Modelling of E. coli distribution in coastal areas subjected to combined sewer overflows

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ABSTRACT

Rivers, lakes and the sea were the natural receiver of raw urban waste and storm waters for a long time in the human history but the scarce sustainability of such practice, the increase of population and a renewed environmental preservation approach increased researcher interest in the analysis and mitigation of the impact of urban waters on receiving water bodies. In Europe, modelling has been promoted as a promising approach for implementing the Water Framework Directive (WFD). A particular interest is given to the fate of Pathogens, and especially of E. coli, in all the cases in which an interaction between population and the RWB is foreseen. The present paper aims to propose an integrated water quality model involving the analysis of several sewer systems discharging their polluting loads near the coast in a sensitive marine environment. From a modelling point of view, the proposed application integrated 1D sewer models with a complex 3D model analysing the propagation in space and time of E. coli in the coastal marine area. The integrated approach was tested in a real case study (the Acicastello bay in Italy) where data were available both for SS model and for Receiving Water Body (RWB) propagation model calibration. The analysis shows a good agreement between the model and monitored data. The 3D RWB model is a valuable tool for investigating the pollutant propagation and to highlight the most impacted areas.

KEYWORDS

Integrated urban drainage modelling, receiving water bodies, E.coli

1   INTRODUCTION

The complexity of the impact of the distribution of pollutants in the marine environment, with reference to coastal regions, requires the use of integrated models able to analyse sewer systems (SS), waste water treatment plants (WWTP) and receiving water bodies (RWB). Integrated urban drainage modelling is often defined as the linkage between at least two of the above mentioned physical systems: SS, WWTP and RWB (Harremoës, 2002). Integrated modelling was made possible by the recent developments in modelling approaches and computer power. An increased interest of scientific

research may be found looking at the importance of such theme in literature (see among other Freni et al., 2010; Fu et al., 2010; and literature therein reported). In Europe, integrated modelling has been promoted as a promising approach for implementing the Water Framework Directive (WFD), and has been applied to many catchments across Member States in order to achieve a good quality status in surface waters by 2015. Scientific literature was mainly focused on receiving rivers both because their higher sensitivity to pollution, the risk of pollution accumulation due to the repeated spills and, secondarily, because the analysis of 2D and 3D natural river bodies is still complex and time-consuming (Candela et al., 2009). The analysis of tri-dimensional receiving water bodies has a relevant importance especially looking environmental preservation of marine coastal areas and the touristic uses of beaches. Particular importance is connected with the analysis of pathogens propagation because their concentration in coastal waters may affect their bathable destination. Faecal coliform bacteria are widely used as indicator organisms to signal the possible presence of faeces and pathogen organisms (Glasoe and Christy 2004). However, there is evidence that faecal coliforms do not reliably predict the occurrence and survival of enteric viruses (Noble and Fuhrman 2001). Total coliforms (TC), faecal coliforms (FC), E. coli and faecal streptococci (FS or enterococci) are used as bacterial indicators for water quality monitoring and health assessment as each group of bacteria is prevalent in the intestines and faeces of warm-blooded mammals, including wildlife, livestock and humans. These indicators are not pathogen; FC, TC and FS are used because they are much easier and less costly to detect and enumerate than the pathogens themselves (Meays et al. 2004). E. coli is recommended as an indicator of faecal contamination in fresh waters (Donnison and Ross 1999); this parameter is proposed as there is a high probable correlation between its occurrence and that of a pathogen. Sinton et al. (1993) suggest the FS as a faecal pollution indicator. E. coli and enterococci are strongly suggested for the assessment of faecal contamination from sewage treatment facilities and for diffuse pollution. The European Directive 2006/7/EC aims to establish more reliable microbiological indicators and a new approach based on an integrated management of water quality. The two faecal indicator parameters retained in the Directive 2006/7/EC are intestinal enterococci (IE) and E. coli, providing the best available match between faecal pollution and health impacts in recreational waters. The choice of the microbiological parameters and values was based on available scientific evidence provided by epidemiological studies; the two parameters are representative of the most frequently reported episodes of contamination and they are correlated with health problems. Although monitoring programs increase the knowledge about the level of microorganisms and other constituents in urban stormwater, the collected data are often used without any consideration of the large uncertainties in such environmental measurements and modelling can play a relevant role to evaluate such uncertainty and for extending the knowledge provided by available data to an area wider than the monitored one (McCarthy et al., 2006; Bertrand-Krajewski, 2007). The objective of the research was the development of an integrated model linking urban drainage models and a detailed 3D RWB model. This tool, once calibrated, can be useful to investigate the propagation of pollutants and the identification of most impacted areas of the RWB. The tool can be also used, coupled with uncertainty analysis, to investigate model reliability and to guide monitoring campaigns. The paper is focused on model description and calibration leaving the possible applications to further studies. In the adopted approach, the urban catchments and the sewer systems were simulated by means of SWMM (Storm Water Management Model) that is a largely adopted open-source software able to simulated both water quantity and quality aspects in dry and wet weather. The marine environment was modeled by means of the fully 3D numerical model PANORMUS (PArallel Numerical Open-souRce Model for Unsteady flow Simulation) (see Napoli, 2011). The integrated approach was tested in a real case study (the Acicastello bay in Italy) where data were available both for SS model and for RWB propagation model calibration. In that case a monitoring campaign was carried out in summer 2004 involving discharge and pathogens loads monitoring in the urban catchments and wind, currents and polluting concentrations in the marine environment.

2   THE NUMERICAL MODEL

The model was obtained by the integration of two parts developed by means of open-source software: the catchment and sewer model developed by means of SWMM (Huber and Dickinson, 1988) and the RWB model developed by means of PANORMUS (De Marchis and Napoli, 2008). The first model was used for simulating the urban areas, the urban drainage system and the small ephemeral rivers that receive urban waters and deliver them to the sea. The latter model was used for simulating the hydrodynamics in the coastal area and the diffusion of pollutants depending on the different driving forces. The two modules can be run separately and they are interconnected by output-input file exchange: the modelling outputs, derived as time series describing water quantity and quality variable at the urban water system outfall, are transferred to the RWB model as input. This one-way (upstream to downstream) approach is very simple but it is justifiable by the negligible back-propagation influence of RWB variables on urban water system ones. The approach allows for the separated run of the two models, being characterised by very different computational demands, and the separation of modelling uncertainty analysis that can be run separately on the two systems considering the urban water system output uncertainty as one of the input uncertainties in the RWB model. In the specific application, the uncertainty analysis was not run because the paper was focused on the model set up and integration but this aspect will be addressed in the future considering the large modelling uncertainty in urban water quality assessment demonstrated in literature (see Freni et al., 2010 and the literature herein cited).

2.1   Catchment and sewer model

The EPA SWMM (5.018 version) was used to simulate the urban drainage network and the small natural catchments that are connected to the sewer outfalls; this model allows the user to select different mathematical models to describe the runoff formation and propagation in sewer systems (Huber and Dickinson, 1988). A distributed "non-linear reservoir" was adopted to simulate the surface runoff, taking into account the surface storage, as an initial hydrological loss, and the infiltration phenomena using the Horton equation. Rainfall – runoff routing was solved by coupling the continuity equation with the Manning equation, thereby obtaining a nonlinear reservoir scheme. This approach is used separately on the pervious and impervious parts of the system, with the only difference that infiltration is not applied to the impervious part of the catchment. The runoff delivered by the two parts of the catchment is routed to the drainage system. The complete 1D De Saint Venant equations were used to simulate the propagation of pollutants into the SS by adopting an iterative explicit mathematical solver. A water quality module was used to simulate the build-up and wash-off of pollutants from the catchment surfaces, where the build-up was simulated by an exponential function (Alley and Smith, 1981) and the solid wash-off, caused by overland flow during a storm event, was simulated using the formulas proposed by Jewell and Adrian (1978). The widely applied Alley-Smith (1981) model (eq. 1) was used in the present paper, in which Ma is an inverse exponential function of ADWP. The model introduces the effect of external causes (such as wind or traffic) that can modify pollutant accumulation by moving part of the accumulated pollutants outside of the catchment area. The symbols have the same meanings as described above. This approach has two calibration parameters, Accu, the unit accumulation rate, and Disp (d^{-1}), the dispersion parameter, which take into account the causes that reduce the pollutant mass availability on the catchment:

$$Ma = Mr \cdot e^{(-Disp \cdot ADWP)} + \left(\frac{Accu}{Disp}\right) \cdot A \cdot IMP \cdot \left(1 - e^{(-Disp \cdot ADWP)}\right) \tag{1}$$

The original formulation by Jewel and Adrian (1978) was used for wash-off analysis, in which Me (kg) is the mass entering the network between t and t+Δt (h), M_a is the mass on the catchment at time t, P_n is the net rainfall intensity (mm/h) and Δt is the time step. The formulation requires the

calibration of two parameters, the wash-off factor, Wh (-), and the wash-off coefficient, Arra (mm$^{-Wh}\cdot$h$^{(Wh-1)}$):

$$Me = Ma \cdot (1 - e^{-(Arra \cdot P_n^{Wh} \cdot \Delta t)}) \qquad (2)$$

E.Coli die-off in the urban water systems was neglected because the runoff travel time in the system is around 60 minutes and the percentage of E.Coli concentration reduction should be in the range of 1% - 3% according to literature values (McCarthy and al., 2008).

2.2 RWB model

The marine environment was modelled by means of the fully 3D numerical model PANORMUS (PArallel Numerical Open-souRce Model for Unsteady flow Simulation). The PANORMUS is an in-house finite-volume model and is able to solve the Reynolds-averaged momentum and mass balance differential equations on a curvilinear structured grid. The Reynolds stresses are modelled according to the standard k-ε turbulence model (De Marchis and Napoli, 2008). The incompressible Reynolds Averaged Navier-Stokes and continuity equations, in the summation convention formulation, read:

$$\frac{\partial u_i}{\partial t} + u_j \frac{\partial u}{\partial x_j} - \nu \frac{\partial^2 u_i}{\partial x_j \partial x_j} + \frac{1}{\rho}\frac{\partial \tau_{ij}}{\partial x_j} + \frac{1}{\rho}\frac{\partial p}{\partial x_i} + g\delta_{ij} + f_i = 0 \qquad i, j = 1,..3 \qquad (3)$$

$$\frac{\partial u_i}{\partial x_i} = 0 \qquad i = 1,..3 \qquad (4)$$

where t is the time, x_i the i-th axis (with the east-west, north-south and vertical directions aligned with the axes x_1, x_2 and x_3, respectively), u_i the i-th component of the Reynolds averaged velocity, ρ the water density, p the Reynolds averaged pressure, g the gravity acceleration, n the kinematic viscosity, δ_{ij} the Kronecker delta, f_i the i-th component of the Coriolis acceleration and τ_{ij} the Reynolds stresses.

The pressure p can be decomposed into an hydrostatic and a non-hydrostatic pressure part, which is independent of the vertical coordinate:

$$p = \gamma[(z_B + h) - x_3] + \rho q \qquad (5)$$

where z_B is the bed elevation from an horizontal plane of reference, h is the depth of the water column and q is the non-hydrostatic pressure. Introducing equation (5) into equation (3), the Reynolds Averaged Navier-Stokes equations can be rewritten as:

$$\frac{\partial u_i}{\partial t} + u_j \frac{\partial u_i}{\partial x_j} - \nu \frac{\partial^2 u_i}{\partial x_j \partial x_j} + \frac{1}{\rho}\frac{\partial \tau_{ij}}{\partial x_j} + \frac{\partial q}{\partial x_i} + g\frac{\partial(z_B + h)}{\partial x_i} + f_i = 0 \qquad i, j = 1,..3 \qquad (6)$$

where the last term is null for i = 3, since the independence of z_B and h from x_3.

As specified above, the turbulent stresses τ_{ij} are calculated using the $k - \varepsilon$ turbulence model in the 'standard' formulation (*Launder* and *Spalding* (1974)), through the calculation of the isotropic eddy viscosity $\mathbf{v}_t$ is obtained as $\mathbf{v}_t = c_\mu \frac{k^2}{\varepsilon}$, *with* c_μ is a closure parameter, k is the turbulent kinetic energy (TKE) per unit mass and ε the dissipation rate of the TKE, and an horizontal eddy viscosity coefficient $v_{h,t}$, acting on the horizontal terms only, calculated through the Smagorinsky model. In fact, in oceanographic applications, due to the high grid anisotropy, with horizontal spacing much larger than the vertical one, the turbulence closure is achieved by using a non-isotropic eddy viscosity coefficient.

In order to simulate coastal processes the free surface movements are calculated according to the *kinematic boundary condition*:

$$\frac{\partial h}{\partial t} + u_{1,s} \frac{\partial(h + z_B)}{\partial x_1} + u_{2,s} \frac{\partial(h + z_B)}{\partial x_2} - u_{3,s} = 0 \qquad (7)$$

where $u_{1,s}$, $u_{2,s}$ and $u_{3,s}$ are the velocities at the free surface along the x_1, x_2 and x_3 directions, respectively.

The code is second-order accurate both in time and space. For the time advancement of the solution the vertical diffusive and turbulent terms are discretized using the Crank-Nicolson implicit method, while the Adams-Bashfort explicit scheme is employed for the remaining terms. The pressure-velocity decoupling, typical of incompressible fluids, is overcome using a fractional-step method. The momentum equations are first solved assuming hydrostatic pressure distribution (*predictor-step*). A Poisson-like equation is then solved to obtain an irrotational velocity field, to be added to the predictor-step to obtain the divergence-free velocity field (*corrector-step*). The numerical code is extensively validated against laboratory results (Napoli et al, 2008), analytical solutions and experimental observations in natural water bodies, confirming that the model is able to reproduce the hydrodynamic mixing in environmental and industrial applications. Further details on the hydrodynamic model can be found De Marchis et al (2012). In order to take into account for the pollutant dispersion in the coastal regions the numerical model resolves, jointly with the system of conservation equations for momentum, the conservation equations for any pollutant present in the water body, which are conveyed and diffused by the currents and by turbulence effects according to:

$$\frac{\partial C}{\partial t} + \frac{\partial\, Cu_i}{\partial x_i} - \alpha\, \frac{\partial^2 C}{\partial x_i \partial x_i} + \frac{\partial\, \Lambda_j}{\partial x_j} - F_c = Q \tag{8}$$

where C is the tracer concentration, u_i are the *i*-th component of the averaged velocity calculated according to the Reynolds averaged Navier Stokes equations (6), α the molecular diffusivity, Λ_j the turbulent diffusive flux modelled as $\Lambda_j = -\, \Gamma \frac{\partial C}{\partial x_j}$, with Γ the turbulent diffusivity obtained as the ratio between the eddy viscosity ν_t and the turbulent Schmidt number. F_c is the horizontal subgrid-scale diffusion term and Q represents the source term or sink of the concentration and takes into account for the bacteria decay time with a first order kinetic law. Following Scroccaro et al. (2010), a cautious decay time (e.folding time) of 1 day was used as the decay parameter. Details on the numerical discretization can be found in Napoli (2011). The computational modelling of pollution dispersion in coastal area, using a 3D Computation Fluid Dynamics (CFD) model, is a key issue in order to preserve the environment of the coastal area.

In order to take into account for a real hydrodynamic circulation in the coastal area the proper conditions have to be assigned at the boundaries. The forces driving the circulation, both variable tidal levels and wind velocity, have been considered as external forces driving the current. The wind speed produces a wind shear stress over the free surface, partially responsible of the hydrodynamic circulation, which is calculated according to:

$$\tau_s = \rho_{air}\, u^{*2} \tag{9}$$

where ρ_{air} is the air density and u^{*2} is the air friction velocity at the free-surface. This friction velocity can be related to the wind speed W_{10} at the standard height of 10 m and to the wind drag coefficient C_{10} through the equation:

$$C_{10} = \frac{u^{*2}}{W_{10}^2} \tag{10}$$

The wind drag coefficient is calculated by the widely employed Wu's formula (Wu, (1982)):

$$C_{10} = (0.8+0.65W_{10}) \cdot 10^{-3}. \tag{11}$$

The wind speed and direction, imposed in the hydrodynamic numerical simulation of the RWB, were registered by the Italian wavemeter and mareoghraphic network. The period corresponding with the

summer period of July-August 2004, when a campaign of velocity measurement has been done by an off shore monitoring station (see figure 1). At the open boundaries, the tidal levels, registered by the Italian wavemeter and mareoghraphic network, has been imposed. At each time step, the free surface elevation, in correspondence of the open boundaries, is updated according to the variation of the registered tidal current. Due to specific calm conditions registered by the Italian wavemeter and mareoghraphic network, in the period here considered, at the external boundaries the possible open sea currents where neglected. Following the recent findings of Roman et al. (2010), when a further current from the open sea is required, on open boundary an Orlanski condition is used. Usually this condition is used adding a buffer layer between the effective grid and the open boundary. The logarithmic wall-law, with a von Karman constant equal to 0,41 and the intercepta B equal to 5, has been used on the solid boundaries (land and island contours and lagoon bottom).

In order to reproduce the RWB, in the numerical simulations the computational domain has been discretized using 64 x 128 x 16 cells in the east-west, north-south and vertical directions, respectively. In the vertical direction a non-uniform grid has been used, with a refinement close to the bottom and the free-surface. Due to the geometrical complexity of the coastal areas and the presence of small islands, a curvilinear boundary-fitted computational grid was used. The coast-line was thus regularized, using a step-like pattern. More accurate description of the coast-line could have been obtained using the Immersed Boundary Method, whose application to curvilinear grids was recently proposed by Roman et al. (2009). Nevertheless the increase in the computational effort required by the use of these methods would not have been justified for the present analysis. The grid size has been chosen by considering the accuracy of result and computing time. In order to this two preliminary simulations were carried out. In the former the domain was discretised using 128 x 256 x 32 cells in the east-west, north-south and vertical directions, respectively, while in latter a coarser resolution was considered discretising the domain with 64 x 128 x 16 cells. The comparison between the velocity fields, not reported here, showed that the two resolutions give rise to the really similar results, thus conforming that the lowest resolution is suitable to provide an acceptable solution accuracy, with a reduction of time consuming.

3   DESCRIPTION OF THE CASE STUDY

The integrated model was applied to a real case study: Acicastello urban area near Catania (Italy). The area is characterized by a relevant coastal urbanization mainly aimed to touristic activities that was not supported by the construction of a organized urban drainage and wastewater treatment. At the moment, the urban area is served by 6 different combined drainage systems collecting both wastewater and stormwater having their outfalls directly in the sea or in small ephemeral rivers near the coast (Figure 1). Due to urbanization, small natural rivers were included in the sewer network becoming a part of drainage system and delivering natural runoff water into the sewer pipes. Figure 1 shows the main sewer pipes in each area and the connected natural river reaches. The whole are is about 633 ha and it can be easily divided in two parts: the coastal area that is totally urbanized with few small pervious areas (less than 1% of the total urban area) and an upstream part that is more pervious with runoff coefficients ranging between 0,3 and 0,5. The main characteristics of the catchments are reported in Table 1 and they are widely described in De Marchis and Freni (2011).

The Acicastello bay is characterized by rocky steep bottom with water depths higher than 80 m at only 150 m from the shore. For this reason, the area is characterized by complex three-dimensional circulation. The specific geometric configuration of the coastal region analysed and the dominant effect of the wind stress over the free surface, in fact, develops a principal circulation in the vertical planes, characterized by upwelling and downwelling processes close to the coastline and at the open sea, respectively.

Table 1: Main characteristics of the analysed catchments

Catchment	Total area [ha]	Pervious area [ha]	Imperious area [ha]	Population [inhab] – year 2004	Nat. catch. average slope [%]	Urban area average slope [%]
1	58.9	24.5	34.4	6134	34.2%	12.2%
2	125.1	95.2	29.9	5867	38.1%	23.6%
3	108.6	81.9	26.7	4322	24.4%	7.2%
4	65.3	23.1	42.2	6543	19.6%	3.6%
5	97.4	60.1	37.3	5545	45.6%	21.3%
6	180.3	136.2	44.1	7143	65%	35%

The vertical mixing induced by the domain complexity, as will be better analysed in the following, suggests the need to use fully three-dimensional non hydrostatic models. The area was subjected to a large monitoring campaign between March 2004 and October 2004 during which both drainage system outfalls and receiving waters where monitored. The monitoring campaign had the following characteristics:

- Rainfall was monitored by Riposto rain-gauge (Figure 1) with a temporal resolution of 5 minutes;
- Sewer flow was monitored by means of ultrasonic probes at the six outfalls with hourly temporal resolution during dry weather periods and with 5 minutes temporal resolution during rainfall events;
- Urban drainage water quality was monitored by means of 24-bottles automatic samplers connected to the rainguage and having a sampling time step of 15 minutes; sampling started after 2 mm rainfall volume was registered in order to collect urban areas first flush. A dry weather monitoring campaign was also carried out with a sampling time step of 1 hour. Monitored water quality included: total suspended solids (TSS), Biological Oxygen Demand (BOD), Chemical Oxygen Demand (COD), ammonia nitrogen (N-NH$_4$), total Kjeldahl nitrogen (TKN), phosphorus (P), Total Coliforms and Faecal Coliforms.
- Wind, water level and wave height were monitored off shore of the analysed coastal area with a temporal resolution of 1 hour. These data were used as boundary conditions for the RWB hydrodynamic model. Water quality was monitored in four points (named with letters between A and D) by means of manual hourly sampling that was used for verifying the model results (Figure 1).

During the period March and October 2004, only four events were monitored with sufficient quality to be usable for model calibration.

4   MODEL CALIBRATION AND ANALYSIS OF RESULTS

Each sub-catchment was fully characterized by seven calibrated parameters. The water quality module was defined by four parameters for each type of land use, while the propagation through the sewer was fully defined by the pipe roughness, which could be defined for each pipe. SWMM is a distributed model and the parameters were calibrated in a lumped way by assuming that the parameters listed above were constant in each urban area. This simplification depends on the availability of a single measuring station for each urban catchment. Each urban area was characterized by seven hydrologic parameters, one hydraulic parameter for the drainage system and four parameters for the water quality (Table 1). These parameters were initially subjected to sensitivity analysis, where the ranges of the parameters were determined using the literature (Huber and Dickinson, 1988). The model proved to be insensitive to infiltration parameters and such parameters were not calibrated. The sensitive parameters were calibrated using the available dataset and maximising the Nash – Sutcliffe criterion (Nash and Sutcliffe, 1970). The water quantity modules were calibrated first and then fixed as the

water quality modules were calibrated (Table 2). No validation steps were performed due to the small available dataset. In all of the SWMM simulations, 1 minute time steps were used for the runoff generation and water quality processes on the catchment surface, while the flow propagation in the sewer system was analysed with a time step of 5 seconds to account for the faster temporal variability of dynamic processes in a sewer during flood propagation.

Figure 1: The analysed area: the station O collects hydrodynamic data and water quality samples; stations A, B, C and D collect water quality samples only.

Calibration of SWMM model, for all the analysed events and for all the six outfalls, provided good results providing N-S efficiencies ranging between 0.65 and 0.73 for outfall discharges and ranging between 0.57 and 0.68 for E. coli concentrations. Calibrated flows and concentrations were transferred as input of the RWB model. E. coli concentrations in the monitoring stations A, B, C, D and O were used to verify the reliability of the model. Figure 2 shows the agreement between modelled and monitored E. coli concentrations in points O and D for the rainfall events between 2nd and 5th July 2004. This event and those monitored points were shown as examples; similar results were obtained for the whole modelled domain.

The figure shows a good agreement between data and simulations even if the model tends to underestimate peak wet weather concentrations. The figure also shows that there is not a strict correlation between rainfall intensity and E. coli concentrations because of the combination of the sewage water dilution and the different hydrodynamic conditions of the RWB. The system is in fact influenced by the combination of urban runoff and natural catchment runoff. The first is the main responsible of E.Coli discharge while the second is not characterised by high E.Coli concentrations, being originated by non urbanised natural areas. At the opposite, natural catchments produce the most part of runoff volume. During intense rainfalls, urban pollutant loads are diluted by natural flows and highest E.Coli concentrations were monitored during short and small rainfall events in which natural catchments are not saturated and do not provide large contributions to runoff.

The model was used to estimate E. coli concentration fields. Figure 3 shows the contour plot obtained in correspondence of the rainfall event of the 04/07/2004 (Figure 3a) and six hours after the end of the rainfall event (Figure 3b). Figure 3 clearly shows how the effect of the rainfall is a reduction of the concentration. The two plots, anyway, emphasize the heavy state of contamination of the marine environment.

Table 2: SWMM parameters subjected to calibration (against sewer outflow and E. coli concentration): underlined parameters have been fixed to average values because not influential

| Parameter	Sensitivity variation range	Urban Areas – Calibrated values					
		1	2	3	4	5	6
Imp. area roughness (Manning)	0.01 – 0.03	0.020	0.022	0.018	0.013	0.021	0.025
Perv. area roughness (Manning)	0.02 – 0.1	0.033	0.035	0.039	0.038	0.042	0.038
Imp. area surface storage [mm]	0.1 – 2	0.3	0.8	0.4	0.6	1.1	0.9
Perv. area surface storage [mm]	0.5 – 4	2.5	2.1	3.3	3.7	2.3	2.1
Initial infiltration [mm/h]	*40 – 100*	*70*	*70*	*70*	*70*	*70*	*70*
Saturated soil infiltration [mm/h]	*8 – 25*	*15*	*15*	*15*	*15*	*15*	*15*
Depletion constant [d^{-1}]	*0.001 – 0.05*	*0.02*	*0.02*	*0.02*	*0.02*	*0.02*	*0.02*
Sewer pipe roughness (Manning)	0.01- 0.03	0.48	0.018	0.41	0.015	0.41	0.015
Unit accumulation rate [kg/(ha*d)]	0.5 – 10	6.7	7.4	6.4	9.8	9.4	7.1
Dispersion parameter [d^{-1}]	0.05 – 1.2	0.51	0.14	0.67	0.16	0.54	0.38
Wash-off coefficient [mm^{-1}]	0.01 – 2.3	0.73	0.85	0.64	0.62	0.44	0.56
Wash-off factor [-]	0.5 – 2.1	1.23	1.15	0.84	1.23	1.18	1.21
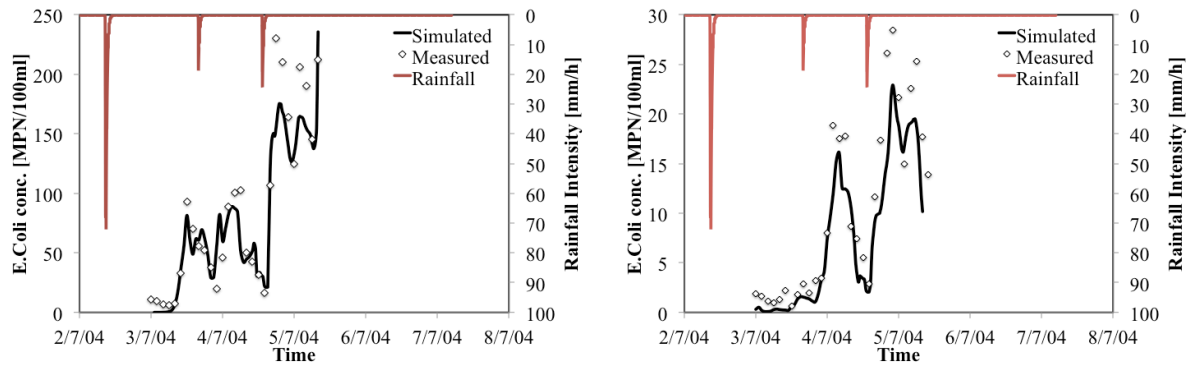
Figure 2: Comparison between monitored and simulated E. coli concentrations in monitoring stations O (on the left) and D (on the right).
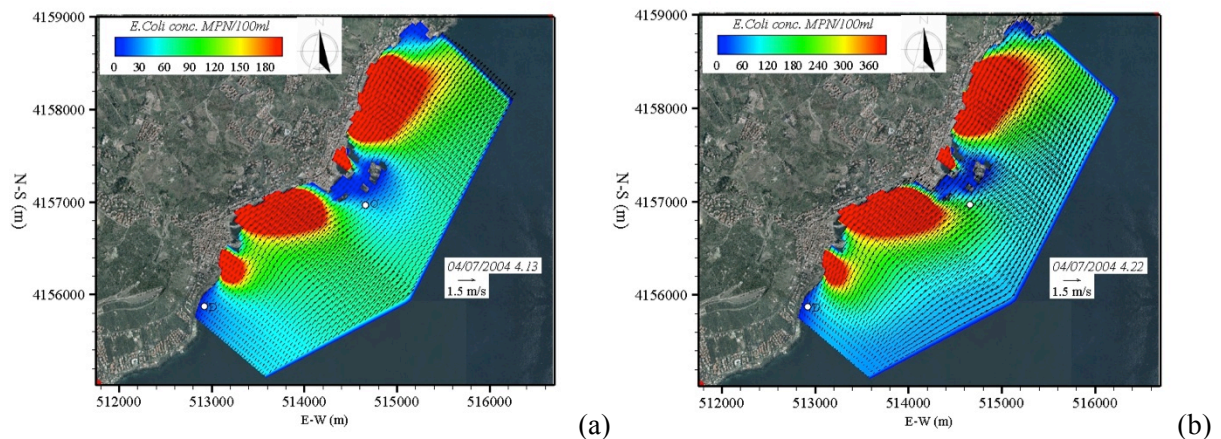

Figure 3: E. coli concentrations in two different periods: (a) during the rainfall event of the 04/07/2004; (b) six hours after the rainfall event. Notice the contour scale of figure (b) is twice (a)
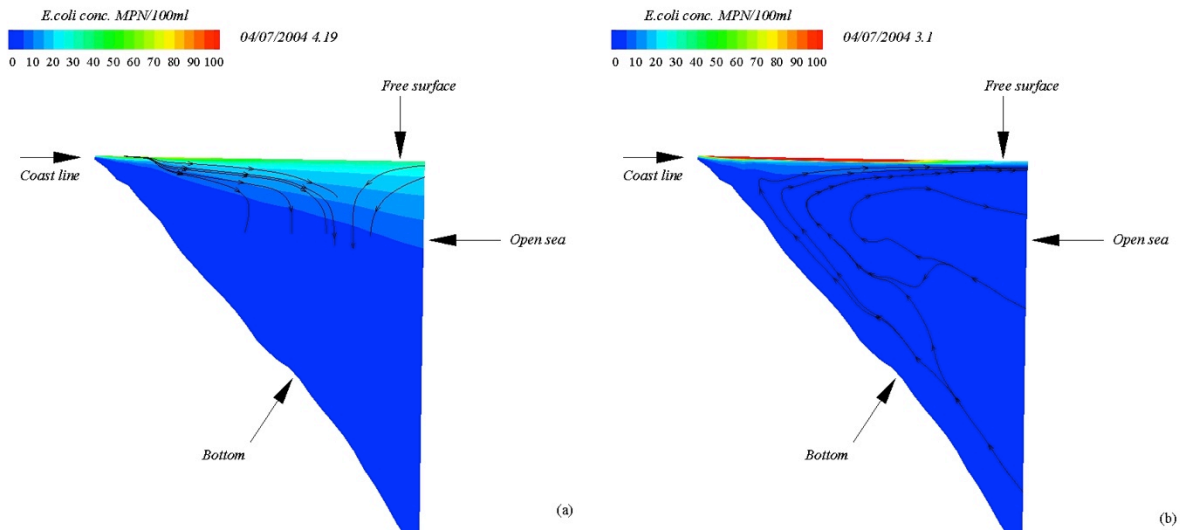
Figure 4: E. coli concentrations in a vertical plane in two different periods. The section here considered is indicated in figure 1. The vectors are indicative of the current direction only.

In order to better analyse the importance to use fully 3D numerical model to reproduce the pollutant dispersion in RWB, showing the mixing of the water column, in figure 4 a plot of the E. coli concentration along in a vertical plane is plotted.

It can be observed that when the current is directed from the free surface toward the bottom layers, a dispersion of the pollutant is achieved, thus increasing the risk to pollute the benthic resources. On the other hand, when the current drives the circulation in the opposite direction, toward the free surface, the E. coli concentration below the free surface is evidently reduced and the pollutant are mainly concentrated in the upper layers. This result confirms the need to use 3D numerical models able to simulate the upwelling and downwelling phenomena in order to improve the knowledge about the possible dispersion of pollutant in coastal water bodies.

5   CONCLUSIONS

The paper presented the application of an integrated water quality model based on open source models. The integrated model has the particular feature of integrating a one-dimensional urban drainage model with a 3D hydrodynamic dispersion model for simulating a coastal area. This study represents a new step towards the description of the microbiological pollution dispersion due to the discharges of the submarine outfalls in the Bay of Acicastello. Thanks to the 3D model PANORMUS, the information coming from the hydrodynamic circulation and the classical methods of analysis to assess possible risks connected to the microbiological parameter E. coli have been linked for future preservation of the environment. From the analysis of the specific case study, some considerations may be drawn. Despite the Coliform mortality rate has been taken into account, the fully 3D hydrodynamic simulations clearly show that the time scale for microbiological parameters' decay could seems to be less sensitive in open water body where the meteo-marine conditions play a fundamental role in the dispersion of the E. coli, thus suggesting the need, also shown by Scroccaro et al. (2010) to further investigate on the time decay parameter in light of environmental parameters such as climate, temperature, and salinity. The numerical results of the RWB model give a more realistic condition of the heavy contamination state of the Bay. In fact, despite the E. coli concentration is slightly reduced in the coast line zone during the rainfall event, the hydrodynamic circulation causes an high level of concentration in the open regions due to the submarine discharges that drive the pollutant dispersion away from the coastline. The application showed that the integrated model can be a precious tool for investigating RWB water quality as pollution concentration is largely dependent on sea conditions providing higher dispersion or the concentration of pollutants on the shore. Up until now, mainly 1D or 2D integrated numerical models

have been developed thus reducing the precision of the model especially in the vertical direction. The 3D model, here considered, represents a preliminary step to improve the environment preservation of the sea water in very complex condition.

Future analysis will be aimed to complete the present study with the integrated areal analysis (Scroccaro et al. 2010) for the definition the bathing water profile required by the directive 2006/7/EC about the management of bathing water quality. Another important aspect that will be addressed in the future is related to the propagation of uncertainty from modelling input to the final RWB concentration distributions.

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