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Water quality modelling and quantitative microbial risk assessment for uMsunduzi River in South Africa

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ABSTRACT

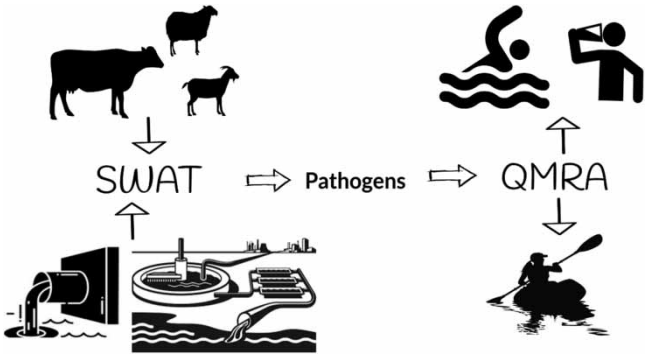
South African rivers generally receive waste from inadequate wastewater infrastructure, mines, and farming activities, among others. The uMsunduzi River in KwaZulu-Natal, South Africa, is among these recipients with recorded poor to very poor water quality. To identify parts of the uMsunduzi River that are polluted by *Cryptosporidium* and *Escherichia coli* (*E. coli*), this study mapped out pollutants emanating from point and non-point sources using the Soil and Water Assessment Tool (SWAT). Streamflow calibration in the upper and lower reaches of the catchment showed good performance with R^2 of 0.64 and 0.58, respectively. SWAT water quality output data were combined with a Quantitative Microbial Risk Assessment (QMRA) to understand the microbial health implications for people using river water for drinking, recreational swimming, and non-competitive canoeing. QMRA results for *Cryptosporidium* and pathogenic *E. coli* showed that the probability of infection for most users exceeds the acceptable level for drinking and recreation as outlined in the South African water quality guidelines, and by the World Health Organization (WHO). The results of this study can be used as a baseline to assess the economic and health implications of different management plans, resulting in better-informed, cost-effective, and impactful decision-making.

Key words: *Cryptosporidium*, *Escherichia coli* (*E. coli*), QMRA, SWAT, uMsunduzi River, water quality modelling

HIGHLIGHTS

- SWAT model identified areas of the uMsunduzi River that were polluted by *Cryptosporidium* and *E. coli*.
- SWAT water quality output data were used in QMRA.
- QMRA investigated the impacts of river water on canoeists, recreational swimmers, and those who drink the water.
- uMsunduzi River water is not suitable for drinking and recreation according to QMRA results.
- These results can inform policies and decision-making within the catchment.

GRAPHICAL ABSTRACT



INTRODUCTION

Surface water pollution is strongly associated with land and water uses within a catchment. These include impact from residences with insufficient sanitation infrastructure in the vicinity of streams, poorly operated and maintained sanitation infrastructure, dense settlements in areas prone to surface runoff, and intensive livestock farming (Luyt *et al.* 2012). Low-income countries continue to have a high incidence of diarrhoea among young children under 5 years of age, which is one of the leading causes of death, ill-health, and disability (Naghavi *et al.* 2015). In such communities with poor service and infrastructure, the spread of diarrhoeal diseases is exacerbated by inadequate and unsafe water supplies and inadequate sanitation and hygiene (Araya *et al.* 2016). Shirley *et al.* (2012) found that most attributable cases of moderate-to-severe diarrhoea in the youngest children were due to four pathogens: rotavirus, *Cryptosporidium*, enterotoxigenic *E. coli* producing heat-stable toxin (ST-ETEC; with or without co-expression of heat-labile enterotoxin), and *Shigella*.

Cryptosporidium has been identified in South Africa as a causative agent of diarrhoeal diseases in immunocompromised, HIV-positive children and adults (Ojuromi & Ashafa 2018). In the study performed in KwaZulu-Natal by Jarmey-Swan *et al.* (2001), *Cryptosporidium* was found to be most prevalent (39.3%) (1,100/2,800) in the <1-year age group. The quality of South Africa's water resources is linked to the inadequate wastewater infrastructure and the subsequent contribution of untreated or poorly treated wastewater discharges to the already deteriorating water sources (Mema 2010; Osuolale & Okoh 2017; Ojuromi & Ashafa 2018; Sekwadi *et al.* 2019). The lack of infrastructure in South Africa forces rural and peri-urban households to supplement their drinking water intake with water of poor microbial quality (Luyt *et al.* 2012).

In response to water-related challenges, hydrological modelling has been developed to analyse, understand, and explore solutions for sustainable water management, to the decision makers and operational water managers (Baffaut & Benson 2009; Mander *et al.* 2017; Abdullah *et al.* 2019). Using models to understand and manage water quality can make it easier to promote better management. By generating more data over time and space, they can supplement limited observational data and allow for scenarios to explore the impact of different management measures on water quality in a watershed. Catchment-scale water quality models have become a key tool for understanding, evaluating, and predicting the adverse effects of contamination on river water quality. Among others, models that provide refined simulation for water quality include Agricultural Nonpoint Source pollution model (AGNPS/AnnAGNPS), Soil and Water Assessment Tool (SWAT), Stormwater Management Model (SWMM), and Hydrologic Simulation Program Fortran (HSPF). Of these, HSPF and SWAT are highly recommended to users (Yuan *et al.* 2020).

In SWAT, there is a microbial sub-model that deals with the fate and transport of both persistent and less persistent microorganisms (Abdullah *et al.* 2019). SWAT can, therefore, simulate microbial transport in catchments as well as identify high-risk areas and estimate peak loads of microorganisms such as *E. coli* and *Cryptosporidium* (Bergion *et al.* 2017; Zhang *et al.* 2019). Output data from a SWAT model was successfully used as input data to conduct quantitative microbial risk assessment (QMRA) by Bergion *et al.* (2020). QMRA assesses the probability of infection due to exposure to microorganisms, based on the dose a person ingests and dose-response models that provide the human response to specific pathogens (Haas *et al.* 2014). The risk of being infected by microbial pathogens is dependent on the level of pollution of the water and the amount of polluted water consumed (Müller *et al.* 2001).

The aim of this study was to utilise SWAT as a water quality modelling tool to map out point and non-point source pollution in the uMsunduzi catchment in KwaZulu-Natal, South Africa, to identify high-risk areas. Concentrations of *Cryptosporidium* and *E. coli* were studied. The concentration outputs from SWAT were used to perform a QMRA to better understand the health implications on communities that use polluted waters for drinking and non-competitive recreation purposes.

MATERIALS AND METHODS

The 875 km^2 uMsunduzi catchment is in the KwaZulu-Natal province, South Africa. The uMsunduzi River, a 115 km watercourse in length, passes through Pietermaritzburg (29°37′S 30°23′E) and is a major tributary of the uMngeni River. It flows through rural and urban dwellings as seen in Figure 1.

Due to the variation in settlement types, the catchment is divided into three units: the upper catchment rural area, urban area in the middle, and lower catchment rural area. The rural areas are characterised by subsistence farming (stock and crops) and rural developments. The urban area is dominated by the urban stretch of Pietermaritzburg that is a mix of formal city, residential and industrial suburbs, and informal settlements. The uMsunduzi River enters a canalised reach at Camps Drift, then passes through the city in an incised narrow channel and is joined by small tributaries before picking up treated effluent from Darvill Wastewater Treatment Plant (WWTP). The lower catchment rural area is dominated by the rural Valley of a Thousand Hills, where the uMsunduzi River confluences with the uMngeni River.

The city of Pietermaritzburg has a dry-winter humid subtropical climate (Köppen climate classification: Cwa), with distinct dry and wet seasons. The average temperatures are ~21 °C in summer (December–February) and ~12 °C in winter (June–August). The monthly average precipitation during dry months (June–August) is 23 and 140 mm during the rainy season (December–February). The dominant soil types in the study area, according to the Food and Agricultural Organisation (FAO 2015) are Hutton, Griffin, Mispah, and Glenrosa. The main land use types are grasslands. It is important to note that these are slowly changing into informal dwellings due to urbanisation.

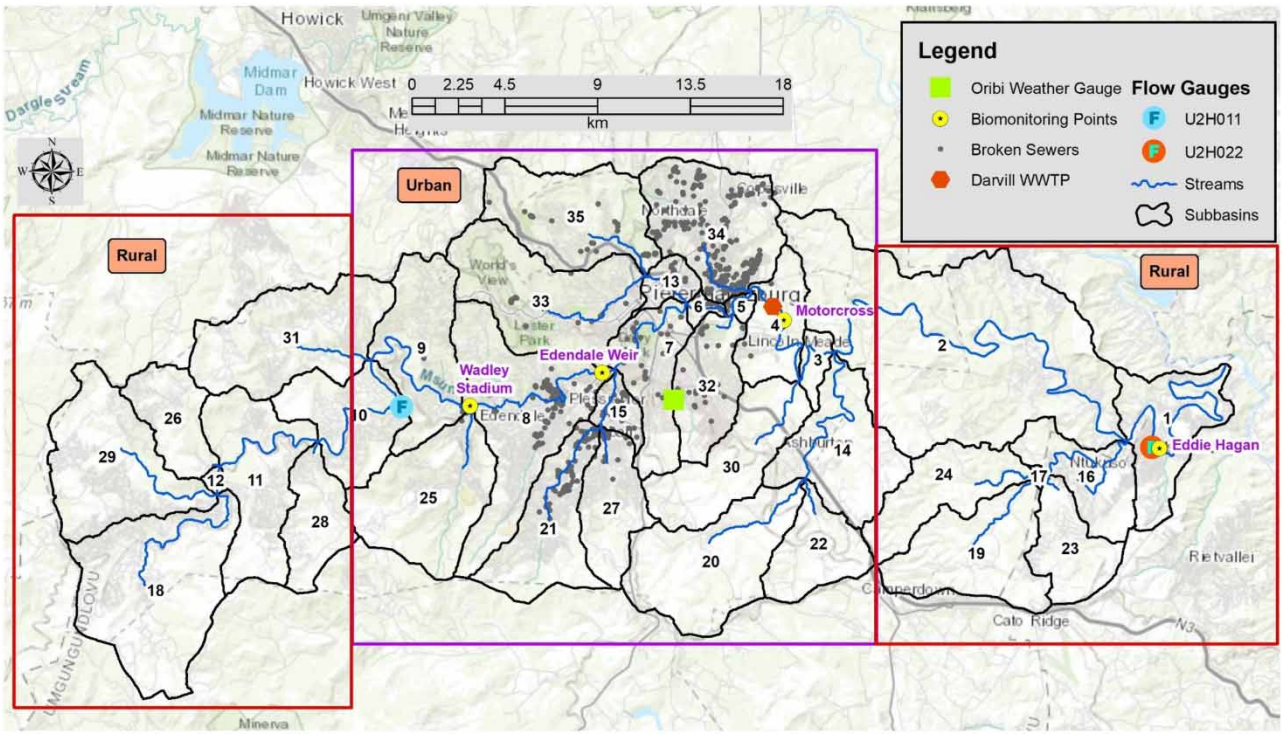


Figure 1 | River flow monitoring gauge positions, water quality monitoring stations, broken sewers, weather station within the uMsunduzi catchment, as well as subbasins as delineated by SWAT. Please follow the link to view in full layout: https://www.google.com/maps/d/viewer?mid=1NQXdFcqDjShMQxRgn4p8ILzpj42NpgdG&ll=-29.63672034343441%2C30.4750729922986928z=11.

Water quality model

QSWAT3 v1.1 for QGIS3 was used to setup the hydrological model for the period 1999–2018, for this study. The years 1999–2003 were reserved for the warm-up period prior to the defined calibration and validation stages to allow the stable performance of the model. Calibration and validation of the model using river flow data were performed for the periods 2004–2011 and 2012–2018, respectively. The input data included topographical, geographical, and geological information as well as meteorological data as indicated in Table 1. The uMsunduzi catchment was divided into 35 subbasins. The water quality output of this model was used in the QMRA as discussed in the subsequent sections.

The hydrological parameter optimisation was performed manually. The Nash–Sutcliffe efficiency coefficient (NSE), the coefficient of determination (R^2), percent bias (PBIAS), and the ratio of the root mean squared error to the standard deviation of measured data (RSR) were used for the evaluation of model performance. These are represented by Equations (1)–(3), respectively.

$$\mathrm{NSE} = 1 - \frac{\sum_{i}^{n} (Q_i^m - Q_i^s)^2}{\sum_{i}^{n} (Q_i^m - \bar{Q})^2} \tag{1}$$

$$\mathrm{PBIAS} = \left[\frac{\sum_{i}^{n} (Q_i^m - Q_i^s) \times 100}{\sum_{i}^{n} Q_i^m} \right] \tag{2}$$

$$\mathrm{RSR} = \frac{\sqrt{\sum_{i}^{n} (Q_i^m - Q_i^s)^2}}{\sqrt{\sum_{i}^{n} (Q_i^m - \bar{Q})^2}} \tag{3}$$

where Q is discharge, the $\bar{Q}$ is the mean discharge, and the indices m and s stand for measured and simulated, respectively and i/n are the ith and nth measured or simulated data.

Sadeghi & Arnold (2002) developed a SWAT bacterial module for predicting microorganism loads and concentrations in rivers. The SWAT microbial sub-model considers the fate and net transport of microorganisms derived from defined pollution sources. The two microorganisms considered in this study were *E. coli* and *Cryptosporidium*, chosen to represent bacteria and protozoan parasites, respectively. Bacteria may be less persistent in the environment due to inactivation by environmental factors compared to the protozoan parasites, whose (oo)cysts are more resistant to environmental conditions (Sadeghi & Arnold 2002). The input parameter values for both microorganisms are shown in Table 2.

The uMsunduzi River is impacted by point and non-point source pollution emanating from rural settlements, informal settlements, townships, formal urban and industrial areas, and the Darvill WWTP (Dabrowski *et al.* 2013). The main

Table 1 | Data for the hydrological and water quality model of the uMsunduzi River

Data	File type	Resolution	Source
Digital Elevation Model	Raster	20×20 m	National Geo-spatial Information (NGI)
Landuse/landcover 2018	Raster	1,700×1,700 m	South African National Biodiversity Institute (SANBI)
Soil type	Raster	250×250 m	Food and Agriculture Organization (FAO)
Meteorological data	Text	One station, Daily[a]	South African Weather Services (SAWS)
River flow data	Text	Two stations, Daily[a]	Department of Water Affairs (DWA)
Water quality monitoring data	Text	Four Points, Weekly[a]	Umgeni Water

[a]Refer to Figure 1 for location within the catchment.

Table 2 | SWAT parameter values for *E. coli* and *Cryptosporidium*

Microorganism	SWAT abbreviation	Unit	Value	Reference
Both	BACTFDDB (Partition coefficient)	Fraction 0≤1	0.9	Coffey *et al.* (2013)
Both	BACTKDQ (Bacteria soil partitioning coefficient)	Constant	175	Arnold *et al.* (2012)
Both	FRT_SURFACE (Fraction of fertiliser applied to 10 mm of soils)	Fraction 0≤1	0.5	Tang *et al.* (2011)
E. coli	WDLPQ (Die-off factor for less persistent bacteria in soil solution at 20 °C)	1/day	0.092	Westrell (2004)
E. coli	WDLPRCH (Die-off factor for less persistent bacteria in streams at 20 °C)	1/day	0.18	Westrell (2004)
E. coli	WDLPS (Die-off factor for less persistent bacteria adsorbed to soil particles at 20 °C)	1/day	0.023	Bougeard *et al.* (2011)
E. coli	WDLPF (Die-off factor for less persistent bacteria on foliage at 20 °C)	1/day	0.016	Bougeard *et al.* (2011)
E. coli	WOF_LP (Fraction of less persistent organisms washed off in rainfall events)	Fraction 0≤1	0.5	Bougeard *et al.* (2011)
Cryptosporidium	WDPQ (Die-off factor for persistent bacteria in soil solution at 20 °C)	1/day	0.005	Westrell (2004)
Cryptosporidium	WDPRCH (Die-off factor for persistent bacteria in streams at 20 °C)	1/day	0.032	Westrell (2004)
Cryptosporidium	WDPS (Die-off factor for persistent bacteria adsorbed to soil particles at 20 °C)	1/day	0.003	Coffey *et al.* (2013)
Cryptosporidium	WDPF (Die-off factor for persistent bacteria on foliage at 20 °C)	1/day	0.03	Tang *et al.* (2011)
Cryptosporidium	WOF_P (Fraction of persistent organisms washed off in rainfall events)	Fraction 0≤1	0.8	Tang *et al.* (2011)

faecal sources in the uMsunduzi catchment can be summarised as: Darvill WWTP, broken sewers in the urban area, dry toilets, and faecal droppings from grazing livestock.

To compare the model output with the measured concentrations, four subbasin output points that closely coincide with water quality monitoring points by the Msunduzi Local Municipality and Umgeni Water along the river were considered in the model. These were: uMsunduzi downstream Henley Dam – rural area (subbasin 10); urban areas (subbasin 8); major stream, city, informal settlements, industrial areas, WWTP, and waste dump site (subbasin 4); and rural areas and confluence with the uMngeni River (subbasin 1) as shown in Figure 1.

The Darvill WWTP serves 51.6% of the 163,993 uMsunduzi Local Municipality households (STATSSA 2011). The Darvill WWTP is currently being upgraded from 65,000 to 100,000 m^3/day to meet current and future demands (Ramnath *et al.* 2019), which means that it often operated above its capacity and the system was overflowing, releasing untreated wastewater directly to the uMsunduzi River. The wastewater from the Darvill WWTP was put into the SWAT model as a point source with an average discharge of 75,000 m^3/day. According to Ramnath *et al.* (2019), the inflows to the WWTP were, for a number of months in 2018, below the 12-month moving average (~75,000 m^3/day), with the average inflow in October 2018 only ~59,000 m^3/day. This is an unseasonal low, and Umgeni Water is concerned that not all the wastewater is reaching the Darvill WWTP. This implies that ~16,000 m^3/day could be attributed to losses due to broken or leaking sewers. This volume of wastewater was distributed over the subbasins within the urban section of the catchment in the SWAT model, where, as shown in Figure 1, there are broken sewers. This data was supplied by Mr Vilakazi of the Duzi-Umgeni Conservation Trust (DUCT). These leaking sewers were included as a point source contribution in each of these subbasins. For this study, the concentration of *Cryptosporidium* used as input in SWAT was 5 (0–17) oocysts/10 L based on 13 samples collected from the Darvill WWTP effluent during 2016–2017 (de Jong 2020, pers. comm.). *E. coli* concentration in wastewater was estimated to be 50,000 counts/100 ml, a median of 3,615 samples from monitoring data supplied by Umgeni Water taken between January 2004 and December 2018.

The remaining 48.4% of the households within the municipality use dry toilets or are without any toilet at home, a small percentage of which may have septic tanks (STATSSA 2011). Data showing the distribution of these dry toilets were not available; however, these toilets are used mainly in informal and rural settlements. Defaecating in open spaces could also be a

human source of microorganisms in this catchment. It was, however, not considered in this study due to difficulties in quantifying its contribution in the runoff.

The SWAT grazing function was used to represent livestock manure production during grazing. Only cattle and sheep/goats were considered for this study as the numbers of other animals were low, and relevant prevalence and infectivity data in Africa are scarce. Due to the warm climate and general subsistence farming practices in Pietermaritzburg, these animals are grazing all year round. No data could be found that account for the distribution of the livestock herds over the catchment, thus, the estimated produced faecal matter was distributed evenly over the modelled grassland landuse area of 15,943 ha. To account for different faecal production depending on age, the fraction of animals that are young was determined using a guide by Vermeulen *et al.* (2017). The average *Cryptosporidium* concentrations in livestock faeces were calculated from the prevalence of infection and the concentration in faeces of infected animals. The assumed input data for *E. coli* and *Cryptosporidium* are presented in Table 3. The *E. coli* concentrations shown in Table 3 do not differentiate between pathogenic and non-pathogenic *E. coli*.

Quantitative microbial risk assessment (QMRA)

The QMRA tool version 2018-06-07 was used for microbial risk assessment to consumers accessed via https://www.analyticacloud.com/view?invite=24&code=56994&subName=Chalmers. This tool was developed by Abrahamsson Ansker & Heinicke (2009) and is managed by the Centre for Drinking Water Research 'DRICKS' on behalf of the Swedish drinking water industry. For the purpose of this study, water treatment steps were omitted in the QMRA tool, to represent untreated water ingested during recreational activities and direct domestic use.

Source water characterisation and exposure

The outputs from the SWAT model in terms of concentration of the selected microorganisms along the river were used as input values for the QMRA. Considering that not all *E. coli* are pathogenic, 8% of *E. coli* was assumed as pathogenic following various studies such as Haas *et al.* (2000) and Mbanga *et al.* (2020). The exposure routes investigated were direct ingestion of the uMsunduzi River water during recreational swimming, canoeing training, and drinking. These exposure scenarios are detailed below.

Recreational swimming. This exposure scenario was considered for subbasins 1, 4, 8, and 10. This is because it was observed during the study that the population along this stretch of the catchment habitually swim in the river during warm periods. The exposed population was categorised into children and adults. The volume ingested during swimming was estimated based on the ranges of 37–47 mL for children and 16–24 mL for adults per 45 minute event as reported in Dufour *et al.* (2006) as shown in Table 4.

Canoeing. This exposure scenario was considered for subbasin 8 as training takes place in this stretch of the river. Training information was obtained from Mr Z. L. Mthalane (pers. comm., 2019), a coach at two canoe clubs and a seasoned Dusi Canoe Marathon and Dusi Non-stop participant. The exposed population was categorised into three: juvenal (10–14 years old), junior (15–18 years old), and senior (>18 years old) based on the competition categories and the required hours for training. These weekly training hours are 3–5 (juvenal), 6–10 (junior), and 11–14 (senior). Based on these schedules, 0.5–1 h (juvenal), 1–1.5 h (junior), and 1.5–2 h (senior) per day were assumed. These hours, together with ingestion volume of 5.8 mL per 45 minute event, were used to estimate the ingested volumes during canoeing events as reported in Dorevitch *et al.* (2011). The subsequently calculated daily minima and maxima are presented in Table 4.

Drinking Water. This exposure scenario was considered for subbasins 10 and 1, which are in the upper and lower rural parts of the catchment, respectively. The exposed population was categorised into three: juvenal, junior, and senior. Table 4 shows daily ingestion volumes for South Africans adapted from Steyn *et al.* (2001).

Dose-response modelling

The dose-response functions provide the important link between exposure dose and the likelihood of occurrence of a negative consequence like infection, illness, severe illness, or death (Smeets *et al.* 2010; Haas *et al.* 2014). The ingested dose (μ) was

Table 3 | *Cryptosporidium* and *E. coli* concentration estimates for SWAT input

Host	Livestock in catchment (#)[a]	Age differentiation (#)[b]	Animal density in grazing area (#/ha)	Manure production (kg/d/1,000 kg)[b]	Animal mass (kg)[b]	Manure production (kg/ha/day)[b]	*Cryptosporidium* prevalence[b]	*Cryptosporidium* concentration infected livestock (oocysts/g)[b]	*Cryptosporidium* concentration total livestock (oocysts/g)[c]	*E. coli* concentration (#/g)[d]
Cattle	35,345	32,517	2.04	86	250	32.07	0.17	$1.0\times10^{+02}$	$1.7\times10^{+01}$	$2.0\times10^{+05}$
Calves		2,828	0.18		125*	1.39	0.29	$7.9\times10^{+04}$	$2.3\times10^{+04}$	$4.2\times10^{+05}$
Goats	29,560	25,126	1.58	41	30	1.42	0.16	$2.0\times10^{+02}$	$3.2\times10^{+01}$	$6.6\times10^{+04}$
Kids		4,434	0.28		15*	0.13	0.11	$2.5\times10^{+04}$	$2.7\times10^{+03}$	$6.6\times10^{+04}$
Sheep	9,920	8,432	0.53	40	28	0.43	0.25	$1.6\times10^{+02}$	$4.0\times10^{+01}$	$6.6\times10^{+04}$
Lambs		1,488	0.09		14*	0.04	0.13	$2.0\times10^{+05}$	$2.6\times10^{+04}$	$6.6\times10^{+04}$

[a]Data source from STATSSA (2011).
[b]Calculations and values after Vermeulen *et al.* (2017).
[c]Calculated using concentration in infected livestock and prevalence.
[d]After Coffey *et al.* (2013).
*0.5 of the adult animal was used to calculate body mass of the respective young as no data could be found.

Table 4 | Water volumes ingested during swimming, canoeing, and drinking

Category	Min	Max
Ingested during swimming (L/d)[a]		
Children	0.0493	0.0627
Adults	0.0213	0.0320
Ingested during canoeing (L/d)[a]		
Juvenal	0.0039	0.0077
Junior	0.0077	0.0116
Senior	0.0116	0.0154
Ingested during drinking (L/d)[a]		
Juvenal	0.0013	0.63
Junior	0.63	0.773
Senior	0.773	0.952

[a]In the QMRA dose-response modelling, the volume was assigned a uniform distribution with minima (Min) and maxima (Max).

estimated using Equation (4).

$$\mu = v \times c \tag{4}$$

where μ is the ingested dose of pathogens, v is the ingested volume, and c is the concentration of the targeted pathogens. Ingested volume was based on the values in Table 4, using a uniform distribution with minima and maxima, according to the values reported in the table. Concentration was defined as the fitted lognormal distribution based on the simulated daily pathogen concentrations output from the SWAT modelling for each subbasin. For each subbasin, fitting was conducted using the @RISK 8 software, and the lognormal distribution was one of the top three ranking distributions according to the Akaike Information Criterion (AIC).

In the QMRA tool, the Exponential dose-response model was used to calculate the probability of infection by pathogenic *E. coli* (O157:H7) and *Cryptosporidium* (Equation (5)).

$$P_{(\text{inf})} = 1 - e^{-r\mu} \tag{5}$$

where $P_{(\text{inf})}$ is the probability of infection, μ is the pathogen dose, and r is a Beta function. In the QMRA tool, applying Monte Carlo simulations and defining a Beta function for r in the Exponential function approximates an exact Beta-Poisson model. The Beta function (α, β) was used for *Cryptosporidium* (0.115, 0.176) (Teunis *et al.* 2002) and pathogenic *E. coli* (0.37, 37.65) (Teunis *et al.* 2004).

RESULTS AND DISCUSSION

Water quality model

The SWAT model's predictions for streamflow were in good agreement with monitored data as shown in Table 5. The good agreement is corroborated by the visual comparison between simulated and observed flow for calibration and validation periods, using monthly data (Figure 2). All evaluation metrics shown in Table 5, i.e., NSE>0.5, RSR≤0.7, and PBIAS≤±0.25, except for the PBIAS for validation for subbasin 10, are referred to as satisfactory, with the PBIAS only slightly out of range (Moriasi *et al.* 2007). Parameters optimised during the calibration process are shown in Table 6.

Figure 2 and 3 show the daily simulated concentrations of *Cryptosporidium* and *E. coli* in subbasins 10, 8, 4, and 1. In Figure 4, these are plotted against weekly monitored *E. coli* concentrations between January 2004 and December 2018. The output from the SWAT model shows slightly low variability, a standard artefact observed in microbiological modelling (Iqbal & Hofstra 2019). In general, the trend in the SWAT model simulations follows the observed data patterns in most subbasins. The model may not have captured some of the extremities observed in the monitoring data; however, the simulated

Table 5 | SWAT calibration and validation results

	Calibration		Validation	
Index	**Subbasin 1**	**Subbasin 10**	**Subbasin 1**	**Subbasin 10**
R^2	0.58	0.69	0.64	0.73
NSE	0.57	0.61	0.58	0.59
RSR	0.31	0.29	0.25	0.28
PBIAS	−0.05	0.24	0.14	0.27

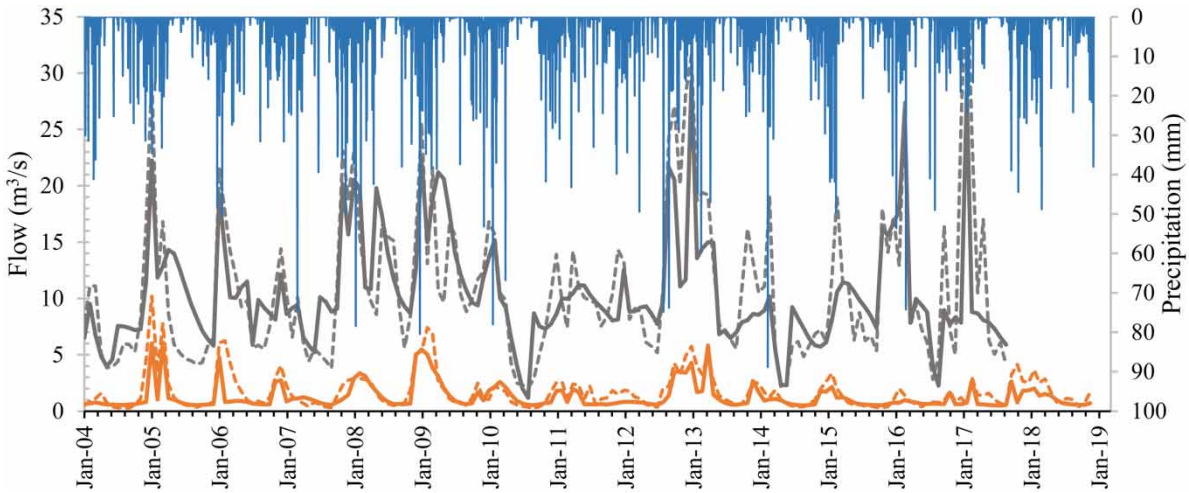


Figure 2 | Graphical comparison of simulated streamflow against observed flow for subbasins 1 and 10 calibration (January 2004–December 2011) and validation (January 2012–December 2018) periods. Subbasin 1 is depicted in grey and subbasin 10 in orange. Continuous and dashed lines represent simulated and observed data, respectively. The blue line in the secondary vertical axis shows the daily precipitation. Please refer to the online version of this paper to see this figure in colour: https://doi.org/10.2166/wh.2022.266.

Table 6 | Parameters optimised during SWAT calibration

Variable		Description	Default Value(s)	Calibrated Value	Input file
Flow	CN2	Initial SCS runoff curve number for moisture condition	45–85	49	.mgt
	SOL_K	Saturated hydraulic conductivity	2.6–52 mm/h	7.54 mm/h	.sol
	SOL_BD	Soil moist bulk density	0.9–2.5 g/cm^3	1.36 g/cm^3	.sol
	ESCO	Soil evaporation compensation factor	0.95	0.9018	.sol
	SOL_AWC	Available water capacity of the soil	0.12–52 mm H_2O/mm Soil	0.127 mm H_2O/mm Soil	.sol
	MSK.CO_2	Coefficient for impact of low flow	3.5	3.65	.bsn
	CANMX	Maximum water storage in canopy	0 mm	0.3 mm	.hru
Base flow	ALPHA-BF	Groundwater recession factor	0.084 d	0.27 d	.gw
	GW-DELAY	Groundwater delay	31 d	28 d	.gw

peaks are one order of magnitude less than those of the monitoring data. Comparing the monitored and simulated mean of the *E. coli* concentration shows that for subbasins 4 and 1 the SWAT model overestimates, while for subbasin 10 and 8, it slightly underestimates the concentrations.

Figure 3 shows that subbasin 4 has the highest *Cryptosporidium* concentration, followed by subbasin 1, then subbasins 8 and 10, respectively. The trends are generally such that concentrations are higher downstream the WWTP than upstream. *Cryptosporidium* concentrations analysed between September 2015 and March 2016 by Adeyemo (2019) in effluents from the wastewater treatment plants in Durban show a range of 50–70 oocysts/10 L (six samples). In the study by Dungeni & Momba (2010) in the Gauteng province (South Africa), the concentrations in effluents were <0.1–4 oocysts/10 L, in 14 samples taken between January and April 2008, weekly. In a study performed between March 2016 and March 2017 with fortnightly sampling, Razzolini *et al.* (2020) detected *Cryptosporidium* concentrations of <0.003–2.6 oocysts/10 L in wastewater effluent in Sao Paulo, Brazil. In Shanghai China, Ma *et al.* (2016) detected 0–0.1 oocyst/10 L *Cryptosporidium* concentrations in wastewater effluent from three WWTPs. In this study, values provided by de Jong (2020, pers. comm.), 5 oocysts/10 L, were used as these were from the Darvill WWTP specifically. Due to a short time-series, there is a possibility of underestimation. More continuous monitoring can help reduce this uncertainty. Robertson *et al.* (2020) suggest that the likelihood of *Cryptosporidium* contamination of drinking water in Africa is probably greater than in many other countries, due to water supplies being limited and a general lack of catchment control for securing the supply of water, as surface waters are both used as drinking water and for livestock drinking.

In reference to *Cryptosporidium*, DWAF (1996a) for domestic use, states that 1 oocyst/10 L (0.1 oocyst/L) is enough to pose the risk of infection. No guidelines are given for protozoa in recreational use.

In the DWAF (1996b) guidelines for recreational use, the effects of *E. coli* on human health based on their concentration are outlined as follows: The concentrations of 0–130 counts per 100 mL may pose a low risk of gastrointestinal illness for full-contact recreational water use; negligible effects are expected if these levels occur in isolated instances only. The concentrations of 200–400 counts per 100 mL may pose some risk of gastrointestinal effects, particularly if this occurs frequently. Above 400 counts per 100 mL will pose increasing risk of gastrointestinal effects. The criteria proposed for full-contact recreation (swimming) are also recommended for intermediate contact (canoeing) (DWAF 1996b).

In the DWAF (1996a) guidelines for domestic use, the effects of faecal coliforms on human health based on their concentration are as follows: 0 will pose negligible risk of microbial infection; 0–10 counts per 100 mL will pose a slight risk of microbial infection with continuous exposure and negligible effects with occasional or short-term exposure; 10–20 counts per 100 mL will pose risk of infectious disease transmission with continuous exposure and slight risk with occasional exposure; and >20 counts per 100 mL will pose significant and increasing risk of infectious disease transmission.

In subbasins 10 and 8, animal sources contribute more than human sources to the combined *E. coli* concentrations, on the other hand, subbasins 4 and 1 show high human source impact. The model has successfully captured the trends of different

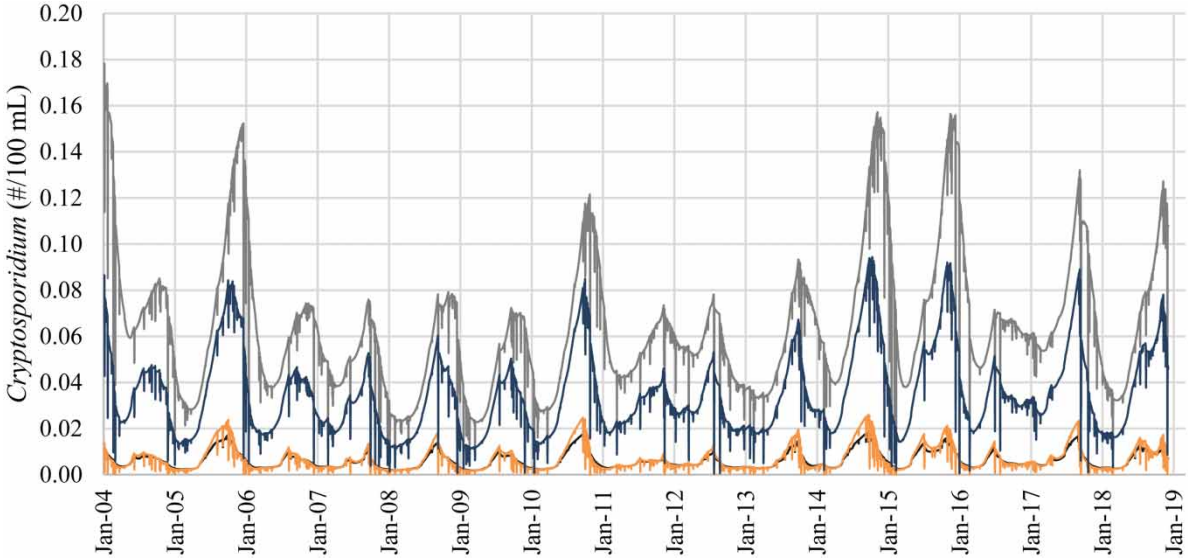


Figure 3 | Simulated *Cryptosporidium* concentrations in basins 1, 4, 8, and 10. The black line represents subbasin 10, the grey line represents subbasin 4, the orange line represents subbasin 8, and the blue line represents subbasin 1. Please refer to the online version of this paper to see this figure in colour: https://doi.org/10.2166/wh.2022.266.

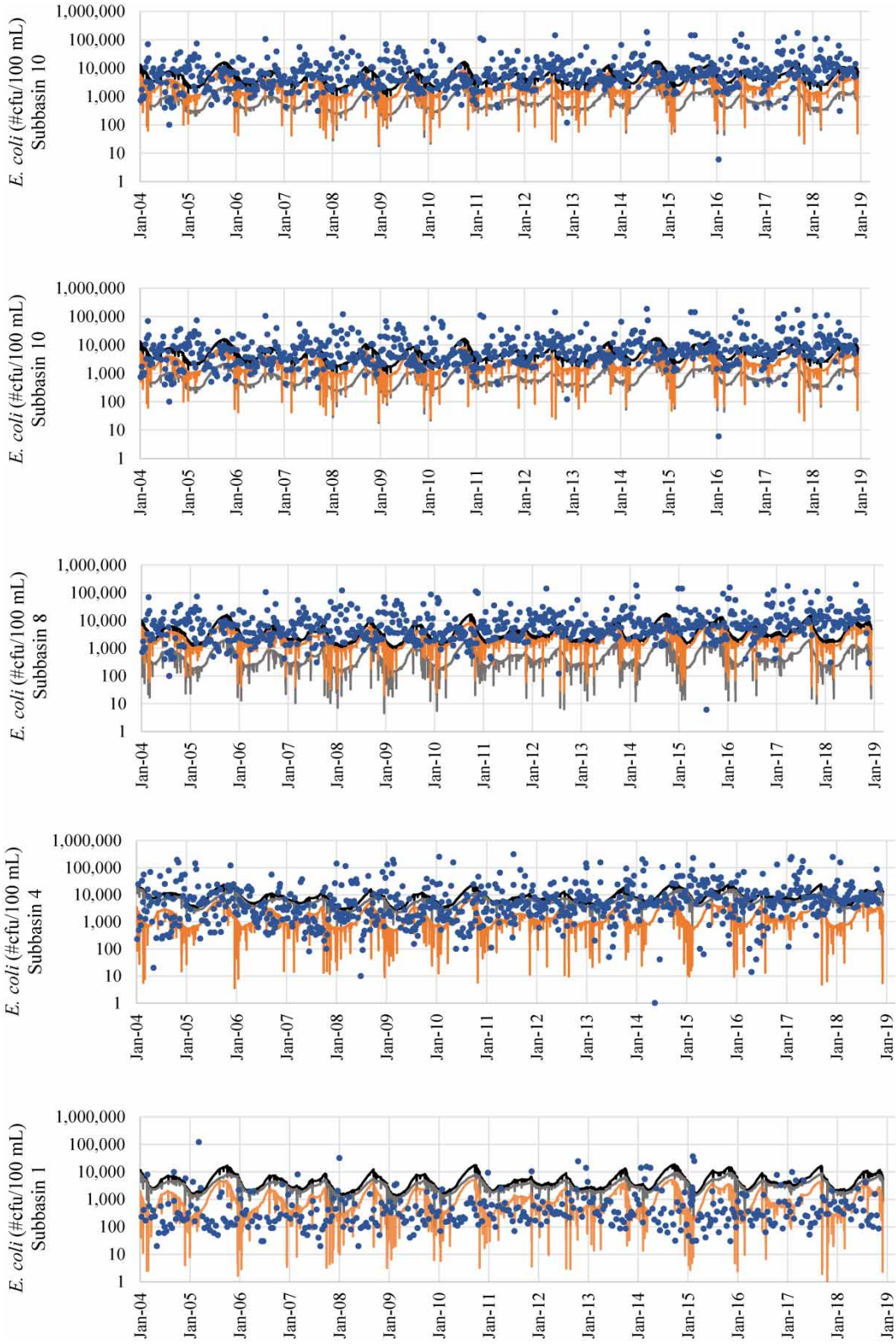


Figure 4 | Simulated and monitored (#cfu/100 ml) *E. coli* concentrations in basins 10, 8, 4, and 1 (respectively from top to bottom). The black line represents combination of sources, the grey line represents human sources, the orange line represents animal sources, and the blue circles represent monitored *E. coli*. Please refer to the online version of this paper to see this figure in colour: https://doi.org/10.2166/wh.2022.266.

sources in different settlements within the catchment. For subbasin 10 (upper rural), human contributions are lower than animal contributions, highlighting the intensive and uncontrolled livestock farming. As subbasin 8 is a transition between rural and urban areas, human contributions are still lower than animal contributions in these parts. Broken sewers do contribute in these parts, albeit low. In subbasin 4, there is a significant contribution from human sources (Figure 3) from the WWTP and broken sewers. Gemmell & Schmidt (2013) pointed out that the water quality of the uMsunduzi River is poor, mainly due to faecal pollution, and deteriorates as the river passes through the city.

The data in Figures 2 and 3 show that *E. coli* concentrations in the uMsunduzi River render the water unsuitable for drinking, swimming, and canoeing due to the high possibility of contracting gastrointestinal diseases, according to DWAF (1996a, 1996b).

There are several limitations to this study in terms of the microbial estimates in the SWAT model. *Cryptosporidium* loads in the effluent from the WWTP were represented as a point value rather than using a time series. The quantification of the contribution from broken sewers was an estimate since the reporting of this did not include any information on discharge. There are also limitations in available data on pathogen parameters, for example, the proportion of pathogenic *E. coli*. In the townships and rural areas, pit latrines are used as sanitation infrastructure, but in the study area, their contribution to the surface waters could not be quantified. Also, microbial contributions from direct stream deposits from livestock and wildlife were not estimated. Defaecating in open spaces could also be a human source of microorganisms in this catchment. It was, however, not considered in this study due to difficulties in quantifying its contribution in the runoff.

Quantitative microbial risk assessment results

Table 7 shows the probability of infection (P_{inf}), 50th and 95th percentile, from different exposure scenarios to *Cryptosporidium* and pathogenic *E. coli*, respectively. Table 7 also shows P_{inf} calculated using monitored *E. coli*. The P_{inf} calculated using simulated data and using the monitoring data generally fall into the same order of magnitude. To put the results of this study into the context of the existing guidelines, the concentrations stated by DWAF were recalculated into the probability of infection. In the DWAF (1996a) guidelines for domestic use, the effect of faecal coliforms on human health based on their concentration is 0 counts/100 mL for negligible risk of microbial infection. In terms of QMRA inputs, uncertainty lies with the ingested volumes for exposure routes.

A value of 0.8 counts per L (based on 1 count/100 mL), accounting for pathogenic portion, was used together with different ingestion scenarios in Table 4, to calculate the associated probability of infection. For domestic use, the recalculated 50th percentile risk values were 6/10,000 (juvenile), 2/1,000 (junior), and 3/1,000 (senior). For swimmers and canoeists, according to DWAF (1996b), low risk of gastrointestinal illness is considered as below 8/1,000.

Table 7 | Probability of infection, P_{inf}, by *Cryptosporidium* and pathogenic *E. coli* across subbasins

Exposure Scenario (per day)	Age group	Location	P_{inf} 50th (95th) Percentile *Cryptosporidium*	P_{inf} 50th (95th) Percentile Modelled *E. coli*	P_{inf} 50th (95th) Percentile Monitored *E. coli*
Swimming	Adults	Sub 10	1.78×10^{-04} (2.59×10^{-03})	3.23×10^{-01} (9.99×10^{-01})	6.34×10^{-02} (9.83×10^{-01})
		Sub 8	2.4×10^{-03} (1.91×10^{-02})	2.51×10^{-01} (9.94×10^{-01})	3.02×10^{-01} ($1.00\times10^{+00}$)
		Sub 4	1.96×10^{-03} (2.37×10^{-02})	4.58×10^{-01} ($1.00\times10^{+00}$)	2.15×10^{-01} ($1.00\times10^{+00}$)
		Sub 1	1.01×10^{-03} (1.41×10^{-02})	2.99×10^{-01} (9.98×10^{-01})	2.12×10^{-02} (5.10×10^{-01})
	Children	Sub 10	3.17×10^{-04} (5.35×10^{-03})	5.79×10^{-01} ($1.00\times10^{+00}$)	1.18×10^{-01} ($1.00\times10^{+00}$)
		Sub 8	5.08×10^{-03} (3.87×10^{-02})	4.33×10^{-01} ($1.00\times10^{+00}$)	5.38×10^{-01} ($1.00\times10^{+00}$)
		Sub 4	3.90×10^{-03} (4.99×10^{-02})	7.55×10^{-01} ($1.00\times10^{+00}$)	4.04×10^{-01} ($1.00\times10^{+00}$)
		Sub 1	2.44×10^{-03} (2.90×10^{-02})	5.23×10^{-01} ($1.00\times10^{+00}$)	4.41×10^{-02} (7.99×10^{-01})
Canoeing	Juvenal	Sub 8	3.14×10^{-05} (6.52×10^{-04})	5.99×10^{-02} (6.7×10^{-01})	7.18×10^{-02} (8.99×10^{-01})
	Junior	Sub 8	6.34×10^{-05} (1.13×10^{-03})	1.00×10^{-01} (8.54×10^{-01})	1.24×10^{-01} (9.74×10^{-01})
	Senior	Sub 8	9.25×10^{-05} (1.58×10^{-03})	1.38×10^{-01} (9.23×10^{-01})	1.69×10^{-01} (9.94×10^{-01})
Drinking	Juvenal	Sub 10	1.34×10^{-03} (3.49×10^{-02})	9.65×10^{-01} ($1.00\times10^{+00}$)	7.78×10^{-01} ($1.00\times10^{+00}$)
		Sub 1	8.15×10^{-03} (1.78×10^{-01})	9.54×10^{-01} ($1.00\times10^{+00}$)	1.71×10^{-01} ($1.00\times10^{+00}$)
	Junior	Sub 10	4.00×10^{-03} (6.47×10^{-02})	$1.00\times10^{+00}$ ($1.00\times10^{+00}$)	9.90×10^{-01} ($1.00\times10^{+00}$)
		Sub 1	2.81×10^{-02} (3.10×10^{-01})	$1.00\times10^{+00}$ ($1.00\times10^{+00}$)	4.22×10^{-01} ($1.00\times10^{+00}$)
	Senior	Sub 10	5.51×10^{-03} (7.87×10^{-02})	$1.00\times10^{+00}$ ($1.00\times10^{+00}$)	$1.00\times10^{+00}$ ($1.00\times10^{+00}$)
		Sub 1	3.28×10^{-02} (3.63×10^{-01})	$1.00\times10^{+00}$ ($1.00\times10^{+00}$)	5.06×10^{-01} ($1.00\times10^{+00}$)

For *Cryptosporidium*, for domestic use, the guidelines by DWAF (1996a) state that 1 oocyst/10 L is enough to pose the risk of infection; and no guidelines are given for protozoa in recreational use. Therefore, a concentration of 1 oocyst/10 L as a point value was used together with different ingestion scenarios in Table 4 to calculate the probability of infection. The 50th percentile by DWAF (1996b) for swimming is 9/10,000 children and adults 4/10,000 adults with respect to *Cryptosporidium*. For canoeing: 8/100,000 Juvenile, 2/10,000 Junior, and 2/10,000 Seniors were calculated based on the DWAF (1996b).

The findings of this study are summarised below in terms of the 50th percentile of probability of infection calculated using the SWAT output data.

Recreational swimming

In reference to *Cryptosporidium*, subbasins 1, 4, and 8 show generally higher probability of infection than the calculated 50th percentile by DWAF (1996b). In reference to pathogenic *E. coli*, the probability of infection is above the DWAF 'low risk category', with the highest risks in subbasin 4.

Canoeing

In subbasin 8, the probability of infection with *Cryptosporidium* is lower than the recalculated values from DWAF (1996b) and higher for Junior and Senior canoeists. The probability of infection with pathogenic *E. coli* during canoeing in subbasin 8 was found to be higher than the risk suggested by DWAF (1996b) of 8/1,000 canoeists across all scenarios.

Drinking water

QMRA results for *Cryptosporidium* indicate that the probability of infection, to the population that drink untreated river water in subbasin 1 is higher than DWAF (1996a) recommendations for protozoa in drinking water. This risk may be reduced by boiling the water before drinking. On the contrary, in subbasin 10, for all groups, the probability of infection is lower than the thresholds in DWAF (1996a). Based on pathogenic *E. coli*, QMRA results show that in subbasin 1 and 10 the probability of infection is higher than the criteria set in DWAF (1996a). Smeets *et al.* (2010) used 2.74×10^{-07} per person per day as the drinking risk benchmark, based on the work by Signor & Ashbolt (2009). Both subbasins 1 and 10, across all drinking scenarios, are above the 2.74×10^{-07} per person per year.

Mbanga *et al.* (2020) quantified *E. coli* contamination and the potential health hazards that workers at the Darvill WWTP and nearby informal communities may face after exposure to waterborne pathogenic *E. coli* in the WWTP and the uMsunduzi River. The authors found that the daily probability of infection with pathogenic *E. coli* following intentional uptake of 100 mL of the river water upstream and downstream from Darvill is 97.6 and 90.8%, respectively.

Several local studies (WRC 2002; Gemmell & Schmidt 2013; Mbanga *et al.* 2020) highlight that the water quality of the uMsunduzi River is poor to very poor, with apparent faecal pollution which impacts on Inanda Dam which serves the eThekwini Municipality with potable water. The findings of this study agree with these studies where they concluded that the uMsunduzi River water did not meet the requirements of DWAF for drinking and recreational uses, in terms of *E. coli*.

Risk of infection during swimming is highest in the urban parts of the catchment for both children and adults, considering both pathogens. This part receives contribution from sewers, WWTP, as well as animals. Canoeing was considered only for training events in this study. Canoeing competitions do take place in the uMsunduzi catchment. One of these is the historic Dusi Canoe Marathon taking place in February each year, along the uMsunduzi River to the uMngeni River all the way to Durban. It attracts 900–1,600 paddlers each year. Hay (2017) reported that a survey done in 2016, immediately after the race, revealed that 40% of the paddlers contracted mild to severe gastrointestinal infections. This illness has been termed the 'Dusi guts' by the paddlers. It is also reported that seasoned paddlers take antibiotics before the race, as a preventative measure.

Those who drink the water from the river are at the highest risk of infection by both studied pathogens. The rural communities and those in peri-urban areas are affected the most. These communities supplement the lack/shortage of treated water with the water from the polluted uMsunduzi. These communities are not always aware of the risks and how they can protect themselves.

This study focused on exposure per event for risk calculations and did not account or relate to the frequency of these events. To extend this approach to annual risk, the frequency of drinking river water may be considered to occur on a daily basis. However, canoeists may not train every day of the year, the training may intensify closer to competitions compared to outside competitions. Additionally, recreational swimming may be more intense during hotter days of the year (September to March)

as opposed to the colder days. These aspects are important to consider if results are reviewed and compared to an annual risk level. Nonetheless, the effect of risk mitigation can be evaluated on a per event or a daily risk level as well (Signor & Ashbolt 2009).

CONCLUSION

The aim of this study was to utilise SWAT and QMRA to highlight areas of high pollution and the risk that the water may pose to the health of consumers. Based on the water quality modelling results of this study, it can be concluded that the uMsunduzi River is highly polluted with pathogens, and the use of untreated water from the river may result in a high risk of infection.

In the urban section, human sources contribute more than animal sources, indicating to the contributions from broken sewers. In terms of the QMRA, people who use the river for domestic and recreational purposes are at risk of infection by *Cryptosporidium* and pathogenic *E. coli*.

Investing in water treatment facilities, regulation of livestock practices, and safe sanitation systems for communities in need are likely to provide long-term and reliable improvement in the uMsunduzi River water quality. Indirect reuse through the river is not often considered when talking about wastewater reuse. The lack of treated water in rural areas and informal settlements usually forces people to reuse water without giving it much thought. Supporting rural communities without reticulated water is only possible if the water is not directly drawn from the river. Controlling how children access the river and educating the people about boiling their drinking water will help reduce the exposure to pathogens.

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DATA AVAILABILITY STATEMENT

All relevant data are included in the paper or its Supplementary Information.

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