Deterministic modelling of integrated urban drainage systems

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Abstract Today, the main concepts required for describing the dynamics of drainage in an entire urban area are known and models are available that can reasonably simulate the behaviour of the urban water system. Still, such integrated modelling is a complex exercise not only due to the sheer size of the model, but also due to the different modelling approaches that reflect the history of the sub-models used and of the purpose they were built for. The paper reviews the state of the art in deterministic modelling, outlines experiences and discusses problems and future developments.

Keywords Integrated urban drainage system; modelling; receiving waters; sewer system; systems analysis; urban wastewater systems; wastewater treatment; water quality

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Introduction

The construction and operation of municipal drainage systems has been historically driven by the two objectives to maintain public hygiene and to prevent flooding. Only later the aspect of pollution control became important and treatment facilities have been introduced to preserve the aquatic ecosystem. The advance of mathematical models as tools for design and operation of the system followed this historical development. Detailed models of sewers, treatment plants and receiving waters have been created that describe the performance according to the individual needs and objectives.

Today's challenge is to move from such individual consideration of system performance to an integrated management of the urban wastewater system. Appropriate numerical tools are required to predict the behaviour of the complete system under historical and future scenarios. Although the basic principles are known (Lijklema *et al*., 1993) the development of integrated models still is a challenging task. The main bottleneck is the complexity of the total system that prevents a simple linkage of the existing detailed deterministic models of the individual subsystems to an entity. The aim of the paper is accordingly threefold: (1) to outline the state of the art in mathematical modelling of the integrated system by focusing on the deterministic description of the processes (2) to provide an overview on modelling experience in practice, indicating feasibility and benefits and (3) to discuss problems and future developments.

Basic principles of integrated modelling

The system

Figure 1 outlines the main elements of the integrated urban drainage system. In the sewer

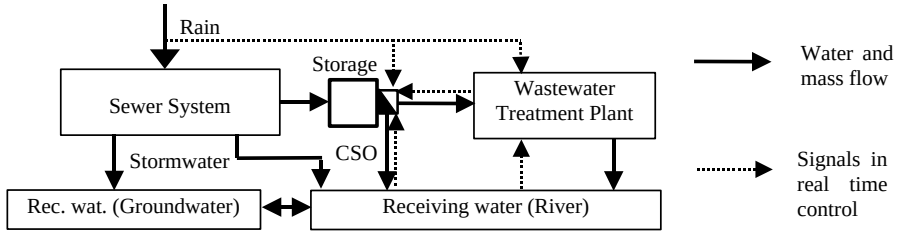

Figure 1 The integrated drainage system: overview on main elements and information flow

system wastewater is transported, that is sewage and runoff, from the catchment area to the wastewater treatment plant (WWTP) for purification and subsequent release into the receiving water. Only if the amount of runoff exceeds the given hydraulic capacity of the plant, wastewater is discharged to the receiving water directly, which can be seen conceptually as a bypass of the WWTP. Given that capacity is available in the system, excess wastewater is also stored for subsequent treatment. Another commonly applied drainage pathway is the separate discharge of minor polluted stormwater either by means of direct infiltration or via a separate storm sewer system. Although several types of receiving water bodies appear in reality (rivers, lakes, sea, groundwater etc.) modelling efforts focus on surface waters, specifically on rivers.

Integrated modelling is defined here as modelling of the interaction between two or more physical systems, i.e. sewer system, treatment plant and receiving water. The governing water quality processes for all systems are essentially alike: water motion, transport and conversion of matter are the key phenomena needed for an accurate description of any aquatic systems behavior. Thus, one might wonder why integrated modelling has not been more common until now. An important explanation is the split responsibilities for the management and planning of sewers, treatment plants and rivers. Hence, the lack of integrated modelling is more due to the administrative fragmentation than to scientific reasoning.

The meaning of information flux in the network

An important aspect in modelling of complex systems is the direction of the flux of information in the network. As long as dynamic phenomena proceed only in a forward (i.e. downstream) direction the system has a tree-like structure (dendritic), and sequential simulation can be applied, i.e. one element after the other. However, when feedback fluxes appear (e.g. return sludge in treatment plants) this procedure is no longer possible and processes in all elements in the systems have to be computed simultaneously. In the integrated drainage system the fluxes of water, compounds and – in case of real time control – of signals have to be considered. Although the overall system with its 3 major sub-systems might be considered dendritic it might involve feedback with its sub-systems. Thus, it depends on the objectives of the study whether complex simultaneous simulation is necessary, but it is usually helpful in understanding system dynamics.

Dominant receiving water impacts

The key purpose of integrated modelling is the evaluation of measures to improve the operation of the system, most important the receiving water quality. For this purpose it is vital to characterise the discharge impacts to the aquatic ecosystem, both with respect to the type of impact (bio-chemical, physical, hygienic, aesthetic, hydraulic etc.) and in terms of duration (e.g. acute, delayed, accumulating) (see e.g. Schilling *et al.*, 1997). Typically, it is not necessary to model the whole variety of effects on the receiving water but to focus on the few dominating ones. Only pollutants and processes that have a direct and significant influence

on the selected impacts need to be described quantitatively, whereas all other processes can be neglected (Rauch *et al*. 1998a). Hence, pragmatism is required to avoid unnecessary complexity of integrated models.

Following the above a key issue in the modelling exercise is the identification of the dominating effect, which then determines the key pollutants, processes and also the necessary timeframe of simulation. As a basic concept the detrimental effects to the aquatic environment can be categorised according to the characteristic time-scale of the impact, which is determined by the specific processes involved e.g. degradation time of pollutants. In receiving waters with long time constants (lakes, sea) individual events have only a little effect to the recipient ecosystem. The pollution is caused above all by accumulative contaminants (e.g. nutrients and heavy metals) and the characteristic time-scale for the effect is in the order of seasons or years. Hence the environmental risk for stagnant water bodies can be evaluated simply as mass load of a relevant contaminant over a certain period of time.

On the other hand, individual rain events produce a significant stress to the river ecosystem. The duration of the acute negative effects (e.g. oxygen depletion and bottom shear stress) is in the order of the duration of the event. According to the source of the environmental impacts, that is rainfall, also the detrimental effects are irregular in occurrence, duration and magnitude. The large variability in conditions has led to a methodology in which long-term time series of water quality variables are calculated. Statistical interpretation of these time series then allows us to compare frequencies of occurrence of critical states with experimentally determined threshold return periods for the avoidance of detrimental effects (see e.g. Harremoës and Rauch, 1996). In principle, both extreme cases of receiving water impacts – acute and accumulative – require long term simulation of systems behaviour. However, with regard to acute effects the dynamics of the single event are much more important than for accumulative effects.

It is interesting to note, that only very few legislations around the world follow the principles outlined above for prescribing a controlled environmental impact. Typically, it is only the volume and/or the number of CSO's that is to be kept below a certain prescribed value. The advance of integrated modelling will be helpful to overcome this obstacle and will promote holistic environmental legislation as already seen in the EU water framework directive.

Determinism, uncertainty and calibration

It is a result of our cultural heritage that the engineering profession applies the formalities of logic to describe the dynamic behaviour of systems by means of cause-effect relationships. In this deterministic approach it is assumed that – provided the basic assumptions are correct and the model parameters are true – a given input on the system will create a unique solution. The virtue is that the model can be used to predict scenarios outside of our experience but the approach needs a complex description of all the phenomena involved. Detailed deterministic models include so many functions and parameters that a stringent calibration of the model is virtually impossible. The pragmatic solution is to use a large set of default parameters and calibrate the model against a few selected ones, which are identified as being decisive for the investigation.

Still, it is an inevitable fact that any deterministic model shows a certain deviation from reality, since underlying processes are either truly stochastic or too complex for a stringent deterministic description. Hence, even if an integrated model is properly calibrated for single rain events, the result of long term simulation will be uncertain. This inadequacy can be taken care of by statistical considerations in design and operation (e.g. use of confidence limits in environmental standards).

State of the art models for sewers, wwtps and recipients

This chapter reviews models typically used for the deterministic description of the fundamental mechanisms and processes in the individual elements of the system. The intention is to give an overview on the important features rather than to provide details. Focus is on water quality processes since the rainfall-runoff-transport process of the water flow is well established and extensively applied and described in the literature (e.g. Chow *et al*., 1988; Schilling, 1991; Harremoës and Rauch, 1999).

Water quality and pollution transport in sewer systems

Since the early 1970s, the most frequent modelling approach used to simulate water quality and pollutant transport in sewer systems takes into account four main steps that are outlined in the following paragraphs (more detailed reviews have been given elsewhere, e.g. Bertrand-Krajewski *et al*., 1993; Mark, 1993; Crabtree *et al*., 1995; Hvitved-Jacobsen *et al*., 1998; Ashley *et al*., 1999):

- surface accumulation and wash-off
- conduit transport including sedimentation and re-suspension,
- conversion processes.

Generally, it is assumed that the best modelling approach describes the conceivable phenomena as far as possible. While this approach is obvious in research, it may be different with regard to other operational objectives. For example, many models have attempted to simulate pollutographs, which is more or less considered as one of the ultimate objectives of any sewer system model. However, for some applications and some operational objectives, i.e. design of retention and settling tanks, it may be observed that these are not sensitive to pollutographs, but rather to total mass per storm event (Bertrand-Krajewski *et al*., 2000), and that simplifications are frequently justified. Therefore, especially in the case of an integrated approach, the definition of the appropriate levels of description and modelling complexity should be clearly defined.

Surface accumulation and wash-off. Pollutant accumulation on catchment surfaces during dry weather is usually described by linear or exponentially asymptotic accumulation functions. In some models, it is assumed that the source of pollutants is not limited, and thus no accumulation process needs to be accounted for. As the available field data, used to calibrate model parameters, suffer from large uncertainties, the choice of the accumulation model depends more on numerical fittings than on physical reasons. Pollutants accumulated on surfaces are washed off during rainfall events. The conceptual models used to simulate this process are diverse, but usually include first order equations where the wash-off process depends linearly on the available accumulated mass, on the rainfall intensity and/or on the overland flow rate.

Transport in sewers. In a typical combined sewer system, the three main sources of pollutants transported during storm events are: 1) pollutants from catchment surfaces transported by runoff and entering into the sewer system through street inlets (with or without gully pots), 2) pollutants accumulated in sewer pipes (deposits, near bed solids and biofilms) during dry weather periods and eroded during flow increases, 3) pollutants transported by the dry weather flow (domestic and industrial effluents). The transport of soluble pollutants can be described by conceptual (reservoir) models and mechanistic models (advection-dispersion equations). Such models are also applied to very fine suspended solids (wash load).

The transport of particulate pollutants is far more complex and many models have been proposed since the late 1980s. The simplest only takes into account suspended solids. More detailed models take into account two or even three particulate phases (suspended load, bed

load, and wash load), where each phase is described by appropriate equations. Interactions between solid transport and hydraulics (e.g. roughness coefficient modification, reduction of cross-section due to deposits, etc.), effects of cohesion due to organic matter, and biochemical processes may also be taken into account.

Conversion processes. Until the middle of the 1980s, most sewer models considered pollutants to be conservative. Only suspended solids transport was simulated, and – since many pollutants are attached to suspended solids – pollutant concentration was simply calculated by applying a certain proportionality factor. This approach has been progressively replaced by a more realistic (but also more complex) one, which takes into account chemical and biological processes and considers the sewer system as a physical, chemical and biological reactor where solid, liquid and gaseous phases interact (see Figure 2). The formalisms of the models (matrix calculations) are similar to those applied in activated sludge modelling (see section below). In such models, decay of oxygen demanding compounds along sewer pipes, re-aeration, exchanges between suspended and bed load fractions, biofilm growth and other biochemical processes may be simulated.

Calibration of water quality related processes in sewers. Owing to the many parameters to estimate, calibration requires appropriate field measurements. In fact, the above four main steps (accumulation, wash-off, transport, processes) are most often estimated only by means of field data collected downstream (e.g. at WWTP inlet, at CSO structures, etc.), since it is impossible to calibrate each step and each parameter individually. Moreover, it seems as if spatial and temporal variability of sediments characteristics are critical aspects and may lead to difficulties with regard to calibration. Nevertheless, modelling of sewer biochemical processes may be very useful in the frame of an integrated approach, e.g. to simulate consequences of discharges in receiving waters by CSO structures where easily biodegradable COD will not have similar effects as refractory or slowly biodegradable COD mainly discharged by the WWTP effluent.

Wastewater treatment

The modelling of the wastewater treatment subsystem is quite different from the modelling of sewer or river systems in two respects: first, the underlying hydraulics can nearly always be approximated crudely and, second, the modelling is built up around unit processes. The mathematical description of the unit processes usually requires specifying a large number of components and numerous interactions. Henze *et al*. (1987) introduced a matrix for the

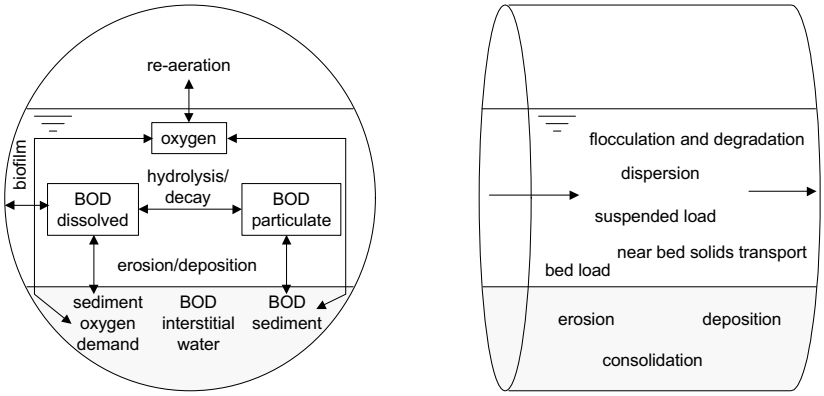

Figure 2 Example of processes model (related to transport and oxygen) applied in sewer systems (adapted from Garsdal *et al*., 1995)

presentation of the model reactions which has become standard in all aspects of water quality modelling.

Only in very particular cases flow propagation through reactors is modelled explicitly (De Clercq *et al.*, 1999). Usually, instantaneous flow propagation is assumed, i.e. the outflow rate is assumed to be equal to the inflow rate at any time. Mixing is typically modelled using the continuously stirred tank reactors (CSTR) in series approach. This approach allows us to reasonably mimic the advection and dispersion of matter in different unit processes, e.g. activated sludge tanks, settlers, trickling filters, etc. Basically, any mixing behaviour can be approximated by properly determining the number of tanks and their respective volumes. The overview given in this paper on the unit processes is limited to the most important ones, i.e primary and secondary clarifiers, activated sludge reactors, biofilm reactors and anaerobic digesters.

Clarifiers. These unit processes act on particulate matter that one either wants to prevent from entering the plant (primary clarification), or from leaving the system (secondary or final clarification). Another objective is thickening, either to increase the biological activity in the bioreactors, or to prepare for waste sludge treatment. Models for these systems are classified according to their spatial resolution (Ekama *et al.*, 1997), going from simple 0- to 3-dimensional models. The most popular clarifier models that can reasonably describe both the separation process and the dynamic mass accumulation in the clarifier are 1-dimensional models. Since usually only 10 layers are applied, the common approach is in fact a reactors-in-series approach rather than a discretisation of a 1-D partial differential equation.

Any clarifier model contains a settling velocity function that describes its dependence on the local concentration (settling is hindered increasingly with concentration above a certain threshold value) and the sludge volume index as an indicator for the settling capacity. While still much debate is ongoing on the best settling function, the empirical model of Takács *et al*. (1991) is widely applied. Finally, it must be stressed that it may be relevant to include a description of the reactions that occur in clarifiers, e.g. denitrification in secondary clarifiers or hydrolysis in primary clarifiers.

Activated sludge. The modelling of the activated sludge process has clearly drawn most of the attention in unit process modelling since the 1950s and many different approaches have been explored. However, since the groundbreaking work of the IAWPRC Task Group on Mathematical Modelling of the Activated Sludge Process in the early 1980s, most model development work has been geared around what can be called the industry standard suite of Activated Sludge Models (Henze *et al.*, 2000). These models have been shown to adequately describe the behaviour of nitrogen and biological and chemical phosphorus removal processes, more particularly in terms of the oxygen demand, sludge production and nitrogen/phosphate removal. Recently, refinements of the models were presented in which storage processes are included and in which the quite recently accepted phosphate uptake process under anoxic conditions were included too. The main success of these models is found in the evaluation of scenarios for upgrading of carbon removal plants to nitrogen and phosphorus removal and in the design of control systems. These models have also stimulated the introduction of simulation software in the consulting and engineering companies and have been a driving force for a more detailed understanding of the processes, leading to considerably improved operation of the treatment plants. The main limitation of the application of these models is the calibration of their parameters to closely reflect plant behaviour. For municipal treatment plants, reasonably accurate predictions of process behaviour can already be obtained with minor adjustments of model parameters, but when a

not negligible fraction of the wastewater is of industrial origin, special measurement campaigns and calibration studies are required (Petersen *et al*., 2001).

Biofilms. Historically, trickling filters were the workhorses of wastewater treatment and, recently, we again observe an increasing use of biofilm-based treatment unit processes. In these systems the conversions of pollutants occur simultaneously in different locations through the biofilm with diffusion of substances in and out of the biofilm and growth of the organisms catalysing the different conversion reactions at different depths of the biofilm. It is therefore mandatory to describe these processes as function of time and space. Henceforth, the state-of-the-art biofilm model of Wanner and Reichert (1996) is a PDE model and includes carbon and nitrogen removal through nitrification/denitrification. The main issue in these models is not the description of the biological conversion processes, but the quantification/prediction of the physical attachment/detachment processes, for which only empirical relationships are available today. An important disadvantage of the PDE-based models is the computational burden they incur. As a result, proposals have been made to simplify the models. The best-known simplification is the one originally proposed by Harremoës and further developed e.g. in Rauch *et al*. (1999). It consists of assuming zero order biodegradation kinetics which allows us to separate the description of the mass transport through diffusion from the mass conversion process.

Anaerobic digestion. For a long time the standard model in anaerobic digestion was the 4-population model, introduced by Mosey (1983) and Rozzi *et al*. (1985). These models describe the main conversions taking place under anaerobic conditions, i.e. acidification and methanogenesis each according to two different pathways: acidogenic bacteria and obligate hydrogen producing acetogens (OHPAs) perform the first step whereas acetoclastic methanogens and hydrogenophilic methanogens form methane from the produced intermediates. These models have been used both to describe the dynamics of anaerobic digestion of sludge and the anaerobic (pre)treatment of different types of wastewater. In the former application they are very relevant for the study of plant-wide interactions between wastewater and waste sludge treatment. Currently research efforts are undertaken to summarise the findings and to publish a standard model for the process.

Rivers (and other receiving waters)

Water quality changes in rivers are due to physical transport and exchange processes (such as advection and diffusion/dispersion) and biological, biochemical or physical conversion processes. The general formulation of the extended transport mechanism (Rauch *et al*., 1998b) offers a useful modelling framework: (1) the transport equation, valid for solute and conservative substances and (2) on top of that, the conversion sub-model for reactive substances which can be developed step-wise and independently of the description of hydrodynamics. However, note that a deterministic model of the water motion is required as input. Finally, it needs to be stressed, that such an extended transport formulation (advection/dispersion equation plus conversion) can also be integrated for subsequent river stretches within which complete mixing is assumed. This approach is conceptually to be seen as a sequence of inter-linked reactors.

A crucial aspect with regard to pollution in river water quality modelling is the role of the sediments. Its importance for integrated drainage management has been demonstrated e.g. by Rauch and Harremoës (1996b). In mathematical terms, sediment is one more compartment for which the transport equation is to be applied. Conversion processes describe changes in the concentration of model constituents due to physical, biological and biochemical processes. The historical development of oxygen, nitrogen and phosphorus

models shows step-by-step extensions and increasing complexity, ranging from the pioneer oxygen sag model of Streeter and Phelps to EPA's former state-of-the-art model QUAL2 (Brown and Barnwell, 1987) which includes nutrient cycling and algae. However, although the fundamental conversion processes in surface waters are similar to the ones in wastewater treatment, the model description is not compatible, which has severe implications when defining interfaces. To overcome this shortcoming, in 1997 IWA (formerly IAWQ) formed a Task Group on River Water Quality Modelling to create a scientific and technical basis from which to formulate standardised, consistent river water quality models. The effort is intended to create models that are compatible with the existing standard models for describing wastewater treatment (Henze *et al.*, 2000) and can be straightforwardly linked to them. The fundamentals of the model as well as guidelines for selecting the appropriate model structure and hydraulic formulation are described in Shanahan *et al.* (2001), Reichert *et al.* (2001) and Vanrolleghem *et al.* (2001).

As rivers are the dominant sinks for urban wastewater, model development focussed accordingly on this type of surface waters. The principles of water quality modeling are equally applicable also to estuaries, coastal waters and lakes (Chapra, 1997). However, different focus is to be put on the spatial dimensions of hydrodynamics and transport. Although detailed models have been developed for those types of surface waters, only little work has been done up to now in terms of integration. In principle, the same holds for groundwater modeling.

Integrated models and program platforms

Even though Beck presented the idea of integrated modelling as early as 1976 and the first integrated model was applied 20 years ago (Gujer *et al.*, 1982), it took until the early 1990s before the concepts started to be disseminated in larger scale (e.g. Triton, 1991; Lijklema *et al.*, 1993; FWR, 1994). Whereas early approaches (Durchschlag *et al.*, 1991) considered only total emissions from sewer system and treatment plant, Rauch and Harremoës (1996b), Schütze *et al.* (1996) and Vanrolleghem *et al.* (1996) applied deterministic models to the total system. These studies revealed the importance of consideration of both, treatment plant effluent as well as CSO discharges for a proper assessment of impacts of storm events on the receiving water body.

The problem of using different state variables in the sub-models was investigated by Fronteau *et al.* (1997), and the first studies on integrated real time control (which requires simultaneous simulation of the complete system) were published by Rauch and Harremoës (1996a, 1999) and Schütze *et al.* (1996, 1999, in preparation). Rauch and Harremoës focused on on-line optimisation of control strategies, whereas Schütze *et al.* performed off-line optimisation by developing the software tool SYNOPSIS. In the following years more and more models have been developed to investigate specific aspects of interaction and dynamics. An overview is given by the series of papers on that issue produced by a EU COST working group (Schilling *et al.*, 1997; Rauch *et al.*, 1998a; Vanrolleghem *et al.*, 1999).

A limit to model development was the fact that appropriate software platforms became available only recently. Hence it was either necessary to develop academic software or to live with the limitations put forward by the existing platforms. An example of the latter is AQUASIM (Reichert, 1994), which allows the implementation of different wastewater treatment and river models in an open simulation environment but lacks the sewer system. Hence, for a study of the total system performance one needs to complement this platform with another model, providing the necessary input from the sewer system (Holzer and Krebs, 1998). Unable to list coherently all models and simulations available, the key characteristics of three commercial software packages for integrated modelling are listed in Table 1.

Table 1 Key characteristics of some of today's commercial packages for integrated modelling

Name of simulator	ICS	WEST	SIMBA
Developer	DHI, Hørsholm/DK, WRc, Swindon/GB	Ghent University/B Hemmis n.v., Kortrijk/B	Ifak Barleben/D
2-directional interactions between submodels	Yes	Yes	Yes
Simulation of control options possible	Yes	Yes	Yes
Simulation of long time series feasible	under development	Yes	under development
Open simulation environment	No	Yes	Yes
Integrated use at a real case study reported	Yes	Semi-hypothetical	Yes

The Danish Hydraulic Institute (DHI) and Water Research Center in GB (WRc) developed an "Integrated Catchment Simulator (ICS)" in a large EU-funded "Technology Validation Project" (Mark and Williams, 2000). ICS is basically a graphical interface for setting up and running integrated models with feed forward/feed back of information. The present ICS version includes existing models for sewers (MOUSE), rivers (MIKE11), wastewater treatment plants (STOAT) and coastal areas. During the course of this project, these fairly complex constituent models were linked in various stages; first in a sequential way, later in a simultaneous way. The complexity of the submodules, however, currently limits the application of ICS.

The Belgian simulator platform WEST follows a different pathway. Although originally developed for wastewater treatment modelling, it can be seen as a general simulation environment for computing the dynamics in a network of interlinked elements. The concept puts a limit to the description of water motion and transport processes in the elements but allows us to implement, more or less freely, different conversion models for the different elements (representing catchments, CSO-structures, reactors, clarifiers, river reaches, etc). Hence, WEST is predominantly an environment for the development of fast surrogate models for the purpose of long term simulation (Meirlaen *et al.*, 2000).

SIMBA® is a simulation platform running on top of MATLAB™/SIMULINK™. Models are available for sewer systems, treatment plants and rivers. The general principle is similar to the network concept already presented for the example of WEST, however, the use of the general purpose simulation environment MATLAB™/SIMULINK™ allows the user to add their own modules to fit the actual modelling needs. Thereby, the distinction between model developer and model user is largely removed. This simulation system is also a convenient tool for control and optimisation of the overall performance of the system (Alex *et al.*, 1999). Real-life applications of integrated control involving SIMBA have been implemented.

In summary, it can be stated that today a number of tools are available which allow the urban wastewater system to be considered in simulation, as what it indeed is – one single system. The practical application of these integrated models appears to be limited more by the lack of data (cf. Vanrolleghem *et al.*, 1999) than by any lack of suitable modelling tools.

Problems and solutions

Interface problems between sub-models

The various subsystems require different approaches since different processes are decisive. Due to the inconsistency of the models of the subsystems, major problems arise at their interfaces. The time resolution varying over several orders of magnitude and enormous differences in complexity to describe flow and mixing (i.e. finite difference method for hydrodynamics in sewers and rivers and stirred tank reactors approach for ASM) are less of a

problem than the fractionation of the key quality descriptors oxygen demand and nitrogen compounds (Rauch *et al.*, 1998a).

Generally, quality modelling of sewers is by far less developed than that in the other compartments. In conventional sewer models, pollutant concentrations are frequently assumed to be equal in different events and constant over the event's duration. Hence conversion processes are neglected. If conversion models are included they are – like river models – often formulated on the basis of BOD, since the biologically degradable part of the oxygen demanding compounds is relevant in those compartments. In ASM the application of COD is absolutely necessary since the models are based on balances. The conversion from particulate and dissolved BOD in the sewer to the COD fractions of ASM and back to BOD of river models is a key problem that limits the reliability of present integrated models aiming at modelling the oxygen content in the receiving waters. The application of conversion factors (see e.g. Vanrolleghem *et al.*, 1996) is a pragmatic compromise to overcome the problem. In fact, those conversion factors depend on the characteristics of the sewer system, on the evolution of the specific event and on the dynamics of the compound transport (Krebs *et al.*, 1999a, 1999b), varying in wide ranges both between various systems and in a certain system over the event. Therefore, one of the major requirements to further develop integrated modelling is to develop consistent sets of model parameters in the various subsystems models in order to dynamically run them without external definition of conversion at the interfaces. However, this task is not as easy as it may appear, since different types of processes need to be described.

Identification and testing of integrated models

Integrated models consist of a sequence of sub models for the various elements of the system. In the model identification and testing phase these models should be built and calibrated both individually and in the integrated mode. Prior individual calibration and testing makes it possible to focus on the specific details of each sub model and ensures that the model predictions are individually satisfactory. Only thereafter the complete model should be tested and verified for the specific situation. Skipping the individual calibration step gives a high risk that – even then the complete model behaves satisfactorily – specific sub models may not give a physically correct representation of the individual subsystem.

Measuring campaigns to support such individual and holistic identification of integrated models may become huge as there is both a temporal and a spatial dimension to consider (Vanrolleghem *et al.*, 1999): (1) the range of time constants in the system (tens of seconds for oxygen and flow dynamics in treatment plant and sewer, respectively, and up to months for population dynamics in treatment plants and rivers) requires long measurement campaigns with very small sampling intervals and (2) the network of sewers and rivers makes it necessary to sample at multiple locations. It is obvious that such effort is hardly possible for all modeling exercises and hence steps are to be taken to get indicative results much faster and cheaper, however, also with less accuracy. This can e.g. be obtained by means of relative studies with the integrated model. For instance, when calibrating the sewer model, the wastewater may be assigned a concentration of 1.0 and the most important weirs can directly be identified together with the relative impact in the recipient. Relative studies like that can be applied to test the order of magnitude of proposed alleviation schemes and they can be used for finding the optimum location of gauges and hence planning of measurement campaigns for detailed integrated modelling studies.

Another guiding principle in the planning of measuring campaigns is the use of simple mass balances. Among other information they can be particularly helpful to evaluate the quality of the measurement data. It is therefore worthwhile to design a measuring set up

that allows us to apply mass balances with the available data, e.g. by including sampling locations at all inputs of a stretch and at its output.

Problem oriented modelling

It is not the most complex model that is the best one, but the least complex that answers the asked question reliably. Therefore, it is vital to analyse carefully what the problem of the system or the receiving water is and based on this formulate the goal of numerical modelling. Three typical problems to be modelled are summarised in Table 2: toxic impact due to unionised ammonia discharged mainly from CSOs, hygienic problems caused by faecal coliforms and oxygen depletion in the river.

- With regard to toxic receiving water impact through unionised ammonia, the peak load in the CSO discharge is caused by short-term hydrodynamic effects in the sewer system which absolutely requires hydrodynamic instead of hydrologic transport modelling (Krebs *et al.*, 1999a) Further, maximum concentrations in the receiving water are induced just after the inflow *to* and the mixing *with* the receiving water. Since the rainfall-runoff process in the natural catchment area is significantly slower than in the urban area, the peak load in the CSO discharge and the minimum dilution capacity at minimum flow rate in the river coincide in the initial phase of the overflow event. The WWTP processes become only significant when the nitrification process or the secondary clarifier is overloaded.
- The presence of faecal coliforms is an indicator for hygienic deficiency in the river. To model this problem, the sewer and WWTP models can be simplified. Faecal coliforms behave like a tracer in the sewer while the effluent concentration of the WWTP is constant, independent from the flow rate and can be determined by measurements. The receiving water needs to be modelled downstream to a section where hygienic deficiency would matter, e.g. a bathing place. All degradation processes and sedimentation can be summarised by applying a single "degradation" rate.
- Assessing of oxygen depletion in the receiving water requires detailed modelling of wastewater treatment and processes in the river. The activated sludge model (e.g. ASM 1, Henze *et al.*, 1987) and the river water quality model must include the processes relevant to the oxygen balance. The latter must thus represent not only the water body but also the sediment inasmuch as oxygen consumption is involved.

Table 2 Minimum requirements for problem-oriented integrated models (after Rauch *et al.*, 1998a)

Goal function		Sewer system	WWTP	Receiving water
Toxic peak loads (NH_3)	Processes	Rainfall-runoff, hydrodynamics, advection/dispersion	Transport, mixing, nitrification	Mixing
	State variables	N_{tot} (= NH_4, "worst case")	NH_4, XBA (autotrophic bacteria)	NH_4, pH (measured)
Hygienic impact (Faecal Coliforms)	Processes	Rainfall-runoff, hydrologic analogy, mixing	–	Transport, Mixing, "decay" incl. Various removal processes
	State variables	FC	FCeffluent = constant	FC
Oxygen depletion	Processes	Rainfall-runoff, hydrol. analogy, mixing, sedimentation in CWRT	Transport, mixing, conversion with ASM1, Sedimentation in SST demand	Transport, mixing, conversion, aeration, sediment oxygen
	State variables	COD, BOD	COD-fractions	BOD-fractions, DO

FC – Faecal Coliforms, SST – Secondary Settling Tank, CWRT – Combined Water Retention Tank

Future developments

Analysing the current trend in modelling of the integrated system, several developments can be observed. That is first the effort to develop the detailed mechanistic models further and to include options also for long-term simulation studies. Emphasis is also put to the development of simpler surrogate models that allow for a fast consideration of long-term effects as well as for the simulation of a large number of different scenarios for planning purposes. Finally, also stochastic modelling is applied, the advantage of which is that uncertainty is taken into account by predicting system performance randomly. All of those different modelling approaches serve specific needs and, hence, have a role to play in integrated modelling.

Another aspect is the management of the large amount of data that is needed for and produced by the modelling exercise. Management and operation of integrated urban drainage systems involves not only the modelling expert but also numerous stakeholders with different backgrounds, methods and objectives. One of the key elements in the decision process is the provision of an appropriate information system that facilitates communication between the experts. As most of the information is given in a spatial context, the use of GIS has become an essential modelling element as well as modern database systems.

Further requirements will emerge with the EU water framework directive (Krebs, 2000). The system boundary will be extended and tools be developed to support river management at basin scale. Models describing point-source and diffuse source pollution will be included, and the rainfall-runoff process in the urban area and in the entire catchment area will be computed simultaneously.

The goal of "good water quality" does not apply to surface waters only, but also to groundwater. Therefore, the effects of different urban water management strategies (e.g. abstraction for drinking water supply, stormwater infiltration) on the quantitative and qualitative state of the resource groundwater should be predictable on a long-term perspective.

Conclusions

The need to improve the present integrated models is obvious. The models of the subsystems should be formulated in a consistent way and run in parallel to consider interactions in both ways or to use information from the downstream system to take control actions in the upstream system. With respect to acute receiving water pollution, modelling of dynamic transport processes and changes in wastewater composition must be improved. This will allow us to progress in the optimisation of the system as a whole rather than just optimising the individual subsystems by using the respective models.

Last, it is worthwhile mentioning that only part of the receiving water impacts caused by the urban water systems can be described adequately. Topics like eco-morphology of the river cannot be predicted by modelling. Hence, a careful evaluation of what the site-specific problems are cannot be replaced by sophisticated modelling but, in turn, is an absolute prerequisite to performing modelling successfully.

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