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Approach to a water safety plan for recreational waters: disinfection of a drainage pumping station as an unconventional point source of fecal contamination

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Abstract

In the context of the management of bathing water quality, the intermittent contamination of rainwater drainage pumps (unconventional point sources) could be controlled by peracetic acid disinfection. Thus, a field experimental study was carried out to set up a water safety plan, determining the monitoring parameters and the critical limit for corrective actions. With a 0.5 mg/l dosage, the average logarithmic microbial reduction was 0.50 ± 0.48 for *Escherichia coli* (EC) and 0.43 ± 0.54 for intestinal enterococci. Among the chemical and physical parameters that could be monitored in real time, the oxidation–reduction potential was the only one able to predict the microbial concentration discharged from a drainage pump and the logarithmic abatement of EC. Considering the possible impact of this source on bathing waters in terms of additional risk of gastrointestinal infections, the critical limit for continuous monitoring was established using a quantitative microbial risk assessment (QMRA) model.

Key words: bathing waters, qualitative microbial risk assessment (QMRA), unconventional point sources, water safety plan

INTRODUCTION

Marine waters used for recreational purposes are sometimes affected by short-term pollution episodes exceeding the safety limits posed by the different national regulations (e.g., EU 2006; USEPA 2012) that are generally based on sanitary inspections and compliance with limit concentrations of fecal indicator organisms, represented by *Escherichia coli* (EC) and intestinal enterococci (IE). These events, beside the health risks, often are responsible for swimming bans and the consequent economic losses. The control of marine water contamination can be carried out according to a specific water safety plan (WSP) as reported by WHO (2003). However, its implementation is hampered by problems in identifying and controlling the various sources of pollution, traditionally divided into 'point' (i.e., effluents of wastewater treatment plants – WWTPs) and 'diffuse' (i.e., surface runoff); moreover, in some areas, also drainage pumping stations could be sources of fecal contamination. Drainage pumping stations are structures designed to collect white waters, which include stormwaters and waters coming from surface runoff, and to discharge them into natural waters (rivers or canals). However, in case of sewage infiltrations or even abusive sewage discharges, they can receive gray or black waters. In this scenario, drainage pumps behave as *unconventional* point sources of pollution because they are conceived to regulate the white-water flow, but they also collect sewage.

In the context of a WSP for recreational waters, the control of drainage pumps is complicated by the difficulty in identifying and closing all the fecal entries in the drainage system. In addition, the seasonal variability of the flow due to rainfall variation is responsible for the variability of the duration of water retention inside the drainage pump basins.

In the study area, the flow discharged by drainage pumps was treated through peracetic acid (PAA) disinfection, which is a promising disinfection method for WWTP effluents and for combined sewage overflows, owing to the favorable association of efficacy and biodegradability of such disinfectant (Chhetri *et al.* 2016; Luukkonen & Pehkonen 2017). The general aim of the study was to describe an approach to WSP for bathing waters considering the PAA disinfection of drainage pump discharges as a critical control point, taking into account also the possible toxicity of the disinfectant. Thus, the disinfection process was carefully monitored considering both microbiological parameters to assess the disinfection efficacy on bacterial indicators and the environmental toxicity through the evaluation of a battery of bioassays. Microbiological analyses require time and do not allow timely corrective actions, while physio-chemical parameters could enable the real-time control of the disinfection process. Therefore, a number of physio-chemical parameters were also monitored with the aim of finding which of them best represented the disinfection efficacy for the definition of a critical limit. In the present study, the critical limit was calculated on the basis of a risk-based approach through a quantitative microbial risk assessment (QMRA) model, considering the additional infection risk for bathers due to drainage pump discharge.

METHODS

Site description

The selected study area is a popular bathing site along the Tuscany coast (Italy) which receives over a half million visitors per year and where rapid urbanization of the coastal environment resulted in a heavy contamination load, especially during the summer season. The geomorphology of the area is characterized by an alluvial plain, most of which is below the sea level, that needs to be drained by a complex stormwater drainage system discharging into canals flowing into the sea. In case of intense rain, the mouths of these canals are affected by short-term pollution events that cause risk for bathers, swimming bans, and poor bathing water classification (Federigi *et al.* 2017). To avoid such events, various actions have been undertaken, such as the limitation of private discharges and the improvement of wastewater treatments. Meanwhile, the PAA disinfection of drainage pump discharges has been planned (see the section 'PAA dosage' for details). After initial monitoring aimed at improving the knowledge of pollution levels, the PAA disinfection was started with an initial dosage setup of 0.5 mg/l of the waterflow entering the tank of the drainage pump.

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PAA dosage

A field PAA disinfection experiment was carried out in the collection tank of the drainage pump employing the commercial product PERACLEAN®15, which is a quaternary mixture in equilibrium containing PAA (14–17%), hydrogen peroxide (20–25%), acetic acid (15–18%), and a stabilizing agent free of sulfuric acid. PAA was applied inside the collection tank using a pump that dosed a volume of PAA on a pulse command. Pulses were sent to the pump in proportion to the waterflow entering the tank (the predictive control based on the influent waterflow). The residual PAA was measured at the middle of the tank, using a stand-alone spectrophotometer based on the chromophore *N*,*N*-diethyl-*p*-phenylenediamine (DPD) method. The PAA was mixed into the collection tank by a mechanical mixer installed at the bottom of the same tank.

Since it was a full-scale test, at the beginning of the experimentation, a precautionary dosage range of 0.5–1 mg/l was chosen. In this initial phase, the PAA dosage control system relied only on the measured influent waterflow because residual PAA values were below the detection limit of the DPD method, owing to the low dosage applied.

Environmental data collection

Since the drainage pump system exhibited complex hydrodynamics, a preliminary spatial and temporal study was carried out in order to highlight the flow variability and to choose the best sampling strategy. Then, the untreated (influent) and treated (effluent) waters were regularly monitored for microbial, chemical, and physical parameters. Bacterial indicators, EC and IE, were enumerated, respectively, with Colilert and Enterolert (IDEXX) according to ISO 9308-3 and ISO 7899-1 and expressed as the most probable number (MPN)/100 ml. For each sample, the following chemical and physical parameters were determined using portable instruments: pH, temperature (T), dissolved oxygen (DO), and oxidation–redox potential (ORP) measured using the Hanna HI98196 portable multiparameter and expressed, respectively, as units of pH, °C, mg/l, and mV. Salinity was measured with a salinity digital refractometer (Atago™ Digital Hand-Held Pocket Refractometer, PAL-06S) and expressed as a practical salinity unit. PAA determination was performed by spectrophotometric measurements, after addition to the sample of a DPD (*N*,*N*-diethyl-phenylenediamine) reagent dose (Hach Method 10070) and expressed as mg/l. The measurements were carried out directly at the sampling site, in parallel to sampling. The ecotoxicity was assessed through several bioassays on different model organisms belonging to different steps of trophic web: bacteria (*Vibrio fischeri* – ISO 11348-3: 2007); fresh- and marine water unicellular algae (*Pseudokirchneriella subcapitata* and *Dunaliella tertiolecta* – ISO 8692: 2012; ISO 10253: 1995); and fresh- and marine water crustaceans (*Daphnia magna* and *Artemia franciscana* – ISO 6341: 1996; Vanhaecke & Persoone 1981).

Data analyses

Microbial concentrations were $\log_{10}$ transformed before the statistical analysis, according to a consolidated approach in the field of microbiology (Wymer & Wade 2007). Statistical analyses were performed using MedCalc (MedCalc Software bvba, Belgium). The disinfection efficacy in drainage pumps was calculated comparing the microbial concentrations of waters entering the tank and the ones at the discharge, and the results were expressed as log reduction (LR) of microbial parameters (WHO 2004; Smeet *et al.* 2006). The collected data were used for the development of a multiple regression model to predict bacterial log reductions (LRs) and bacterial concentrations in the discharged waters (dependent variables, considered one at a time) using physio-chemical parameters

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(independent variables). The regression model including all predictors was then simplified by stepwise selection with the Akaike criterion.

QMRA methodology

A simple point QMRA model was developed to assess infection risk resulting from swimming in drainage pump-impacted bathing waters, considering the potential presence of pathogenic EC as index pathogen. Pathogen concentration was calculated, assuming that 8% of the total EC level is pathogenic (WHO 2016, see table C.6 on page 154). This value is a published ratio widely used in other QMRA studies on recreational waters (Federigi *et al.* 2019). In the absence of pathogen surveillance data, the estimation of the pathogen load based on indicators is allowed within the QMRA framework (WHO 2016). Moreover, although the 8% value could not accurately represent pathogenic *E. coli* concentration, it is used to provide a conservative estimate of the health risk derived from bacterial pathogens (Howard *et al.* 2006). The volume of ingested water during recreation was a point estimate of 30 ml, according to the average volume swallowed during swimming in seawater by Schets *et al.* (2011). The probability of infection (P_{inf}) per swimming event was estimated using the beta-Poisson dose–response relationship for diarrheagenic strains of EC in the following equation.

$$P_{\text{inf}}(\text{dose}; \alpha, \beta) = 1 - \left(1 + \frac{\text{dose}}{\beta}\right)^{-\alpha} \tag{1}$$

where the dose is obtained as the product between the concentration of pathogenic EC and the swallowed volume during each swimming event, and the parameters α and β characterize the infectivity of the microorganism (and assume the values $\alpha = 0.395$, $\beta = 2.473$) as described by Strachan *et al.* (2005).

QMRA approach for setting up critical limits

In the context of WSP for recreational waters, point pollution sources, such as drainage pumping stations, can be considered as ‘critical control points’. Consequently, once a continuously measurable control parameter is detected (as a prediction of fecal bacterial concentration), critical limits must be determined. To this aim, an acceptable additional risk of infection of 1 for 1,000 bathers was defined, and QMRA was used to estimate the EC concentration in bathing waters corresponding to the defined tolerable risk. Then, the EC concentration at the exit of the drainage pump was calculated by taking into account a 4 $\log_{10}$ dilution factor from the drainage pump to the bathing site, based on a conservative estimation performed in a previous study carried out in a location with hydrological features similar to the study area (Federigi *et al.* 2017). With this criterion, a critical limit for the predictive chemical variable of the EC concentration in discharged waters was established.

RESULTS AND DISCUSSION

Evaluation of disinfection efficiency and toxicity

On the whole 25 samples, microbial monitoring before and after the treatment process showed a high variability of bacterial and viral abatement, showing in some cases a very poor efficacy (Figure 1). In this figure, besides bacterial parameters, ORP is also represented as a possible physio-chemical parameter for continuous monitoring. Meanwhile, the toxicity was always undetectable in treated waters.

The efficiency of the disinfection over the period of time was 0.50 ± 0.48 for EC and 0.43 ± 0.54 for IE, but the variability of bacterial abatement was very high, and in some cases very poor. The

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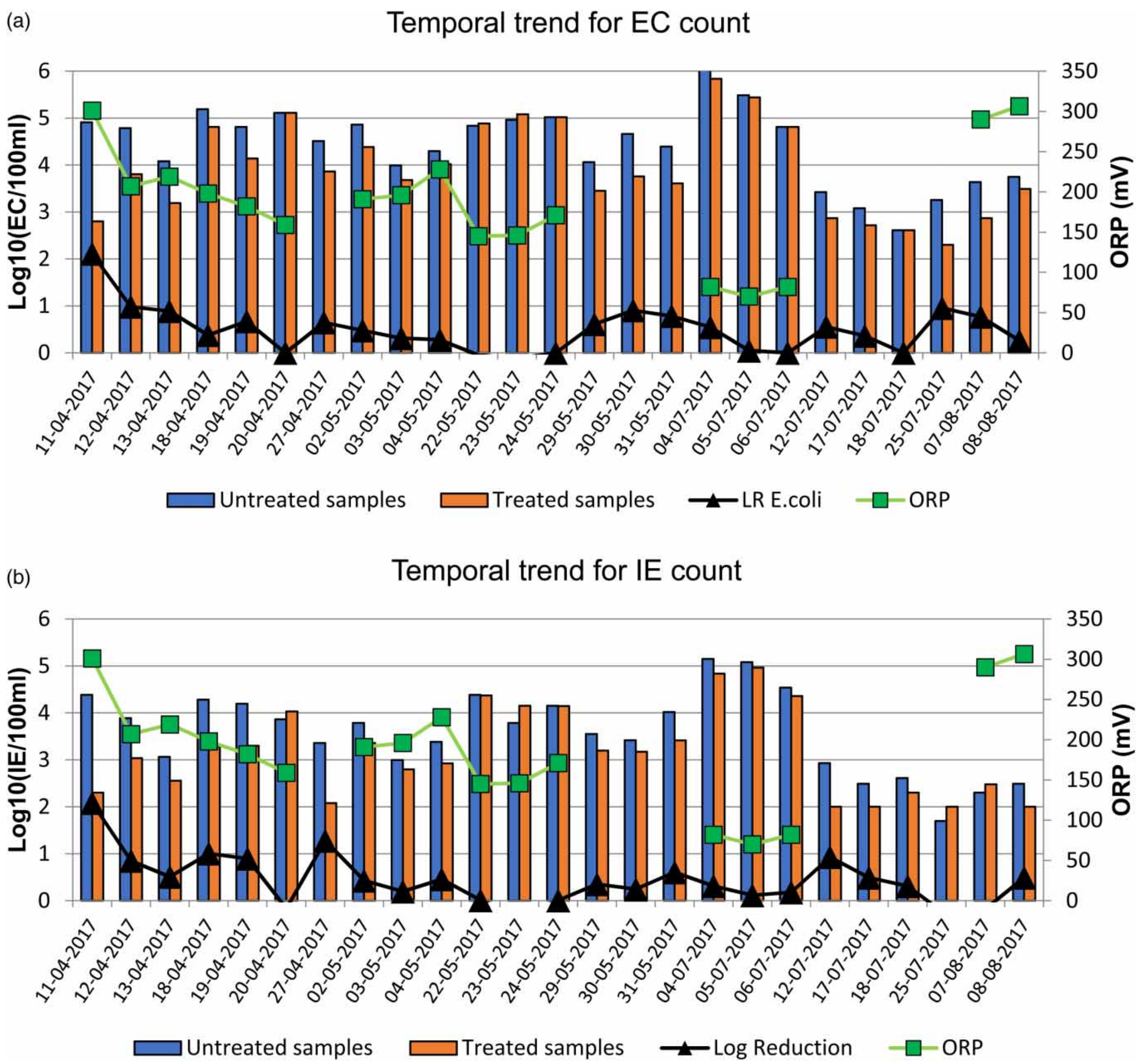

Figure 1 | Temporal trends for microbial parameters in the drainage pump.

differences observed among log reductions of the two microbial parameters were not statistically significant (one-way analysis of variance (ANOVA), $P > 0.05$).

Identification of a predictive variable for PAA effectiveness

Unlike microbiological analyses that require a time of about 24 h between sampling and results, monitoring of chemical parameters gives results in real time, thus allowing timely corrective actions on the PAA disinfection system. Physio-chemical parameters (pH, T, DO, ORP, PAA, and salinity) were used as independent variables to develop the multiple regression model for predicting EC and IE log reductions, EC and IE levels in discharged waters (used one at a time as the dependent variable).

The regression statistics indicated the ORP as the only predictive variable for the concentration of both EC and IE in discharged waters and of the EC LR (Table 1). Models for bacterial concentrations were able to explain the variability in the dependent variable with reasonable accuracy: 77% in the case of EC levels ($R^2 = 0.77$) and 93% in the case of IE levels ($R^2 = 0.93$), whereas the model for EC LR left a large amount of the variability in EC LR as unexplained ($R^2 = 0.28$).

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Table 1 | Regression statistics of four models, using six physio-chemical parameters as independent variables (pH, T, DO, ORP, PAA, and salinity)

Dependent variable	Independent variables retained in the model	R^2 of the model
EC in the discharged waters	ORP	0.77
IE in the discharged waters	ORP	0.93
LR of EC	ORP	0.28
LR of IE	none	–

R^2, the fraction of the variation in the dependent variable that is explained by the model.

Considering the waters discharged from the drainage pump, the observed EC data were plotted against ORP values, as depicted in Figure 2. In the following equation, this relationship was represented by a linear regression model.

$$EC = -0.0112\ ORP + 6.4023 \tag{2}$$

Therefore, in this experimental system, ORP was chosen for continuous monitoring since it is able to give timely advice for corrective actions, in case of sudden changes in the inflow waters.

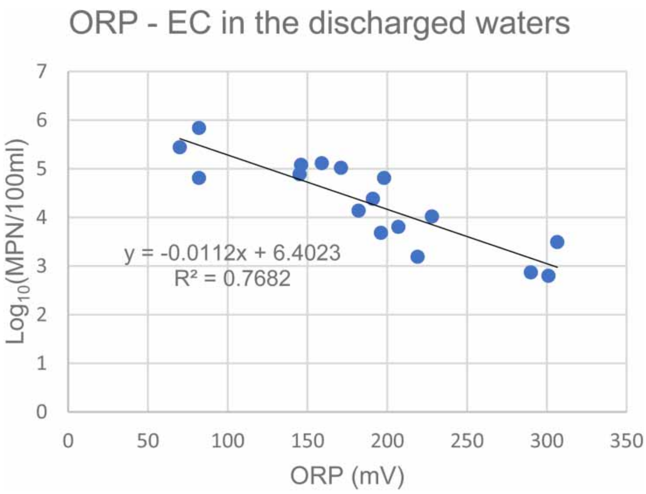

Figure 2 | Predictive model for EC (dependent variable) based on ORP (explanatory variable).

This result is supported by the literature on the sewage treatment, where ORP is used as a measurement of the wastewaters' capacity to permit the occurrence of specific biochemical reactions. In particular, the ORP-based regulation is mainly applied for controlling activated sludge processes, such as the control of nitrification and denitrification processes (Klapwijk 1998; Lu *et al.* 2000) and of nutrient removal conditions (Yu *et al.* 1997; Li & Bishop 2002), but it is also applied for monitoring wastewater disinfection (Harp 2000). Moreover, in our study, the relationship described in Equation (2) can be considered reliable over the time in the full-scale plant because the nature of entering waters is not subjected to high fluctuations in the ORP. This is confirmed by the ORP values measured in waters entering into the drainage pump (data not shown) where the ORP assumes a low mV value due to organic loading, with negative values ranging from −89 to −22 mV. On the other hand, the ORP values were positive in the discharged waters (Figure 2), indicating that the oxidizing conditions were attributable exclusively to the disinfectant application.

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Calculation of the critical limit based on QMRA

To set critical limits at the drainage pump discharge, the point QMRA model was used in conjunction with a regression model for EC, assuming an acceptable additional risk per swimming event of 1 infection for 1,000 bathers deriving from the drainage pump. The QMRA model was used to calculate the EC concentration in the bathing water at the acceptable infection risk (Equation (1)). Subsequently, this EC value was used to calculate EC levels at the drainage pump discharge taking into account the dilution factor. Finally, the EC value at the drainage pump was converted into ORP using the linear regression model built to predict EC concentration from ORP at the exit of the drainage pump (Equation (2)). A schematic framework of these calculations is shown in Figure 3.

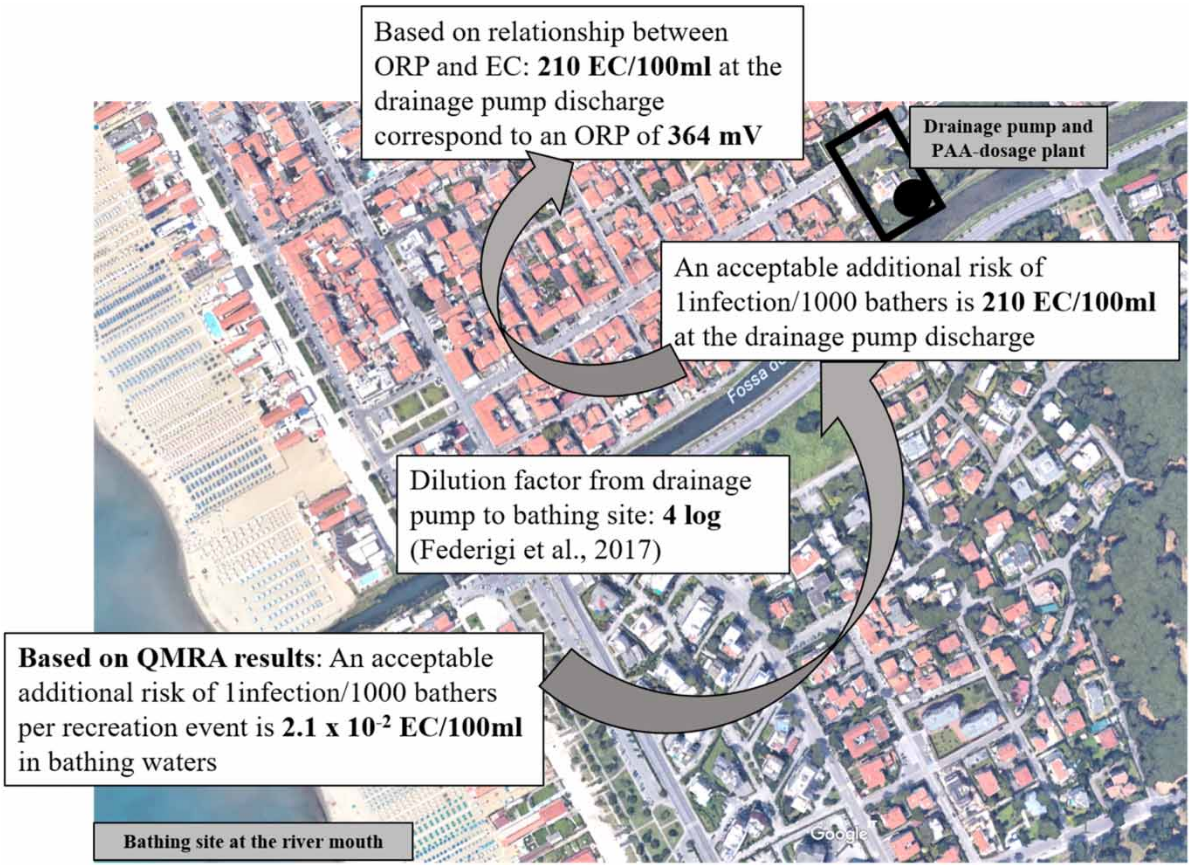

Figure 3 | Framework for performing QMRA to define the critical limit at drainage pump discharge, integrated with a predictive model of EC concentration for rapid corrective actions.

At this benchmark infection risk (1/1,000 bathers), the estimated microbiological limit was 0.021 MPN/100 ml in bathing waters, equivalent to 210 MPN/100 ml at the exit of the drainage pump. Using the relationship found between EC and ORP, EC concentration corresponded to an ORP value of 364 mV in the discharged waters. This value represents the critical limit for corrective actions on the PAA disinfection system, in order to achieve the tolerable risk target in bathing waters.

CONCLUSION

This study focused on the development of a methodology for the implementation of WSP for bathing waters with QMRA, using the data collected during full-scale experimentation on the effluent of a

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drainage pumping station disinfected with PAA. This approach allowed us to set up the PAA disinfection procedure in order to reduce the probability of infection below the defined tolerable infection risk. The microbial contamination and the efficacy of disinfection turned out to be highly variable and unpredictable using the fecal indicators monitoring. Nevertheless, ORP proved to be strong related to the bacterial concentration in treated waters and was chosen for continuous monitoring. Subsequently, the application of QMRA allowed us to calculate the critical limit for timely corrective actions. This investigation showed how QMRA can be a useful tool for supporting WSP applied to bathing waters, thus enabling the implementation of the methodology to other unconventional point sources characterized by high variability in flow and contamination. However, the current study exhibits a series of limitations because the QMRA model was based on a single pathogen and worked with point-estimated values. Therefore, the model needs further in-depth studies such as (i) the inclusion in the QMRA of multiple fecal pathogens associated with waterborne recreational illness in the study area or detected in the waters under investigation during monitoring (in the case of lack of epidemiological data) and (ii) the representation of variability and uncertainties of input parameters (i.e., conversion factors of pathogen to indicators, ingestion volume) using a stochastic approach, in which they are modeled as probability density functions rather than as point values.

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