Combining modeling and monitoring to study fecal contamination in a small rural catchment

Morgane Bougeard, Jean-Claude Le Saux, Anna Teillon, Jérôme Belloir, Cécile Le Mennec, Sterenn Thome, Gael Durand and Monique Pommepuy

ABSTRACT

The present study sought to identify *Escherichia coli* sources in a small catchment and to use the agro-hydrological model soil and water assessment tool (SWAT) to estimate their impact on river water quality. The innovative aspects of this research are to assess the hourly variations of fecal contamination and to take these variations into account in the model to provide a better evaluation of river quality. Thus, water samples were taken weekly at the river outlet ($n = 4$) and 24-h monitoring sessions were performed during low and high-flow periods ($n = 74$). *E. coli* variations were found to be primarily linked to rainfall and not to resuspension mechanisms. Subdaily fluctuations and deviations were $\pm 0.33\ \log_{10}$ cfu/100 mL and $\pm 0.70\ \log_{10}$ cfu/100 mL for dry (<3 mm/day) and wet (>3 mm/day) weather, respectively. After river flow calibration, all known pollution sources (septic systems, manure spreading, farm discharges) were introduced into SWAT. The model reproduced the fecal contamination in the river and the use of subdaily deviations allowed us to evaluate the simulation quality and compare grab samplings with simulated daily *E. coli* concentration, thus confirming that the performance of the model is better when additional information on hourly concentration variations is used.

Key words | catchment, *E. coli*, modeling, monitoring, non-point sources

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INTRODUCTION

When water quality exceeds safety standards, actions are taken aimed at protecting public health and the environment. In a recent review, Boehm *et al.* (2009) identified the uncertainties and shortcomings of the current regulation criteria and proposed that new ones should be defined, based on the water quality framework developed by the World Health Organization (WHO), which recommends a harmonized approach to risk assessment and risk management for microbial hazards in recreational waters (WHO 2003). The opinion of the WHO experts led the European Community, on 15 February 2006, to adopt a new bathing water directive imposing stricter standards for *Escherichia coli* and enterococci in recreational bathing waters (Directive 2006/7/EC, CEU 2006).

To examine the causes of temporary low water quality impairment, research has been conducted to identify and quantify all major point and non-point sources of targeted pollutants (Hunter *et al.* 1992; Kay *et al.* 2004; Pohlert *et al.* 2005; Shehane *et al.* 2005; EPA 2009). Non-point sources include many diffuse sources and cause pollution via rainfall and surface runoff. As the runoff moves, it picks up and carries away natural and human-made pollutants, finally depositing them into rivers and coastal waters (EPA 1994).

Over the years hydrologists have developed numerous predictive tools (e.g. empirical models, lumped models, distributed models and statistical models) that aid decision making with respect to water resources and water quality management. For a long time, applications of more sophisticated models continued to meet difficulties owing to lack of data or restrictive assumptions (insufficient database size for calibration, spatially heterogeneous diversity of landscape, etc.).

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However, recently gained knowledge and new technological advances presently offer solutions that could improve models currently available for water quality prediction, by improving understanding of the processes or by offering more advanced theories, new measurement technologies (satellites, environmental or microbial tracers), and advanced data processing, archiving and visualization technologies (Sivapalan *et al*. 2003). Panels of tools have recently been successfully applied to compare and display how potential scenarios will impact watersheds, especially with regard to runoff and water quality. These include forecasting models, stochastic models, agro-hydrologic models (MIKE-SHE, SWAT), Automatic Geospacial Watershed Assessment tool, and Analytical Tools Interface for Landscape Assessment (EPA 1999; Steets & Holden 2003; Jamieson *et al*. 2004; Parajuli *et al*. 2009). These models are often academic or commercially available and a few, such as SWAT, have open free access and are thus of interest for application in areas where a watershed model is not yet in use.

With regard to water quality, one promising method is the application of watershed models to fecal quality, which provides an interesting approach to assess agricultural practices and to propose alternatives to improve river quality (Ferguson *et al*. 2007; Bougeard *et al*. 2011). Jamieson *et al*. (2004) reviewed the evolution of this approach over recent decades; 'first loading' models were developed to simulate landscape microbial pollution processes: MWASTE (Moore *et al*. 1989), COLI (Walker *et al*. 1990) and SEDMOD (Fraser *et al*. 1998). Then, models took into account the survival and transport of fecal bacteria in receiving waters such as lakes, rivers and coastal areas (Canale *et al*. 1993; Wilkinson *et al*. 1995; Riou *et al*. 2007). More recently, a SWAT (soil and water assessment tool) model developed by Sadeghi & Arnold (2002) was built incorporating both landscape and in-stream microbial processes, a well as a watershed model (Tian *et al*. 2002). The present model now incorporates various complex parameters and processes such as a land base budget of diffuse pollution and occasional urban sources (sewage treatment plant, septic tanks, etc.). It also calculates the fecal load, which is then routed through the different sub-catchments using hydrological models, including bacterial behavior sub-models (Ferguson *et al*. 2007; Parajuli *et al*. 2007; Baffaut & Benson 2009; Bougeard *et al*. 2011).

In parallel, fecal catchment budgets were successfully generated: total maximum daily load (TMDL) approaches, recommended by US EPA regulation (EPA 1997), were established in water bodies not meeting the water quality standards; land use water quality models were set up with a large extension in UK watersheds (Kay *et al*. 2005, 2010); and a process-based mathematical modeling was also applied to catchment pathogen budgets (Ferguson *et al*. 2007).

The models have proved to be extremely useful for simulating pollutant sources, enabling the reduction of pollution from those sources and improvement of water quality to meet the applicable state water-quality standards (EPA 1999, 2009). They are also an important tool for hypothesis building in the search for significant diffuse sources. Watershed models have the capability of predicting the spatial pattern of various hydrological factors and contaminant outflows within a watershed, and are thus widely used for simulating microbial fate and transport in watersheds (Steets & Holden 2003; Parajuli *et al*. 2007; Baffaut & Benson 2009; Ferguson *et al*. 2009).

Such models range in complexity, from simple, empirical ones to highly complex models with extensive data requirements. On a daily time scale, relatively simple models can give good results. Nevertheless, most of the time, they fail to take hourly variations into account; hourly temporal resolution could sometimes be of interest, to simulate short storm events (Micovic & Quick 2009). For example, recent research has suggested that, under such circumstances, a rapid increase of *E. coli* concentration occurs in rivers. Changes of one or two orders of magnitude have been observed within hours (Jamieson *et al*. 2005; Davies-Colley *et al*. 2008). The short-term response during storm events must, therefore, be taken into account. However, hourly temporal distribution of bacterial loads, which has to be introduced in the model, represents a major difficulty, given the limited information available and the fact that good-quality input data are usually only available on a daily or monthly time step. For these reasons, daily time scale models are often taken as a good compromise between the difficulty in getting precise information on bacterial input and the necessity to simulate short events.

The focus of this paper is to present an innovative approach to obtaining a good representation of the hourly variations of *E. coli* concentrations observed *in situ*, investigated using simple methodology and a modified daily time step model. Thus, the study objectives were: (1) to determine the hourly range of variation of *E. coli* concentration as a function of two major factors (rainfall and sediment resuspension); and (2) to combine field results with the daily modeling of *E. coli* concentrations in rivers using the agro-hydrological model SWAT. The application was set up in a small rural catchment named Sainte Anne (SA) located in Brittany (western France).

METHODS

Study site presentation

The study was conducted in the 5 km^2 SA catchment located on the western French Atlantic coast. Figure 1 shows the map of the watershed indicating monitoring stations and the wastewater treatment plant.

The river system has a main river, Le Nevent, which is 6.1 km long. Land uses across the total catchment are primarily pasture (28.1%), urban areas (18.3%), forest (15.8%), corn silage (14.8%), gardens (14.7%) and bare ground (8.3%). Climate is of the oceanic temperate type and annual rainfall ranged from 988 to 1,530 mm in the period 2000–2008 (Guipavas Station, Meteo-France). The elevation for the catchment ranges from 0 to 93 m.

Agriculture is one of the land uses, with 28.1% of the catchment covered by pasture and grassland. The catchment contains three livestock farms (two bovine and one swine), which have land holdings wholly within the hydrological drainage basin. Three types of agricultural source have been identified: swine manure spreading, bovine pasture and discharges from farmyards. At the time of the investigation, the catchment supported approximately 60 cattle. Most of the time, because of the oceanic temperate climate, cattle are allowed to graze on the pasture (from February to November) and have unrestricted access to the stream channel. Bovine and swine manure is applied to land as fertilizer, with timing and application rates for both manures based on guidelines specified by local regulation (J.C. Le Saux, IFREMER, personal communication).

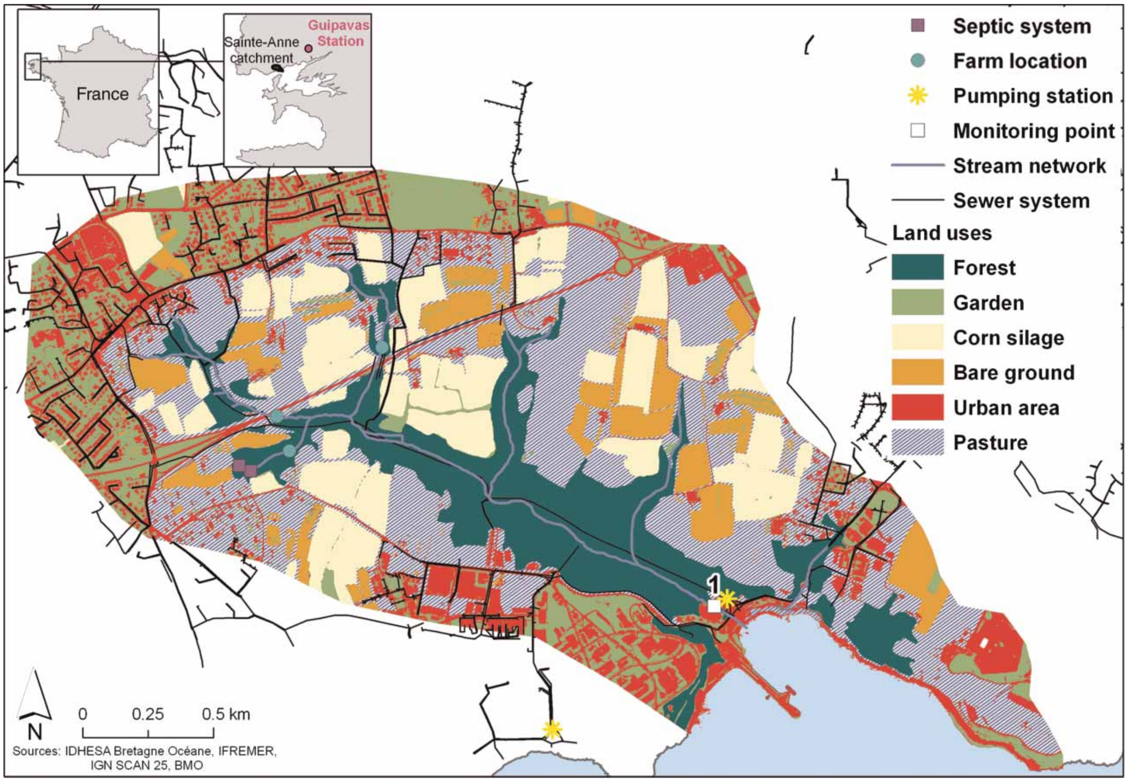

Figure 1 | SA catchment location, land uses, monitoring points and main fecal contamination sources.

The catchment is located in the town of Plouzané, which has a human resident population of 13,000. For the SA catchment, the population is located mostly within the upper catchment. Waste from the habitations is transported out of the catchment via a sewer system (Figure 1), whilst that from the remaining dispersed population passes via septic tanks and soaks away into the stream system. Non-agricultural bacteria sources include failing septic systems. Three failed septic systems are known of in the catchment; these dispose of their sewage through a pipe going directly into the stream (an illegal straight pipe discharge).

Sampling

A data set was collected at monitoring point 1 (Figure 1) from April 2008 to October 2009. The monitoring collected two types of data: river flow and *E. coli* concentrations.

River flow

River flow was measured weekly from April 2008 to February 2009 (Table 1) by a portable flow meter (OTT acoustic digital current (ADC)). From February 2009, a permanent flow meter (Mainstream Hydreka) was fixed on the bottom of the river to obtain continuous measurements (time step: 5 min). The accuracy of the permanent flow meter was 1 mm/s.

Escherichia coli analyses

We collected water at point 1: daily samplings from April 2008 to July 2009 and weekly samplings from July to October 2009 (Table 1). *E. coli* enumeration was realized by the miniaturized, most-probable-number method (ISO 9308-3) using microplates. Briefly, 200 μL of several decimal dilutions of the sample were added to each of the 96 wells of the microplate containing the substrate (4-methylumbelliferyl-β-D-glucuronide) (MUGlu) in dehydrated form. The microplates were incubated for 36–48 h at 44 °C. The hydrolysis of MUGlu was detected under ultraviolet light. The number of positive wells after incubation allowed the calculation of the *E. coli* abundance using a statistical analysis (Servais *et al.* 2007). Moreover, an automatic sampler yielded four 24-h monitoring sessions at point 1 with one liter sampled each hour (Table 2); one 24-h monitoring session was conducted during dry weather (DW) and three were done during separate rainfall events (WW: wet weather).

Table 1 | Dates for sampling and model calibration and validation

Sampling period	Dates	n	Use for modeling
River flow			
→ Weekly	From 17 April 2008 to 24 February 2009	48	River flow validation
→ Continuous	From 25 February 2009 to 30 June 2009	126	River flow calibration
E. coli concentrations	From 9 April 2008 to 19 June 2009	94	*E. coli* concentrations calibration

Resuspension

To evaluate the resuspension of fecal bacteria from sediment to the water column, artificial resuspension of sediment was created by releasing 20 L of river water dyed with fluorescein in the small riverbed (2 m large and 15 cm deep, average river flow 0.067 m^3/s). The *E. coli* resuspension procedure used in the study was described by Muirhead *et al.* (2004). In our study, this procedure was adapted to characteristics of the river, which has a limited size, and tracer additions were made according to a slightly modified procedure (Graf 1995). For this, 5 mL of fluorescein concentrate were diluted in the 20 L of river water before the experiment

Table 2 | General features of 24-h monitoring in the SA catchment

Weather conditions	Date and time	Antecedent dry days	Rain depth (mm/day)	n
Dry weather (DW)	20 May 16.30 to 21 May 15.30, 2008	1	0	24
Wet weather 1 (WW1)	28 April 16.30 to 29 April 15.30, 2008	0	23.8	24
Wet weather 2 (WW2)	2 July 20.00 to 03 July 20.00, 2008	0	5.4	13
Wet weather 3 (WW3)	11 May 15.30 to 12 May 16.30, 2009	2	3	13

(i.e. concentration 1.5 mg/L). The resulting water plume was monitored for *E. coli* and turbidity, 50 m downstream. Dye tracing was used to ensure sampling into the created plume. A first sample was collected before resuspension at T0, and then at times T1 (2 min), T2 (5 min) and T3 (10 min). For each sample, *E. coli* concentration and turbidity measurements were realized. Turbidity of the water samples was measured by nephelometry (Hach 2100AN Ratio Nephelometer, Hach Co, Loveland, CO, USA) and the results reported in nephelometric turbidity units (NTU). We conducted eight resuspension experiments at different monitoring sites on the main river of the SA catchment on 22 April 2009 and 5 May 2009.

Meteorological data

Meteorological data (daily precipitation, minimum and maximum air temperatures, wind speed, solar radiation and relative humidity) was obtained from the Guipavas meteorological station managed by Meteo France (Figure 1).

Statistical analyses

All statistical analyses were performed on $\log_{10}$-transformed values of *E. coli* concentrations. The study used the very robust Shapiro–Wilk test before the parametric test to ensure the data ($\log_{10}$ *E. coli* concentrations measured over 24-h monitoring sessions) had normal distributions. Fisher and Student tests were then used to compare average *E. coli* concentrations and variance in dry and WW conditions.

SWAT model

SWAT is a continuous time model that operates on a daily time step; it was developed by the United States Department of Agriculture, Agricultural Research Service (Arnold & Fohrer 2005). Briefly, the model simulates the hydrological processes of a catchment. In the hydrodynamic component, the model estimates the runoff separately in each sub-basin and obtains total runoff for the catchment. The runoff model uses a modified SCS (soil conservation service) curve number method, and peak runoff rates are predicted from a modified rational formula. The Penman–Monteith method calculates the estimation of potential evapotranspiration. Sub-models, including microbial survival and transport, have also been added. The SWAT microbial sub-model was explained in Bougeard *et al.* (2010, 2011).

ArcSWAT (Di Luzio *et al.* 2002) was developed as an interface between SWAT 2005 and ArcGIS 9.3, allowing the model to be run on a geographical information system.

Model procedures

There were different steps to modeling *E. coli* fluxes at the outlet of the catchment and then in coastal waters. Each step is important and the details are explained below.

Implementation of SWAT with SA catchment characteristics: digital elevation model, river system, land uses, soils and climatic data

The topography of the catchment was derived from a 15 × 15 m (digital elevation model); land use information was obtained from infrared orthophotos by a clustering algorithm (Nasca Geosystems); and for soil data, we used two profiles made in soil permeability studies (S. Cabillic, Brest Métropole Océane, personal communication). The average depth of the soil profiles ranged from 1.20 to 1.50 m and textures were generally silty or sandy loam. Daily climatic data input into SWAT concerned rainfall, humidity, wind, solar radiation and minimal and maximal temperature.

Calibration and validation of river flow at point 1

SWAT was calibrated daily for river flow at point 1 from 25 February 2009 to 30 June 2009 (Table 2). For this, autocalibrations used the PARASOL method by changing each parameter ten times within the allowable range of values for the specific parameter (van Griensven & Meixner 2007; Green & van Griensven 2008). Some parameters were adjusted from the initial SWAT values to match the simulated and observed daily flows; these are presented in Table 3.

Validation was then performed from 01 April 2008 to 24 February 2009. The coefficient of determination (r^2) and the Nash–Sutcliffe efficiency (Ens) were used to evaluate model predictions of flow (Nash & Sutcliffe 1970). Using this validation, if the values of r^2 and Ens are

Table 3 | Parameters for river flow and bacterial calibration in SWAT

Variable name	Description	Value
SURLAG	Surface runoff lag coefficient	0.001
ESCO	Soil evaporation compensation factor	0.138
EPCO	Plant uptake compensation factor	0.916
SLSUBBSN	Average slope length (m)	−9.12%
SLOPE	Average slope steepness (m/m)	+25.0%
ALPHA_BF	Baseflow alpha factor (days)	0.027
GW_DELAY	Groundwater delay times (days)	19.4
RCHRG_DP	Deep aquifer percolation fraction	0.001
GWQMN	Threshold depth of water in the shallow aquifer required for return flow to occur (mm H_2O)	78.7
SOL_K	Saturated hydraulic conductivity (mm/h)	100/50/20
SOL_AWC	Available water capacity of the soil layer (mm H_2O/mm soil)	+24.7%
CN2	Initial SCS runoff curve number for moisture condition II	+29.2%
Parameters for fecal bacteria simulations		
WDLPQ	Die-off factor for less persistent bacteria in soil solution at 20 °C (1/day)	2.01
WGLPQ	Growth factor for less persistent bacteria in soil solution at 20 °C (1/day)	0
WDLPS	Die-off factor for less persistent bacteria adsorbed to soil particles at 20 °C (1/day)	0.023
WGLPS	Growth factor for less persistent bacteria to soil particles at 20 °C (1/day)	0
WDLPRCH	Die-off factor for less persistent bacteria in streams (moving water) at 20 °C (1/day)	0.35
WDLPRES	Die-off factor for less persistent bacteria in water bodies (still water) at 20 °C (1/day)	1.030
BACTKDQ	Bacteria runoff extraction coefficient (m^3/mg)	90
BACTKDDB	Bacteria partitioning coefficient	0.90
THBACT	Temperature adjustment factor for bacteria die-off/growth	1.070
WDLPF	Die-off for less persistent bacteria on foliage at 20 °C (1/day)	0.016
BACT_SWF	Fraction of manure applied to land areas that has active colony-forming units	1

equal to one, then the model prediction is perfect, and the model's efficiency is considered satisfactory if r^2 is superior to 0.6 and Ens superior to 0.5 (Santhi *et al.* 2001; Gassman *et al.* 2007).

Integration of contamination sources

Integration with point source (farm discharge locations and septic system discharges) and non-point source (manure spreading) functions of real sources were defined for the catchment. The next section describes the simulated scenario chosen for the SA catchment. The sewer system discharges were not integrated into SWAT because there was no observed overflow during the simulated period.

Analysis of simulation results for *E. coli* concentrations and loads at point 1

A frequency curve analysis method was used to compare measured and predicted data (Baffaut & Benson 2003; Pachepsky *et al.* 2006; Guber *et al.* 2007; McGechan *et al.* 2009; Parajuli *et al.* 2009). Moreover, to improve this frequency analysis, the dry and wet subdaily deviations of *E. coli* concentrations observed in the river were considered.

Scenario for contamination sources simulated on the catchment

The study simulated the fecal contamination in the river from 1 April 2008 to 30 June 2009. Two source types were added to simulate the majority of the known sources of contamination (Figure 1): human and animal.

Human sources

Human sources correspond to septic systems and they were simulated as direct discharges in the river. We selected only the septic systems located within 200 m of streams because of their possible impact on river water quality, and those defined as polluting by the local survey done by the regulator. Thus, SWAT integrated two septic systems (Figure 1). The study assumed a discharge defined by an *E. coli* concentration equal to 6.3×10^6 *E. coli*/100 mL and a flow equal to 0.10 m^3/day per capita (Parajuli 2007).

Animal sources

The study considered two types of animal source in the simulation: manure spreading and farm discharge locations.

It assumed that three farm locations caused pollution in streams from washing on hard-surfaced areas. After wash water analyses, farm location discharges were defined as those producing 1 m^3/day with 3×10^5 *E. coli*/100 mL. This value was chosen from a study conducted by Lewis *et al.* (2005), analyzing *E. coli* in runoff from dairy housing and barns (*E. coli* concentrations from driveways and parking areas, gutters and drains are 1.95×10^5, 2.62×10^5 and 2.98×10^2 *E. coli*/100 mL, respectively).

Manure spreading was simulated during DW (no rainfall), and during only the authorized spreading period from 15 January to 30 June, according to local agricultural practices:

- from 15 January to 30 April on corn silage before seedbed preparation;
- from 1 May to 30 June on pasture.

Spreading occurred at the rate of 30 metric tons (wet weight) of manure per hectare, i.e., 4.5 tons dry weight, which corresponds to one working day. From these data, we assumed a spreading calendar to perform spreading on different sub-catchments with:

- two manure spreadings on corn silage from January to April; and
- two manure spreadings on pasture from May to June on each sub-basin.

The manure spread corresponded to fresh swine manure with an *E. coli* concentration equal to 10^6 *E. coli*/g (Geldreich 1966).

RESULTS AND DISCUSSION

Monitoring

Two 24-h monitoring sessions at point 1, realized in April and May 2008, are presented in Figures 2(a) DW and (b) WW1. During DW, *E. coli* concentration levels were very low, between 1.6 and 3.1 $\log_{10}$ *E. coli*/100 mL. DW baseflow concentrations varied over time but this variation was very limited.

The limited variability observed in rivers has already been reported and could correspond to the weak activity of non-point sources and runoff during such wet periods (Bougeard *et al.* 2011).

During WW1, the *E. coli* level and its variations were greater than in DW. Figure 2(b) shows the *E. coli* concentration and amount of rainfall observed during WW1 monitoring: 23.8 mm of rainfall were recorded over 24 h, and *E. coli* concentrations ranged from 1.6 to 4.7 $\log_{10}$ *E. coli*/100 mL. A main trend in the wet period surveys showed that very high *E. coli* levels were evident at the start of a rainfall event and that, as the event progressed, these concentrations decreased. For example, the maximal value (Hour 8) occurred three hours after a major rainfall event (2.6 mm/h at Hour 5). Similar results have already been reported (Jamieson *et al.* 2005; McCarthy 2008) and this phenomenon, known as the first flush effect, is based upon the hypothesis that a significant amount of the catchment's available surface pollutant load is washed during rainfall (Soupir *et al.* 2006). In this study, most of the *E. coli* peaks occurred simultaneously with flow increase, but other causes could also contribute, such as subsurface accumulated load in urban storm water (McCarthy 2008), resuspension of contaminants in the sediment (Nagels *et al.* 2002; Ashbolt & Roser 2003; Signor *et al.* 2005) or discharges of sewer systems close to the river.

During the following hours (Hours 9–15), a decrease in concentration was observed; then, a heavy rainfall event (5 mm at Hour 15) was responsible for a new contamination peak two hours after the peak rainfall (4.3 $\log_{10}$ *E. coli*/100 mL at Hour 17). This end-flush is less important in terms of contamination than the first peak, probably because the surface runoff or riverbed sediments were less rich in fecal coliforms following the first rainfall event. The WW1 monitoring showed that the lag time of the SA catchment was very short: between 2 and 3 h, depending on previous climate conditions.

It is generally accepted that fecal concentrations in rivers are higher during wet periods (Ashbolt & Roser 2003) and that overland flow is the major contributor of

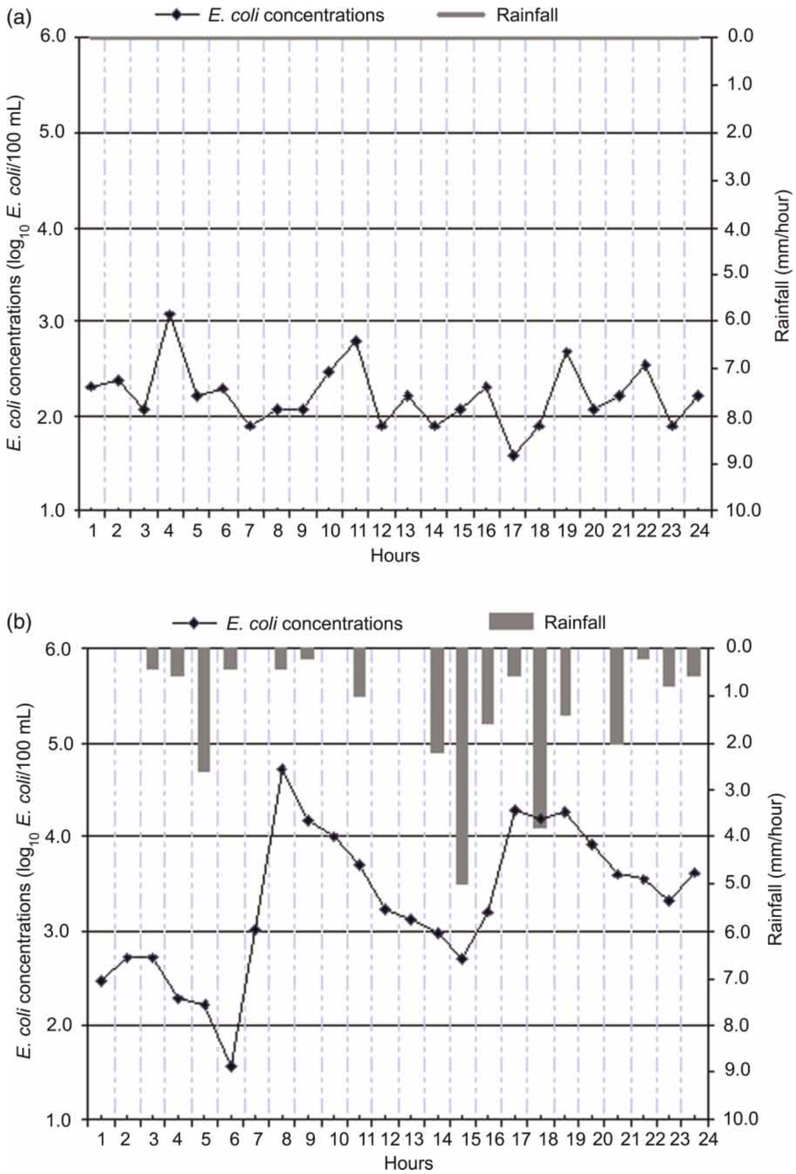

Figure 2 | Hourly monitoring (a) during DW from 4.30 p.m. 20 May 2008 to 3.30 p.m. 21 May 2008 (DW monitoring) and (b) during a rainfall event from 4.30 p.m. 28 April 2008 to 3.30 p.m. 29 April 2008 (WW1 monitoring).

bacteria to streams (Hunter *et al*. 1992). Under these circumstances, rainfall mobilizes and transports non-point source microbial particles via runoff. Microbial densities correlate significantly with increased rainfall and stream flow in estuaries. Heavy rainfall can also lead to direct fecal input into the watershed by reducing field draining when soils are saturated (Lipp *et al*. 2001; Shehane *et al*. 2005). In some cases, it is possible that the increased flow also leads to resuspension of contaminants from the river sediments (Nagels *et al*. 2002; Muirhead *et al.* 2004; Signor *et al*. 2005);

The current study investigated the role of sediment on fecal water contamination. Resuspension of river sediment at different sites was created by artificial water release. Figure 3 shows the *E. coli* concentration and turbidity evolutions during the eight experimentation periods (mean and standard deviation (SD)). We observed very little increase in *E. coli* concentration ($<0.5\ \log_{10}$ colony-forming units (cfu)/100 mL) due to sediment resuspension, while turbidity was multiplied by a factor of 18. Turbidity ranged from 1.7 to 31.5 NTU for T0 and T2 while *E. coli* concentration varied from 2.1 to 2.4 $\log_{10}$ cfu/

100 mL for T0 and T2. Moreover, in the resuspension experiments, the relationship between *E. coli* and turbidity was very weak ($y = 6.3146x + 2.1405$, $r^2 = 0.0209$).

By no means were the largest increases in *E. coli* concentration observed during the eight experiments, and concentrations in water were significantly inferior to those observed at point 1 during rainfall events (Figure 2, increase of two or three orders of magnitude). Thus, sediment contribution to water contamination seems to be limited in our study site. This weak impact is probably because sediment was composed of muddy sand (95% of particles >10 μm) that is poorly contaminated with *E. coli.* Indeed, it is currently reported that fecal bacteria concentration in sediment is related to high level of organic matter and small (mainly silt) grain size (Haller *et al.* 2009) and it was found that 90.5% of bacteria are associated with small particles (<10 μm) (Auer & Niehaus 1993).

To conclude, these results showed that, in SA catchment, *E. coli* concentration increases during rainfall events were probably not related to resuspension mechanisms. This conclusion contrasts with that of Gentry *et al.* (2006) who observed that the presence of *E. coli* in a faster response was due to a non-runoff source. Davies-Colley *et al.* (2008) showed a close correlation ($r = 0.98$) between *E. coli* concentration and turbidity during such events. They concluded that, most of the time, the major fraction of the total coliforms in agricultural streams resided in the streambed, from where it could be released during high-flow events. Moreover, they also showed that the fecal matter washed-off from manure spread on the land cannot be responsible for storm flow peaks because wash-in would tend to arrive much later than the observed peaks of *E. coli* and flow peaks (Davies-Colley *et al.* 2008). Furthermore, Nagels *et al.* (2002) and Muirhead *et al.* (2004) demonstrated an increase of two orders of magnitude during artificial flood experiments carried out in New Zealand, in the absence of rainfall, and a good relationship as well between *E. coli* and turbidity. In contrast, our results indicate that, in the SA catchment, only a small fraction of contamination can be attributed to resuspension, and sediment contribution to water contamination seems to be limited. Several hypotheses explain the differences observed between the French and other sites, among them climate, type of vegetation, size of catchments, stream morphology and mainly the nature of sediments. This is consistent with observations indicating that the site-specific magnitude of the streambed *E. coli* effect on river water quality would depend on the concentration of *E. coli* in the sediments (Kim *et al.* 2010). Thus, we did not introduce streambed sediment release processes into SWAT in our study in agreement with Kim *et al.* (2010) who stated that, when sediment contribution is limited, fecal contamination, taking into account fate and transport, is the principal bacteria input to rivers during rainfall events. In this study, we indeed observed a

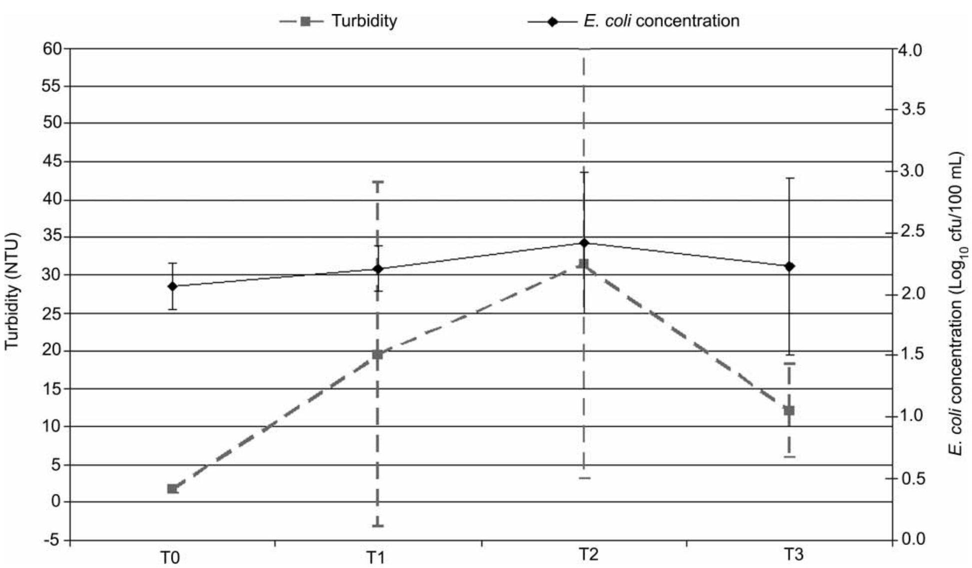

Figure 3 | *E. coli* concentrations and turbidity during the eight resuspension experiments in the Nevent river.

significant effect of precipitation on *E. coli* concentration in water during WW.

The subdaily variations of fecal contamination in the river were calculated from the four 24-h monitoring sessions realized at point 1. Figure 4 presents the average *E. coli* concentrations with SD for DW, WW1, WW2 and WW3, associated with the observed cumulative rainfall.

According to a Student's *t*-test ($P = 0.05$), there was no significant difference between average *E. coli* concentration during DW and rainfall events (WW). However, according to a Fisher test ($P = 0.05$), there was a significant difference between the variance of the DW monitoring sessions (variance = 0.11 for rainfall = 0 mm) and that of the three other monitoring sessions performed during rainfall events, WW3, WW2 and WW1 (variance = 0.60, 0.61 and 0.31 for rainfall = 3.0, 5.4 and 23.8 mm, respectively). Moreover, there was no significant difference between the variances of these three WW sessions. Therefore, the subdaily variation of *E. coli* concentration on the SA catchment was different between periods during dry days (SD = $\pm 0.33 \log_{10}$ *E. coli*/100 mL for rainfall <3 mm/day) and periods during rainfall events (average SD = $\pm 0.70 \log_{10}$ *E. coli*/100 mL for rainfall $\geq$3 mm/day). Results of monitoring sessions showed that the threshold to observing an impact of rainfall on variation in river contamination ranged from 0 to 3 mm rainfall/24 h for this small rural catchment. In the model, we ran simulations at a daily time step and these dry and wet SD were introduced into SWAT calculations to take into account the subdaily variation of *E. coli* concentrations in the river.

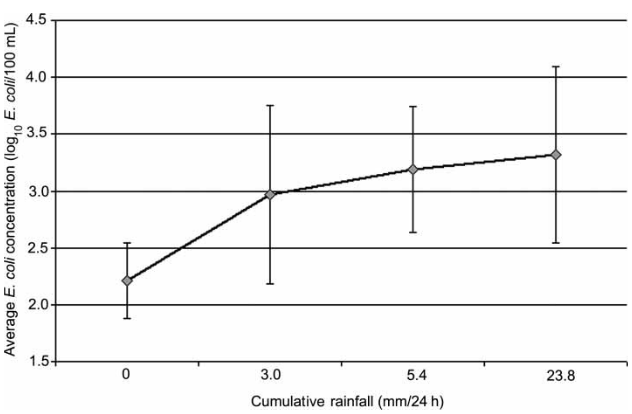

Figure 4 | Relation between average *E. coli* concentrations (± SD) and cumulative rainfall for 24-h monitoring sessions.

Modeling

Calibration and validation of SWAT for river flow at point 1

Hydrological calibrations were made from 25 February 2009 to 30 June 2009, using autocalibrations. The results of these calibrations indicate a good reproduction of daily river flow (Figure 5) according to efficiency criteria ($r^2 = 0.86$, Ens = 0.83). Several river flow peaks were not accurately reproduced by the model, but the calibration was done on only 4 months of continuously measured data and the greater the flow data set, the more the calibration was efficient with SWAT. This calibration would improve with longer measurements on the river in the SA catchment.

The validation of the model took place from 17 April 2008 to 24 February 2009, using the same calibration parameters. The study used this period to validate the model because, owing to the limited database, it was more efficient to use continuously measured data for calibration. The validation was acceptable according to our efficiency criteria results ($r^2 = 0.63$, Ens = 0.62) (Pohlert *et al.* 2005; Michaud *et al.* 2006).

Calibration of SWAT for *E. coli* concentrations in river at point 1

There are a number of different methods for assessing model performance; this study chose the comparison of the frequency curves of simulated and observed concentrations as proposed by Parajuli (2007). This method overcomes problems posed by the uncertainty of factors and mechanisms implicated in biological models (Baffaut & Benson 2009; Bougeard *et al.* 2010, 2011). The calibration of *E. coli* simulation with SWAT was realized at point 1 with observed data (2008–2009, $n = 94$). Table 4 presents the results of frequency curve analysis for observed and simulated concentrations.

Concentrations less than $2.3 \log_{10}$ *E. coli*/100 mL and more than $3.7 \log_{10}$ *E. coli*/100 mL reproduced well, but there was an underestimation of *E. coli* concentrations ranging between 2.7 and $3.3 \log_{10}$ *E. coli*/100 mL. It is interesting to underline the good evaluation of the peaks by the model, because these peaks proved to be the main factor when considering microbial risk assessment. The overall performance of the model was found to be reasonable as

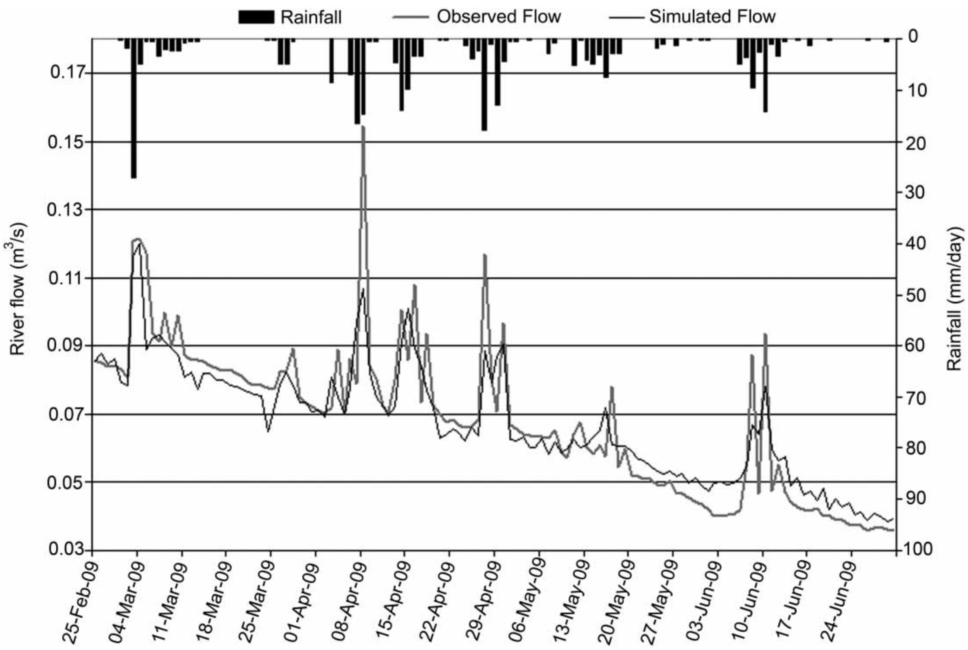

Figure 5 | Calibration of river flow at point 1 from 25 February 2009 to 30 June 2009.

the correlation between simulated and observed concentration frequency was good ($r^2 = 0.87$).

Combining modeling and monitoring

The performance of SWAT was assessed using the entire data set of measured *E. coli* concentrations at point 1. For this purpose, subdaily DW and WW variations of *E. coli* concentration in the river were used to compare with discrete measured concentrations.

Figure 6 shows the performance results of the model at point 1 from April 2008 to June 2009. The simulation took into account fecal contamination sources and the subdaily variations calculated with the DW and WW deviations.

Simulated *E. coli* concentrations ranged from 1.9 to 5.2 $\log_{10}$ *E. coli*/100 mL and observed concentrations from 1.6 to 6.2 $\log_{10}$ *E. coli*/100 mL. The majority of observed concentrations (54.3%) were within the variation interval (±dry/wet deviation), thus confirming that the performance of the model is better when additional information is used.

Even though the model produced quite reliable results, subdaily variations did not explain 45.7% of the observed concentrations. McCarthy (2008) found a result of 33% for storage and analytical uncertainty for *E. coli* measurements in a project modeling urban storm water, without taking into account the subdaily variation and the difference between a grab sample and a daily average concentration simulated by the model.

Three observed contamination peaks broke away from the simulation with *E. coli* concentrations above 4 $\log_{10}$ cfu/100 mL (Figure 6). For example, on 29 July 2008, a major contamination was observed in the river: 4.1 $\log_{10}$ *E. coli*/100 mL. This value did not correspond to a rainfall event (rainfall = 0 mm). When we tried to reproduce this concentration by SWAT, it corresponded to a point source equivalent of 11.7 $\log_{10}$ *E. coli*/day (0.5 m^3/day and 8.0 $\log_{10}$ *E. coli*/100 mL). This could be due to unknown factors such as uncontrolled sewage or manure discharge, a defective septic system discharge or an outflow of a pumping station. In terms of quantity, this source would be

Table 4 | Observed and simulated *E. coli* concentrations frequency during the simulation period

Concentrations	>**2**	>**2.3**	>**2.7**	>**3.0**	>**3.3**	>**3.7**	>**4.0**
Observed frequency	83	63	38	23	15	6	3
Simulated frequency	99	54	8	5	4	2	2

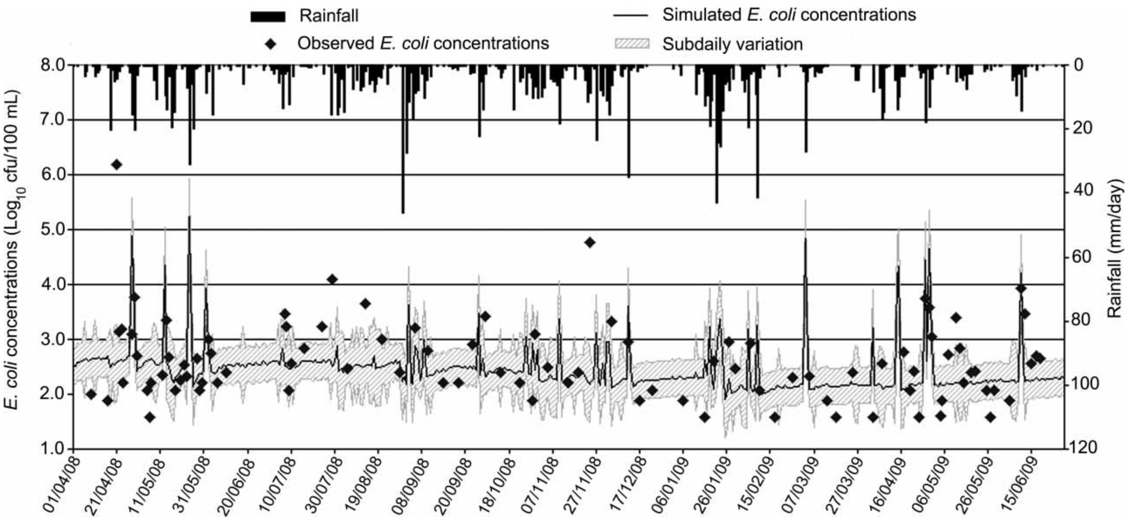

Figure 6 | Observed and simulated *E. coli* concentrations at point 1 from 1 June 2008 to 20 November 2008, subdaily variation ($\pm 0.67 \log_{10}$ *E. coli*/100 mL for rainfall > 3 mm, $\pm 0.33 \log_{10}$ *E. coli*/100 mL for rainfall <3 mm) and daily rainfall.

comparable to a raw sewage discharge of about 1,000 inhabitant-equivalents, a capacity that is comparable with those living in the upper region of this rural catchment.

The integration of additional data clearly improves the overall performance of predictive models (Kelsey *et al.* 2010). The lack of knowledge of fecal contamination sources is one of the major limits for modeling projects. In order to overcome the problem, it is necessary to have a good knowledge of study catchment characteristics concerning hydrological processes, time of concentration, climatic conditions and contamination events.

Some authors have suggested that using a modeling approach in conjunction with laboratory experiments and field observations, can improve understanding of fate and transport of fecal indicator bacteria in water bodies (Cho *et al.* 2010). In our study, the 24-h monitoring at point 1 showed that rainfall runoff substantially increased *E. coli* concentrations in a very short time (2 or 3 h), potentially threatening bathing water quality during WW conditions. Moreover, the subdaily variation of *E. coli* concentration in the river of this small rural catchment was clearly different between dry (rainfall <3 mm/day) and WW conditions (rainfall >3 mm/day). Thus, it is possible to distinguish a difference between simulated concentrations and measurements due to sampling time or uncertainties in analytical techniques, with a suspicious difference due to an unknown isolated discharge into the river.

Hourly modeling can be useful in a small watershed because of the short lag time between a rainfall event and the response of the river in terms of contamination. Nevertheless, the difficulties of obtaining real hourly information on contamination sources hamper this approach. Micovic & Quick (2009) investigated the effect of model complexity on temporal resolution by experimenting with different simulation periods and computational time steps. They indicated that daily time scale calibration forms the basis of an hourly model with a short-term response factor. They demonstrated that, on a daily time scale, a simple model gives good results. Therefore, the hourly simulation can benefit from more complex modeling (especially the simulation of high-intensity rainfall events), but high-quality input data is also required.

CONCLUSIONS

The present study is a full approach combining monitoring and modeling of *E. coli* concentration in a recreational area. The paper focuses on the possibility of introducing the hourly variations observed *in situ* into a daily time scale model. It was applied to a small rural catchment with a lag time response shorter than the model's computational time step. The SWAT model, running on a daily time step, was used to reproduce river flow and *E. coli* concentration in a river with point and non-point agricultural and urban sources. In parallel, the intense field monitoring gave a better understanding of processes in the catchment.

The findings of this study are relevant at the European scale and concern the recent Water Framework Directive (CEU 2000) and the Bathing Waters Directive (Directive 2006/7/EC, CEU 2006). To meet the expectations of EU Directives in terms of water quality, a better understanding of loads occurring over a short lag time is need. Greater confidence in our models is also essential to take into account the important development of activities in coastal areas. Investigation of microbial quality of rivers, lakes and estuaries has become more and more common recently. Coupled with the possibility afforded by modeling, this will aid understanding of the wider impact of activities on river contamination. In this context, in small watersheds, a shorter model time step (i.e. hourly) will become available.

The main issues arising from the present study are as follows:

- The fecal washing from stream sediments represents a weak source of water contamination in the catchment, and the main source of contamination of the river was the wash-in of fecal matter from deposits on land with overland flow. This observation contrasts with other studies (Jamieson *et al.* 2005; Davies-Colley *et al.* 2008) and underlines the necessity of collecting additional information before starting any study, in order to have the opportunity to choose the best modeling approach.
- The results underscore the high variation in fecal contamination due to rainfall events, and the importance of first flush on water quality. This is consistent with results obtained in other regions (Nagels *et al.* 2002; Jamieson *et al.* 2005; Davies-Colley *et al.* 2008; Cho *et al.* 2010). The contamination occurs in a few hours after the rain, most of the time within less than 2–3 h. This field information provides a guide for choosing the time step of the model and confirms the need for an hourly approach.
- The introduction of subdaily variations is a way of improving modeling. We demonstrated that the use of a daily step model on a small catchment, combined with adequate monitoring, offers the best approach for modeling water contamination in a river.

To conclude, our approach is a good compromise for achieving the bathing directive goals. Daily modeling is less time consuming than hourly modeling and can be done without the hour-by-hour definition of fecal contamination sources. As Cho & Olivera (2009) observed, there is a threshold beyond which increased model complexity does not lead to better model results, and makes a model computationally more extensive.

Nevertheless, although the model produces promising results and the selected approach has allowed us to progress, further investigations are necessary to test, verify and improve this method. Benham *et al.* (2006) emphasize that fecal bacteria simulation using water quality models needs more research to improve source characterization of both animal (behavior patterns, habitat and population density, plus accurate estimations of bacteria types produced and their variability) and human sources (reliable surveys of septic/sewage locations and bacterial production of different populations). In this study, in order to improve our model, an extensive database would be helpful to better calibrate and validate the model over a longer period. Concerning the sources, more precise information on the calendar of manure spreading, the *E. coli* concentration in manure and the rate of feces deposited by animals with direct access to the river would be of great interest. Parajuli (2007) demonstrated that bacteria concentration in manure, for example, has a direct relationship with bacteria concentration in water and bacteria prediction, except at low input values. Furthermore, a survey of sewer system overflows would also be helpful if we want to run the model in real time in the near future and thus use it as a 'warning tool'.

A second step of further work would include sensitivity testing concerning, for example, hydrological parameters, bacterial partition coefficient, bacterial die-off, etc. Concerning model-parameter sensitivity, Parajuli (2007) analyzed the relative sensitivity of about one hundred model runs; the results showed varied sensitivity of each model run for different parameters used in the study. Four model parameters and one input parameter were tested: the bacteria partition coefficient in surface runoff; the temperature adjustment factor; the less persistent bacteria die-off factor in solution; the less persistent bacteria die-off factor for sorbed bacteria; and the fecal coliform bacteria concentration in manure. This author demonstrated that, in the studied catchment, the role of the partition coefficient on sensitivity is significantly important compared with the other factors. Moreover, uncertainty source analysis could also be introduced into the sensitivity study to provide guidance for further reductions in pollutant loading. Novotny (2003) underlines that the variations of *E. coli* concentration, including bacterial estimation errors, were the highest uncertainty in modeling. Once validated, the model could be further coupled with a hydrodynamic model and applied to provide predictive information for effective public health measures, to improve the management of fecal loads arriving in the bathing area and to minimize health risks.

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