Urban water quality modelling: a parsimonious holistic approach for a complex real case study

Gabriele Freni, Giorgio Mannina and Gaspare Viviani

ABSTRACT

In the past three decades, scientific research has focused on the preservation of water resources, and in particular, on the polluting impact of urban areas on natural water bodies. One approach to this research has involved the development of tools to describe the phenomena that take place on the urban catchment during both wet and dry periods. Research has demonstrated the importance of the integrated analysis of all the transformation phases that characterise the delivery and treatment of urban water pollutants from source to outfall. With this aim, numerous integrated urban drainage models have been developed to analyse the fate of pollution from urban catchments to the final receiving waters, simulating several physical and chemical processes. Such modelling approaches require calibration, and for this reason, researchers have tried to address two opposing needs: the need for reliable representation of complex systems, and the need to employ parsimonious approaches to cope with the usually insufficient, especially for urban sources, water quality data. The present paper discusses the application of a bespoke model to a complex integrated catchment: the Nocella basin (Italy). This system is characterised by two main urban areas served by two wastewater treatment plants, and has a small river as the receiving water body. The paper describes the monitoring approach that was used for model calibration, presents some interesting considerations about the monitoring needs for integrated modelling applications, and provides initial results useful for identifying the most relevant polluting sources.

Key words | urban drainage integrated modelling, water quality management, water quality monitoring, wet weather pollution

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NOTATION

ADWP	Antecedent dry weather period
SS	Sewer System
WWTP	Wastewater treatment plant
RWB	Receiving water body
CSO	Combined sewer overflow
SWT	Storm water tank
TSS	Total suspended solids
BOD	Biochemical oxygen demand
COD	Chemical oxygen demand
TKN	Total Kjeldahl Nitrogen
NH_4-N	Ammonia
P	Phosphorus
DO	Dissolved oxygen

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INTRODUCTION

Integrated urban drainage modelling is defined as the modelling of the interactions of two or more physical systems, which may include a sewer system (SS), wastewater treatment plant (WWTP), and receiving water body (RWB) (Rauch *et al.* 2002). When a catchment-scale analysis is performed, a model can integrate several units;

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this is especially useful when more than one urban area is present. In these cases, the complexity of the integrated model quickly increases, and may involve numerous parameters; therefore, large measurement databases are necessary to calibrate the model, and to effectively describe the complex system. In such cases, the use of parsimonious approaches can be fundamental in providing useful and reliable modelling results (Willems & Berlamont 2002; Freni *et al.* 2008*a*). The use of a holistic approach is also required by the EU Water Framework Directive, which suggests the adoption of integrated analyses for river basin management in order to meet environmental and ecological objectives (Chave 2001).

The idea of the joint analysis of urban drainage systems and related natural systems has been present since the Seventies in both research and technical practice (Beck 1976). The INTERURBA Workshop in 1992 stated that the planning and management of SS and WWTP should be based on their combined impacts on the RWB (Lijklema *et al.* 1993). It was concluded that the criteria for judging the performance of urban drainage systems should be linked to realistic objectives for the sustainable use of the receiving waters. The only way to accomplish this objective is the adoption of numerical models that are able to describe all of the different components of the system, as well as their interactions, in a reliable way.

Although the integrated approach is theoretically effective for both the planning and the management of urban drainage systems, a few shortcomings prevent its effective application. Specifically, the main issues hampering such applications may be summarised as (taken from Butler & Schütze 2005; Vanrolleghem *et al.* 2005; Mannina *et al.* 2006; Matthies *et al.* 2006; Achleitner *et al.* 2007; Dorner *et al.* 2007; Letcher *et al.* 2007; Xu *et al.* 2007; Benedetti *et al.* 2007, 2008; Barthel *et al.* 2008; Freni *et al.* 2008*a*; Freni *et al.* 2009*a,b*):

- The responsibilities regarding the planning and management of sewer systems, WWTP, and water bodies have been split between different authorities in most European countries.
- Models for the different sub systems have been developed independently. Thus, different concepts, model approaches, and state variables are used to describe processes in each of the sub systems, hampering the linkage of models.
- Data requirements increase dramatically with the inclusion of more sub systems. The calibration of an individual model requires many samples, which, in turn, can increase the economical cost of the projects to a prohibitive level. In addition, when using integrated models, it may be necessary to add a second calibration phase to verify the system operation.
- Model complexity in the context of low data availability introduces uncertainties in the modelling process that, sometimes, are not clearly identifiable or assessable.
- Different temporal and spatial scales are relevant for the analysis of the processes. The simulation time scales for sewer, WWTP, and river models can be different and they depend on the simulation goal.
- Uncertainty accumulation: the uncertainty generated in each sub-model is propagated through the integrated models when each downstream model uses the outputs of the upstream model as inputs.

The aforementioned points are common to several environmental modelling applications, and in a general framework of best practice, the modeller should provide an evaluation of the confidence in the model, possibly assessing the uncertainties associated with the modelling process with the results of the model itself (Freni *et al.* 2009*a–c*).

There are approaches that couple existing models, usually via the simulation of different system components using the output of each of them as input for the downstream components. As an example, the Integrated Catchment Simulator (ICS) basically links commercial models, such as MOUSE for drainage systems, STOAT for WWTP simulation, and MIKE11 for rivers (Tomicic *et al.* 1999). One of the most relevant problems regarding the adoption of detailed models involves the significant parameterisation that requires lengthy and expensive data acquisition activities, as well as the costly computational resources necessary for running long-term simulations (Vanrolleghem *et al.* 1999; Willems & Berlamont 2002; Mannina & Viviani 2009*a*). In fact, these models are usually developed as stand-alone detailed representations of the different parts of the integrated system, and

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they have not been really developed for working together as a single modelling approach. In recent years, the development of models or modelling environments specifically aimed to integration became more frequent in literature according to the pragmatic consideration that not all the effects of urban pollution on RWB need to be analysed but only the dominant ones (Achleitner *et al.* 2007). Similar concepts pushed the development of modelling packages aimed to reduce model computational needs in order to allow for real time analysis (Schütze *et al.* 2002). The high number of developed models and modelling applications required standardization efforts that recently led to new modelling environments aimed to define standards for interconnecting models not necessarily developed for integrated analysis (Reußner *et al.* 2009).

The complexity of those models is often excessive in comparison with the simulation needs and data availability, thus a good balance between model detail level and computational effort minimization is difficult to obtain. Such aspects are crucial if compared with common data availability for such applications, and with the efforts needed for data monitoring.

Although connections between different detailed models have been operationally successful, in most cases, problems are still present regarding model connectors that "translate" the state variables of one model to the input variables of the downstream one (Meirlaen *et al.* 2001). On the other hand, problems of uncertainty accumulation as well as model complexity unbalance may lead to very uncertain and sometimes useless results (Willems & Berlamont 2002; Freni *et al.* 2008*b*).

Available data for integrated urban drainage analysis are often piecemeal because contemporary monitoring approaches for the SS, WWTP, and RWB are complex, and they require considerable technical and economic efforts. When monitoring an entire rural basin that contains more than one urban catchment, monitoring efforts should be scaled according to system complexity, which markedly increases the number of instruments needed for a complete system analysis.

The situation is even worse when looking at the wet weather impact on RWB because of the small scale of pollution, which increases the difficulties in providing reliable monitoring data due to the impacts of wet weather flows on WWTP processes and the consequent increase in WWTP emissions (Artina *et al.* 1999).

Several studies in this field have demonstrated that the monitoring of integrated urban drainage systems is complex, and full-scale case studies are limited. More specifically, Bertrand-Krajewski *et al.* (1995) focused on monitoring needs for WWTP analysis during wet weather. Rouleau *et al.* (1997*a*,*b*) highlighted the failure of WWTP processes during rain events, demonstrating that pollution impacts on RWB in wet weather cannot be limited to CSO emissions.

To bypass this kind of problem, Schütze *et al.* (1999) proposed the use of "semi-hypothetical" case studies for analysing integrated modelling potentialities. Different parts of the system are simulated in detail using data from real and well-documented case studies, but without a physical connection between them (see, among other, Harremoës & Rauch 1996; Lau *et al.* 2002). This approach is useful for understating the reliability of the modelling tools, however, it could be misleading and not helpful for understanding the interactions between the different parts of real integrated systems and their particular peculiarities. Furthermore, the use of semi-hypothetical case studies could prevent the use of practical methodology for integrated catchment management, i.e. identifying polluting sources having greater impact on RWB, depending on its self-depuration capacity, or it may prevent the evaluation of specific catchment management practices with respect to RWB quality objectives.

The present paper is aimed at developing a reliable integrated tool for analysing the impact of urban drainage systems on the RWB quality. The model is applied to a complex integrated catchment located in Italy, which has been monitored intensely for several years. The model was used to obtain some interesting information about the contribution of different sub-systems to the quality characteristics of the RWB.

The paper will briefly describe the applied model; it will discuss the monitoring campaign carried out on the Nocella basin (Italy), and the initial results of the application of the integrated model.

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METHODS AND MATERIALS

The integrated model

The system was modelled employing a bespoke integrated model developed during previous studies (Mannina *et al.* 2004; Mannina 2005; Mannina *et al.* 2006; Freni *et al.* 2009*c*). The structure of the adopted model will be briefly described here and the cited literature can be referenced for a more detailed description of the chosen algorithms.

In brief, the model is able to estimate both the interactions between the different systems, i.e. SS, WWTP, Combined Sewer Overflow (CSO) and RWB and the impacts in terms of water quality that urban storm water has on the RWB. The modelling structure can be adapted to the specific application by removing or adding sub-models or parts of them, such as the Storm Water Tank (SWT), or the Combined Sewer Overflow (CSO), as depicted in Figure 1. Such an integrated system is made up mainly of three sub-models:

- the rainfall-runoff and flow propagation sub-model, which evaluates the qualitative-quantitative features of the storm water;
- the WWTP sub-model, which is representative of the treatment processes;
- the receiving water body sub-model, which simulates the pollution transformations inside the water body.

The first sub-model describes the physical phenomena that occur both in the catchments and in the sewers during wet and dry weather. Dry weather flow is simulated according to the Fourier series (Mannina & Viviani 2009*b*). The sub-model produces the hydrographs and pollutographs in the sewer

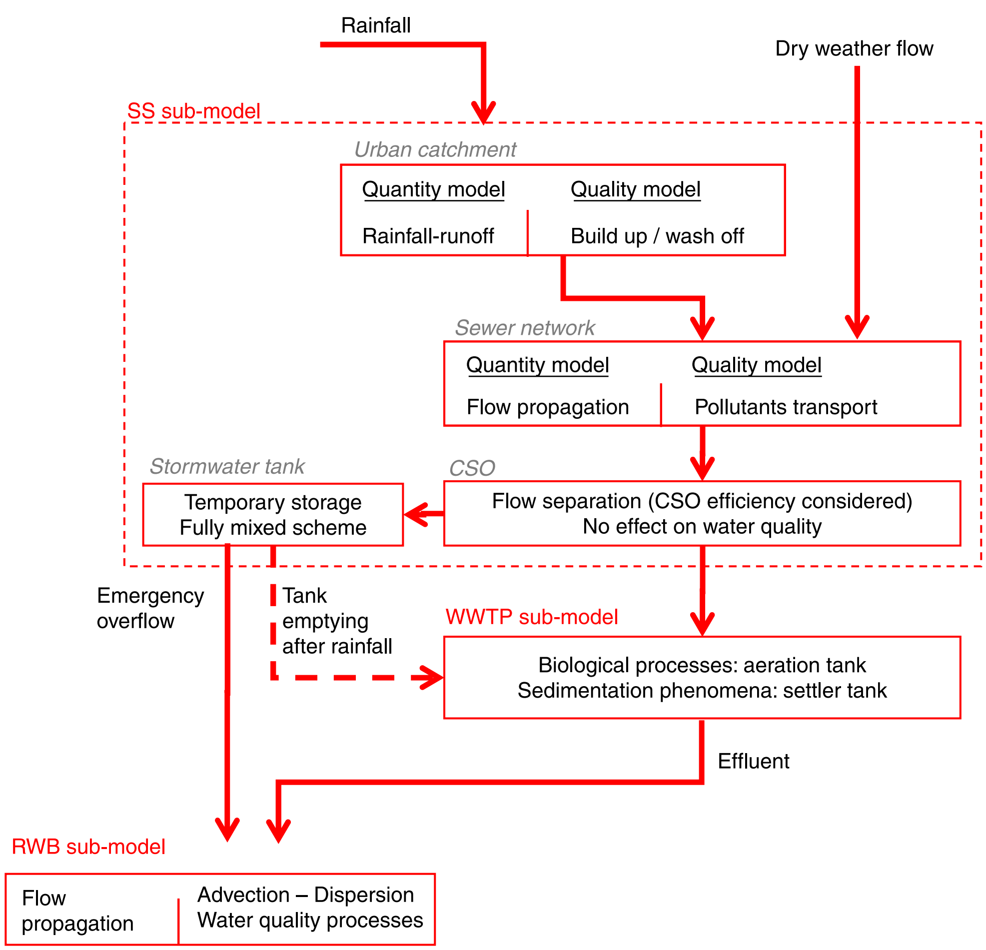

Figure 1 | Schematic overview of the different sub-models, analysed processes, and interconnections.

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as outputs that can be passed to the downstream sub-models. For the assessment of pollutant concentrations, particular attention was given to the sediment transformation in sewers, considering their cohesive-like behaviour linked to organic substances, and to the physical-chemical changes during sewer transport (Mannina 2005; Freni *et al.* 2008*c*).

The urban sewer system was modelled considering two reservoirs and one channel; the main physical-chemical phenomena that take place during both dry and wet periods were taken into account. Specific attention was paid to the antecedent dry weather period (ADWP) that is responsible of the first flush event.

As stated above, the second sub-model is aimed at the analysis of WWTP. In particular, this sub-model simulates the behaviour of the activated sludge tank and the secondary sedimentation tank. In the activated sludge tank model, the equations derived from Monod's theory were used to describe the removal of BOD and NH_4-N. The sedimentation tank was simulated using the modelling approach of Takács *et al.* (1991). In particular, the model predicts the solids concentration profile in the settler by dividing the settler into a number of layers of constant thickness, and by performing a solids balance for each layer.

The third sub-model assesses RWB discharges and water quality. More specifically, the modelling approach has been focused on rivers characterised by few field data and ephemeral characteristics (i.e. rivers characterised by a long dry season and intense flow for short periods following precipitations). This latter aspect is relevant since the phenomena generally involved in the evaluation of the RWB quality state play a different role with respect to the perennial streams commonly presented in the literature (Freni *et al.* 2008*a*; Mannina & Viviani 2009*a*). Such rivers are also found frequently in Mediterranean areas that are characterised by semi-arid climates. Due to the highly non–stationary conditions typical of the ephemeral streams, a dynamic model was employed for the propagation of the flow in the river. The quality phenomena are greatly affected by natural mixing processes derived by river ephemeral behaviour so the 1D advection-dispersion equation was implemented accordingly to model these characteristics. For the same reason, nutrients chemical modifications in the RWB were neglected because of the short residence time of the pollutant.

The adopted case study and the monitoring campaign

The analysis was applied to a complex integrated system: the Nocella catchment is a partially urbanised catchment located nearby Palermo in the north-western part of Sicily (Italy). The entire natural basin has a surface of 99.7 km^2, and has two main branches that flow primarily east to west (Figure 2).

The two main branches join together at 3 km upstream from the river estuary. The southern branch is characterised by a smaller elongated basin, and receives water from a large urban area (Partinico) and a smaller one (Borgetto). Both urban areas are characterised by relevant industrial activities partially served by a WWTP, and partially connected directly to the RWB.

The northern branch was monitored in the present study (Figure 3). The basin closure is located 9 km upstream from the river mouth; the catchment area is 66.6 km^2. The catchment end is equipped with a hydro-meteorological station (Nocella a Zucco). This river reach receives wastewater and stormwater from two urban areas (Montelepre, with a catchment surface equal to 70 ha, and Giardinello, with a surface of 45 ha) drained by combined sewers.

Both urban areas are characterised by concrete sewer pipes with steep slopes. The Montelepre sewer is characterised by circular and egg-shaped pipes with maximum dimensions of 100×150 cm. The sewer system serves 7,000 inhabitants, and it is characterised by an average dry weather flow equal to 12.5 l/s (water supply 195 l/capita/d), and an average dry weather BOD concentration of 223 mg/l. The Giardinello sewer system is characterised by circular pipes with a maximum diameter equal to 800 mm. The served population is 2,000 inhabitants, and it has an average dry weather flow equal to 2.5 l/s (water supply 135 l/capita/d) and an average dry weather BOD concentration of 420 mg/l. The calculated BOD unit loading factors for the two urban catchments are 35 and 45 g/capita/d for Montelepre and Giardinello, respectively. These values are lower than those typically observed in Italy (60 g/capita/d), likely due to the industrial activities present in the urban catchments. Furthermore,

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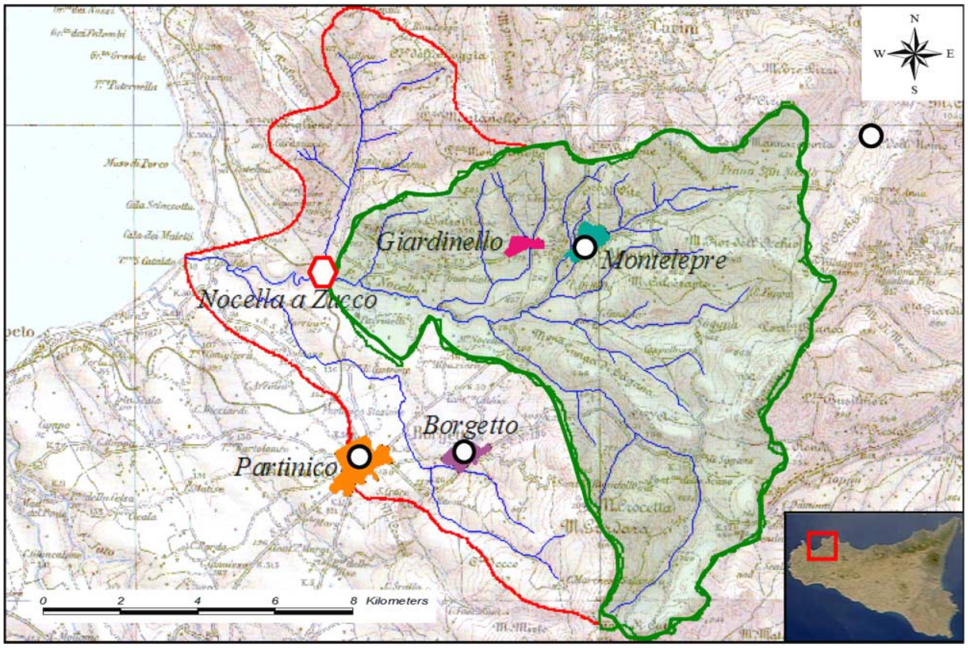

Figure 2 | Overview of the Nocella basin with indication of the four included urban areas: the monitored sub-catchment is shaded, rain gauges are indicated by white dots, and the flow gauge station is indicated by a white hexagon.

the lower concentration of BOD in Montelepre's urban catchment is due to the presence of infiltration flow into the sewer system.

Each sewer system is connected to a WWTP protected by CSO devices. The WWTPs are characterised by simplified activated sludge processes with preliminary mechanical treatment units, an activated sludge tank, and a final circular settler. In particular, attention in data acquisition was given to the activated sludge tank, and to the sedimentation tank, according to the modelling scheme. Moreover, such units are the most sensitive to flow and concentration variations during wet weather periods. Indeed, wet weather loads affect activated sludge settling tanks, and could significantly affect the effluent quality

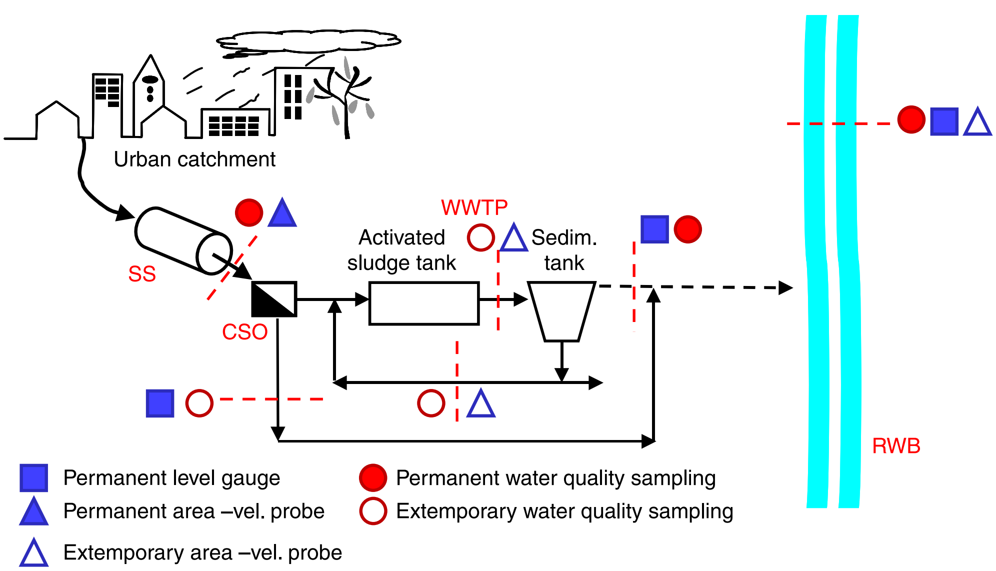

Figure 3 | Schematic of the urban drainage system monitoring methodology performed on the Montelepre and Giardinello urban areas.

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(Harremoes *et al.* 1993). In particular, the impact of stormwater can be summarised as follows (Stricker *et al.* 2003): an increase in sludge handling from the primary and secondary treatment processes, an increase in oxygen demand, a decrease in the removal efficiencies, and sludge blanket overflows in the plant effluent.

The activated sludge tank and the settler are 668 and 328 m^3, for the Montelepre WWTP, respectively, and 231 and 46 m^3, for Giardinello. The returned activated sludge recirculation for both plants is equal to the 100% of the dry weather flow. Additionally, the sludge retention times are 12 and 15 d^{-1}, for the Giardinello and Montelepre WWTP, respectively. The average mixed liquor suspended solids are, 2.5 and 3 kg VSS/m^3, respectively.

Rainfall was monitored by four rain gauges distributed over the basin (Figure 2): the Montelepre rain gauge is operated by Palermo University, and is characterised by a 0.1 mm tipping bucket and a temporal resolution of 1 min; the other three rain gauges are operated by the Regional Hydrological Service, and they are characterised by a 0.2 mm tipping bucket and a temporal resolution of 15 min. The hydro-meteorological station ("Nocella a Zucco") located at the catchment end is characterised by an ultra-sonic level gage operated by the Regional Hydrological Service, and has a temporal resolution of 15 min. The instruments were integrated by Palermo University by installing an area–velocity submerged probe that provides water level and velocity data with a 1 minute temporal resolution. An ultrasonic external probe was used to give a second water level measurement for validation, and as a backup in case the submerged probe failed; an automatic 24-bottle water quality sampler was used for water quality data collection. Each urban drainage system was monitored according to the scheme presented in Figure 3.

The monitoring was carried out considering both permanent and extemporary measures:

- permanent monitoring stations were established at the outflow of the SS, as well as at the outflow of the WWTP and at a RWB site upstream from the WWTP outlet.
- extemporary monitoring stations were set up for short periods to study specific WWTP processes, such as sludge recirculation, or for calibrating level discharge curves at specific cross sections.

Flow measurements have been carried out using area–velocity probes with 1 min temporal resolution, which allow the inflow and outflow volumes for each element in the system to be defined. Water quality sampling was performed using automatic 24-bottle samplers, and grab sampling was used for defining pollutant loads and treatment efficiencies.

Dry weather automatic sampling was performed at hourly time steps; wet weather samples were taken every 15 minutes in the urban drainage systems, and every 20 minutes in the river. These intervals were chosen to address two opposing needs:

- the need for high-resolution data in order to identify phenomena, like the first flush, that occur a short time after the beginning of the rainfall event.
- the limitation of the 24-bottle sampler that, in case of narrow sampling interval, may not adequately represent long rainfall events.

The differences between the urban drainage and river sampling interval is connected to the hydraulic distance between the urban areas and the Nocella a Zucco station (18 km downstream from the Montelepre WWTP, and 15 km downstream from the Giardinello WWTP), which is expected to dilute the polluted load originating from the urban areas. The water quality parameters monitored were Total Suspended Solids (TSS), Biological Oxygen Demand (BOD), Chemical Oxygen Demand (COD), Ammonia (NH_4-N), Total Kjeldahl Nitrogen (TKN), and Phosphorus (P); the Dissolved Oxygen (DO) was only monitored for the river. All analyses were carried out according to *Standard Methods* (APHA 1995).

The monitoring campaign began in December 2006 and was still in progress at the time of submitting of this paper. Rainfall and discharge were monitored continuously, while water quality was measured during specific periods:

- Dry weather water sampling was done for 5 days in April 2007, May 2007, June 2007, February 2008, and March 2008: for each sampling event, 24 hourly samples were taken in the cross-sections in the urban drainage system and in the river;
- Seven rainfall events were monitored between April 2007 and May 2008, as described in Table 1; an automatic sampler that contained 24 1-litre bottles was

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Table 1 | Main features of the monitored rainfall events

	Event 1 21/04/07	Event 2 02/06/07	Event 3 05/03/08	Event 4 06/03/08	Event 5 24/03/08	Event 6 27/04/08	Event 7 03/05/08	Dry Weather
Rainfall duration [min]	260	780	1080	630	190	130	200	–
Rainfall depth [mm]	8.3	18.4	7.2	17.6	36.8	23.0	19.2	–
Rainfall max. intensity [mm/h]	13.6	8.8	5.6	12.3	36.4	45.0	18.2	–
Rainfall av. intensity [mm/h]	2.0	1.4	0.4	1.6	12.2	11.1	5.8	–
ADWP [h]	1268	642	260	8	485	79	135	–
SS No. of water quality data points	48	48	48	24	42	40	36	48
CSO No. of water quality data points	48	48	48	24	34	38	40	48
WWTP No. of water quality data points	24	24	24	12	24	48	24	144
RWB No. of water quality data points*	72	108	108	48	48	64	72	72

*Five cross sections.

used to collect samples. The sampler was automatically activated by the Montelepre rain gauge to start sampling immediately upon the beginning of a rainfall event, with the exception of the sampler located at Nocella a Zucco, which was programmed to start sampling 20 min after the beginning of the rainfall event to account for the natural catchment concentration time.

MODEL APPLICATION AND RESULTS ANALYSIS

The model was evaluated for its applicability in a complex integrated catchment comprised of two SS, two CSO, two WWTP, and five RWB reaches, according to the simplified scheme presented in Figure 4. Five river reaches were analysed to account for the contribution of natural basins discharges in the dilution and propagation of pollutant loads in the RWB. Integrated models exhibit high levels of parameterization, and in the present application, 64 parameters are used to define the entire integrated system.

The calibration parameters are:

- for each SS sub-model: linear reservoirs constants, build-up and wash-off parameters, resuspension parameters in the sewer system, CSO efficiency;
- for each WWTP sub-model: kinetic constants for in Monod's model;
- for each RWB sub-model: river bed roughness, de-oxygenation and re-oxygenation coefficients.

A preliminary sensitivity analysis was performed to reduce the number of parameters to 48. Even after the sensitivity analysis, the number of parameters to be calibrated was still quite high, and this consideration highlights the main limiting condition for the application of integrated models, i.e. data availability. Several modelling applications in the hydrology literature suggest that a reliable calibration

Figure 4 | Integrated model conceptual scheme indicating two simulated urban catchments (hexagon), two SS (arrow from hexagons), two WWTP (biggest circle), two CSO (semi-circular arrow), and five RWB reaches (linear arrow).

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may need a number of data points ten times higher than the number of parameters to be calibrated, and this number may grow depending on data intrinsic uncertainty (Ghizzoni *et al.* 2007). Assuming that this assertion can be extended to water quality modelling, several rainfall events should be monitored along with dry weather periods to obtain a reliable model calibration.

After these initial considerations, the model calibration was performed by progressively calibrating the upstream to downstream sub-models, thus obtaining the best outcome from the monitoring approach over the different parts of the integrated system.

The calibration method was based on the classical Monte Carlo approach (Bertrand-Krajewski *et al.* 2002); simulation efficiencies were computed using the Nash & Sutcliffe (1970) criterion for each sub-model, and model calibration was performed via efficiency maximization:

$$E_i = 1 - \frac{\sigma_e^2}{\sigma_o^2} \tag{1}$$

where σ_e^2 is the variance of the error, defined as the difference between the measured and simulated values, and σ_o^2 is the variance of the observations; the i subscript represents model variables, such as Q, TSS, BOD, COD, NH_4-N, and DO. Monte Carlo analysis has been preferred even if highly computationally demanding because it can be more robust in case of the calibration of complex models characterised by high equifinality (Beven & Binley 1992) and it provides information for the entire investigated parameter region, rather than only in the vicinity of the 'optimum' region (Beven & Freer 2001). Like other likelihood measures, the Nash–Sutcliffe (N–S) index is equal to or less than zero for all simulations that are considered to exhibit behaviour dissimilar to the system under study, and it increases monotonically as the similarity in behaviour increases with a limit of one. Because of the presence of observational variance in the denominator of Equation (1), the N-S index values have a good fit with regard to the whole model output, meaning that model output peaks can be poorly simulated. The N–S index has the advantage of requiring no specific hypotheses regarding the formulation of the error between simulation and measurement. For this reason, it has been widely used in previous studies, and has been adopted in our study to be used as a screening

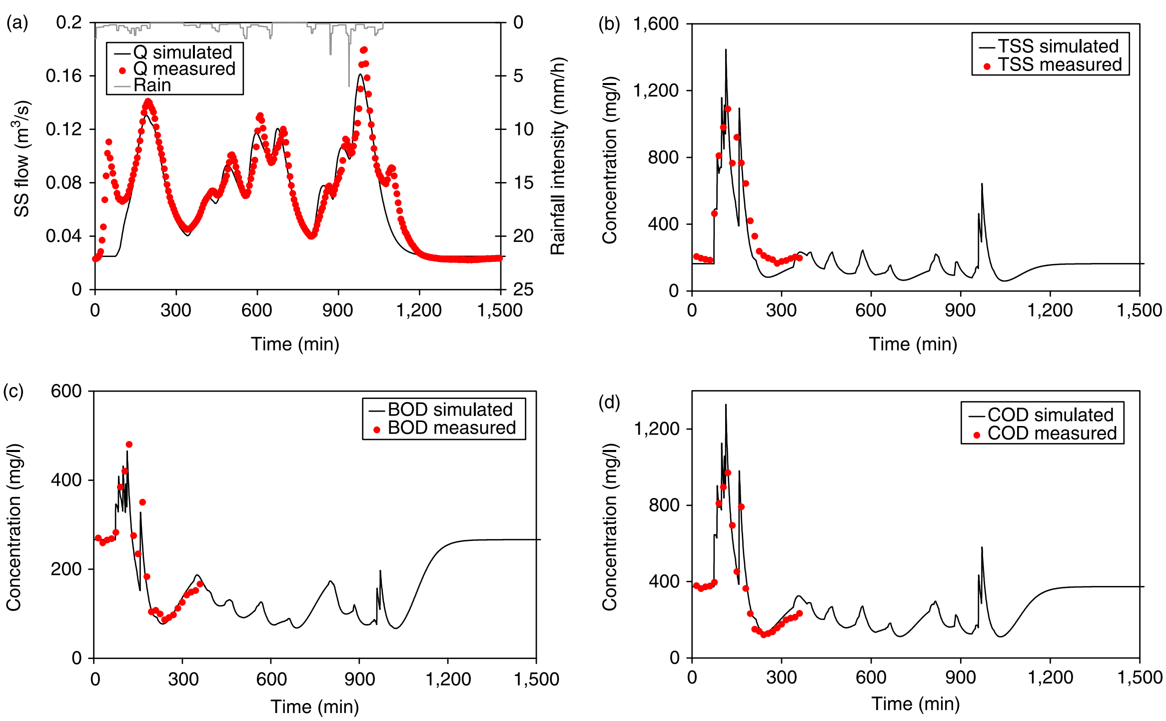

Figure 5 | Comparison between monitored data and modelling outputs at the Montelepre SS outlet (event 5/03/2008): (a) rainfall pattern and SS outflow discharge; concentrations of TSS (b), BOD (c) and COD (d).

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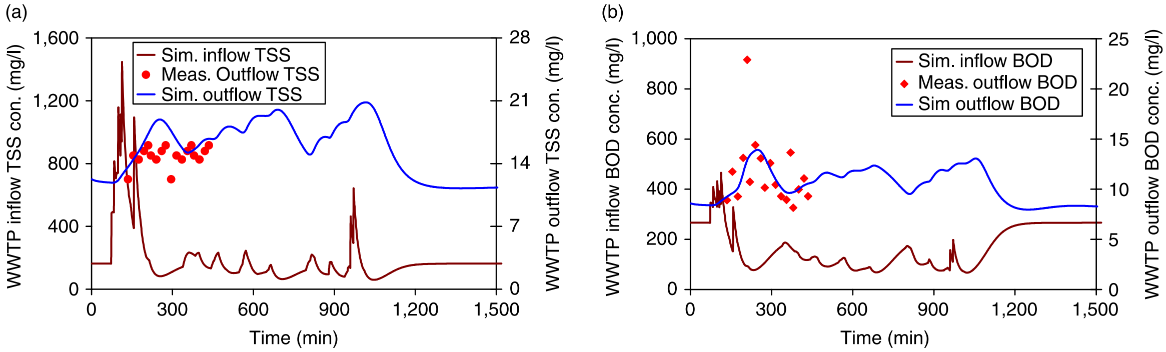

Figure 6 | Comparison between monitored data and modelling outputs at the Montelepre WWTP (event 5/03/2008): inflow and outflow TSS (a) and BOD (b) concentrations.

technique (Freni *et al.* 2009*d*). A total of twenty-nine intermediate and final modelling outputs were analysed (five for quantity and twenty-four for quality modules).

As previously mentioned, sub-models were calibrated in a cascade from upstream to downstream, first addressing quantity, and then water quality modules. For calibration purposes, 10,000 simulations were carried out. With respect to wet weather conditions, stormwater flow coming from the natural catchment upstream of the monitored Nocella reach was neglected because the concentration time of the natural catchment is much higher than that of the experimental urban area.

Figures 5, 6, and 7 show the main results of model calibration applied to the rainfall event of 5th March 2008. Figure 5 shows the model application to the Montelepre SS. The presence of multiple peaks is related to the rainfall pattern (Figure 5a) that was characterised by relatively low intensities and a long duration; the rainfall was characterised by three rainfall periods with less than two hours in between, and therefore, these cannot be considered as separate events, as demonstrated by the SS outflow graph (Figure 5a). The SS water quantity sub-model demonstrated a good capacity to fit real data (Figure 5a); an evident calibration problem is related to the initial part of the discharge graph (Figure 5a), which is due to a very fast catchment response that is not correctly represented by the model. This flaw is mainly a result of the simplification adopted for initial hydrologic losses, which are assumed in the model to be uniformly distributed over the catchment, while it is instead likely that a significant part of the urban catchment generates runoff immediately after the rainfall begins.

The SS water quality sub-model calibration showed good agreement with measured data, and it is able to represent both the first flush characteristics and the following dilution of dry weather flow. Figures 5b, 5c,

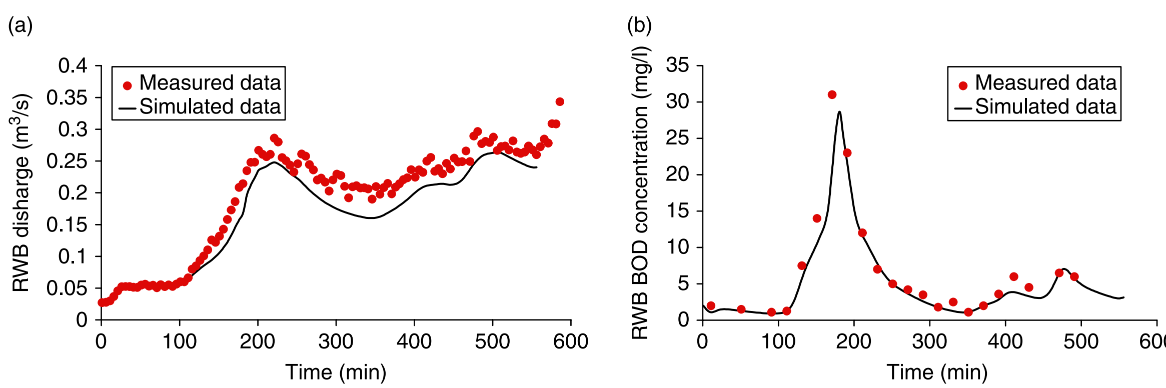

Figure 7 | Comparison between monitored data and modelling outputs at the "Nocella a Zucco" hydro-meteorological station (event 5/03/2008): river discharge (a) and BOD concentrations (b).

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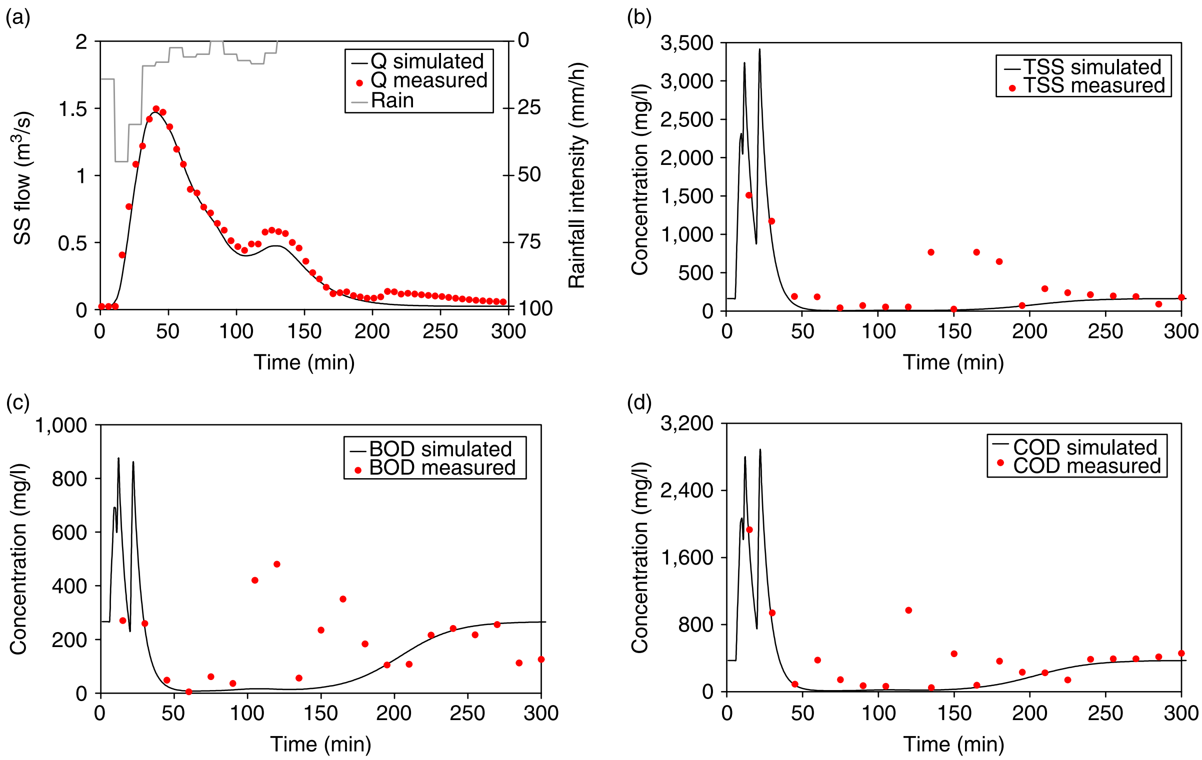

Figure 8 | Comparison between monitored data and modelling outputs at the Giardinello SS outlet (event 27/04/2008): (a) rainfall pattern and SS outflow discharge; concentrations of TSS (b), BOD (c) and COD (d).

and 5d show the limitations correlated with the use of automatic samplers, which provide much information on the first flush mechanism, but are not able to track the pollution propagation process for a long period.

Figure 6 displays the results of the Montelepre WWTP sub-model calibration. In Figure 6, only WWTP outflow measured data are presented, since inflow data are given in Figures 5b and 5c. The figures compare the WWTP inflow and outflow, highlighting the impact of rainfall discharges on WWTP depuration capacity. The graphs also show the translation of concentration peaks between the WWTP inflow and outflow that are connected with the detention capacity of the treatment plant. The duration of the storm event is long (1,080 min), and consequently, the WWTP inflow is higher than the dry weather flow for a long period, leading to possible problems with the rising of the sludge blanket; nevertheless, the TSS and BOD concentrations at the effluent site are relatively low, which indicates that the

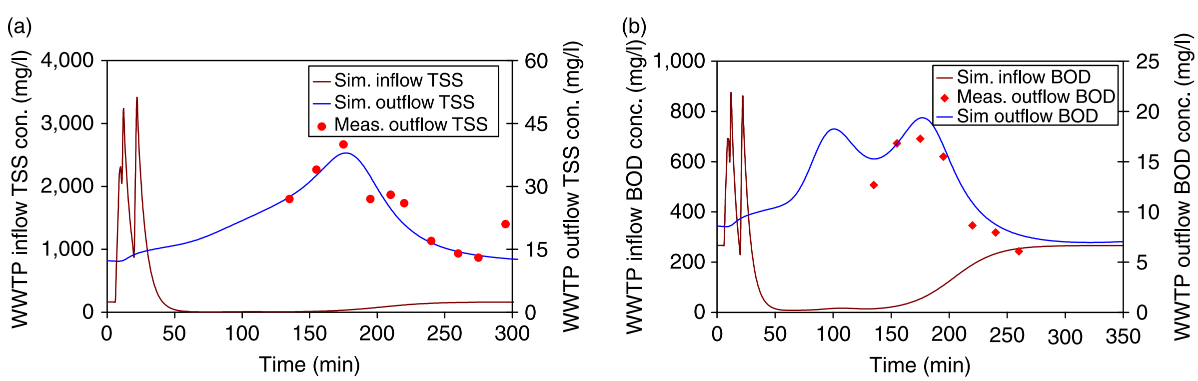

Figure 9 | Comparison between monitored data and modelling outputs at the Giardinello WWTP (event 27/04/2008): inflow and outflow TSS (a) and BOD (b) concentrations.

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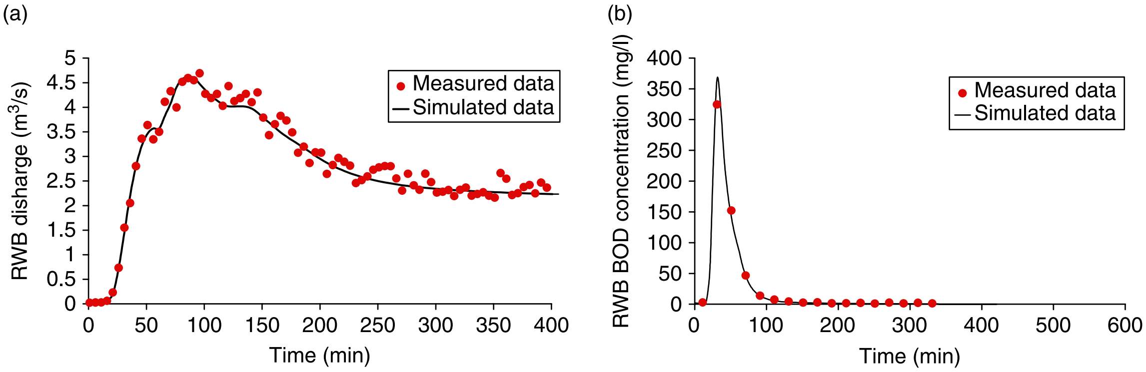

Figure 10 | Comparison between monitored data and modelling outputs at the "Nocella a Zucco" hydro-meteorological station (event 27/04/2008): river discharge (a) and BOD concentrations (b).

WWTP has a good capacity to equalise the storm events (Rouleau *et al.* 1997*a*).

The analysis of the discharge at the most downstream river cross-section (Nocella a Zucco) shows that the natural river has a relevant diluting and flow peak damping effect. The lag time between the SS/CSO outflow peak and RWB peak is around 130 min, which is compatible with the high flow velocities (between 2 m/s and 3 m/s) exhibited by this small river that is generally characterised by a long dry season and intense flow for short periods following precipitation. The model represents the measured data well considering both the water quantity and water quality.

Even if the RWB self-depuration capacity is able to reduce BOD peak concentrations, the concentration remains several times higher than the dry weather levels (Figure 7). Figure 7 shows only the first part of the analysed event (particularly the first 600 min) to better show the adaptation of simulated results to measured data. The simulation results confirm the importance of using a non-steady model approach in simulating ephemeral rivers (see Freni *et al.* 2008*a*). Indeed, for short rivers, pollution propagation and self-depuration phenomena generally play a different role compared to their roles in the perennial stream often described in the literature.

Figures 8, 9, and 10 show the main results of model calibration applied to the rainfall event of 27th April 2008. This event is shorter and characterised by rainfall intensities higher than the previous one. The maximum rainfall intensity is 45 mm/h, and the event duration is approximately 2 h. Figure 8 shows the correlation between water quantity and quality variables in the Giardinello SS. The quality module provides good results that are similar to the event discussed above. The water quality module provides poorer results for the pollutographs peak; this may be partially justified by the uncertainty related to sampling in the sewer system during the passage of very high discharges.

Figure 9 shows the impact of the flood event on the WWTP. The outflow concentration increases at the end of the rainfall event depend on the surcharge of the aeration and settling tanks. The difference between the TSS (Figure 9a) and BOD (Figure 9b) concentrations is attributed to the role played by dissolved BOD, which

Table 2 | Nash–Sutcliffe coefficient values for the SS sub-models and different variables (Q, TSS, BOD, COD and NH_4-N)

No event	Urban catchment	E_Q	E_{TSS}	E_{BOD}	E_{COD}	$E_{NH_4\text{-}N}$
1	Montelepre	0.701	0.527	−0.381	0.523	0.765
	Giardinello	0.812	0.692	0.711	0.687	0.456
2	Montelepre	0.810	–	–	0.632	−0.231
	Giardinello	0.854	0.724	0.691	0.750	0.972
3	Montelepre	0.888	0.823	0.864	0.878	0.940
	Giardinello	0.920	0.271	0.870	0.569	0.948
4	Montelepre	0.862	0.038	−0.328	0.557	0.623
	Giardinello	0.901	–	–	0.497	0.767
5	Montelepre	0.694	0.384	0.746	0.619	0.534
	Giardinello	0.787	0.533	0.602	0.397	0.184
6	Montelepre	0.867	0.438	0.528	0.215	0.638
	Giardinello	0.834	0.527	−0.123	0.668	0.704
7	Montelepre	0.833	0.555	0.499	0.602	0.872
	Giardinello	0.873	0.752	0.634	0.679	0.902

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Table 3 | Nash-Sutcliffe coefficient values for the CSOs and different variables (Q, TSS, BOD, COD and NH_4-N)

No event	Urban catchment	E_Q	E_{TSS}	E_{BOD}	E_{COD}	$E_{NH_4\text{-}N}$
1	Montelepre	0.832	0.567	0.752	0.824	0.543
	Giardinello	0.791	0.795	0.645	0.801	0.605
2	Montelepre	0.812	–	–	0.451	0.342
	Giardinello	0.920	–	–	0.503	0.520
3	Montelepre	0.930	0.532	0.053	0.751	0.487
	Giardinello	0.960	0.394	0.071	0.222	0.482
4	Montelepre	0.867	0.456	0.586	0.119	0.119
	Giardinello	0.952	–	–	0.212	–
5	Montelepre	0.765	0.238	−0.311	0.619	0.523
	Giardinello	0.652	0.612	0.823	0.497	0.487
6	Montelepre	0.821	0.511	–	0.324	–
	Giardinello	0.902	0.734	0.102	0.705	0.681
7	Montelepre	0.792	0.606	−0.436	0.458	0.758
	Giardinello	0.703	0.710	0.641	0.534	0.834

generates a secondary peak several minutes after the arrival of the discharge peak.

Figure 10 shows the results of the RWB sub-model application. The maximum flow rate in the RWB is 4.5 m^3/s, and the model is able to accurately describe the measured discharge. The flow rate in the RWB becomes more than an order of magnitude higher with respect to the dry weather one. This causes the establishment of dynamic conditions in the RWB, and hence, high turbulence conditions inside the water column. Such phenomena play a relevant role with respect to water quality variables: the initial peak is mainly due to advective characteristics of the flow, the polluting load discharge by the urban areas is diluted and rapidly transferred to the final RWB cross-section. After the peak has passed, the kinetic parameters in the RWB play a larger role in quickly reducing the BOD concentration, because of the ephemeral characteristics of the river.

Table 4 | Nash-Sutcliffe coefficient values for the WWTP sub-models and different variables (TSS, BOD and NH_4-N)

No event	WWTP	E_{TSS}	E_{BOD}	$E_{NH_4\text{-}N}$
1	Montelepre	0.710	0.580	0.511
	Giardinello	0.795	0.642	0.787
2	Montelepre	–	–	0.439
	Giardinello	–	–	0.521
3	Montelepre	0.354	0.596	0.922
	Giardinello	0.655	0.491	0.770
4	Montelepre	0.838	0.678	0.619
	Giardinello	0.733	0.523	0.597
5	Montelepre	0.639	0.434	0.692
	Giardinello	0.712	0.622	0.745
6	Montelepre	0.681	0.528	0.643
	Giardinello	0.702	0.623	0.527
7	Montelepre	0.812	0.312	0.481
	Giardinello	0.608	0.733	0.324

Table 5 | Nash-Sutcliffe coefficient values for the RWB sub-model and different variables (Q, BOD, DO)

No event	River section	E_Q	E_{BOD}	E_{DO}
1	Nocella a Zucco	0.912	0.602	0.687
2	Nocella a Zucco	0.934	–	0.220
3	Nocella a Zucco	0.855	0.762	0.781
4	Nocella a Zucco	0.812	–	0.419
5	Nocella a Zucco	0.901	0.781	0.912
6	Nocella a Zucco	0.822	0.528	0.290
7	Nocella a Zucco	0.845	0.412	0.422

In Tables 2–5, the model results in terms of N-S coefficients are reported for the different sections of the integrated models. More specifically, in Table 2, the Nash–Sutcliffe index is given for the last section of the drainage systems connected with the CSOs at the WWTP inlet. Some values were not computed because measurements were not available for the specific event and variable.

Table 2 shows that generally the water quantity processes are better simulated (average efficiency of 0.90) than the quality ones (average efficiency of 0.65). This is mainly due to the higher complexity of water quality processes and higher uncertainty in measurements. Negative maximum efficiency values were obtained for some water quality variables, and they were kept because they may be due to measurement errors or specific modelling issues not affecting the general performance of the model.

CONCLUSIONS

The present paper discusses the application of an integrated urban drainage model to a complex integrated catchment characterised by two urban areas and a relevant natural

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catchment. Each urban area is served by combined sewer systems and a WWTP. The application required the implementation of extensive monitoring, which required simultaneous data collection over the two urban drainage systems and over the RWB.

The first model application verified the ability of the model to describe the major characteristics of the analysed processes. With respect to the specific case study, the analysis allowed the following considerations:

- Even if inflow polluting peak concentrations to the WWTP during wet weather are 4–5 times higher than those during dry weather, treatment plants are able to stabilise these polluting loads, and outflow concentrations are only marginally higher than in dry weather;
- WWTPs usually return to normal dry weather behaviour a few hours after the end of the rainfall event, depending on the rainfall discharge tail;
- The RWB is characterised by moderate polluting loads during dry weather, and depends mainly on the WWTP treatment efficiencies; however, peak concentrations during wet weather can be several times higher, depending on the polluting contribution of the urban areas that cannot be mitigated by the self depuration capacity of the river;
- Water quantity aspects are generally better simulated than water quality ones; this result may be connected to the higher uncertainty of water quality processes that need more data for reliable calibration of the model or to the simplifications introduced by the model with respect to the complexity of the physical system.
- Even if modelling efficiencies for some sub-systems and some events are not satisfactory, the model is often able to compensate such error in the sequent sub-models thus demonstrating the importance of having simultaneous measuring points in different cross-sections of the system; such model behaviour shows the presence of over-parameterization and equifinality in the model as the wrong estimation of a parameter can be compensated by the others.

The initial results from the study show that more research should be carried out to evaluate the uncertainty connected with the modelling application and the measurement campaign, in order to increase operator confidence for such a powerful analysis tool.

This study is mainly focussed on acute pollution, whose effects last for a period comparable to that of the rainfall (see, Harremoës & Rauch 1996). Therefore, the evaluation of the RWB quality state was carried out at the scale of the wet weather periods. The time scale and the corresponding monitoring campaigns were performed accordingly. However, complementary research is ongoing to assess the RWB quality state as a result of accumulated pollution (Candela *et al.* 2008*a*,*b*; 2009). During long periods characterised by continuous dry weather, discharges, and occasional rainfall events, pollutants accumulate in the RWB and gradually build to a level that can be toxic (Harremoës & Rauch 1996). In this latter case, the discharge per individual event may be not relevant, but instead, the sum of discharges from a series of events is important. Other sources discharging to the RWB must be taken into account for the RWB quality state assessment, i.e. no point sources.

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