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Faecal bacterial contamination of rivers: Evolution, suitability, and health risk implications for recreational use

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Abstract

The suitability of rivers for recreational purposes is an important issue in public health and water management. A quantitative microbial risk assessment was conducted using a dataset of *E. coli* concentrations in three Sinaloa rivers from 2013 to 2018 to define the level of faecal contamination and estimate the risks of acquiring gastrointestinal infections (GI) from recreational exposure. Faecal contamination was conditioned by river flow or time ($p > 0.05$) and increase during summer in urban areas. The national laws classify these rivers as suitable for recreational activities. However, the dose-response model estimated the probability of acquiring GI during recreational use ($>1.0 \times 10^{-4}$), which implies the occurrence of 283 and 788 GI annually in adults and children, respectively. This research exposes the risk for the development of GI in the population of the region, especially in children; and justifies the controlling microbiological quality of rivers used for recreational use.

KEYWORDS
gastrointestinal illness, pollution, QMRA, rivers, water recreation

1 | INTRODUCTION

Fresh and marine waters are frequently affected by microbiological contamination resulting from anthropogenic activities, including agricultural, industrial, urban (CONAGUA, 2018), and livestock activities (Hathaway et al., 2011). Similarly, the influence of effluent from wastewater treatment plants as a vehicle for the dissemination of microorganisms has been described (Kotlarska et al., 2015). Microbiological contamination can prevail when receiving water bodies compromise the environmental and sanitary quality of these resources (Stone et al., 2008). Exposure to water contaminated with infectious agents increases the risk of contracting gastrointestinal (GI) infections during recreational or irrigation use (World Health Organization, 2016). The World Health Organization (WHO) estimates that 842 000 cases of deaths occur worldwide due to GI diseases associated with inadequate water, which represents 1.5% of the total disease burden and 58% of diarrhoeal diseases (Prüss-Ustün et al., 2014; WHO, 2020). *Escherichia coli*, *Shigella*, *Cryptosporidium*, and *Giardia* have been identified as the main aetiological agents of faecal origin (Clarke et al., 2017; Hlavsa et al., 2015; WHO, 2003).

Several authors have recognized the problem of chemical and microbiological contamination of surface water and its potential relationship with GI diseases in Mexico (Rubio-Arias et al., 2016). The National Epidemiology Department (NED) reported 5 360 604 cases of GI in 2019 nationally (DGE, 2020), presumably related to contact with contaminated food or water. In Sinaloa, 146 890 cases of GI diseases were reported in 2019. In addition, the NED reported cases of salmonellosis, shigellosis, cholera, giardiasis, and hepatitis A (DGE, 2020), which are considered pathogens of waterborne diseases worldwide (Hlavsa et al., 2015). However, this epidemiological association has not yet been clarified in Mexico.

Given the presence of infectious agents in recreational waters, and the use of this environmental resource for anthropogenic activities, research has been conducted to determine the exposure-response relationship that links the concentrations of microorganisms in water (accidentally ingested) with reported GI disease rates (Colford et al., 2007). Quantitative microbial risk assessment (QMRA) is a mathematical model that predicts the risks of infection, disease, and death due to exposure to pathogens via the environment (Smith et al., 2015; WHO, 2016). The success of a QMRA model is based on the correct description and argumentation of its elements: (i) hazard identification, (ii) exposure assessment, (iii) evaluation of the dose-response, and (vi) risk characterization (EPA, 2010; Haas et al., 1999).

Faecal contamination of water with pathogenic bacteria, viruses, and protozoa remains one of the main causes of waterborne diseases worldwide (Prüss-Ustün et al., 2014). QMRA for specific pathogens is not always feasible or the data are not available. *E. coli* and *Enterococcus* spp. are used as faecal indicator bacteria (FIB), since both bacteria are of intestinal origin (Korajkic et al., 2018). In particular, *E. coli* is recognized as a regulatory microbiological limit for water safety (Meals et al., 2013; WHO, 2016). In contrast, certain limitations have been described for the use of *E. coli* as a FIB, including its vulnerability to survival in natural environments and the geographical and temporal variability (Gitter et al., 2020). Although the absence of this indicator does not guarantee the absence of faecal pathogens (Van Lieverloo et al., 2007), high concentrations of *E. coli* have been linked to waterborne diseases (Clarke et al., 2017). FIB can also contain pathogenic subsets (e.g., *E. coli* O157:H7) that can be considered for a QMRA (Clarke et al., 2017).

Regulations for sanitary and ecological water quality in Mexico include NOM-001-SEMARNAT-1996, NOM-003- SEMARNAT-1997, and CE-CCA-001/89, which are based on faecal coliform limits (1000 MPN·100 mL^{-1}) as a microbiological standard to evaluate the ecological quality of water and its suitability for use (DOF, 1989, 1997). FIB concentration was measured owing to their local regulatory importance for recreational water usage.

In Sinaloa, some studies have reported that surface water bodies are reservoirs of various pathogenic microorganisms, such as *Salmonella* spp. (Fuhrimann et al., 2016; Jiménez & Chaidez, 2012; López et al., 2009), hepatitis A virus (Hernández-Morga et al., 2009), *Giardia*, and *E. coli* (Ahumada-Santos et al., 2014; López et al., 2009). The presence of FIB denotes contamination by faecal origin and anthropogenic activity (Chagas et al., 2010). The presence of FIB does not necessarily denote the presence of pathogens (Gitter et al., 2020), but is an indicator of faecal pollution and therefore, is a potential health risk during recreational activities.

Various water bodies provide many economic activities in Sinaloa. The largest amount of water is used for agricultural irrigation, livestock, and aquaculture farms in the region (CONAGUA, 2018). All these activities generate effluents contaminated by physicochemical and microbiological agents before reaching bays and estuaries (Ahumada-Santos et al., 2014). In addition, waste discharges from rural populations that do not have adequate drainage and sanitation systems affect the local water resources and can contribute to faecal contamination (Ahumada-Santos et al., 2014).

Highlights

1. First report of the evolution of the faecal pollution of freshwater of the north-central rivers of Sinaloa and its implication in health due to recreational.
2. The behaviour of faecal pollution warns the increase of the concentration de *E. coli* in warm months and towards the urban and agricultural areas.
3. QMRA denote that the recreational use of the rivers in Sinaloa could represent an important exposure route for the development of GI in the population of the region.

Water resources have been traditionally used as economic and recreational sources in Sinaloa, Mexico. However, the microbiological quality of these resources has been questioned (Ahumada-Santos et al., 2014; Fuhrimann et al., 2016; Hernández-Morga et al., 2009; Jiménez & Chaidez, 2012; López et al., 2009), and the information available is limited to its epidemiological relationship with waterborne diseases. For a better understanding of the effect of faecal contamination of water resources in Sinaloa on human health, the integration of microbiological water quality monitoring and estimation of health risks is needed to fill this gap. This study aimed to determine the microbiological quality and health risk associated with the recreational use of major rivers in Sinaloa, Mexico, from 2013 to 2018.

2 | MATERIALS AND METHODS

2.1 | Location and characteristics of the study site

A longitudinal study was conducted to describe the fate of faecal pollution of the Culiacan, Sinaloa, and El Fuerte rivers, and its implication in the risk of acquiring GI diseases in Sinaloa. The rivers of Culiacan, Sinaloa, and El Fuerte are in the 010, 074, and 075 irrigation districts of Sinaloa, respectively (Figure 1). The districts are in the northern part of the Pacific coast, between meridians 105° 23′ 32″ and 109° 26′ 52″ OL, and parallels 22° 28′ 02″ and 27° 02′ 32″ NL. The districts are characterized by developing agricultural activities near the rivers mentioned. Sinaloa is characterized by a semi-dry warm climate with an annual average temperature and precipitation of 22°C and 729 mm, respectively (INEGI, 2016).

2.2 | Dataset of *E. coli* concentrations

This study used a dataset of microbial water quality of the Culiacan River, El Fuerte River, and Sinaloa River provided by the National Water Commission of Mexico. The microbiological

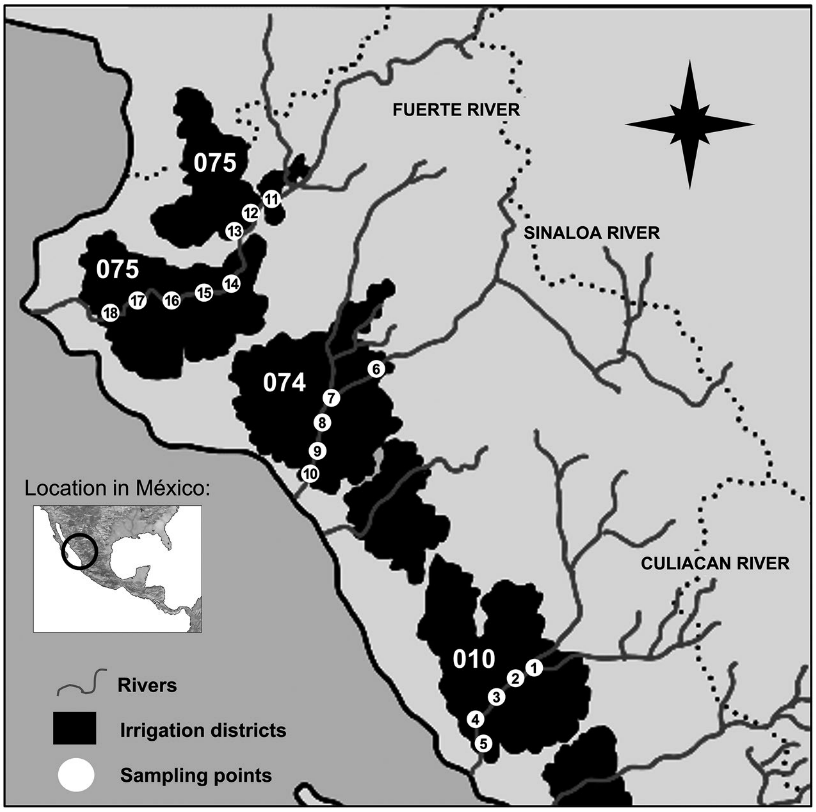

FIGURE 1 Map of Rivers located in Sinaloa, Mexico

quality of the water was determined using *E. coli* as a faecal indicator (MPN·100 mL^{-1}), which was measured at different sites along with the flow of the Culiacan (n = 5), Sinaloa (n = 5), and El Fuerte rivers (n = 8) during 2013–2018. The dataset contained 596 observations of *E. coli* quantified from surface water samples collected from the Culiacan (n = 165), Sinaloa (n = 168), and El Fuerte (n = 263) rivers. Monitoring points were selected from the downtown area of the city (0 km) to the coast. Table 1 summarizes the *E. coli* data recorded during this period in the three rivers. Quantification of *E. coli* in water samples was performed using the IDEXX Colilert™ Most Probable Number (MPN) method according to the manufacturer's instructions. This methodology was selected because it is highly reproducible for the determination of *E. coli* in a natural water matrix (Kinzelman et al., 2005). The *E. coli* values of <1 MPN·100 mL^{-11} and >2419.6 MPN·100 mL^{-1} were considered as 1 MPN·100 mL^{-1} and 2419.6 MPN·100 mL^{-1}, respectively. The difference in the number of water samples between the rivers depended on the availability of the collection point.

2.3 | Water volume ingested (WVI)

It has been previously reported that the WVI and exposure time varies with the age of the swimmer (Dufour et al., 2006). Studies have reported that the estimated WVI for children and adults is 37 mL and 16 mL, respectively, per event when swimming (Dorevitch et al., 2011; Dufour et al., 2006). The WVI during recreational activities for both populations was fitted by a lognormal statistical distribution (Ln). The parameters of the WVI are listed in Table 2.

2.4 | Exposure assessment

The exposure assessment of this study assumes a scenario to estimate the risk of acquiring GI infections during recreational use of a certain river in Sinaloa, Mexico. Certain factors were considered to integrate this QMRA: (i) the concentration of *E. coli* from the rivers of Sinaloa during 2013–2018 (dataset of CONAGUA), (ii) the volume of water ingested by children or adults, and (iii) the exposure frequency of the population.

For the analysis of the population exposure, the data of WVI (mL) and *E. coli* concentrations (MPN·100 mL^{-1}) were fitted to a normal distribution using the Anderson Darling, Kolmogórov-Smirnov (K-S) or chi-square (χ^2) tests in the Oracle Crystal Ball software (vs. 11.1.2.3.500). Table 2 presents a summary of the data used in this study.

For exposure assessment, the total population of Sinaloa was considered to estimate annual GI cases. According to the Mexican population census in 2015 (INEGI, 2020), Sinaloa has a total population of 2 966 321 inhabitants, of which 26.5% are considered to be children.

A Monte Carlo simulation was used to generate 10 000 iterations for a fitted WVI and *E. coli* concentration dataset (Table 2). The exposure was calculated using Equation (1):

$$D = VC \quad (1)$$

TABLE 1 Descriptive analysis of *E. coli* (MPN·100 mL^{-1}) levels monitored in the rivers of the north-center of Sinaloa between 2013 and 2018

River		Site	Sample	2013	2014	2015	2016	2017	2018	GMS[a]	DE	% CV	IC_{95}	A	U	AC
Culiacan	1	Puente negro	32	57	33	107	161	227	36	**104**	78	75	76.9–131.0		X	
	2	USE	35	26	8	41	14	11	5	19	14	74	14.4–23.6		X	
	3	San Pedro	35	29	17	19	30	24	46	**29**	9	31	26.0–31.9		X	
	4	El Limoncito	35	20	176	131	14	17	25	**64**	71	111	40.5–87.5	X	X	
	5	El Molino	28	11	6	6	7	4	4	16	14	85	10.6–2 1.4	X	X	
Sinaloa[b]	6	Bamoa	34	26	16	**43**	19	9	3	19	14	73	14.3–23.7	X	X	
	7	Guasave	33	17	16	**68**	18	21	6	23	23	98	15.2–30.9		X	
	8	Tamazula	34	17	29	**60**	21	16	23	28	16	60	22.6–33.4	X	X	
	9	La Brecha	34	16	35	**27**	20	12	23	22	8	38	19.3–24.7	X	X	
	10	Alamito	33	6	8	**23**	14	12	7	12	6	54	9.9–14.1		X	X
El Fuerte[a]	11	El Mahone	34	4	13	3	3	4	3	6	6	108	3.9–8.0		X	
	12	El Fuerte	34	4	5	2	3	2	2	3	1	37	2.7–3.3	X	X	
	13	Baroten	35	21	4	3	3	4	2	6	7	120	3.7–8.4	X	X	
	14	Mochicahui	34	65	98	33	27	39	35	52	30	57	41.9–62.1	X	X	
	15	San Miguel	35	17	20	31	12	29	23	**22**	8	34	19.4–24.7	X	X	
	16	Cohuibampo	34	15	16	27	9	5	17	15	8	53	12.3–17.7	X	X	
	17	San Jose de A.	35	10	11	11	7	4	7	8	3	33	7.0–9.0	X	X	
	18	Las Grullas	22	25	ND	17	10	16	32	17	11	68	12.1–21.9	X		

Abbreviations: CV, coefficient of variation (%); DE, standard deviation; GMS, geometric mean for the six-year dataset; IC_{95}, confidence interval 95%; ND, not determined.

[a]Within the GMS, values in bold denote statistical differences between sampling points of each river.

[b]Within the flow of each year, values in bold denote statistical differences between each year evaluated.

TABLE 2 Model data and distributions

Parameter	Distributions/Values	References
a	0.000105	EPA (1986)
k	0.0000511	
V	Ln (0, 0.02, 1.51) [a]Adults	Dorevitch et al. (2011)
	Ln (0, 0.04, 1.39) [a]Children	
$C_{E.coli}$ Puente negro	Ln(0, 4.49, 1.94)[a]	CONAGUA (2018)
$C_{E.coli}$ USE	Ln (0, 2.70, 2.24)[a]	CONAGUA (2018)
$C_{E.coli}$ San Pedro	Ln (−1.49, 3.22, 1.84)[a]	CONAGUA (2018)
$C_{E.coli}$ El Limoncito	Lg (3.60, 0.80)[b]	CONAGUA (2018)
$C_{E.coli}$ Sataya	U (−0.16, 4.78)[c]	CONAGUA (2018)
$C_{E.coli}$ Bamoa	W (−2.12, 5.40, 3.72)[d]	CONAGUA (2018)
$C_{E.coli}$ Guasave	Lg (2.96, 0.79)[b]	CONAGUA (2018)
$C_{E.coli}$ Tamazula	Ln (−3.00, −3.00, 1.41)[a]	CONAGUA (2018)
$C_{E.coli}$ La Brecha	Lg (2.96, 0.72)[b]	CONAGUA (2018)
$C_{E.coli}$ Alamito Caimanero	Lg (2.96, 0.87)[b]	CONAGUA (2018)
$C_{E.coli}$ El Mahone	Ga (−0.06, 0.90, 1.70)[e]	CONAGUA (2018)
$C_{E.coli}$ El Fuerte	ME (0.69, 0.67)[f]	CONAGUA (2018)
$C_{E.coli}$ Baroten	β-PERT (−0.11, 0, 9.15)[g]	CONAGUA (2018)
$C_{E.coli}$ Mochicachui	Lg (3.80, 0.67)[b]	CONAGUA (2018)
$C_{E.coli}$ San Miguel Zapotitlan	β (1.62, 5.39, 1.20, 2.01)[h]	CONAGUA (2018)
$C_{E.coli}$ Cohuibampo	β (−1.29, 4.88, 5.03, 3.02)[h]	CONAGUA (2018)
$C_{E.coli}$ Son José de Ahome	Lg (2.09, 0.52)[b]	CONAGUA (2018)
$C_{E.coli}$ Las Grullas	β (0.88, 6.92, 1.61, 3.36) [h]	CONAGUA (2018)

Note: a and k: parameters that characterize the dose-response relationship referred to as *E. coli* infectivity constants. V: Volume of water ingested; MC: Monte Carlo simulation; C: *E. coli* concentration. Distribution (parameters): [a]lognormal (location, mean, and standard deviation); [b]Logistics (mean, scale); [c]Uniform (minimum and maximum); [d]Weibull; [e]Gamma (location, scale, and shape); [f]Maximum extreme (most likely, scale); [g]β-PERT (minimum, most likely, and maximum); [h]β (min, max, α, and β).

where *D* is the exposure dose, *V* is the volume of water ingested, and *C* is the concentration of *E. coli* (MPN·100 mL^{-1}) ingested during a water recreational event.

2.5 | Dose-response model and risk characterization

This is an exploratory risk assessment study not based on a conventional dose-response framework, where the dose-response model is non-threshold in nature. This alternative model considers the presence of a threshold dose that is required to be ingested to produce infection or disease. Using this model and the ingested dose calculated above, the risk of illness for an individual exposed to a single event of swimming was estimated. The inferior and superior threshold values used were 1 MPN and 2419.6 MPN, respectively.

To address a single recreational exposure to water contaminated with FIB, the data of *E. coli* were fitted in a dose-response model by using the exponential Equation (2) (Sunger & Haas, 2015):

$$P = a + (1 - a)(1 - e^{-kd}) \quad (2)$$

where *P* is the probability of risk of infection for an individual exposed to *E. coli* dose *d* through ingestion. *a* and *k* are parameters that characterize the dose-response relationship referred to as *E. coli* infectivity constants.

The annual infection risks are estimated using Equation (3):

$$P\,(year) = 1 - (1 - P)^n \quad (3)$$

where *P* (year) is the annual cumulative risk of infection, P is the probability of illness for an individual exposed during recreational activities on a certain number of days "n" in a year. For this QMRA, "n" is assumed to be 6 days per year (Clarke et al., 2017; Fuhrimann et al., 2016).

2.6 | Disability-adjusted life-years

Risk characterization was conducted to integrate hazard identification, exposure assessment, and the dose-response relationship to determine a health outcome (risk of infection, illness, and mortality) (Haas et al., 1999). The final risk was expressed in disease burden, that is, disability-adjusted life-years (DALYs) per year. DALYs

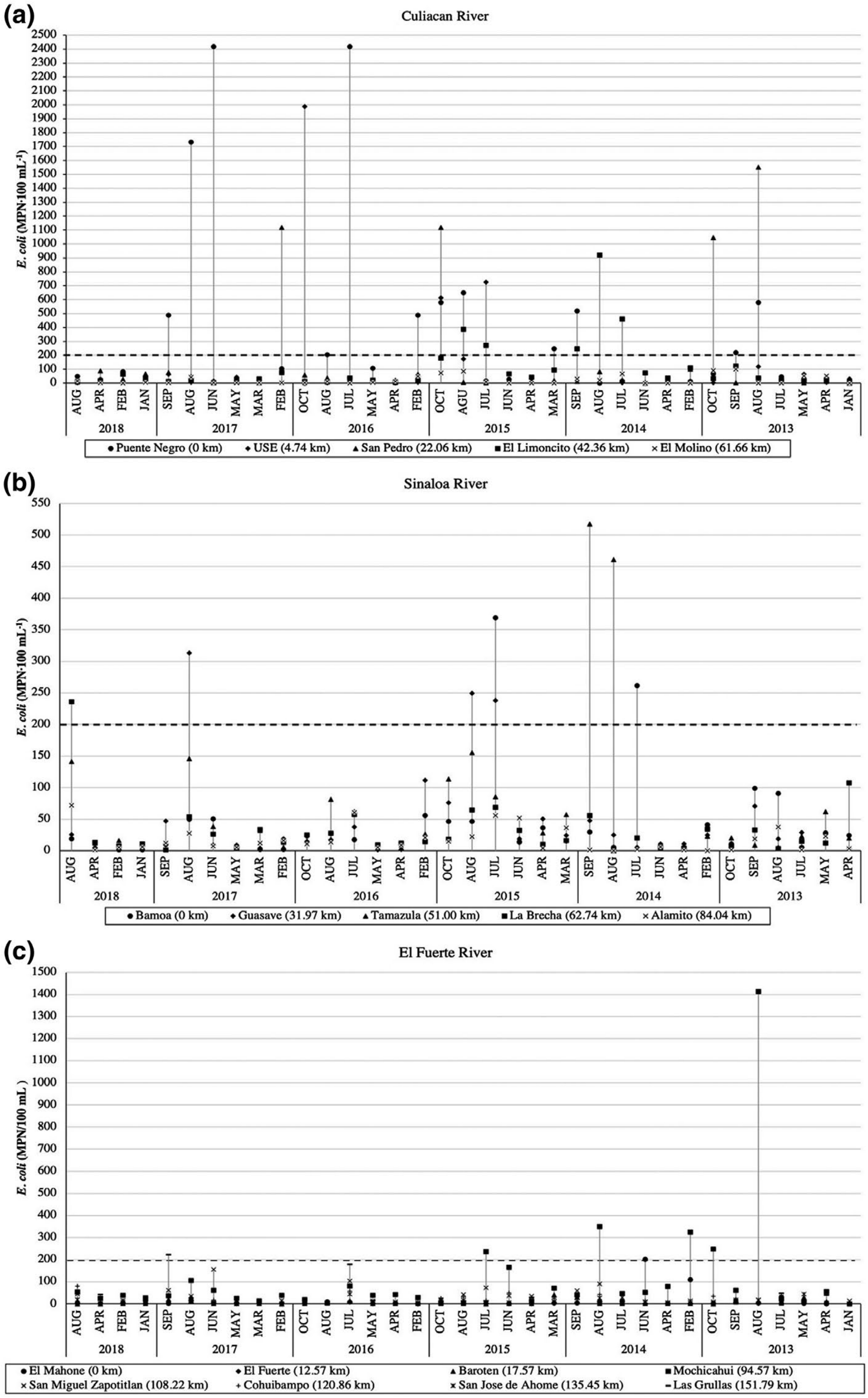

FIGURE 2 Evolution of *E. coli* levels monitored in rivers of the north-centre of Sinaloa during 2013–2018. The concentration of *E. coli* is illustrated for the Culiacan River (a), Sinaloa river (b), and El Fuerte River (c). The dashed line represents the national limit of *E. coli* acceptable for recreational activities in rivers (DOF, 1989)

are the possible adverse health effects on humans from exposure to pathogens (Katukiza et al., 2014). For this assessment, the DALYs for pathogenic *E. coli* were based on the strain with the most severe outcomes, *E. coli* O157:H7 (Howard et al., 2006). The severity weights were taken from Fuhrimann et al. (2016) and the duration for the different outcomes was taken from Katukiza et al. (2014). The average life expectancy at birth of 75.8 years in Sinaloa was obtained from the National Institute of Statistics and Geography. DALY is calculated using Equations (4) and (5) (Chhipi-Shrestha et al., 2017)

$$Risk\ of\ disease\ (Pill) = P_{inf,y} * P_{ill/inf} \tag{4}$$

$$\text{DALY} = Pill * mdf * fs \tag{5}$$

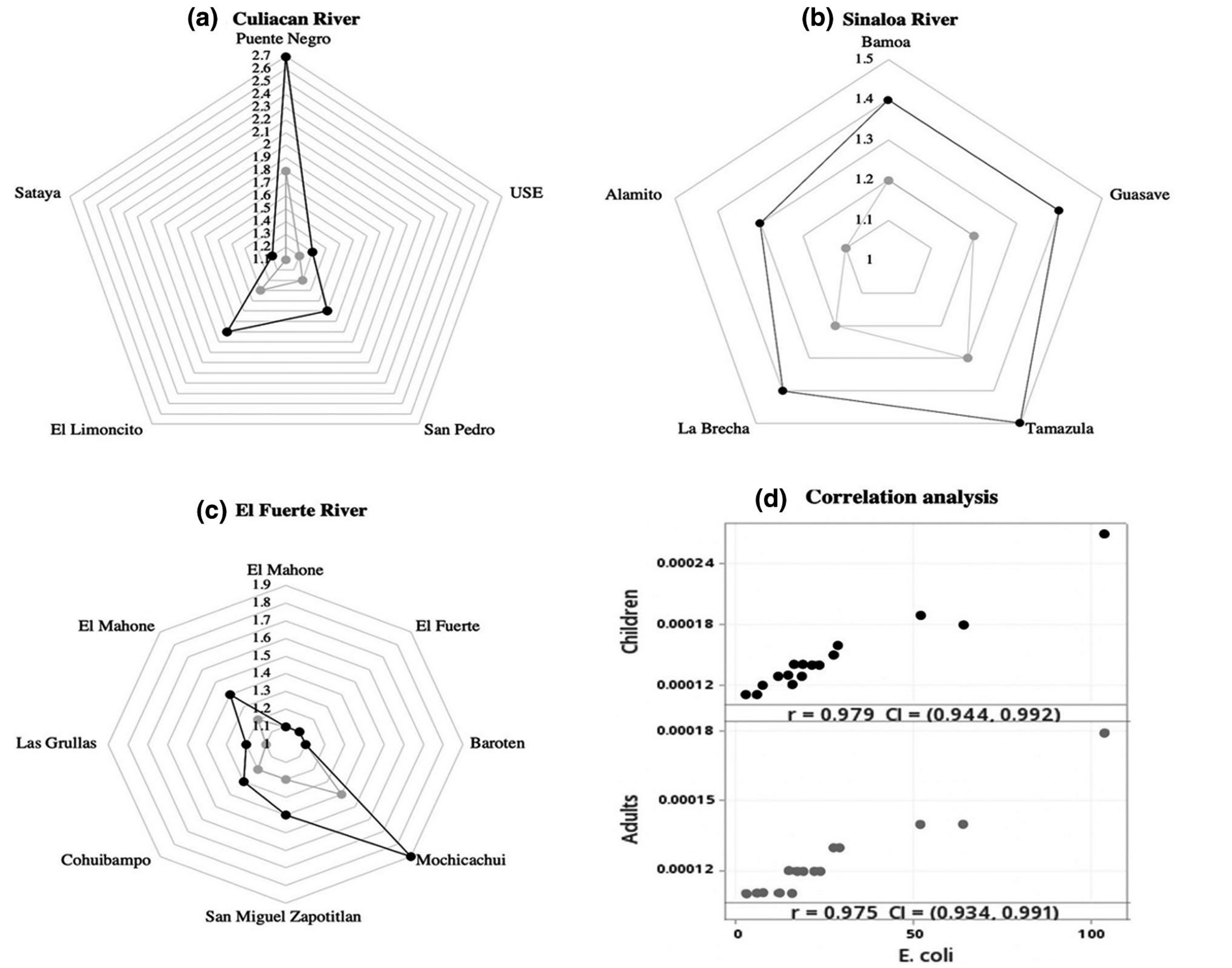

FIGURE 3 Estimation of the health risks associated with exposure of *E. coli* due to water intake during the recreational use of the rivers of the north-centre of Sinaloa. The values of health risk are expressed in a base of 10^{-4}. Figure a (Culiacan River), b (Sinaloa River), and c (El Fuerte River) show the health risk along with the flow of rivers. Figure d describes the correlation between *E. coli* levels and health risk exposure. The black and gray dots represent children and adults, respectively

where $P_{inf,y}$ is the risk of infection per year, $P_{ill/inf}$ is the risk of disease given infection, DBPC is disease burden per case (DALY/year), and *fs* is the susceptibility fraction. The values 0.53 and 0.9, for $P_{ill/inf}$ and *fs* parameters respectively, were obtained from the literature (Chhipi-Shrestha et al., 2017; Havelaar & Melse, 2003).

2.7 | Statistical analysis

The dataset of *E. coli* was analysed using descriptive statistics estimators: geometric mean, standard deviation, coefficient of variation (%CV), and confidence interval (IC_{95}). ANOVA and non-parametric tests were used to estimate the difference in *E. coli* levels between the rivers and time. A value of $p \leq 0.05$ was considered statistically significant. Excel (vs.16.39) and Minitab (vs.19.2020.1.0) were used for the analyses.

3 | RESULTS

3.1 | Microbiological quality of the rivers in northwest Sinaloa during 2013–2018

Figure 2 and Table 1 summarize the behaviour and descriptive analysis of the *E. coli* levels monitored in the rivers of the north-centre of Sinaloa during 2013–2018, respectively. The geometric mean limit of *E. coli* amongst the sampling points of the three rivers ranged from 2.0 to 227 MPN·100 mL^{-1}, with higher *E. coli* concentrations in the Culiacan River (Table 1). The %CV indicated high fluctuation in the *E. coli* values at each sampling point of the rivers throughout the 6 years period evaluated (33%–120%). The IC_{95} determined at the river flow predicted *E. coli* average values below the national limit (200 MPN·100 mL^{-1}) (DOF, 1989) (Table 1). However, 15.8% (26/165), 4.8% (8/168), and 2.3% (6/263) of the total water samples of the Culiacan River, Sinaloa River, and El Fuerte River, respectively, were found to be above this limit (DOF, 1989) (Figure 2).

Statistical analysis showed that the level of faecal pollution was different between the studied rivers (H = 11.92, p = 0.003), whose maximum values were mainly observed in the summer or warm months and the urban area (Figure 2). In addition, it was observed that the concentration of *E. coli* in the Culiacan River (F = 4.16, p = 0.010) and El Fuerte River (F = 10.47, p = 0.000) varied along the flow, but not overtime ($p > 0.05$). Conversely, the contamination levels in the Sinaloa River remained unchanged in the flow (F = 1.04, p = 0.408), but varied with time (F = 6.03, p = 0.001) (Table 1).

3.2 | Health risk estimation

Figure 3 summarizes the estimated health risk associated with exposure to *E. coli* due to water intake during the recreational use of the rivers of the north-centre of Sinaloa for adults and children. The dose-response model estimated the probability of acquiring GI infections during the recreational use of the rivers ranging from 1.1 to 2.7×10^{-4} for children and 1.1 to 1.8×10^{-4} for adults. In addition, a positive correlation was observed between the *E. coli* levels of the rivers and the risk of GI (r = 0.979, p = 0.000) (Figure 3). The estimated GI risk was statistically significant between the population evaluated (F = 4.81, p = 0.035), but not between the rivers (H = 5.49, p = 0.064).

Table 3 shows the risk and number of estimated cases of GI infections associated with exposure to *E. coli* due to water intake during recreational use of the rivers of the north-centre of Sinaloa. The average health risk estimated per river was 1.5×10^{-4}, 1.3×10^{-4}, and 1.2×10^{-4} for the rivers of Culiacan, Sinaloa, and El Fuerte, respectively. The annual cumulative risk was estimated to be 9.2×10^{-4} for the Culiacan River, 7.8×10^{-4} for the Sinaloa River, and 7.5×10^{-4} for El Fuerte River. These risk values represented up to 26 and 99 cases of GI infections in the region. In addition, average DALYs were estimated as 2.7×10^{-2} for adults and 3.1×10^{-3} for children (Table 3).

4 | DISCUSSION

One of the most important aspects of proper management of national water resources is the reduction and evaluation of the effects of public health threats associated with microbiological hazards. Traditionally, the quantification of faecal indicators, such as *E. coli* has been used to assess the microbial quality of freshwater (EPA, 1986). The microbiological quality has been previously questioned in some rivers located in the north (El Fuerte River) and central (Humaya, Tamazula, and Culiacán river) areas of Sinaloa because of the detection of faecal indicators and *Salmonella* spp., respectively (Jiménez & Chaidez, 2012; Rodríguez et al., 2016). This study shows the evolution of the faecal pollution of freshwater from the Culiacan, Sinaloa, and El Fuerte rivers and their health effects because of recreational use.

TABLE 3 Risk and estimated cases of GI infections associated with exposure to *E. coli* due to water intake during the recreational use of the rivers of the north-centre of Sinaloa

				Adult[a]				Children[b]			
River	Cities	Adult population	Children population	Per event	Yearly	DALYs	Cases	Per event	Yearly	DALYs	Cases
Culiacan	Culiacan and Navolato	762 924	296 693	1.3×10^{-4}	8.0×10^{-4}	2.9×10^{-2}	99	1.7×10^{-4}	1.0×10^{-3}	3.6×10^{-2}	50
Sinaloa	Guasave	212 654	82 699	1.2×10^{-4}	7.2×10^{-4}	2.6×10^{-2}	26	1.4×10^{-4}	8.6×10^{-4}	3.1×10^{-2}	12
El Fuerte	Ahome and El Fuerte	395 765	153 909	1.2×10^{-4}	7.4×10^{-4}	2.5×10^{-2}	48	1.3×10^{-4}	7.0×10^{-4}	2.7×10^{-2}	20

Abbeviation: DALYs, disability-adjusted life-years.

[a]Total population during 2015: 1 059 617 inhabitants of Culiacan and Navolato, 295 353 in Guasave, and 549 674 in Ahome and El Fuerte.

[b]Children population represented 28% of the population of Sinaloa during 2015.

Previous studies in Mexico have documented the contamination of water resources with pathogens of faecal origin (Ahumada-Santos et al., 2014; Fuhrimann et al., 2016; Hernández-Morga et al., 2009; Jiménez & Chaidez, 2012; López et al., 2009), and the effect of water as a contamination source on horticultural production has also been reported (González-Mendoza et al., 2015). Particularly in the Culiacan Valley, the concentration of *E. coli* has been determined, which varies from 4 UFC·mL^{-1} to 4.5 × 10^{5} UFC·mL^{-1} in natural water bodies (López et al., 2009). Canizalez-Roman et al. (2019) indicated that >33% (n = 472) of water samples collected from water resources from Sinaloa state showed *E. coli* concentrations above the permissible level for agricultural use (200 MPN·100 mL^{-1}). In the understanding of our methodology is restricted to a microbiological limit of quantification (1 MPN·100 mL^{-1} and 2419.6 MPN·100 mL^{-1}) and that could be exceeded, it should be noted that our results (Figure 2) agree with previous data in the regions. In addition, Abia et al. (2016) and Ebomah et al. (2019) reported similar values of *E. coli* in water samples of rivers, which have been linked to health risks for recreational water use.

The presence of *E. coli* in these rivers indicates a constant pattern of faecal contamination in the region, which can be associated with urban or agricultural practices where the rivers are located (Table 1). Many studies have mentioned agriculture and urban activities as contamination sources of natural water bodies in Sinaloa (Ahumada-Santos et al., 2014; Canizalez-Roman et al., 2019; Fuhrimann et al., 2016; Hernández-Morga et al., 2009; Jiménez & Chaidez, 2012; López et al., 2009). However, we do not dismiss the potential participation of other proposed sources, such as cattle/domestic animals and wildlife (Gitter et al., 2020). Ahumanda-Santos et al. (2014) have explained the effect of wastewater discharges on the microbiological quality of water resources in Sinaloa. Although the amount of waste discharged into these water bodies is not estimated, CONAGUA (2018) pointed out that 63% and 38% of water generated by municipalities (218.1 m^3/s) and industries (215.2 m^3/s), respectively, were treated. In developing countries, it has been estimated that 80% (300–400 tons/annually) of water wastes (domestic, urban, agricultural) are discharged without treatment on natural waterbodies (Ahumanda-Santos et al., 2014).

It can even be assumed that the contamination pattern could be varied along the flow or time, but with an increment in the *E. coli* concentration in the summer or warm months. The seasonality of high concentrations of the indicators of faecal pollution in rivers during summer has been evidenced, and it is mainly associated with warm environmental temperature and wastewater discharges (Sabae & Rabeh, 2007). Jacob et al. (2015) described the risk of acquiring waterborne pathogens during the summer season due to the frequency of recreational practices. These results show the faecal pollution of river waters and highlight the importance of restoring and controlling microbiological quality, especially when microbial density and recreational practices tend to increase.

As expected, faecal contamination predominated in the urban and agricultural areas (Table 1), and interestingly, the microbial load was diluted towards downstream flow. The endemic vegetation of aquatic ecosystems can act as a removal agent for physicochemical contaminants (Maine et al., 2016) and FIB (Hathaway et al., 2011). The natural dilution of freshwater resources can vary and depend on the volume of the water body, flow rate, and other factors. In addition, the growth rates of microorganisms in aquatic ecosystems vary with the temperature and intensity of solar radiation (Haas, 1983). Chittoor Viswanathan and Schirmer (2015) have demonstrated that the improvement in water quality should be carried out through the adoption of a combination of fluvial restoration measures (widening of the riverbed, creation of wetlands, and improvement of flow), the implementation of engineering alterations to basin infrastructure (stormwater controls and wastewater treatment plants), and active public participation in water management.

Water quality is a public health and environmental concern; therefore, the specific pathogens responsible for the contamination and their exposure routes should be investigated. The recreational suitability of the rivers of Sinaloa was explored in this study. The geometric mean and IC$_{95}$ of *E. coli* in the river water samples showed levels below the microbiological limits established by Mexican legislation (DOF, 1989, 1997), allowing their use for recreational activities (200 MPN·100 mL^{-1}), and public reuse service (1000 MPN·100 mL^{-1}). Similarly, the regulatory standard of the Environmental Protection Agency of the United States of America, applied to assess recreational water activities (126 CFU·100 mL^{-1} geometric mean over a month), would allow the recreational uses of these rivers. However, QMRA indicates a relevant health risk (Table 3).

Although the determination of a QMRA is based on analysing waterborne pathogens (Gitter et al., 2020), the regulations in Mexico, based the microbiological quality of water on the determination of FIB (DOF, 1989, 1997), therefore the monitoring of waterborne pathogens could be limited. Therefore, this study presents a dose-response model based on the use of *E. coli* as a reference microorganism for estimating GI infection risks during recreational water exposure. And the interpretation of the results is limited to the concentration of *E. coli*, assumed values of the input parameters, and chosen scenario of recreational exposure with accidental water ingestion (Korajkic et al., 2018; Petterson et al., 2016). The use of FIB should take with caution, while Korajkic et al. (2018) described the usefulness of FIB as a general faecal contamination indicator of freshwater and in wet weather, some factors may limit its measurement, such as its weak relationship with waterborne viruses or the values could be conditioned due to nature of water, faecal contamination source, and detection rates.

Moreover, Sunger and Haas (2015) predicted a health risk of 5.2 × 10^{-2} to 10^{-3} for recreational water use in the rivers of Philadelphia (USA). Some studies have exhibited an increased health risk (0.28–0.52) for aquatic recreational activities in South Africa (Ebomah et al., 2019). The variation in estimated health risk depends on the data and model employed. The studied rivers showed risk values (Table 3, Figure 3) higher than the limit (1.0 × 10^{-4}) declared by the WHO for aquatic recreational activities (WHO, 2003), meaning that the calculated risk of *E. coli* in adults (1.2 × 10^{-4} to 1.7 × 10^{-4}) and children (1.0 × 10^{-3} to

8.6×10^{-4}) implied 1–2 in 10 000 people and 8–10 children in 10 000 people getting infected. DALYs risk in adults (2.7×10^{-2}) and children (3.1×10^{-2}) implied 20 (adults) to 30 (children) in 10 000 people getting infected. This indicated that the recreational use of the Sinaloa rivers represents an important exposure route for the development of GI infection in the population of the region, especially in children (Table 3). The Environmental Protection Agency declared that the acceptable risk level has been adjusted from 8 to 36 cases of GI infections per 1000 individuals (EPA, 2012). Conducting QMRA with other detected pathogens in these rivers could better understand their suitability for use.

There is evidence that GI infections are an important cause of morbidity and mortality in Mexico (DGE, 2020), where water could be a vector based on the microbiological quality reported and the estimation of the health risk observed. The NED reported that GI infections represent the second or third most common diseases in the entity from 2013 to 2019, with an average of 145 082 annual cases (DGE, 2020). The highest occurrence of GI infections in Sinaloa was in 2018 (155 820), followed by 2017 (156 793), and 2013 (155 035) (DGE, 2020), and its occurrence could be linked to the high concentrations of *E. coli* observed in the studied rivers. In summer, swimming is a frequent practice because of the hydration conditions of the surface water bodies in the region (Jacob et al., 2015; Rodríguez et al., 2016). Therefore, the calculated risk in the Culiacan River, Sinaloa River, and El Fuerte River have epidemiological relevance to justify the occurrence of the annual GI infections reported in Sinaloa and propose the restoration of water quality to maintain microbiological levels within safe limits.

5 | CONCLUSIONS

Our results provide an overview of the evolution of the faecal pollution of freshwater from the north-central rivers of Sinaloa during 2013–2018 and its implication in public health because of recreational use. Faecal pollution of rivers seems to be stable over time and is influenced by regional practices and nature of each irrigation district, but it can be noted that the concentration of *E. coli* in all rivers increases in the warm months and towards urban and agricultural areas. Since swimming in natural water bodies is a regional custom mainly in summer, it should not be underestimated in terms of public health. This study proposes a QMRA model that exposes the importance of recreational use of the rivers of Sinaloa and justifies the potential of water as a vector of GI infections in the region.

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CONFLICT OF INTEREST

The authors declare no conflict of interest for this article.

DATA AVAILABILITY STATEMENT

The authors confirm that the data supporting the findings of this study are available within the article.

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