Hydrodynamic modelling and forecasting of microbial water quality in a drinking water source

Ekaterina Sokolova, Thomas J. R. Pettersson and Olof Bergstedt

ABSTRACT

Faecal contamination often enters drinking water sources through emergency discharges, which occur as a result of technical malfunctions or a hydraulic overload of the sewer system during periods of heavy rain. In October–November 2012, several emergency discharges entered Lake Rådasjön – a drinking water source for Gothenburg, Sweden. To describe and forecast the influence of these emergency discharges on the microbial water quality, the spread of *Escherichia coli* (*E. coli*) within the lake was simulated using a three-dimensional hydrodynamic model. The model was run for a period of 4 months using the observed data, and for a period of 9 days using meteorological forecast data. The modelling results showed how much every contamination source contributed to the total *E. coli* concentrations at the water intakes. The agreement between the modelling results and the measured concentrations was satisfactory. The results of this study led to the decision to use the lake for drinking water production. This study demonstrated that the proposed modelling approach can be used to provide short-term forecasts of the microbial water quality in drinking water sources.

Key words | *E. coli*, faecal contamination, lake, MIKE 3 FM, sewer overflows, water quality modelling

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INTRODUCTION

The faecal contamination of drinking water sources can pose health risks for consumers. Faecal contamination often enters drinking water sources through emergency wastewater discharges during sewer overflow events, which occur mainly due to a hydraulic overload of the sewer network during periods of heavy rain or due to technical malfunctions. The emergency wastewater discharges may increase the levels of faecal contamination in raw water used for drinking water production.

In order to ensure that the drinking water treatment processes are sufficient to manage the increased levels of faecal contamination in the raw water, it is necessary to describe the influence of the wastewater discharges on the microbial water quality at the intake of a drinking water treatment plant. However, regular monitoring of the raw water quality may not be sufficient to capture the rapid changes in the faecal contamination levels. To complement the monitoring, hydrodynamic modelling can be used to simulate the spread of faecal contamination within the water source. For example, hydrodynamic modelling studies of microbial water quality were performed by Thupaki *et al.* (2010) on a freshwater lake, Liu & Huang (2012) on an estuary and by Chan *et al.* (2013) on marine beaches. Moreover, hydrodynamic modelling was used to provide a forecast regarding the microbial water quality at bathing sites (Chan *et al.* 2013).

Lake Rådasjön is a drinking water source for the cities of Gothenburg and Mölndal in Sweden. Between 1 October and 3 November 2012, several emergency wastewater discharges into the drinking water source Lake Rådasjön occurred; these discharges were caused by hydraulic overloads and, on one occasion, by a power failure. The goal of this study was to implement an existing hydrodynamic model of the lake to assess and forecast the influence of these wastewater discharges on the microbial water

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quality at the raw water intakes. A three-dimensional hydrodynamic model of Lake Rådasjön (Sokolova *et al.* 2012a, 2013) was used to simulate the spread of the faecal contamination from the emergency overflows and other contamination sources within the lake. In this study, we show how a hydrodynamic model can be used to describe and forecast the water quality impacts of extreme weather-related events. The novelty of this study lies in suggesting hydrodynamic modelling as a method to forecast the microbial water quality in drinking water sources.

METHODS

Study area

Lake Rådasjön is located on the west coast of Sweden. The surface area of the lake is approximately 2.0 km^2 and its catchment comprises an area of 268 km^2. The maximum water depth is 24 m and the main inflow into the lake is the river Mölndalsån (Figure 1). The water flow in the river Mölndalsån varies from 1 to 20 m^3/s and the average water flow is approximately 4 m^3/s. The raw water intakes

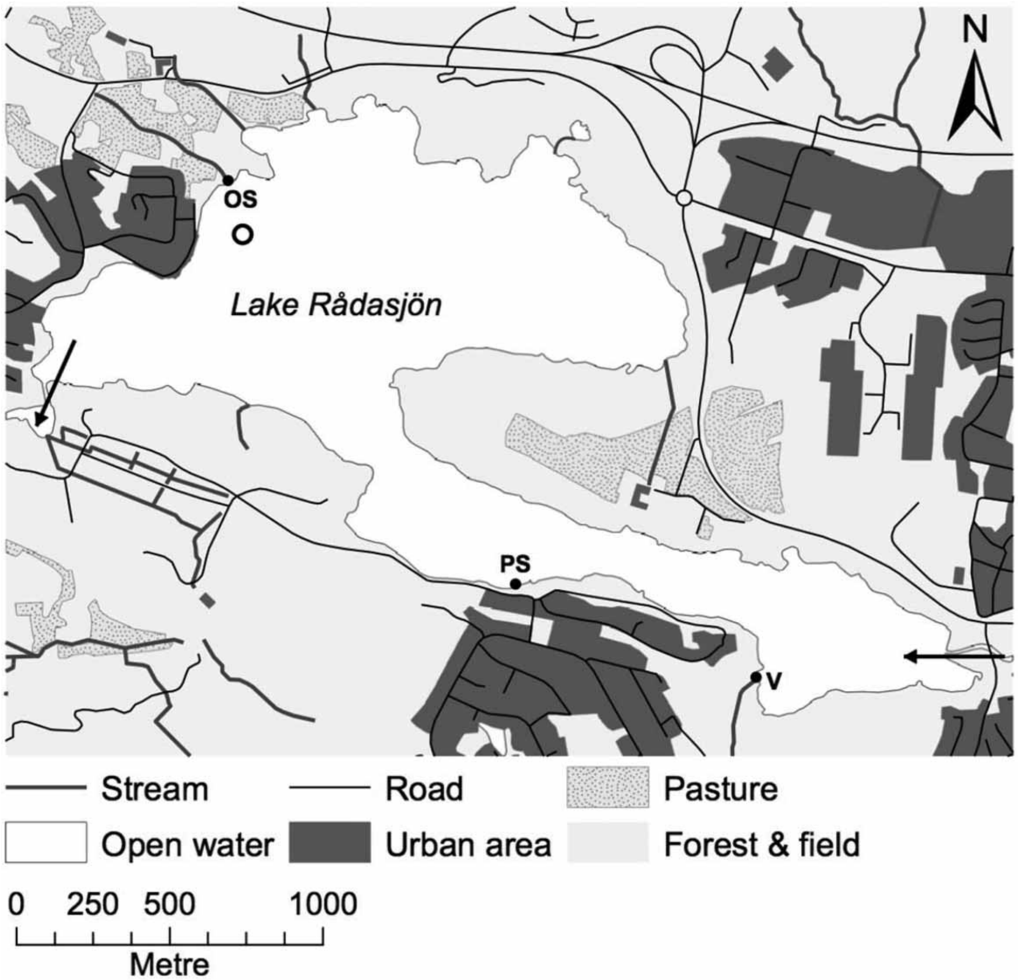

Figure 1 | Map of Lake Rådasjön. Symbols: OS – the on-site sewage treatment systems, PS – the pumping station Pixbo Päls, V – the stream Vällbäcken that transports emergency wastewater discharges. Black arrows represent the inflow to the lake from the river Mölndalsån and the outflow from the lake to Lake Stensjön. The circle represents the location of the raw water intakes.

for the cities of Gothenburg and Mölndal are located in the north-western part of the lake at depths of 8 and 15 m, respectively (Figure 1).

Lake Rådasjön serves as the main water source for the city of Mölndal (60,000 consumers, annual withdrawal of water 5 million m^3) and as one of the reserve water sources for the city of Gothenburg (500,000 consumers, annual withdrawal of water 5 million m^3). The city of Mölndal has a reserve surface water source that could provide enough water for approximately 1 week. For the city of Gothenburg, the main drinking water source is the river Göta älv. However, the water intake in the river is closed for up to one-third of the time, often due to suspected high levels of faecal contamination (Åström et al. 2007). The reserve water sources for the city of Gothenburg could provide drinking water for approximately 3 weeks. Water from Lake Rådasjön is also used regularly to complement the river water.

There are no longer any national raw water quality standards in Sweden. The drinking water treatment plants supplied via this lake are operated according to the following arbitrary guidelines regarding the *Escherichia coli* (*E. coli*) concentration in the raw water: 3–10 No/100 mL – increased need for microbial barrier efficiency and >10 No/100 mL – severely increased need for barrier efficiency. There is no upper limit regarding the microbial raw water quality, the decisions are based on the aim to use the best available raw water for drinking water production.

Lake Rådasjön is subject to contamination from various faecal sources, which were identified in earlier studies of this lake (Sokolova *et al*. 2012a, b). The faecal contamination enters the lake through emergency wastewater discharges from the pumping station Pixbo Päls (location PS, Figure 1) and from the pumping station located near the stream Vällbäcken (location V, Figure 1). These emergency wastewater discharges occur several times a year, due to hydraulic overloads or technical failures within the sewer network. The faecal contamination in the lake can also originate from the on-site sewage treatment systems (hereafter referred to as on-site systems), which were designed for sludge removal. These on-site systems do not meet legal requirements regarding treatment performance and provide little to no microbial reduction. The effluent from these on-site systems is continuously released into a stream that enters the lake close to the raw water intakes (location OS, Figure 1). Moreover, faecal contamination can enter the lake through the inflow from the river Mölndalsån (Figure 1), which transports contamination from various sources, mainly emergency sewer overflows and on-site systems, located in its upstream catchment area.

In addition, the faecal contamination from a cattle grazing area located to the east of the lake and an urban area located to the north-east of the lake could enter the lake with the surface runoff (pasture and urban areas in Figure 1). In an earlier modelling study of the lake (Sokolova 2011), the contributions from the cattle grazing area and the urban area to the total *E. coli* concentrations at the water intake were estimated to be approximately 2 and 1 $\log_{10}$ units less than from the river Mölndalsån, respectively. According to the results of another study (Sokolova *et al*. 2012b), the contribution from the cattle grazing area to the concentrations of *Bacteroidales* genetic markers at the water intake was approximately 1 $\log_{10}$ unit less compared to the contributions from the on-site systems and from the Pixbo Päls pumping station. Furthermore, in a study on the pathogen concentrations in this lake (Sokolova *et al*. 2012a), it was found that the contributions from the cattle grazing area and the urban area to the *Cryptosporidium* concentrations at the water intake were approximately 1 and 3 $\log_{10}$ units less than from the Pixbo Päls pumping station, respectively. In summary, based on the results of the earlier studies of this lake, we have concluded that the contribution from these nonpoint sources is generally much smaller than from the other identified contamination sources. Therefore, we do not consider the contribution from these nonpoint sources in this article.

Model implementation

To simulate the water flows in Lake Rådasjön, the three-dimensional time-dependent hydrodynamic model MIKE 3 FM (DHI 2011a) was used. In order to simulate the fate and transport of the faecal contamination in Lake Rådasjön, the microbial water quality model ECO Lab (DHI 2011b) was coupled to the hydrodynamic model of the lake.

The model was set up to study the influence of the emergency sewer overflows that occurred between 1 October and 3 November 2012 (Table 1) on the microbial water quality at

Table 1 | Emergency sewer overflows into Lake Rådasjön (data provided by the Municipality of Härryda)

Date	Time	Contamination source	Duration, hours	Volume, m^3
1 October 2012	16:30–20:00	Pixbo Päls	3.50	30
3 October 2012	21:00–24:00	Pixbo Päls	3.00	30
25 October 2012	16:00–02:00	Vällbäcken	10.00	170
25 October 2012	16:00–02:00	Pixbo Päls	10.00	15
3 November 2012	18:30–20:15	Pixbo Päls	1.75	13

the raw water intakes in Lake Rådasjön. The first simulation was performed on 12 and 13 November, shortly after the overflow events, in order to enable timely decisions regarding the drinking water production. The model was run for a period in the past (1 September–11 November 2012) using observed hydrometeorological data, and a forecast period (12–20 November 2012) using forecast data (Figure 2). Later, the simulation was repeated for 12–20 November 2012 using observed data, and was extended to include the period 21 November–31 December 2012 (Figure 2).

Hydrodynamic model

The hydrodynamic model for this lake was developed in earlier studies (Sokolova *et al.* 2012a; 2013). In brief, the MIKE 3 FM model is based on the numerical solution of three-dimensional incompressible Reynolds averaged Navier–Stokes equations using Boussinesq and hydrostatic assumptions (DHI 2011a). The model consists of continuity, momentum, temperature, and density equations, and is closed using a turbulent closure scheme. Horizontal and vertical eddy viscosities were modelled using the Smagorinsky and k-epsilon formulations, respectively.

The modelling domain was approximated with prisms (triangles in the horizontal plane), using a flexible mesh approach, i.e. the size and shape of the mesh elements in the horizontal plane could vary to describe the complex geometry of the modelling domain. The mesh consisted of 611 nodes and 1,015 elements. The length of the triangles' sides varied from approximately 40 to 90 m, and was adjusted to describe the coastline and bathymetry. Vertically, the lake was approximated with 37 layers of varying thickness (from 0.5 m in the thermocline zone to 3 m at the bottom). For more information on the spatial discretisation of the modelling domain and model calibration the reader is referred to Sokolova *et al.* (2013).

The model was set up to account for the hydrometeorological conditions and to simulate heat exchange between the atmosphere and the lake. The data regarding air temperature, relative humidity, wind speed and direction were obtained from Weather Underground (Weather Underground 2013) for the meteorological station of Landvetter, located approximately 10 km from Lake Rådasjön. The observed and forecast data were available with 0.5 and 3 hour temporal resolution, respectively. The clearness

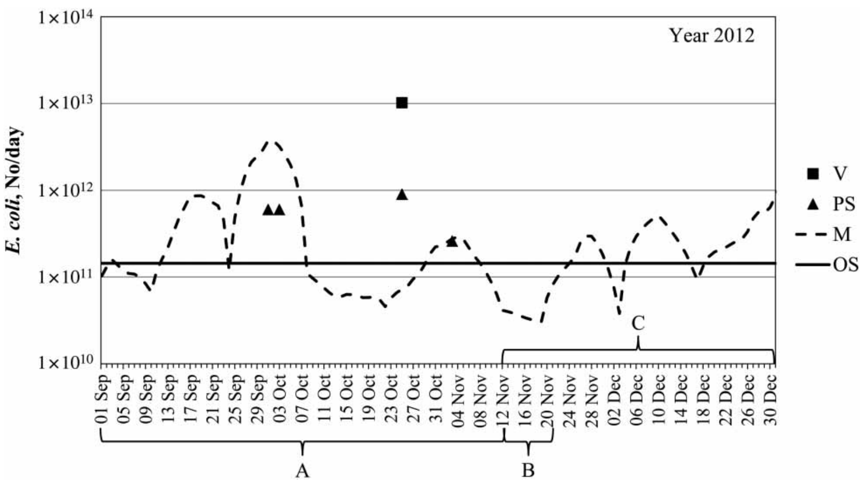

Figure 2 | *E. coli* load to Lake Rådasjön and simulated periods. The contamination sources are: the river Mölndalsån (M), the on-site sewage treatment systems (OS), the stream Vällbäcken (V) and the Pixbo Päls pumping station (PS). (A) and (C) represent the periods for the hindcast simulations, (B) represents the period for the forecast simulation.

coefficient (cloud coverage) was specified as an average of the observed data for the period 2007–2011 (data provided by the Swedish Meteorological and Hydrological Institute, SMHI). Precipitation on the lake surface was not accounted for in the model. The temperature distribution in the lake at the beginning of the simulation on 1 September 2012 was assumed to be the same as on 1 September 2011. Therefore, the initial conditions regarding the temperature distribution in the lake were specified using the modelling output for the year 2011, obtained in an earlier study (Sokolova *et al.* 2013). The temperature on the open boundaries was described as zero gradients. The lake was assumed to be covered by ice starting from 30 November 2012. In the model, the heat exchange and the wind stress were excluded for the areas covered by ice.

The conditions on the inflow and outflow boundaries were specified using time series (1 day temporal resolution) of the flow in the river Mölndalsån and the water level in Lake Stensjön, respectively. The data regarding flow variations in the river Mölndalsån were obtained from the SMHI (2013); the data on the water level variations in Lake Stensjön were obtained from Mölndals Kvarnby – the association that controls and regulates the water flow in the water system of the river Mölndalsån (Mölndals Kvarnby 2013). For the forecast period, the flow in the river Mölndalsån and the water level Lake Stensjön were specified as constant values; the values used were the average values for November 2007–2011. The initial conditions in the lake were defined by the constant surface elevation; the flow velocity was set to zero. The land boundary was defined by zero normal velocity.

Microbial water quality model

The microbial water quality model ECO Lab uses the flow fields from the hydrodynamic model to calculate the concentrations of faecal indicators in the lake. The fate and transport of the faecal contamination were simulated using *E. coli* bacteria as a faecal indicator. In the ECO Lab model, the inactivation of the *E. coli* in the lake due to temperature and sunlight was described by Equation (1):

$$\frac{\mathrm{d}C}{\mathrm{d}t} = -k_0 \cdot \theta_\mathrm{S}^{Sal} \cdot \theta_\mathrm{I}^{Int} \cdot \theta_\mathrm{T}^{Temp-20} \cdot C \qquad (1)$$

where t is the time; C is the *E. coli* concentration; k_0 (1/day) is the decay rate at 20 °C for a salinity of 0‰ and darkness; θ_s is the salinity coefficient for the decay rate; *Sal* (‰) is the salinity; θ_I is the light coefficient; *Int* (kW/m^2) is the light intensity integrated over depth; θ_T is the temperature coefficient for the decay rate; *Temp* (°C) is the water temperature.

The decay of *E. coli* and other faecal indicators in Lake Rådasjön was studied earlier during outdoor microcosm trials, which were performed in different seasons (March, August and November 2010) in light exposure and darkness (Sokolova *et al.* 2012b). Since no statistically significant differences between the persistence of *E. coli* in light and dark incubations were identified (paired samples *t*-test, $p > 0.05$), the light coefficient (θ_I) in Equation (1) was set to 1. The salinity coefficient (θ_s) in Equation (1) was also set to 1, as Lake Rådasjön is a fresh water lake. The temperature (θ_T) and the decay rate (k_0) coefficients for *E. coli* were set to 1.04 and 0.2, respectively. To distinguish between the influences of different contamination sources on the water quality at the intake, the contamination spread from each source was modelled separately.

Several emergency discharges were registered within the sewer network located in the vicinity of Lake Rådasjön during the study period (Table 1). As a result of these emergency discharges, untreated wastewater entered Lake Rådasjön from the pumping station Pixbo Päls and from the pumping station located near the stream Vällbäcken. The discharges on 1 October, 3 October and 3 November 2012 occurred due to a hydraulic overload of the sewer system, as a result of heavy rains, while the discharges on 25 October 2012 were caused by an extensive power failure. It can be assumed that, due to dilution by stormwater, the *E. coli* concentration in the discharged wastewater was lower when the discharges were caused by heavy rain than by a power failure.

To account for the influence of the on-site systems on the microbial water quality in Lake Rådasjön, it was assumed that these systems provide no microbial reduction and discharge untreated wastewater. The discharge of wastewater from the on-site systems was calculated based on the estimation that there are 36 people connected to these systems and that the average water consumption is 200 L/person/day.

The *E. coli* concentrations in the discharges from the contamination sources were assigned using data on the *E. coli* concentrations in untreated wastewater from the Pixbo Päls pumping station measured under dry weather conditions (four measurements). The data collected under dry weather conditions were used in order to provide a worst-case scenario and to prevent possible underestimation due to uncertainties regarding the degree of wastewater dilution during wet weather conditions. It was assumed that the *E. coli* concentration in the untreated wastewater was 2×10^6 No/100 mL, which was the median value. The *E. coli* concentration in the emergency discharges on 25 October 2012 was assumed to be higher than on the other occasions, since these discharges were caused by a power failure and not by a hydraulic overload. Therefore, to account for the worst-case scenario, it was assumed that the *E. coli* concentration in the emergency discharges on 25 October 2012 was 6×10^6 No/100 mL, which was the maximum measured concentration.

The concentrations of *E. coli* bacteria in the river Mölndalsån were measured every week during September–December 2012 (18 measurements). The *E. coli* concentrations in the river Mölndalsån during this period varied between 6 and 370 No/100 mL, and the median and average concentrations were 31 and 84 No/100 mL, respectively. It was assumed that the concentrations of *E. coli* in the river Mölndalsån varied linearly between the measured values. For the forecast period, the concentration of *E. coli* in the river was the average value for November 2009–2011.

Based on the aforementioned assumptions and data, the *E. coli* load from different sources to Lake Rådasjön was calculated (Figure 2).

The *E. coli* concentrations at the water intakes were monitored by the cities of Gothenburg and Mölndal through laboratory analysis of regularly collected grab samples.

RESULTS AND DISCUSSION

Modelling microbial water quality

The spread of the faecal contamination in Lake Rådasjön during September–December 2012 was simulated using the developed hydrodynamic model. The modelling results showed how much every considered contamination source contributed to the total *E. coli* concentrations at the water intakes (Figure 3). According to the modelling results, the highest peaks at the water intakes were caused by the river Mölndalsån and the emergency discharges caused by a power failure (Figures 3 and 4). These discharges occurred on 25 October 2012 and were transported to the lake by the stream Vällbäcken. The modelling results showed that, in this case study, the contribution from the emergency discharges caused by heavy rainfall was lower than the contribution from the other contamination sources (Figure 3).

The contribution from the river Mölndalsån to the *E. coli* concentrations at the water intakes fluctuates over time (Figure 3), due to variations in the *E. coli* load (Figure 2) and in the hydrodynamic situation in the lake. After the contamination enters the lake with the water flow from the river Mölndalsån, it is mixed during the transport in the narrow and shallow part of the lake (Figure 5). The contaminant spread in the wide part of the lake is largely driven by wind (Figure 5).

Since the discharges from the pumping stations Pixbo Päls and Vällbäcken also enter the lake in its narrow part, the contaminant spread from these sources is similar to that from the river Mölndalsån. The discharge from the Vällbäcken pumping station caused high concentrations at the water intakes (Figure 3), mostly due to the assumption of high *E. coli* load during this event (Figure 2).

Since the *E. coli* load from the on-site systems was assumed to be constant (Figure 2), the fluctuations in the contribution from this source to the *E. coli* concentration at the intakes are only dependent on the hydrodynamic situation, which is largely driven by wind.

A comparison of the modelling results and the measured *E. coli* concentrations at the water intakes indicated that, taking into account the measurement uncertainties of the *E. coli* analyses (Köster et al. 2003), the agreement between the simulated and measured values was satisfactory (Figure 4). The model performance was quantified in a similar manner as by Chan et al. (2013): the correlation coefficient (R) and root-mean-square-error (RMSE) between decimal logarithms of the simulated and measured *E. coli* concentrations ($\log_{10}$(*E. coli*)) were calculated. For the 8 m

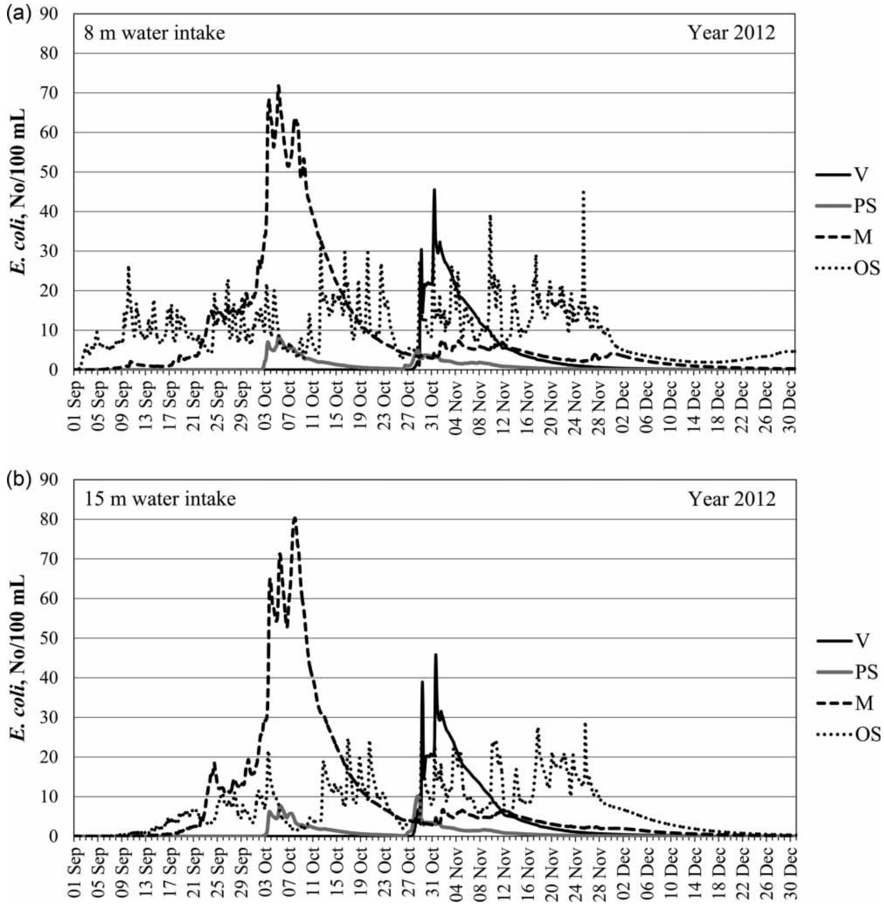

Figure 3 | Modelling results: the contribution from the different contamination sources (V – the stream Vällbäcken, PS – the pumping station Pixbo Päls, M – the river Mölndalsån, OS – the on-site sewage treatment systems) to the total *E. coli* concentrations at the 8 m (a) and 15 m (b) water intakes in Lake Rådasjön.

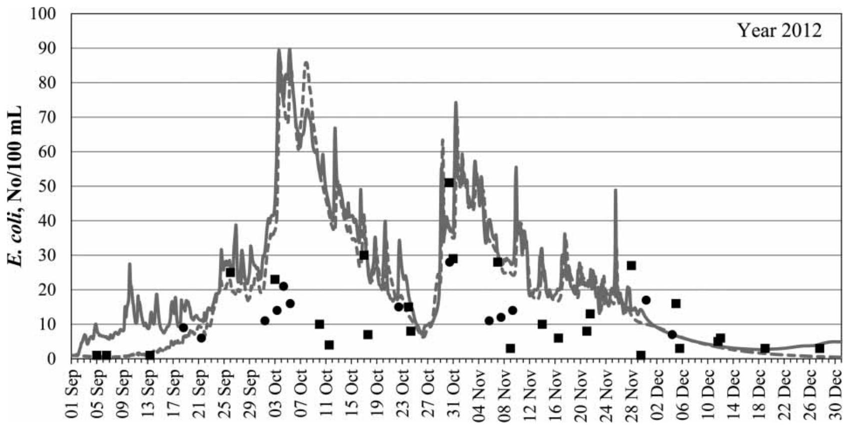

Figure 4 | Simulated (continuous and dotted lines) and measured (circles and squares) *E. coli* concentrations at the 8 and 15 m water intakes, respectively.

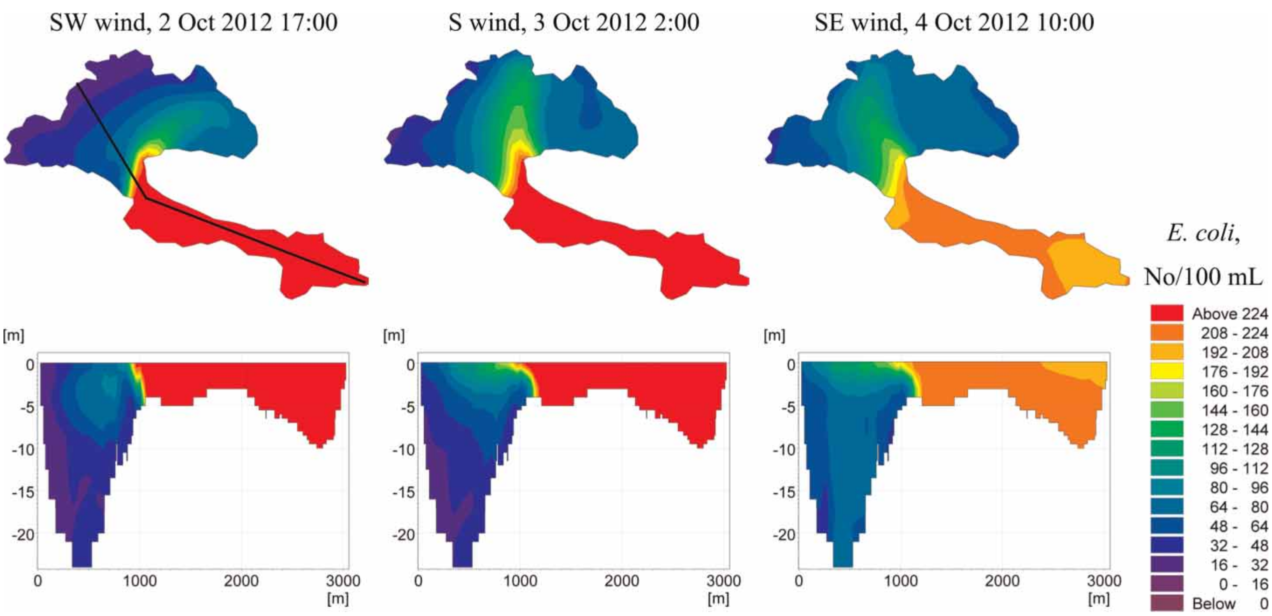

Figure 5 | Modelling results: the spread of the faecal contamination (*E. coli*, No/100 mL) from the river Mölndalsån within Lake Rådasjön on 2, 3 and 4 October 2012. Top and bottom rows represent horizontal and cross-section views, respectively. The approximate location of the cross-section is shown in the top left figure.

water intake, the model performance was ($N = 13$): $R = 0.64$ and RMSE $= 0.42$ $\log_{10}$(*E. coli*). For the 15 m water intake, the model performance was ($N = 17$): $R = 0.71$ and RMSE $=$ 0.47 $\log_{10}$(*E. coli*).

The model described the peak in the *E. coli* concentrations observed at the end of October and the low concentrations observed in December (Figure 4). However, on some occasions, the simulated concentrations were higher (up to approximately 1 $\log_{10}$ unit) than the measured *E. coli* concentrations at the intakes, for example, in the first half of October (Figure 4). These high simulated concentrations of *E. coli* at the water intakes in the first half of October were caused by the river Mölndalsån (Figure 3), due to the high *E. coli* load from the river in the beginning of October (Figure 2). However, the *E. coli* load from the river was likely overestimated on this occasion, due to linear interpolation between the weekly measured concentrations (24 September: 17 No/100 mL; 1 October: 370 No/100 mL; 8 October: 12 No/100 mL).

The simulated *E. coli* concentrations at the intakes were linearly dependent on the input data regarding the *E. coli* load from the contamination sources. Therefore, the uncertainties in the modelling results originated (i) from the assumptions regarding the *E. coli* concentrations in the wastewater discharges and (ii) from the linear interpolation between the weekly measured concentrations in the river Mölndalsån.

Forecasting microbial water quality

The hydrodynamic model was used to provide a forecast of the *E. coli* concentrations at the raw water intakes. For this purpose, the model was run for the period 12–20 November 2012 using meteorological forecast data and the assumptions regarding the boundary conditions and the *E. coli* load (forecast run). Then, this simulation was repeated using the observed data as input for the model (hindcast run). The comparison of the results of both runs showed that the magnitude of the simulated concentrations was similar; but with some discrepancies in terms of temporal variations (Figure 6). For example, on 17 November 2012, the difference between the *E. coli* peaks at the water intakes predicted by the forecast and hindcast runs was less than 2 No/100 mL (Figure 6). However, for the forecast run, the predicted *E. coli* peaks occurred 11 and 10 hours earlier at the 8 and 15 m water intakes, respectively, than for the hindcast run (Figure 6). The most prominent differences were observed for the simulated contribution from the on-site

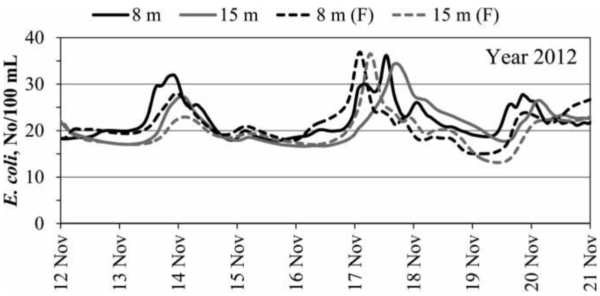

Figure 6 | Modelling results: the *E. coli* concentrations at the 8 and 15 m water intakes simulated using the observed input data ('8 m' and '15 m', respectively) and the forecast input data ('8 m (F)' and '15 m (F)', respectively).

systems (Figure 7). This can be explained by the fact that the spread of contamination from the on-site systems was strongly dependent on the wind forcing; thus, the differences between the forecasted and observed meteorological data (Figure 8) were reflected in the modelling results. Some minor differences (up to 2 No/100 mL) were also noticed for the simulated contribution from the river Mölndalsån (Figure 7); these differences originated from the differences in the underlying assumptions regarding the *E. coli* load from this source. No differences (less than 1 No/100 mL) were observed for the simulated contributions from the emergency discharges.

The results of this study demonstrated that a hydrodynamic model can be used to simulate the raw water quality in a near real-time regime and to forecast the microbial water quality in a drinking water source. However, in order to utilise this modelling approach to facilitate everyday drinking water management, a better integration of the input data is needed. In the case of Lake Rådasjön, most of the required input data (observed and forecasted meteorological data,

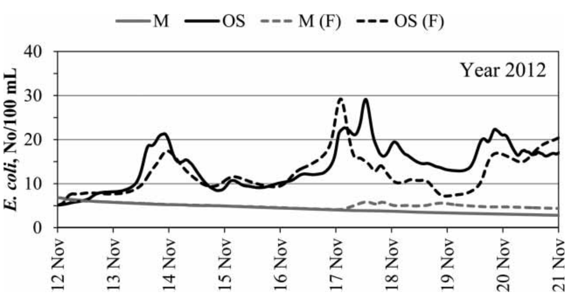

Figure 7 | Modelling results: the contributions to the *E. coli* concentrations at the 8 m water intake from the on-site sewage treatment systems and the river Mölndalsån simulated using the observed input data ('M' and 'OS', respectively) and the forecast data ('M (F)' and 'OS (F)', respectively).

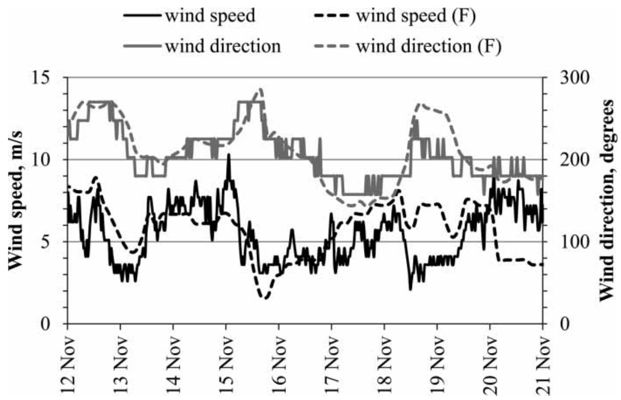

Figure 8 | Observed and forecasted (F) wind speed and direction (source: Weather Underground).

water flow in the river Mölndalsån and water level in Lake Stensjön) were available from different online sources. The data regarding the emergency discharges, the *E. coli* concentrations in the river Mölndalsån and at the water intakes were available through personal contact with the municipalities. However, preparation of the input data for the model was still a labour intensive and time-consuming process. A solution could be to construct a unified database, in which the data are stored in a suitable format and regularly updated. This database could be used to generate input data for the model, which then could be run continuously and used to provide short-term forecasts of the *E. coli* concentrations at the water intakes. Nevertheless, careful and extensive validation of the model is of the utmost importance for the implementation of this modelling approach for drinking water management.

According to long-term climate predictions, the intensity of precipitation events will increase (Olsson *et al.* 2009; Willems *et al.* 2012). Considering the current capacity of the sewer networks, this would lead to more frequent sewer overflow events. In this context, hydrodynamic modelling constitutes a suitable tool to describe and forecast the impact of these extreme weather events on the microbial water quality in drinking water sources.

Outcomes for drinking water suppliers

After the emergency overflow events that took place between 1 October and 3 November, on 10 November, the at-line monitoring equipment of the drinking water supplier in Gothenburg detected an increase in the *E. coli* concentrations at the raw water intakes in Lake Rådasjön. Consequently, the Gothenburg raw water intake was closed. However, on 13 November, when provided with the modelling results, which showed that the peak in the *E. coli* concentrations had already passed (Figure 4), the water supplier decided to re-open the water intake. This example shows that such modelling results can provide helpful information for drinking water suppliers and decision makers. For more examples of the practical outcomes from the modelling studies of Lake Rådasjön, the reader is referred to Sokolova *et al*. (2013).

CONCLUSIONS

The hydrodynamic model was successfully used to provide an assessment and forecast of the influence of emergency sewer overflows and of other sources on the faecal contamination levels at the water intakes. The comparison of the modelling results with the measured *E. coli* concentrations at the water intakes showed satisfactory agreement. The modelling results provided helpful decision support data for the drinking water suppliers.

It was demonstrated that the proposed modelling approach can be used to provide short-term forecasts of the microbial water quality in a drinking water source. Such forecasts are of particular importance in the context of the predicted increase of rainfall intensity and, consequently, the expected increase in frequency of emergency sewer overflows. A better system for collection and integration of the necessary input data for the model is suggested as the next step towards the implementation of this tool on a regular basis. This modelling approach can be used to facilitate drinking water management and to address the health risks for consumers.

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