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Establishing acceptable exposure limits for Escherichia coli in WWTP tailwater discharge for recreational activities

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Abstract

Tailwater discharged from wastewater treatment plants (WWTPs) can contain pathogenic microorganisms, posing potential health risks during recreational water activities. While Quantitative Microbial Risk Assessment (QMRA) is commonly used to evaluate these risks, its complex outputs were not easily translated into operational standards. To address this gap, this study introduces the concept of a threshold limit value (TLV) defined as the maximum acceptable *E. coli* concentration in tailwater that ensures compliance with specific health risk benchmarks. TLVs were derived using reverse QMRA for four age groups (children, early teens, teens, and adults) under two risk criteria: the U.S. EPA annual infection risk (10^{-4}) and the World Health Organization disease burden benchmark (10^{-6} DALYs per person per year). Results showed that TLVs decrease with age, as adult individuals inhale or ingest larger volumes, resulting in higher exposure doses under identical conditions. Consequently, lower TLVs indicate stricter health protection requirements. WWTPs with higher treatment capacity and larger receiving water flows exhibited lower TLVs, reflecting more stringent acceptable concentrations due to reduced exposure risk. Swimming TLVs (4.43E+01–7.72E+02 CFU/100 ml) were about three times lower than rowing TLVs (1.25E+02–1.09E+03 CFU/m³), based on WHO and U.S. EPA benchmarks, due to more direct exposure and higher contact frequency. TLVs based on the WHO benchmark were consistently lower than those based on the EPA benchmark, emphasizing the impact of risk criteria on regulatory limits. Sensitivity analysis identified annual exposure frequency as the dominant variable for both rowing and swimming, with exposure time also being a key determinant for swimming exposure. This study provides a practical, risk-based framework for defining site-specific microbial limits, supporting evidence-based water quality management.

Keywords: Wastewater treatment plants; Tailwater discharge; Threshold limit value; Reverse quantitative microbiological risk assessment; Sensitivity analysis

1. Introduction

Tailwater refers to the treated effluent discharged from wastewater treatment plants (WWTPs) into natural water bodies such as rivers, lakes, or oceans. Although treated, this water can still contain contaminants, including pathogenic microorganisms, depending on the treatment processes employed and environmental factors (Peng et al., 2024). Studies have shown that tailwater from WWTPs can contain high concentrations of pathogenic organisms (Dias et al., 2019; Pantanella et al., 2020; Lu et al., 2022; Wang et al., 2023; Peng et al., 2024). These microorganisms, such as diarrheagenic *Escherichia coli* (*E. coli*), *Salmonella spp.*, and *Cryptosporidium parvum*, indicate fecal contamination and can pose risks to public health, particularly in recreational water use scenarios (Pernet-Coudrier et al., 2012). For example, *E. coli* levels in tailwater have been recorded in the range of 10^5 to 10^7 colony-forming units per 100 milliliters (CFU/100 mL), depending on factors such as the treatment efficiency of the plant and the characteristics of the receiving water body (Price et al., 2017, Pernet-Coudrier et al., 2012; Kowalski et al., 2017; Amoah et al., 2022). Although not all strains of *E. coli* are pathogenic, their presence is widely used as an indicator of fecal contamination, signifying the potential presence of other harmful pathogens (Bivins et al., 2017). Tailwater directly discharged into the surface water system is one of the primary sources of pollution for receiving water (Pernet-Coudrier et al., 2012). Each year, diarrheal diseases linked to contaminated water, poor sanitation, and inadequate hygiene contribute to an estimated 485,000 deaths globally (WHO, 2019), underscoring the significant public health burden associated with waterborne pathogens (WHO, 2019). Furthermore, it is estimated that the global economic losses due to inadequate water supply and sanitation amount to $260 billion annually.

Recreational activities in contaminated receiving water bodies pose direct health risks to nearby residents, primarily through direct water ingestion during swimming, as well as through droplet ingestion and bioaerosol inhalation during activities such as rowing (Ikehata et al., 2012; Shi et al., 2018). The risk assessment focuses on nearby residents because they are more likely to have frequent and prolonged exposure to these waters through daily activities such as bathing, fishing, or washing (Kowalski et al., 2017). In contrast, other water users, such as tourists, typically have more sporadic or limited contact. For instance, studies have shown that individuals swimming in waters with high levels of fecal contamination have an increased risk of gastrointestinal illnesses and other waterborne diseases (Kowalski et al., 2017; Amoah et al., 2022). The prevalence of

Escherichia coli (*E. coli*) in domestic wastewaters is widely used as an indicator of pathogenic microorganisms in tailwater quality assessment and related health risk assessments (Jamieson et al., 2004; Lieverloo et al., 2007; Hunter et al., 2011; Machdar et al., 2013; Szyłak-Szydłowski et al., 2016; Bivins et al., 2017; Shi et al., 2018). Both the United States Environmental Protection Agency (U.S. EPA) and the European Environment Agency recommend its use in assessing the microbiological quality of recreational waters (Mwanamoki et al., 2014; EEA, 2022).

Quantitative microbiological risk assessment (QMRA) is commonly employed to estimate the health risks of illnesses and diseases, typically quantified using the annual probability of infection and disease burden (DB) (Haas et al., 2014). However, QMRA results often present health risks in a dual-structure form, making direct application challenging for local utilities in managing tailwater quality on-site (Haas et al., 2020). Additionally, fixed standards for controlling pathogens in tailwater lack specific guidelines based on various exposure scenarios, which hinders the immediate identification of health risks at receiving water sites (U.S. EPA, 2002). To address this, the reverse QMRA framework is typically used to calculate the threshold limit value (TLV) of acceptable exposure concentrations of pathogens in tailwater, using initial risk levels published by the U.S. EPA or the World Health Organization (WHO) (Dean et al., 2020). The TLV serves as a boundary between satisfactory and unsatisfactory exposure concentrations, facilitating risk assessment under various exposure scenarios (Carducci et al., 2018; Dean et al., 2020).

While previous studies have provided insights into TLV determination (Hamilton et al., 2019; Rasheduzzaman et al., 2019; Dean and Mitchell, 2020; Shoults et al., 2021), there is no consensus on the most appropriate TLV benchmarks for *E. coli* in WWTP tailwater, especially concerning the recommendations of regulatory agencies such as the U.S. EPA and WHO. Furthermore, understanding the sensitivity of TLV calculations to input parameters is crucial for robust risk assessment (Dias et al., 2019). However, previous studies have not extensively investigated the sensitivity of TLVs to factors such as relative humidity, temperature, exposure frequency, and dermal absorption rate, especially in the context of recreational activities like rowing and swimming. Moreover, the diversity of exposure scenarios, including different recreational activities and environmental conditions, has not been comprehensively examined until now, leading to uncertainties in assessing the actual health risks associated with various exposure pathways. In this study, reverse QMRA was used to derive TLV benchmarks of *E. coli* concentration in WWTP tailwater under different exposure scenarios of nearby residents, based on

U.S. EPA and WHO risk levels. Additionally, sensitivity analysis was employed to assess the influence of the input parameters on the uncertainty of TLV calculations in scenarios involving residents rowing and swimming in receiving water.

2. Method and materials

2.1 Exposure scenarios of receiving water

This study was performed on the receiving waters of tailwater from three municipal WWTPs located in the central part of P.R. China. These plants, designated as plants A, B, and C, have treatment capacities of 1×10^5, 2×10^5, and 5×10^5 m^3/day, respectively (Supplementary Material Table 1). They have the same treatment processes, including screening, primary setting, biochemical reaction, secondary clarifier, and disinfection, to treat similar inlet domestic wastewater (Supplementary Material Table 2). However, their biochemical reaction tank types differ (see Supplementary Material, Table 1). The treated tailwater was discharged into the nearby receiving water.

2.2 Reverse QMRA

Residents living nearby are exposed to *E. coli* in the tailwater during activities such as rowing or swimming. The exposed population was categorized into four age groups (children, early teens, teens, adults) (Tables 1 and 2), with primary exposure routes including ingestion of contaminated water and inhalation of bioaerosols generated from the water surface. Dermal contact was excluded, as *E. coli* infections are not typically associated with skin exposure. Two widely recognized health risk levels, the U.S. EPA annual infection risk and the WHO Disability-Adjusted Life Years (DALY), were used as an initial assessment of the risk (Table 2). Table 1 shows the primary exposure routes of rowing and swimming. Detailed calculation methodologies and exposure parameters are provided in the Supplementary Materials. The analysis incorporates: (1) activity-specific water ingestion rates, with swimming (I_s) using age-stratified values (50 mL/hr for children, 37.5 mL/hr for early teens, 25 mL/hr for teens, and 20 mL/hr for adult) from U.S. EPA (2019) recreational exposure data, and rowing (I_r) applying a lognormal-distributed droplet ingestion rate (3.9–4.68 mL/hr, all ages) per field measurements (Dorevitch et al., 2011; Sümpelmann et., al. 2017); (2) background *E. coli* concentrations (100–500 CFU/100 mL) (Price et al., 2017); and (3) conservative health risk assumptions, including a 30% illness probability given infection (*Pill/inf* = 0.3) based on clinical data (Teunis et al., 2008). This 30% pathogenic

fraction was uniformly applied to all dose-response calculations, ensuring protective public health margins for WWTP-derived risks while accounting for distinct exposure pathways (direct ingestion for swimming vs. aerosol/splash ingestion for rowing).

The study analyzed bioaerosols in receiving waters downstream of WWTP discharges by evaluating the aerosolization rate (V_{a2}), which was determined by meteorological conditions, including temperature (T), relative humidity (RH), and absolute humidity (AH), as shown in Supplementary (Eqs. 6 and 7). For rowing exposure (Eq. 5), the model calculated a bioaerosol-based threshold limit value (C_R in CFU/m³) by incorporating the droplet ingestion rate (IR), Et is exposure time (min visit^{-1}), V_{a2} was aerosolization rate and bioaerosol ingestion ratio (Ag) to estimate pathogen intake from inhaled aerosols. In contrast, the swimming exposure model (Eq. 8) derived a waterborne TLV (C_S in CFU/100 mL) based on direct water ingestion, using age-specific ingestion rates (I_S) and exposure time (t) according to EPA (2019) guidelines, without considering aerosol-specific parameters (Sümpelmann et. al., 2017). Because C_S represents a health-based water quality target for recreational use of the receiving water, a mass-balance approach was applied to translate this target into an allowable discharge concentration from the WWTP. The allowable discharge concentration (C_t) was then determined through a mass-balance model (Eq. 9), which incorporates the WWTP effluent flow (Q_E), the receiving water flow (Q_0), and the background *E. coli* concentration (C_0) to ensure that the mixed concentration in the river does not exceed the recreational safety standard (C_S). The model assumes that complete mixing of effluent and receiving water ensures that the resulting downstream *E. coli* concentration does not exceed the recreational safety standard (C_S). The analysis demonstrated that higher receiving water flows (Q_0) provide greater dilution capacity, permitting increased discharge concentrations (C_t) while maintaining water quality. Although the study examined TLVs across typical climate ranges, future research could explore extreme scenarios to enhance public health guidance. The separation of aerosol (rowing) and water ingestion (swimming) exposure pathways in the risk assessment framework ensures the appropriate evaluation of each route, accounting for their distinct mechanisms and parameters.

Table 1 Descriptions of exposure scenarios

Exposure population	Exposure events	Primary route of exposure	Annual exposure frequency (visits/year)	Exposure time (min/visit)	References
Nearby residents	Rowing	Bioaerosol nasal inhalation; Splashed tailwater droplets ingestion	Lognormal distribution (3.1, 6.6)	Lognormal distribution (66.8, 74.4)	Zhang et al. 2022, Dufour et al. 2017, EPA 2019
	Swimming	Direct water ingestion; Dermal contact	Lognormal distribution (12.4, 29.7)	Lognormal distribution (55.2, 46.8)	

Table 2 Parameters for reverse-QMRA calculation

Items	Description	Unit	Values			References
HB	Health burden	DALYs case^{-1}	*E. coli*: 0.0455			Shi et al. 2018
$P_{ill/inf}$	Specific conditional probability of illness given an infection	Unitless	0.3			Teunis et al. 2008
n	Annual exposure frequency	-	Table 1			-
α	Beta-Poisson dose-response model parameter	Unitless	1.55×10^{-1}			Shi et al. 2018
D_{50}			2.11×10^{6}			
AG	Aerosol ingestion rate	%	50			Jahne et al. 2015
RH	Relative humidity	%	Uniform distribution (77, 87)			-
T	Temperature	°C	Uniform distribution (3.7, 25.1)			
I_R	Droplets ingestion rate	mL h^{-1}	Lognormal distribution (3.9, 4.68)			Dorevitch et al. 2011
I_S	Water ingestion rate (swimming)	mL h^{-1}	Lognormal distribution (20, 50)			Dufour et al. 2017, EPA 2019
ET	Exposure time	min visit^{-1}	Table 1			-
Q_P	Flow rate of receiving water	m^3 s^{-1}	-			Supplement materials Table 1
U.S. EPA	United States Environmental Protection Agency annual infection risk	pppy	10^{-4}			U.S. EPA, 2002
WHO DB	World Health Organization disease burden	DALYs pppy	10^{-6}			WHO, 2008
Br	Breathing rate	L d^{-1}	Four age groups	Children, 9–12 years old	0.55	Zhao et al., 2014
				Early Teens, 12–15 years old	0.56	
				Teens, 15–18 years old	0.58	
				Adults, ≥18 years old	0.65	
C~0	Background *E. coli* concentration	CFU/100mL	100-500			Price et al., 2017

2.3 Monte Carlo simulation and sensitivity analysis

Monte Carlo simulation and sensitivity analysis were performed using Oracle Crystal Ball and Microsoft Excel 2010 to calculate the threshold limit value (TLV) benchmarks for acceptable *E. coli* exposure concentrations (Devleesschauwer et al., 2014; Liu et al., 2019; Dean & Mitchell, 2020). Input parameters were sampled from their respective probability distributions (Tables 1 and 2), and each simulation was run for 10,000 iterations to ensure convergence and stability of the output distributions. For the rowing exposure model, which considers bioaerosol inhalation and droplet ingestion, the uncertain input parameters included exposure time (*ET*), breathing rate (*Br*), aerosol ingestion ratio (*Ag*), droplet ingestion rate (*IR*), relative humidity (*RH*), temperature (*ET*), and annual exposure frequency. For the swimming exposure model, which focuses solely on direct water ingestion, the uncertain parameters were limited to annual exposure frequency, exposure time (*Et*) and the water ingestion rate (I_S). All other uncertain variables were assigned uniform distributions based on plausible ranges derived from literature or monitoring data. Sensitivity analysis was conducted using the Spearman rank correlation coefficient to quantify the relative contribution of each input parameter to the variance in the TLV outputs; a higher absolute coefficient indicates a greater influence on model uncertainty (Dean et al., 2020). Detailed computational procedures are provided in the Supplementary Materials.

3. Results and discussion

3.1 TLV of acceptable exposure concentration

Figure 1 and Supplementary Table 3 present the threshold limit values (TLVs) for acceptable *E. coli* bioaerosol concentrations (CFU/m³) originating from the tailwater of WWTPs A, B, and C. These TLVs were estimated using reverse QMRA for four age groups (children, early teens, teens, and adults) engaged in rowing activities in the receiving water. The TLVs reported in this study were modeled health-based thresholds derived from the U.S. EPA annual infection risk benchmark and the WHO disease burden (DB) criterion.

Table 3 Exposure dose of rowing and swimming (CFU day^{-1})

Item		Rowing	Swimming
Risk level	U.S. EPA	3.24	0.75
	WHO	0.72	0.17

These values are not actual concentrations observed in the WWTP effluent or receiving water but represent the maximum *E. coli* levels permissible to meet defined health risk thresholds for different age groups. The use of both EPA and WHO benchmarks ensures that the TLVs reflect internationally recognized risk criteria, providing a realistic basis for assessing safe exposure levels during recreational activities such as rowing and swimming.

TLVs were estimated under two health risk benchmarks: the annual infection risk level set by the U.S. EPA (Fig. 1a) and the disease burden (DB) metric established by the WHO (Fig. 1b). According to the U.S. EPA benchmark, TLVs across all three WWTPs showed a consistent trend of decreasing values with increasing age: children exhibited the highest TLVs, followed by early teens, teens, and adults, respectively. This age-dependent pattern reflects physiological differences in exposure parameters (Zhao et al., 2014). For rowing exposure, the breathing rate plays a significant role in determining the inhaled bioaerosol volume, which contributes to the ingested dose via mucociliary clearance (Zhang et al., 2022; Dufour et al., 2017). In contrast, for swimming, ingestion is the primary exposure route, and age-related TLV differences are driven by ingestion volume rather than respiratory activity (Dufour et al., 2017; EPA, 2019). While children exhibit higher TLVs in both scenarios due to lower exposure intake parameters used in the model, they remain biologically more vulnerable to gastrointestinal infections, particularly during swimming, as supported by existing literature (Sunger et al., 2019).

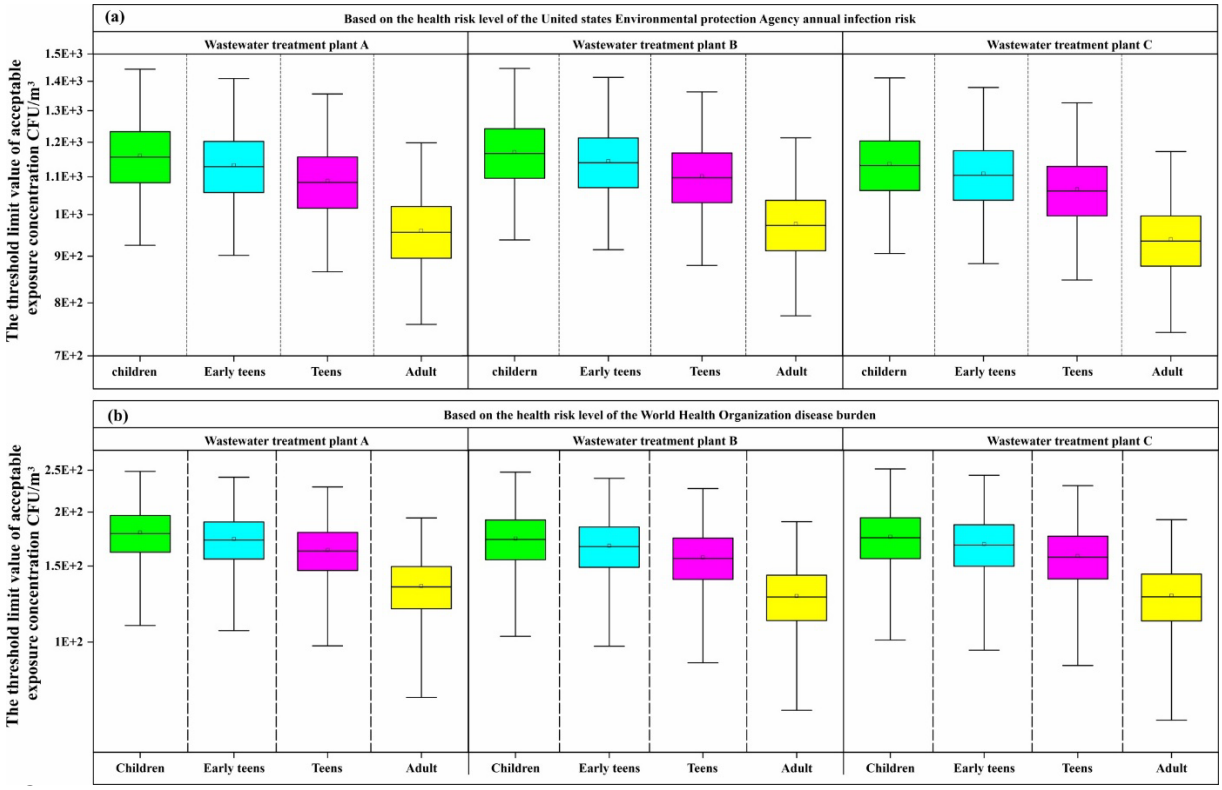


Fig. 1 Threshold limit values (TLVs) of acceptable *E. coli* exposure concentrations (CFU/m^3) for residents engaged in rowing, based on discharges from WWTPs A, B, and C. (a) U.S. EPA annual infection risk; (b) WHO disease burden benchmark. The simulation was run for 10,000 iterations; the bottom and top of the box represent the first and third quartiles (25th and 75th percentile values, respectively). The band inside the box represents the second quartile (median), and the tetragon denotes the average. The bottom and top of the whiskers represent the 5th (optimistic estimate in the best-case scenario) and 95th (conservative estimate in the worst-case scenario) percentiles, respectively.

It is essential to clarify that these TLVs are not regulatory thresholds, but rather modeled concentrations derived from exposure-dose calculations based on specific health risk benchmarks (Viau 2021). For the four age groups analyzed based on the annual infection risk level, the average TLV of WWTP C was (1.06E+03 CFU/m^3), which was approximately 2% and 5% lower than those of WWTPs A (1.08E+03 CFU/m^3) and B (1.11E+03 CFU/m^3), respectively (Supplementary Material Table 3). Meanwhile, based on the DB risk level, the average TLV of the WWTP C (1.58E+02 CFU/m^3) was roughly 3% and 8% lower than those of WWTP A (1.62E+02 CFU/m^3) and B (1.70E+02 CFU/m^3), respectively (Supplementary Material Table 3). These slightly lower TLVs may appear inconsistent with prior literature suggesting tighter

discharge thresholds near sensitive ecosystems. However, the TLV benchmark here is derived not from regulatory limits, but from exposure-dose calculations under defined risk levels. In this study, WWTP C yielded lower TLVs not because of stricter standards, but because it had a higher treatment capacity and a significantly greater dilution potential (due to the high flow rate of the receiving water), which led to reduced environmental concentrations of *E. coli*. Thus, TLV differences between plants reflect variations in treatment capacity and receiving water characteristics, rather than regulatory stringency. As a result, lower modeled TLVs are needed to maintain the same health risk threshold. These findings are explained by the 1.5–4.0 times higher treatment capacity of WWTP C (Supplementary Table 1) and a three-orders-of-magnitude greater flow rate of its receiving water (Supplementary Table 2) compared with the other two WWTPs. Both factors jointly influence TLVs: a higher treatment capacity reduces the effluent microbial load, while a higher receiving water flow rate enhances dilution, resulting in a lower concentration of *E. coli* in the environment. This directly reduces the exposure dose experienced by downstream populations. Thus, TLVs are influenced not only by the characteristics of the receiving water but also by the treatment plant operational scale and performance. It is therefore not accurate to attribute variations in TLVs or discharge regulation needs solely to flow rate.

Other contributing factors, such as ecosystem sensitivity, pathogen persistence, recreational water use, population density, and cumulative upstream discharges, must also be considered. This finding coincides with those in the literature. The discharge standard for WWTPs located in the Taihu watershed in China is considerably stricter than that applied nationwide (Department of Ecology and Environment of Jiangsu Province, 2007; Department of Ecology and Environment of Anhui Province, 2016) because Taihu is a notably more sensitive receiving water body than others. WWTPs in Hong Kong implement different drainage standards when discharging into various kinds of receiving water (coastal, inshore, and marine waters) (Environmental Protection Department of the government of the Hong Kong Special Administrative Region, 1997). In the United States, the Federal EPA authorizes local EPAs in each

state or county to establish specific discharge standards for regional WWTPs based on the local receiving water environment (U.S. Environmental Protection Agency, 2023). The urban wastewater treatment guidelines issued by the European Union mandate member states to set specific, stringent WWTP discharge standards depending on the individual characteristics of the receiving water in different countries (European Union, 2000). In the global environmental regulation for WWTP discharge standards in Italy (D. Lgs. n. 152/2006), destination-specific environmental quality targets are specified for various receiving water bodies, and varied limits are identified for the same water quality parameters of discharge standards.

Furthermore, the TLV based on the risk level of annual infection (9.30E+02–1.23E+03 CFU/m^3) was approximately one order of magnitude higher than that based on the DB (1.25E+02–2.00E+02 CFU/m^3) (Supplementary Material Table 3). This finding illustrates that the TLV derived from the WHO DB risk level is more stringent and less tolerable in practical applications. The WHO DB threshold of 10^{-6} DALYs per person per year (pppy) is derived by considering the morbidity and mortality associated with different pathogenic *E. coli* subtypes and thus reflects a more conservative health-protection target. In contrast, the U.S. EPA threshold of an annual infection risk of 1 in 10,000 (10^{-4}) assumes equal weight across all *E. coli* subtypes. It aligns with the general assumptions of the QMRA framework applied in this study. The choice to use the EPA risk level in our primary analysis was supported by the underlying assumptions of the QMRA model, which considers uniform infectivity across *E. coli* subtypes, an approach consistent with the EPA framework. Moreover, this study focuses on exposure scenarios where infection risk serves as the primary decision-making criterion, such as recreational water use, aligning with applications commonly addressed in U.S.-based regulatory frameworks. The EPA threshold is also frequently adopted in QMRA literature for *E. coli*, making it a practical reference point for comparison and benchmarking (Heida et al., 2025; Devane et al., 2021; Shoult et al., 2021). Nonetheless, our results explicitly highlight the contrast in TLV outcomes under both WHO and EPA criteria, reinforcing that the selected risk metric significantly influences the tolerable microbial load in receiving waters. These

findings are consistent with previous studies (Hamilton et al., 2019; Rasheduzzaman et al., 2019), which also demonstrated that initiating QMRA using the WHO DB threshold yields more stringent outcomes than using the EPA infection-based threshold.

The acceptable *E. coli* concentration thresholds for swimming are presented in Figure 2 and Supplementary Material Table 4 for four age groups across WWTPs A, B, and C. Units for swimming are expressed as CFU/100 mL to reflect water ingestion exposure. In contrast, rowing TLVs are expressed as CFU/m^3 for inhaled bioaerosols. Results are shown separately for the U.S. EPA (Fig. 2a) and WHO disease burden (Fig. 2b) risk levels. A consistent age-related trend in TLVs is observed across both exposure pathways. Similar to the results for rowing, the permissible *E. coli* concentrations for swimming decrease with increasing age under both the U.S. EPA and WHO risk benchmarks. This pattern holds for all three WWTPs studied, with children exhibiting the highest TLVs and adults the lowest Fig. 2a and b). Overall, TLVs for swimming were approximately 2.2-fold lower than those for rowing (Figure 2 and Supplementary Table 4). This reduction is attributed to higher exposure frequencies and more direct exposure routes during swimming, such as full-body contact and incidental ingestion of water, which increase the exposure dose. As a result, lower acceptable concentrations (i.e., TLVs) are needed to meet the same health risk thresholds. These findings are supported by previous studies emphasizing the influence of recreational activity type on microbial health risk. For instance, Kundu et al. (2013), Fuhrimann et al. (2016), and Fang et al. (2018) reported higher microbial risks associated with full-contact activities, such as swimming, compared to low-contact activities, such as rowing. Cui et al. (2017) likewise concluded that playing or swimming in water posed a greater health burden than rowing. Van Abel et al. (2017) also found that the annual probability of illness is typically higher for swimmers than rowers.

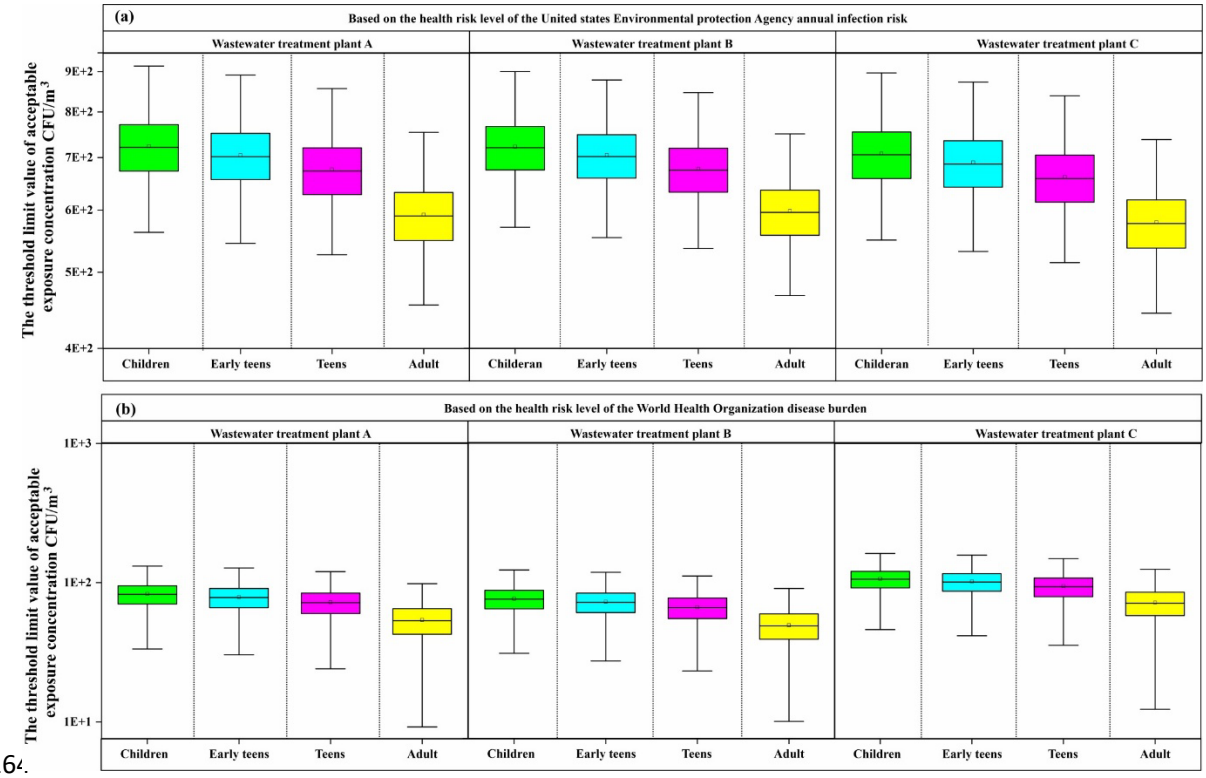


Fig. 2 Threshold limit values (TLVs) of acceptable *E. coli* exposure concentrations (CFU/ 100ml) for residents swimming in the receiving waters downstream of WWTPs A, B, and C. (a) U.S. EPA annual infection risk; (b) WHO disease burden benchmark. The simulation was run for 10,000 iterations; the bottom and top of the box represent the first and third quartiles (25th and 75th percentile values, respectively). The band inside the box represents the second quartile (median), and the tetragon denotes the average. The bottom and top of the whiskers represent the 5th (optimistic estimate in the best-case scenario) and 95th (conservative estimate in the worst-case scenario) percentiles, respectively.

Additionally, although children exhibited higher TLVs in our results due to lower inhalation or ingestion rates and exposure assumptions used in the QMRA, many studies have highlighted their heightened biological vulnerability to microbial contamination. Wade et al. (2003) identified children as highly susceptible to E. coli, while Sinclair et al. (2009) and Schets et al. (2011) observed higher illness rates among child swimmers compared to adults. Kundu et al. (2013) demonstrated that the youngest age groups are at the highest risk from ingesting surface water. Similarly, Vergara et al. (2016) reported a 46% higher mean risk in children than adults, and Cui et al. (2017) found elevated health burdens in child swimmers across various recreational water settings. These studies further validate the importance of age-

specific analysis when assessing microbial risk thresholds, particularly under the WHO DB framework, which accounts more directly for disease severity and age sensitivity. Although this study incorporated climate parameters (temperature and relative humidity) into the QMRA framework via ranges for key exposure variables, particularly aerosolization, the TLVs presented were not separated by distinct climate scenarios. This was due to the lack of empirical or site-specific data linking specific climate conditions to multiple pathogen fate and transport parameters. Nevertheless, the results suggest that warmer and drier conditions may lower TLVs due to increased aerosolization efficiency. In comparison, cooler or more humid conditions may reduce the potential for airborne transmission, potentially allowing higher TLVs to be set under the same risk benchmarks. Moreover, climate variables may also affect other exposure-relevant parameters such as microbial survival in water, UV inactivation, or regrowth rates, which were not explicitly modeled in this analysis. Future studies could extend this framework by quantifying TLVs under specific climate scenarios (e.g., high temperature vs. low temperature; high vs. low humidity) and incorporating their effects on additional model parameters to support climate-responsive risk management strategies.

3.2 Sensitivity analysis for reverse QRMA results

Figure 3 illustrates the sensitivity ranking of key exposure input parameters influencing the calculated TLVs of acceptable *E. coli* exposure concentrations. For rowing, the uncertain input parameters included exposure time (*ET*), breathing rate (*Br*), aerosol ingestion ratio (*Ag*), droplet ingestion rate (*IR*), relative humidity (*RH*), temperature (*T*), and annual exposure frequency. In contrast, the swimming exposure model considers only direct water ingestion; therefore, the sensitivity analysis was limited to annual exposure frequency, exposure time (*ET*), and the water ingestion rate (I_S). Fig. 4 shows the specific contribution to variance of the input variable parameters that influence the output TLVs of acceptable exposure concentration. Similar results were observed for WWTPs A, B, and C, with consistent sensitivity patterns across all four exposed age groups (children, early teens, teens, and adults).

The sensitivity analysis for rowing exposure identified annual exposure frequency as the most influential parameter, accounting for 59.4% (Fig. 4) of the total variance and exhibiting a strong negative Spearman correlation (-0.755) (Fig.3).

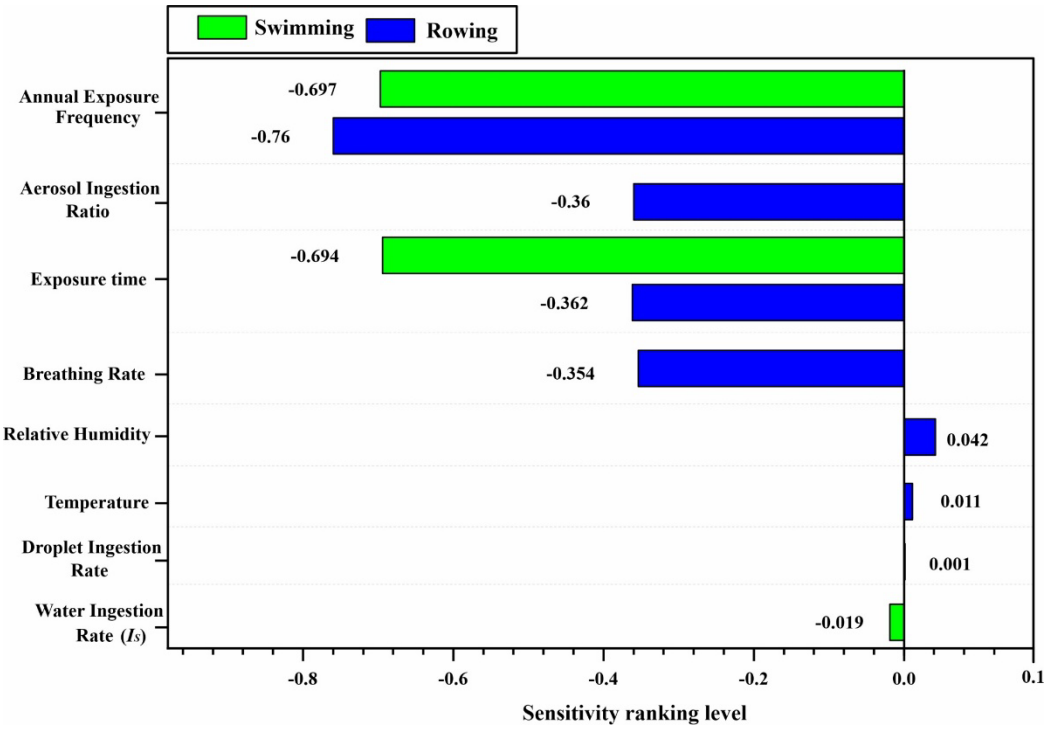


Fig. 3 Sensitivity analysis based on the ranking of input variable parameters that affect the TLVs of acceptable exposure concentration when nearby residents are rowing and swimming.

This was followed by the aerosol ingestion ratio (13.7%), exposure time (13.7%), and breathing rate (13.1%), which contributed comparably to the overall variance. In contrast, environmental factors, including relative humidity (0.2%) and temperature (<0.1%), as well as the droplet ingestion rate (<0.1%), showed negligible influence (Fig. 3). These results indicate that the frequency of rowing activity is the primary determinant of acceptable bioaerosol concentrations. Consequently, managing microbial risk in receiving waters impacted by WWTP discharges may be most effectively achieved by regulating the number of rowing days per year, for example, through seasonal restrictions or activity permits. The sensitivity analysis for swimming exposure revealed that annual exposure frequency (50.2%) and exposure time per event (49.7%) were the dominant parameters, together accounting for nearly all variance in the swimming TLV (Fig. 4).

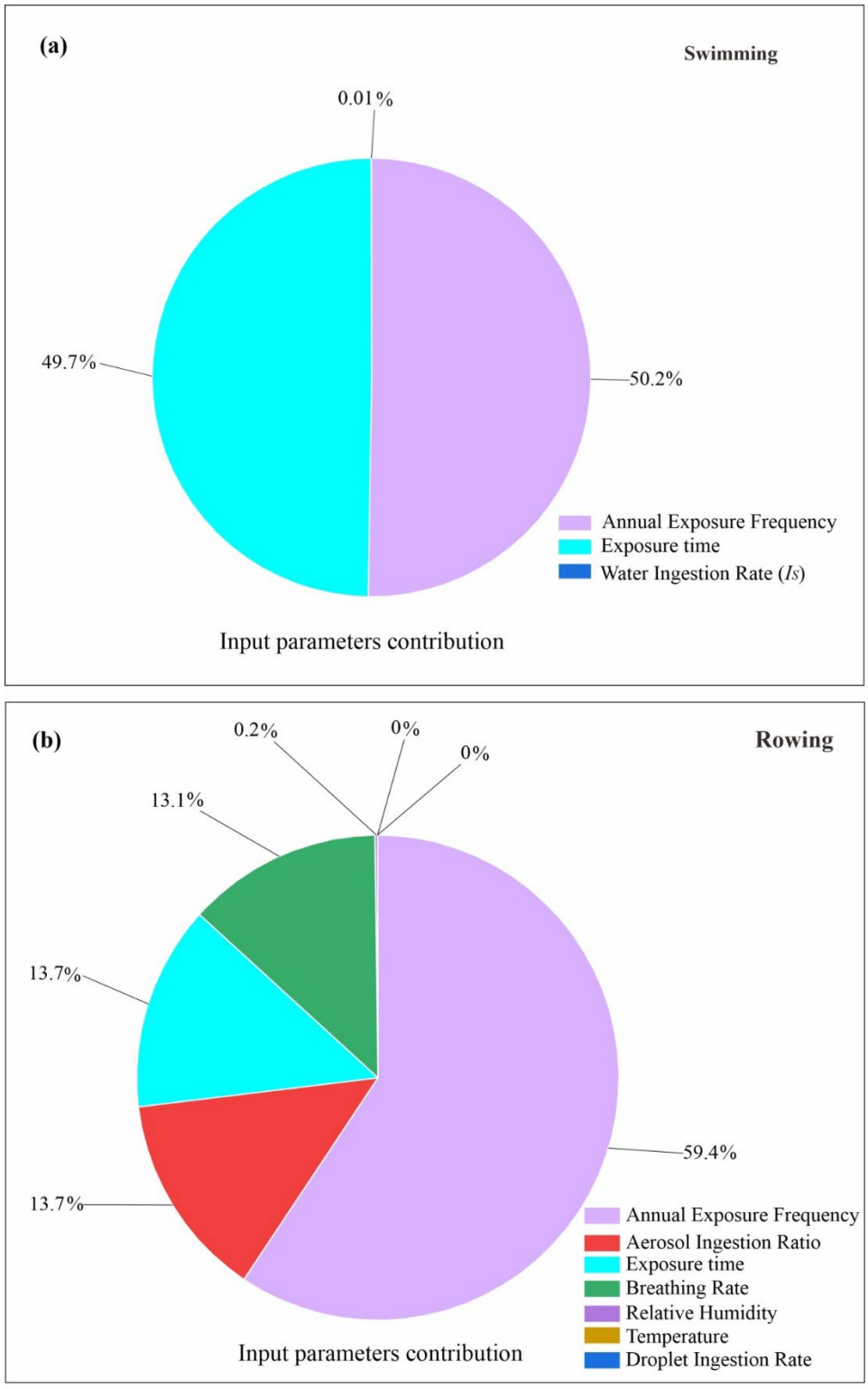


Fig. 4 Sensitivity analysis based on the specific contribution of input variable parameters that affect the output TLVs of acceptable exposure concentration when nearby residents are (a) rowing and (b) swimming.

Both exhibited strong negative Spearman correlations (-0.697 and -0.694, respectively), indicating that increased exposure duration or frequency necessitates

lower permissible *E. coli* concentrations to meet health risk benchmarks (Fig. 3). In contrast, the water ingestion rate (I_S) showed negligible influence (0.0% contribution), reflecting its limited variability in the modeled age-stratified distributions. These results underscore that behavioral factors, such as the frequency and duration of swimming, are the primary determinants of acceptable microbial levels in recreational waters, rather than the volume of water ingested per hour. It should be noted that the sensitivity analysis presented here was based on plausible ranges of exposure parameters rather than discrete environmental scenarios. While annual frequency and exposure time emerged as key drivers for swimming, this study did not explicitly partition climate variables (e.g., RH, T) within the swimming ingestion model. Future work could extend this framework by evaluating TLVs under specific climatic conditions and incorporating climate-sensitive parameters, such as pathogen decay, regrowth, or seasonal variations in recreational water use, thereby enabling more dynamic, climate-informed risk management.

4. Conclusion

Our study sheds light on the varying tolerance levels of different age groups to acceptable concentrations of *E. coli* in WWTP tailwater, based on two distinct risk levels. Younger populations exhibit a higher tolerance compared to adults, highlighting the importance of age considerations in risk assessments. Moreover, adherence to the prudent risk level recommended by the WHO DB results in a stringent TLV for *E. coli* in tailwater, posing challenges for populations engaging in recreational activities in receiving water. Furthermore, we observed significant disparities in acceptable exposure concentrations between swimming and rowing activities, with swimming exhibiting considerably lower thresholds. Sensitivity analysis highlighted the pivotal role of regulating the annual exposure frequency of rowing, or limiting swimming activities during high relative humidity conditions, in extending the TLV benchmarks for WWTP tailwater. Overall, this study addresses critical gaps in the immediate assessment of acceptable exposure concentration limits for tailwater exposure, providing valuable insights for stakeholders. By elucidating

the most effective measures for extending TLV benchmarks, our findings empower policymakers to optimize WWTP discharge standards tailored to the unique characteristics of local receiving water environments. Overall, our research contributes to informed decision-making and underscores the importance of proactive strategies in safeguarding public health and environmental quality.

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