closely linked to long-term exposure rather than short-term peaks [27]. Table 3 also reveals that, for brief to moderate CSO duration during summer low flows, a slight reduction in river flows tends to decrease the mean concentration of *E. coli* at the studied DWI. This could be explained by the diminished dilution capacity, which fails to compensate for the prolonged inactivation time caused by the slower flow velocities. In contrast, longer overflow durations (>690 min) lead to higher *E. coli* loads from CSOs, exceeding the river's dilution capacity, thus causing an increase of *E. coli* concentrations at DWI, despite the longer inactivation time.

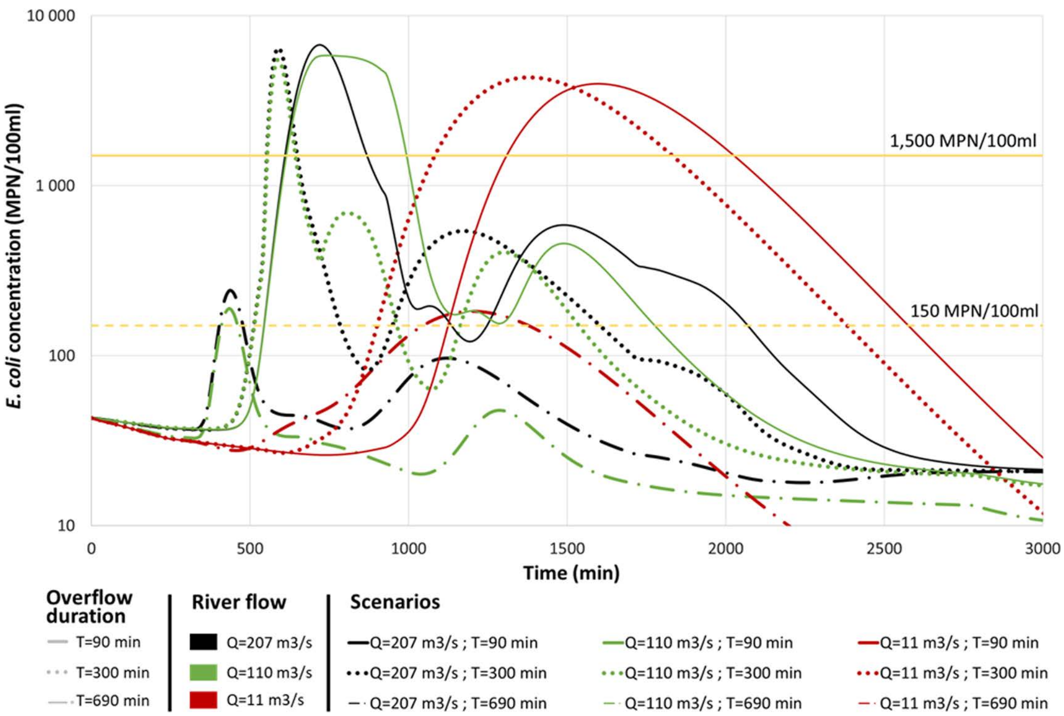

Fig 6. Impact of river flow (Q = 207, 110, 11 m3/s) and overflows duration (T = 90 min, 300 min, 690 min) on the *E. coli* concentration at DWI-3 for PFc = 1.

https://doi.org/10.1371/journal.pwat.0000352.g006

Table 2. Maximum *E. coli* concentration (MPN/100 mL) simulated at DWI-3 for different duration of overflows (T = 90, 300, 690 min) and river flows (Q = 207, 110, 11 m^3/s), assuming all 33 CSOs are active, with discharge PF$_c$ = 1. Bracket values indicate the relative deviation from values obtained at the mean flow of 207 m^3/s.

	Q = 207 m^3/s	Q = 110 m^3/s (-47%)	Q = 11 m^3/s (-95%)
T = 90 min	223	189 (-15%)	183 (-18%)
T = 300 min (+233%)	6418	5464 (-15%)	4337 (-32%)
T = 690 min (+667%)	6748	5828 (-14%)	3974 (-41%)

https://doi.org/10.1371/journal.pwat.0000352.t002

Table 3. Mean *E. coli* concentration (MPN/100 mL) simulated at DWI-3 for different duration of overflows (T = 90, 300, 690 min) and river flows (Q = 207, 110, 11 m^3/s), assuming all 33 CSOs are active, with discharge PF$_c$ = 1. Bracket values indicate the relative deviation from values obtained at the mean flow of 207 m^3/s.

	Q = 207 m^3/s	Q = 110 m^3/s (-47%)	Q = 11 m^3/s (-95%)
T = 90 min	30	26 (-14%)	47 (+55%)
T = 300 min (+233%)	244	214 (-12%)	840 (+244%)
T = 690 min (+667%)	484	621 (+28%)	773 (+60%)

https://doi.org/10.1371/journal.pwat.0000352.t003

It is worth mentioning that on the studied river, the presence of islands has a significant influence on the transport of contaminants. The integration of these islands into the model reveals that contaminants tend to concentrate on the shores, particularly in areas with a high number of islands. This highlights the significance of coupling hydrodynamic modeling with quality modeling to identify potential issues that would not be visible using empirical or conceptual models.

From a broader perspective, the range of flow conditions examined in this study (Q = 11, 110, 207 m^3/s) provides valuable insight into potential future risks associated with microbial contamination at DWIs. This bottom-up approach ensures that the selected scenarios account for a wide range of hydrodynamic conditions in summer, complementing traditional top-down approach that relies solely on climate model projections, as explained in section 1. While additional flow scenarios could have further refined the analysis, the selected range is sufficiently broad to assess the impact of global changes on microbial contamination. Future work could expand on these findings by incorporating additional environmental factors, such as temperature-dependent inactivation rates and sediment resuspension dynamics, to refine predictions under changing climate conditions.

3.5. Future improvement to the bottom-up vulnerability assessment approach

To improve the bottom-up approach used in this study, several improvements can be made to the semi-probabilistic model of *E. coli* loads and water quality model, ultimately strengthening the overall bottom-up vulnerability assessment approach, proposed in this study. The Manning-Stricker formula (S1 Text) used to model the peak flow at the output of overflow structures has limitations and is only valid for rough, turbulent flows. Other factors such as elbows or falls in the sewerage network can induce the dissipation of energy, and this should be considered in the model. Additionally, dry weather deposits can increase local load losses, which is not modeled in this study.

Setting up electronic overflow recorders at the CSOs' structures could provide valuable data on overflows' duration, a critical variable for assessing microbial contamination. This data could help determine if longer overflow durations lead to higher levels of *E. coli* concentration at DWIs. Also, a better understanding of the mean and maximum overflow duration specific to each CSO is crucial for optimizing the proposed methodology in this study. By assigning realistic duration ranges for each CSO and running several simulations with stochastically varying overflow durations, researchers could gain a more comprehensive understanding of the impacts of overflows on water quality under global change. A specific duration for each CSO during a rain event could also be determined through a watershed hydrological model, coupled with the hydrodynamic model of the river.

Low flows significantly influenced the *E. coli* concentrations, the timing of the *E. coli* peak, as well as the shape of the DWI pollutograph (Section 3.4). There is therefore a real interest in developing hydroclimatic projections for the studied river, despite its strong regularization that complicates the direct use of output climate models. In this context, an approach using neural networks would be relevant, as proposed by Huaringa Alvarez et al. [70], who have developed a neural model simulating the future reservoir management in the context of climate change. Given the projected decrease in summer low flows in southern Quebec [48], there could be significant impacts on both the mean and the maximum *E. coli* concentration at DWI, as highlighted in Tables 2 and 3. Given the potential consequences of a sharp river flow rate, it is necessary to undertake adaptation measures to climate change as soon as possible, as the vulnerability of DWI-3 to microbial contamination could increase with global change. Urbanization trends, such as increased population growth and infrastructure development, are expected to exacerbate these impacts. As urban areas expand, more impervious surfaces will increase runoff and exacerbate overflows, potentially increasing *E. coli* concentrations during extreme weather events, such as longer dry weather periods followed by extreme rainfall [41]. In addition, urbanization can put pressure on wastewater systems reducing their capacity to manage overflows effectively.

Climate policy changes, particularly those aimed at mitigating greenhouse gas emissions, could influence the severity of climate change impacts. For example, more stringent policies to reduce emissions could slow the rate of warming and reduce the frequency of extreme weather events like droughts and heavy rainfall. In contrast, delays in climate action may lead to an exacerbation of the conditions that increase microbial contamination risk at DWIs, particularly during prolonged droughts. In such scenarios, the probability of occurrence of critically low flows in the river would increase, and overflows could occur when the river's dilution capacity is already compromised. This would result in higher and longer-lasting *E. coli* concentrations at DWIs (Section 3.4).

In addition to mitigation policies, climate adaptation measures, such as restoring natural environments and implementing nature-based solutions like blue-green infrastructure in cities, could help reduce the negative impact of urbanization. Therefore, both urbanization and climate policy decisions—whether focused on mitigation or adaptation—will play crucial roles in shaping the future vulnerability of the DWI system to microbial contamination.

Finally, the methodology proposed in this study has the potential to be readily extended to other pollutants discharged during overflows (nutrients, organic and inorganic substances, and other fecal microorganisms), which are generally included in vulnerability analyses of drinking water sources. Furthermore, perspectives include further developing the model by integrating the effects of sedimentation and resuspension, as it is recognized that sediments play a very important role in the transport and fate of *E. coli* in streams [62,71]. Since soil erosion is expected to intensify due to global change (urbanization, increased intensity of extreme rainfall) [48], its impact on the microbial contamination at DWIs and on the risk of DWI obstruction should be addressed. The consequences of these changes on the vulnerability of DWIs to microbial contamination need to be quantified to guide land-use planning as early as possible, which must evolve in line with scientific advances on the impacts of climate change on the vulnerability of drinking water sources to contamination.

Potential directions for future research include expanding the study to larger geographic areas, particularly those where jurisdiction is shared between two countries, such as Canada and the United States. Such studies would be valuable for surface water systems that cross international borders, helping to understand the compounded impacts of climate change on these shared water resources. Examining these transboundary water bodies could provide insights into how climate change influences water quality parameters, such as microbial contamination, and could inform the development of joint regulations and management strategies. By studying these shared boundaries, researchers can guide vulnerability and risk assessments in regions where political and regulatory complexities exist. This research would be critical in shaping effective, coordinated policies to address climate change impacts and safeguard water quality across international lines.

3.6. Mitigation strategy

To protect public health under global change, the Quebec government has put in place a regulatory framework to limit the contamination of surface water by CSO structures. This regulation is in response to Canadian regulation, which prohibits developing territories from increasing the frequency of overflows in the absence of compensatory measures [72].

Given the risk of microbial contamination of surface waters during and after CSOs, as assessed from *E. coli* concentrations, the Quebec government requires municipalities to conduct a vulnerability assessment of their DWIs [47]. Incorporating the expected impacts of global change at the local level is particularly critical. However, few studies have quantitatively assessed the impacts of global change on the risk of microbial contamination at local levels [14].

Our findings highlight the significant impact of simultaneous overflows on both mean and peak *E. coli* concentrations at DWIs, particularly during prolonged CSO events and when discharged *E. coli* concentrations are high. Furthermore, extreme low-flow conditions notably amplify *E. coli* concentrations, reshaping pollutographs at DWIs. Future climate projections suggest a decrease in summer low flows, potentially exacerbating water source vulnerability to CSOs contamination. These findings emphasize the need for adaptive management strategies that consider vulnerable periods characterized by reduced dilution capacity, high simultaneous overflow occurrences, and elevated contaminant loads.

From a water resource management perspective, our results underscore the necessity of refining source water protection strategies. Jalliffier-Verne et al. [41] showed that reducing peak *E. coli* concentrations at overflow structures is more effective in safeguarding raw water quality than solely reducing the frequency of CSOs. Postulating an identical frequency of overflows by 2050 and based on a business-as-usual global change scenario, the authors reported an over 87%-increase in mean *E. coli* concentrations entering DWIs located on a river in the Greater Montreal Area (Quebec, Canada).

Green infrastructure has been put forth as a potential CSO mitigation strategy [73–75], as they also bring additional socio-environmental benefits. Yet, their efficiency to reduce contaminants loads has to be further demonstrated, including continuous simulations that take into account dry weather periods during which the accumulation of contaminants on the surface and in sewer systems is significant [73]. Green infrastructure responds very well to low-intensity precipitation, helping to reduce the frequency of overflows [73], but their effectiveness for extreme precipitations is based on limited evidence [76]. With the expected future increase in dry weather periods precipitation intensities, green infrastructure will thus likely not be the sole solution to CSO mitigation [73], although it would likely be part of a more integrated and resilient urban water strategy given their multiple co-benefits [77].

4. Conclusions

The objectives of this work were to propose a broadly applicable bottom-up approach to assess the vulnerability of DWIs to the risk of microbial contamination and to analyze semi-qualitatively the impacts of global change on a chosen study site during summer low flows. Our results show that it is not the number of simultaneous overflows that dictates the amplitude of peak *E. coli* concentration at the DWI, but rather that a high number of simultaneous overflows induces prolonged high levels of mean *E. coli* concentration at the DWI downstream. During critical low river flows, high levels of *E. coli* concentration lasted longer than during high flows, which is problematic in terms of source water protection.

Based on these findings, several mitigation strategies are suggested. First, identifying priority sources of contamination, such as overflows in high-risk area, can help target interventions. Additionally, recognizing scenarios that create high risk level, such as extreme whether events or prolonged dry periods, can guide the development of early warning systems and improve communication among municipalities to coordinate responses effectively. Nature-based solutions that can increase stormwater infiltration and reduce pressures on sewer systems, particularly during low flow periods, should be considered as part of a broader adaptation strategy to address these challenges in the context of global change.

Urbanization, demographic growth coupled with reduced per capita water consumption, which causes a deposition in sewers, and changes in summer/autumn precipitation patterns, are responsible for an increase in *E. coli* loads in overflows. Therefore, adaptation efforts to global change should be planned at the local level with regards to contaminants loads, and at the regional level regarding river hydraulicity during summer. These efforts require interactions at different spatial scales between decision makers.

While the bottom-up approach proposed in this study provides a better understanding of local conditions, its applicability may be limited in regions with fewer data. In such areas, simplified hydraulic models or alternative methods to estimate impacts, such as proxy indicators or expert assessments, could be explored. Despite the potential limitations, the bottom-up approach remains a valuable tool for identifying vulnerabilities and informing strategies to protect DWIs from microbial contamination. Our proposed approach is not limited to the current study's focus on microbial contaminants and can be readily adapted to investigate other pollutants, offering a flexible framework for water quality management.

Supporting information

S1 Text. Overflow discharge flow rates.
(DOCX)

S2 Text. *E. coli* concentrations in overflows.
(DOCX)

S3 Text. Assessing the impacts of climate change.
(DOCX)

Author contributions

Conceptualization: Baptiste Leveque, Mouhamed Ndong, Raja Kammoun, Jean-Baptiste Burnet, Sarah Dorner, Françoise Bichai.

Data curation: Baptiste Leveque.

Formal analysis: Baptiste Leveque, Sarah Dorner, Françoise Bichai.

Funding acquisition: Sarah Dorner, Françoise Bichai.

Investigation: Baptiste Leveque, Sarah Dorner, Françoise Bichai.

Methodology: Baptiste Leveque, Albert Irakiza Shyaka, Mouhamed Ndong, Raja Kammoun, Jean-Baptiste Burnet, Sarah Dorner, Françoise Bichai.

Project administration: Sarah Dorner, Françoise Bichai.

Resources: Sarah Dorner, Françoise Bichai.

Software: Baptiste Leveque, Albert Irakiza Shyaka, Mouhamed Ndong, Raja Kammoun.

Supervision: Sarah Dorner, Françoise Bichai.

Validation: Baptiste Leveque, Mouhamed Ndong, Raja Kammoun, Sarah Dorner, Françoise Bichai.

Visualization: Baptiste Leveque, Raja Kammoun, Sarah Dorner, Françoise Bichai.

Writing – original draft: Baptiste Leveque, Mouhamed Ndong, Sarah Dorner, Françoise Bichai.

Writing – review & editing: Baptiste Leveque, Mouhamed Ndong, Raja Kammoun, Jean-Baptiste Burnet, Sarah Dorner, Françoise Bichai.

References

1. Botturi A, Ozbayram EG, Tondera K, Gilbert NI, Rouault P, Caradot N, et al. Combined sewer overflows: A critical review on best practice and innovative solutions to mitigate impacts on environment and human health. Critical Reviews in Environmental Science and Technology. 2020;51(15):1585–618. https://doi.org/10.1080/10643389.2020.1757957
2. Gooré Bi E, Monette F, Gasperi J, Perrodin Y. Assessment of the ecotoxicological risk of combined sewer overflows for an aquatic system using a coupled "substance and bioassay" approach. Environ Sci Pollut Res Int. 2015;22(6):4460–74. https://doi.org/10.1007/s11356-014-3650-9 PMID: 25315929
3. Jalliffier-Verne I, Heniche M, Madoux-Humery A-S, Galarneau M, Servais P, Prévost M, et al. Cumulative effects of fecal contamination from combined sewer overflows: Management for source water protection. J Environ Manage. 2016;174:62–70. https://doi.org/10.1016/j.jenvman.2016.03.002 PMID: 27011341
4. Anne-Sophie M-H, Dorner SM, Sauvé S, Aboulfadl K, Galarneau M, Servais P, et al. Temporal analysis of *E. coli*, TSS and wastewater micropollutant loads from combined sewer overflows: implications for management. Environ Sci Process Impacts. 2015;17(5):965–74. https://doi.org/10.1039/c5em00093a PMID: 25816314
5. Mascher F, Mascher W, Pichler-Semmelrock F, Reinthaler F, Zarfel G, Kittinger C. Impact of Combined Sewer Overflow on Wastewater Treatment and Microbiological Quality of Rivers for Recreation. Water. 2017;9(11):906. https://doi.org/10.3390/w9110906
6. Ordulj M, Jozić S, Baranović M, Krželj M. The effect of precipitation on the microbiological quality of bathing water in areas under anthropogenic impact. Water. 14(4).
7. Passerat J, Ouattara NK, Mouchel J-M, Rocher V, Servais P. Impact of an intense combined sewer overflow event on the microbiological water quality of the Seine River. Water Res. 2011;45(2):893–903. https://doi.org/10.1016/j.watres.2010.09.024 PMID: 20934197
8. Mailhot A, Talbot G, Lavallée B. Relationships between rainfall and Combined Sewer Overflow (CSO) occurrences. Journal of Hydrology. 2015;523:602–9. https://doi.org/10.1016/j.jhydrol.2015.01.063
9. World Health Organization. COP24 Special Report: Helath And Climate Change. Geneva, Switzerland 2018.
10. Jalliffier-Verne I, Leconte R, Huaringa-Alvarez U, Madoux-Humery A-S, Galarneau M, Servais P, et al. Impacts of global change on the concentrations and dilution of combined sewer overflows in a drinking water source. Sci Total Environ. 2015;508:462–76. https://doi.org/10.1016/j.scitotenv.2014.11.059 PMID: 25506909
11. Asif A, Chen J-S, Hussain B, Hsu G-J, Rathod J, Huang S-W, et al. The escalating threat of human-associated infectious bacteria in surface aquatic resources: Insights into prevalence, antibiotic resistance, survival mechanisms, detection, and prevention strategies. J Contam Hydrol. 2024;265:104371. https://doi.org/10.1016/j.jconhyd.2024.104371 PMID: 38851127
12. Jiménez Cisneros B, Oki T, Arnell N, Benito G, Cogley J, Döll P. Freshwater resources. Cambridge, United Kingdom & New York, NY, USA: GIEC. 2014.
13. Abbott BW, Bishop K, Zarnetske JP, Minaudo C, Chapin FS, Krause S, et al. Human domination of the global water cycle absent from depictions and perceptions. Nat Geosci. 2019;12(7):533–40. https://doi.org/10.1038/s41561-019-0374-y

14. Hofstra N. Quantifying the impact of climate change on enteric waterborne pathogen concentrations in surface water. Current Opinion in Environmental Sustainability. 2011;3(6):471–9. https://doi.org/10.1016/j.cosust.2011.10.006

15. World Health Organization. Protecting surface water for health. Identiying, assessing and managing drinking-Water quality risks in surface-water catchments. Switzerland: World Health Organization; 2016. https://apps.who.int/iris/bitstream/handle/10665/246196/9789241510554-eng.pdf?sequence=1

16. Fortier C, Mailhot A. Climate Change Impact on Combined Sewer Overflows. J Water Resour Plann Manage. 2015;141(5). https://doi.org/10.1061/(asce)wr.1943-5452.0000468

17. Bi EG, Monette F, Gachon P, Gaspéri J, Perrodin Y. Quantitative and qualitative assessment of the impact of climate change on a combined sewer overflow and its receiving water body. Environ Sci Pollut Res Int. 2015;22(15):11905–21. https://doi.org/10.1007/s11356-015-4411-0 PMID: 25869430

18. Roseboro A, Torres M, Zhu Z, Rabideau A. The impacts of climate change and porous pavements on combined sewer overflows: A case study of the city of Buffalo, New York, USA. Front Water. 2021;3.

19. Gogien F, Dechesne M, Martinerie R, Lipeme Kouyi G. Assessing the impact of climate change on Combined Sewer Overflows based on small time step future rainfall timeseries and long-term continuous sewer network modelling. Water Res. 2023;230:119504. https://doi.org/10.1016/j.watres.2022.119504 PMID: 36621275

20. Salerno F, Gaetano V, Gianni T. Urbanization and climate change impacts on surface water quality: Enhancing the resilience by reducing impervious surfaces. Water Res. 2018;144:491–502. https://doi.org/10.1016/j.watres.2018.07.058 PMID: 30077910

21. Abdellatif M, Atherton W, Alkhaddar RM, Osman YZ. Quantitative assessment of sewer overflow performance with climate change in northwest England. Hydrological Sciences Journal. 2015;60(4):636–50. https://doi.org/10.1080/02626667.2014.912755

22. Bendel D, Beck F, Dittmer U. Modeling climate change impacts on combined sewer overflow using synthetic precipitation time series. Water Sci Technol. 2013;68(1):160–6. https://doi.org/10.2166/wst.2013.236 PMID: 23823552

23. Nie L, Lindholm O, Lindholm G, Syversen E. Impacts of climate change on urban drainage systems – a case study in Fredrikstad, Norway. Urban Water Journal. 2009;6(4):323–32. https://doi.org/10.1080/15730620802600924

24. Mahaut V, Andrieu H. Relative influence of urban-development strategies and water management on mixed (separated and combined) sewer overflows in the context of climate change and population growth: A case study in Nantes. Sustainable Cities and Society. 2019;44:171–82. https://doi.org/10.1016/j.scs.2018.09.012

25. Iqbal MS, Islam MMM, Hofstra N. The impact of socio-economic development and climate change on *E. coli* loads and concentrations in Kabul River, Pakistan. Sci Total Environ. 2019;650(Pt 2):1935–43. https://doi.org/10.1016/j.scitotenv.2018.09.347 PMID: 30286359

26. Islam MMM, Iqbal MS, Leemans R, Hofstra N. Modelling the impact of future socio-economic and climate change scenarios on river microbial water quality. Int J Hyg Environ Health. 2018;221(2):283–92. https://doi.org/10.1016/j.ijheh.2017.11.006 PMID: 29217424

27. Taghipour M, Shakibaeinia A, Sylvestre É, Tolouei S, Dorner S. Microbial risk associated with CSOs upstream of drinking water sources in a transboundary river using hydrodynamic and water quality modeling. Sci Total Environ. 2019;683:547–58. https://doi.org/10.1016/j.scitotenv.2019.05.130 PMID: 31146060

28. Madoux-Humery A-S, Dorner S, Sauvé S, Aboulfadl K, Galarneau M, Servais P, et al. Temporal variability of combined sewer overflow contaminants: evaluation of wastewater micropollutants as tracers of fecal contamination. Water Res. 2013;47(13):4370–82. https://doi.org/10.1016/j.watres.2013.04.030 PMID: 23764588

29. Anne-Sophie M-H, Dorner SM, Sauvé S, Aboulfadl K, Galarneau M, Servais P, et al. Temporal analysis of *E. coli*, TSS and wastewater micropollutant loads from combined sewer overflows: implications for management. Environ Sci Process Impacts. 2015;17(5):965–74. https://doi.org/10.1039/c5em00093a PMID: 25816314

30. Prévost M, Madoux-Humery A-S, Dorner S. Mesures de protection des prélèvements d'eau de surface effectués à des fins de consommation humaine: aires de protection et vulnérabilité des sources - revue bibliographique. Rev Bibliogr. 2017.

31. Sylvestre E, Burnet JB, Smeets P, Medema G, Prevost M, Dorner S. Can routine monitoring of *E. coli* fully account for peak event concentrations at drinking water intakes in agricultural and urban rivers? Water Res. 2020;170:115369.

32. Rodriguez M, Fu G, Butler D, Yuan Z, Cook L. The effect of green infrastructure on resilience performance in combined sewer systems under climate change. J Environ Manage. 2024;353:120229. https://doi.org/10.1016/j.jenvman.2024.120229 PMID: 38310790

33. Bhave AG, Mishra A, Raghuwanshi NS. A combined bottom-up and top-down approach for assessment of climate change adaptation options. Journal of Hydrology. 2014;518:150–61. https://doi.org/10.1016/j.jhydrol.2013.08.039

34. Brown C, Ghile Y, Laverty M, Li K. Decision scaling: Linking bottom-up vulnerability analysis with climate projections in the water sector. Water Resources Research. 2012;48(9). https://doi.org/10.1029/2011wr011212

35. Conway D, Nicholls RJ, Brown S, Tebboth MGL, Adger WN, Ahmad B, et al. The need for bottom-up assessments of climate risks and adaptation in climate-sensitive regions. Nat Clim Chang. 2019;9(7):503–11. https://doi.org/10.1038/s41558-019-0502-0

36. Kelly PM, Adger WN. Theory and practice in assessing vulnerability to climate change and facilitating adaptation. Clim Change. 2000;47(4):325–52.

37. Hallegatte S, Shah A, Lempert RJ, Brown C, Gill S. Investment decision making under deep uncertainty: Application to climate change. Washington, DC: World Bank. 2012.
38. Stainforth D, Allen MR, Tredger ER, Smith L. Confidence, uncertainty and decision-support relevance in climate predictions. Climatic Change. 81:2145–61.
39. García L, Matthews J, Rodriguez D, Wijnen M, DiFrancesco K, Ray P. Beyond downscaling: A bottom-up approach to climate adaptation for water resources management. AGWA Report 01. Washington, DC: World Bank Group. 2014.
40. Wilby RL, Dessai S. Robust adaptation to climate change. Weather. 2010;65(7):180–5. https://doi.org/10.1002/wea.543
41. Jalliffier-Verne I, Leconte R, Huaringa-Alvarez U, Heniche M, Madoux-Humery A-S, Autixier L, et al. Modelling the impacts of global change on concentrations of *Escherichia coli* in an urban river. Advances in Water Resources. 2017;108:450–60. https://doi.org/10.1016/j.advwatres.2016.10.001
42. Brown C, Wilby RL. An alternate approach to assessing climate risks. Eos Trans AGU. 2012;93(41):401–2. https://doi.org/10.1029/2012eo410001
43. Pulido-Velazquez M, Marcos-Garcia P, Girard C, Sanchis-Ibor C, Martinez-Capel F, García-Prats A. A top-down meets bottom-up approach for climate change adaptation in water resource systems. Cham: Springer International Publishing. 2022.
44. Robert C, Bolduc A. Bilan de la qualité de l'eau potable au Québec 2005-2009. Direction des politiques de l'eau. Ministère du Développement durable, de l'Environnement et des Parcs. 2012.
45. Côté H, Logan T, Charron I. Vers l'adaptation. Synthèse des connaissances sur les changements climatiques au Québec. Partie 1: évolution climatique au Québec. Ouranos. 2015.
46. Petrucci J, Jalbert J, Dorner S, McQuaid N, Bichai F. Strategic prioritization of sewersheds to mitigate combined sewer overflows under climate change. Environmental Challenges. 2025;18:101088. https://doi.org/10.1016/j.envc.2025.101088
47. Government of Quebec. Water withdrawal and protection regulation- Q-2, r. 35.2. Quebec Official Publisher. 2014.
48. Leveque B, Burnet J-B, Dorner S, Bichai F. Impact of climate change on the vulnerability of drinking water intakes in a northern region. Sustainable Cities and Society. 2021;66:102656. https://doi.org/10.1016/j.scs.2020.102656
49. Ottawa RIVERKEEPER. Watershed facts. https://ottawariverkeeper.ca/watershed-fact/. 2023. 2024 December 17.
50. Brouillette D. Qualité de l'eau de la rivière des Mille Îles 2000-2005. Québec. Québec: ministère du Développement durable dlEedP. 2007;36.
51. COBAMIL. Portrait de la zone de gestion intégrée de l'eau par bassins versants du COBAMIL. Sainte-Thérèse, Québec; 2013.
52. Burnet J-B, Sylvestre É, Jalbert J, Imbeault S, Servais P, Prévost M, et al. Tracking the contribution of multiple raw and treated wastewater discharges at an urban drinking water supply using near real-time monitoring of β-d-glucuronidase activity. Water Res. 2019;164:114869. https://doi.org/10.1016/j.watres.2019.114869 PMID: 31377523
53. Kammoun R, McQuaid N, Lessard V, Prévost M, Bichai F, Dorner S. Comparative study of deterministic and probabilistic assessments of microbial risk associated with combined sewer overflows upstream of drinking water intakes. Environmental Challenges. 2023;12:100735. https://doi.org/10.1016/j.envc.2023.100735
54. Kammoun R, McQuaid N, Lessard V, Goitom EA, Prévost M, Bichai F, et al. Risk assessment of drinking water intake contamination from agricultural activities using a Bayesian network. PLOS Water. 2023;2(7):e0000073. https://doi.org/10.1371/journal.pwat.0000073
55. Government of Quebec. Regulation respecting the quality of drinking water-Q-2, r. 40 (Environment Quality Act). Quebec Official Publisher. 2012.
56. Garcia-Armisen T, Prats J, Servais P. Comparison of culturable fecal coliforms and *Escherichia coli* enumeration in freshwaters. Can J Microbiol. 2007;53(6):798–801. https://doi.org/10.1139/W07-033 PMID: 17668040
57. Hathaway JM, Krometis LH, Hunt WF. Exploring Seasonality in *Escherichia coli* and Fecal Coliform Ratios in Urban Watersheds. J Irrig Drain Eng. 2014;140(4). https://doi.org/10.1061/(asce)ir.1943-4774.0000700
58. Shyaka AI. Développement d'un modèle hydrodynamique couplé qualité de l'eau pour l'étude de la contamination des prises d'eau de la rivière des Mille-Îles. Polytechnique de Montréal.
59. Taghipour M, Tolouei S, Autixier L, Prévost M, Shakibaeinia A, Dorner S. Normalized dynamic behavior of combined sewer overflow discharges for source water characterization and management. J Environ Manage. 2019;249:109386. https://doi.org/10.1016/j.jenvman.2019.109386 PMID: 31421478
60. Madoux-Humery A-S, Dorner S, Sauvé S, Aboulfadl K, Galarneau M, Servais P, et al. The effects of combined sewer overflow events on riverine sources of drinking water. Water Res. 2016;92:218–27. https://doi.org/10.1016/j.watres.2015.12.033 PMID: 26866859
61. Environment and Climate Change Canada. Climate data for resilient. Canada. 2019 [2024 December 19]. https://climatedata.ca/.
62. Dorner SM, Anderson WB, Slawson RM, Kouwen N, Huck PM. Hydrologic modeling of pathogen fate and transport. Environ Sci Technol. 2006;40(15):4746–53.
63. Deltares. Delft3D-Flow: simulation of multi-dimensional hydrodynamic flows and transport phenomena, including sediments- user manual. Netherlands: Deltares; 2023.
64. Deltares. D-Water Quality Processes Library Description: Technical Reference Manual: Version: 5.01. Netherlands: Deltares; 2023.
65. Deltares. D-water Quality: Versatile water quality modelling in 1D, 2D or 3D systems including physical, (bio)chemical and biological processes: User manual. Netherlands: Deltares; 2023.

66. MELCC. Banque de données sur la qualité du milieu aquatique (BQMA). Québec: QC Direction générale du suivi de l'état de l'environnement; 2019.
67. Brookes JD, Antenucci J, Hipsey M, Burch MD, Ashbolt NJ, Ferguson C. Fate and transport of pathogens in lakes and reservoirs. Environ Int. 2004;30(5):741–59. https://doi.org/10.1016/j.envint.2003.11.006 PMID: 15051248
68. Burnet J-B, Faraj T, Cauchie H-M, Joaquim-Justo C, Servais P, Prévost M, et al. How does the cladoceran Daphnia pulex affect the fate of *Escherichia coli* in water?. PLoS One. 2017;12(2):e0171705. https://doi.org/10.1371/journal.pone.0171705 PMID: 28178322
69. Tromas N, Goitom E, Chin T, Dinh QT, Dorner SM, Khawasik OS, et al. Impact of grazing by multiple Daphnia species on wastewater bacterial communities. Sci Total Environ. 2025;962:178364. https://doi.org/10.1016/j.scitotenv.2024.178364 PMID: 39799645
70. Huaringa Alvarez UF, Trudel M, Leconte R. Impacts and Adaptation to Climate Change Using a Reservoir Management Tool to a Northern Watershed: Application to Lièvre River Watershed, Quebec, Canada. Water Resour Manage. 2014;28(11):3667–80. https://doi.org/10.1007/s11269-014-0694-z
71. Kiefer LA, Shelton DR, Pachepsky Y, Blaustein R, Santin-Duran M. Persistence of *Escherichia coli* introduced into streambed sediments with goose, deer and bovine animal waste. Lett Appl Microbiol. 2012;55(5):345–53. https://doi.org/10.1111/j.1472-765X.2012.03296.x PMID: 22897753
72. Conseil Canadien des Ministres de l'Environnement (CCME). Stratégie pancanadienne pour la gestion des effluents d'eaux usées municipales Canada. 2009;23.
73. Autixier L, Mailhot A, Bolduc S, Madoux-Humery A-S, Galarneau M, Prévost M, et al. Evaluating rain gardens as a method to reduce the impact of sewer overflows in sources of drinking water. Sci Total Environ. 2014;499:238–47. https://doi.org/10.1016/j.scitotenv.2014.08.030 PMID: 25192930
74. Cavadini GB, Rodriguez M, Cook LM. Connecting blue-green infrastructure elements to reduce combined sewer overflows. J Environ Manage. 2024;365:121465. https://doi.org/10.1016/j.jenvman.2024.121465 PMID: 38901320
75. Cavadini GB, Rodriguez M, Nguyen T, Cook LM. Can blue–green infrastructure counteract the effects of climate change on combined sewer overflows? Study of a swiss catchment. Environ Res Lett. 2024;19(9):094025. https://doi.org/10.1088/1748-9326/ad6462
76. Johnson T, Butcher J, Santell S, Schwartz S, Julius S, LeDuc S. A review of climate change effects on practices for mitigating water quality impacts. J Water Clim Chang. 2022;13:1684–705. https://doi.org/10.2166/wcc.2022.363 PMID: 36590233
77. Bichai F, CabreraFlamini A. The water-sensitive city: implications of an urban water management paradigm and its globalization. Wiley Interdiscip Rev Water. 2018;5(3).