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Review

A critical review of integrated urban water modelling – Urban drainage and beyond

Peter M. Bach [a,*], Wolfgang Rauch [b], Peter S. Mikkelsen [c], David T. McCarthy [a], Ana Deletic [a]

[a] Monash Water for Liveability, Department of Civil Engineering, Building 60, Monash University, Clayton, 3800 VIC, Australia
[b] Unit of Environmental Engineering, University of Innsbruck, Technikerstraße 13, 6020 Innsbruck, Austria
[c] Department of Environmental Engineering, DTU Environment, Technical University of Denmark, Miljøvej, Building 113, 2800Kgs., Lyngby, Denmark

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ABSTRACT

Modelling interactions in urban drainage, water supply and broader integrated urban water systems has been conceptually and logistically challenging as evidenced in a diverse body of literature, found to be confusing and intimidating to new researchers. This review consolidates thirty years of research (initially driven by interest in urban drainage modelling) and critically reflects upon integrated modelling in the scope of urban water systems. We propose a typology to classify integrated urban water system models at one of four 'degrees of integration' (followed by its exemplification). Key considerations (e.g. data issues, model structure, computational and integration-related aspects), common methodology for model development (through a systems approach), calibration/optimisation and uncertainty are discussed, placing importance on pragmatism and parsimony. Integrated urban water models should focus more on addressing interplay between social/economical and biophysical/technical issues, while its encompassing software should become more user-friendly. Possible future directions include exploring uncertainties and broader participatory modelling.

1. Introduction

Today, more than half of the world's population has migrated to urban areas (United Nations, 2001). The ever-increasing stress that this poses on the urban infrastructure and utility provision has compelled governments and institutions to find new ways to adapt. Researchers and practitioners are beginning to recognise the importance of multiple benefit solutions, multi- and inter-disciplinary work and broad perspectives (e.g. Fratini et al., 2012). Water management paradigms have greatly evolved since the birth of cities, from the most fundamental objective of a secure water supply to sanitation, flood protection and a *Water Cycle City* (Brown et al., 2009). This transition is accompanied by increasing acknowledgement of the inherent complexity of the urban environment. As such, we are moving towards combined management of the various urban water system *components* (i.e. water treatment, distribution, sewerage and storm drainage, wastewater treatment, environmental compartments) and have become considerate of their interactions and feedbacks: the concept known as 'integration'. The principles of Integrated Urban Water Management (IUWM) (Vlachos et al., 2001; Mitchell, 2004) and the vision of a Water Sensitive City (Wong and Brown, 2009) have emerged from recurring dilemmas that we have perceived in previous decades, such as persistent floods and droughts, degradation of natural waterway health at the expense of urban growth, and frequent upgrades and rehabilitation of existing centralised water infrastructure required without end.

Traditional management of urban water systems considers all components independent of each other in a fragmented manner (e.g. Rauch et al., 2005). Due to its ongoing success in delivering water supply and improving public health, there is unwillingness to acknowledge and improve our limited understanding of feedbacks, non-linearity and time delays – all of which alter the nature and state of systems – alongside a diversifying water management agenda and changing societal perceptions (Pahl-Wostl, 2007). Global challenges and the framing of additional water management objectives (e.g. setting water recycling targets or delivering liveable city through stream protection, micro-climate improvements and amenity improvements using green water infrastructure) have now limited the success of single-objective optimisation (Erbe and Schütze, 2005). As such, the need to accept the intrinsic complexity of our environment instead of continuously simplifying

* Corresponding author. Tel.: +61 4 3217 5283.
E-mail addresses: peter.bach@monash.edu, peterbach@gmail.com (P.M. Bach).

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and isolating the problems (Beck, 1976) is slowly being recognised. We now strive to adopt an integrated approach to urban water systems management that: (1) considers all parts, or components, of the system, (2) involves water conservation and diverse fit-for-purpose water supplies, (3) works at a range of scales (both central and decentralised) and (4) allows establishment of links with other environmental cycles (e.g. energy and nutrients) (Mitchell, 2004; Brown et al., 2009).

Modelling the urban water system has progressed along the same path as management. Countless software packages are available for different parts of the urban water system, each developed to the greatest detail, but excluding dynamic interactions with the surrounding environment. The shift towards IUWM has, however, been reflected in model development, with integration seen as necessary to analyse the whole system without neglecting important physical phenomena in each component and their interactions (Gujer et al., 1982; Durchschlag et al., 1992; Holzer and Krebs, 1998; Chocat et al., 2001; Harremoës, 2002; Meirlaen et al., 2002; Muschalla, 2008; Candela et al., 2012). Institutional barriers (Brown, 2008), expensive data requirements (Vanrolleghem et al., 1999; Rauch et al., 2005) and limitations in computational hardware (Vanrolleghem et al., 2005b) have greatly challenged integrated modelling of urban water systems. With rapid advancements in technology, improving collaboration among water management authorities and researchers, the field has gained momentum in recent years. Yet another roadblock has been the shortcomings of traditional single-system modelling approaches, such as those demonstrated by Box and Draper (1987), Wagener et al. (2004). We frequently apply most, if not all, of our pre-conceived and widely accepted modelling principles, which we will henceforth refer to as '*classical modelling*', to integrated models and consequently suffer from repercussions of poor performance and unwieldy models (Ashley et al., 1999; Freni et al., 2008). The urban water system is very complex and whilst integration can simply be accomplished by combining individual model packages, achieving an overall sound system description requires not only knowledge of the sub-models, but also their interactions. Fundamental complexity theory would describe this in terms of the 'big picture' that cannot be seen simply as the sum of its parts (De Haan, 2006; Schmitt and Huber, 2006; Mitchell et al., 2007; Pahl-Wostl, 2007). Consequently, we encounter complexity on two possible fronts: (1) either through the introduction of additional behaviours into the model through interactions between components or (2) new emergent features that result from combining interacting sub-models, which we need to make sense of to ensure that our models are adequate representations of the real system. However, as we do not have a complete understanding of all possible interactions between various processes within the urban water system 'integration' should be regarded as the pathway for achieving model objectives rather than the objective itself. With this comes a suite of new modelling approaches.

Greater focus on interactions between overall system components has been frequently stressed as the primary characteristic of integrated models (Schütze et al., 1999; Meirlaen et al., 2002; Rauch et al., 2002a; Olsson and Jeppsson, 2006). Yet, decades have passed and we still lack sound knowledge on feedback loops in urban water systems. Moreover, we are limited in our choices of suitable algorithms or sub-models for integration due to (1) their excessive complexity (noted by Harremoës and Rauch, 1996; Muschalla, 2008 among others), (2) different aims at their time of development (Rauch et al., 1998) (integrating these models is therefore also an integration of objectives as was highlighted by Schütze et al., 1999) and (3) incompatible parameters and variables (e.g. different primary pollutants between models, different conceptualisation) – an ongoing issue as pointed out by Fronteau et al. (1997) and Erbe et al. (2002). Fortunately, research is addressing these and many other challenges, to make integrated approaches to systems analysis more feasible.

Over the past few decades, a growing and diverse body of integrated urban water modelling literature has emerged, uniquely influenced by historical development in different parts of the world, fraught with linguistic uncertainty (i.e. confusion with semantics due to a lack of transparent terminology as there have been contributions from a variety of academic disciplines) and with localised schools of thought. Consequently, we feel that researchers (particularly those not experienced in this field) delving into this literature may find it lacking clarity and confusing. At the same time, we perceive adoption of integrated models in practice appears to progress slowly. Even though many barriers against adoption have been identified 20 years ago (Lijklema et al., 1993), this perceived slow uptake could be attributed to difficulties of overcoming some of these barriers as well as new emergent challenges, which are documented, yet sprawled across the literature. As such, there is an urgent need for better order in this modelling discipline and an assessment of where these models may find value in practice, so that constructive changes in the current research trends can be made. We aim to clear some of the confusion across the literature in this review by approaching the field from the following aspects: (1) understanding the historical context of integrated modelling of urban water systems, (2) clarifying linguistic uncertainty by looking at how extensive the 'integration' term can be and has been used, (3) making clear distinctions between integrated and classical modelling and where overlaps can be identified, (4) the state of adoption of integrated urban water models (referred to in this review as the entire collection of integrated models of urban water systems regardless of scope and complexity) in research and practice and (5) their likely future role. This review also aims to establish a platform for communication across many disciplines that have been working in this field.

This critical review focuses only on urban water systems. Any mention of 'integration' therefore implies these systems unless otherwise mentioned. A lot of the reviewed literature is, furthermore, centred on the urban drainage systems (e.g. focussing on collection of both sewage and stormwater and their treatment) since it was historically the focal point of 'integrated modelling' in much of the early urban water literature. The authors, however, find many concepts developed in this sector useful for extending the integration principle beyond the urban drainage system, which is reflected in the title of the paper. An attempt will therefore be made to generalise many of these concepts to other parts of the urban water system where required. The authors are aware of the extensive body of literature on integrated river basin modelling and integrated ecological models and will draw upon these in the discussion where deemed relevant.

2. Brief historical overview

The first notable study that aimed to develop a combined understanding of several components of an urban water system was conducted in the late 1970s in the Glatt Valley, Switzerland (Gujer et al., 1982). Whilst this study did not specifically report any modelling results, it recognised the impact of wet weather events on secondary wastewater treatment processes and the temporal patterns of pollutant loads at every point of the integrated drainage, treatment and receiving water body systems. Acknowledging the lack of complex studies at the time, the authors promoted widespread adoption of integrated approaches. This statement has had far-reaching impact, evidenced by some of the most influential and highly-cited papers on the topic, e.g. Schütze et al. (1999),

Harremoës (2002), Rauch et al. (2002a; 2005), Erbe and Schütze (2005), Vanrolleghem et al. (2005a) and Mitchell et al. (2007).

The INTERURBA I conference (Lijklema et al., 1993) can be regarded as a milestone in research and development of integrated urban water models (Fig. 1). The conference focus was on the integrated analysis of sewer, wastewater treatment plant and receiving water systems – the *integrated urban drainage system* – as well as the benefits therein. Integrated real-time control (RTC) followed almost immediately (Linde-Jensen, 1993; Schilling et al., 1996; Schütze et al., 1999). Much of the basis for this was laid by the 'thought experiments' of Beck and Reda (Beck and Reda, 1994), who developed a dynamic model for wastewater treatment plant – river system interaction. Studies in Belgium also began integrating sewer system, treatment plant and river models to assess long-term water quality impacts on receiving waters according to four alternative water management scenarios (Bauwens et al., 1996; Vanrolleghem et al., 1996a). Further possibilities in integrated modelling of the urban drainage system were seen with the conception of the Activated Sludge Model No. 1 (ASM1) (Henze et al., 1993), which adopted process engineering approaches (e.g. mass balances and Gujer matrices) to describe dynamic biological processes.ASM1 not only formed the basis for many other wastewater quality models thereafter (Rauch, 2006), but also introduced new ways of thinking about integration among urban drainage modellers. Since 2000, legislation in Europe became more stringent. The EU Water Framework Directive (EU, 2000) places focus on the entire river basin. The directive requires achievement of 'good chemical and ecological status' in all natural water bodies within and advocates a combined approach comprising emission control through best available technology and best environmental practice, as well as environmental quality standards defined by annual average and maximum allowable concentrations (Ostrowski and Schröter, 2004; Achleitner et al., 2005; De Toffol et al., 2005; Candela et al., 2012). Meeting these requirements has had researchers and practitioners turn further towards integration. As a matter of clarity and semantics, ambient environmental (water) quality standards were also often referred to as 'immission' standards (derived from the German language) by several researchers (Bauwens et al., 1996; Ostrowski and Schröter, 2004; De Toffol et al., 2005; Erbe and Schütze, 2005; Vanrolleghem et al., 2005a; Reußner et al., 2008; Freni et al., 2010). The term, however, has not actively been used outside Europe.

The INTERURBA II conference was held in 2001 (Harremoës, 2002), with an expanded scope that included sustainable stormwater management. At the same time, River Water Quality Model No. 1 (RWQM1) (Reichert et al., 2001; Shanahan et al., 2001; Vanrolleghem et al., 2001) was developed to complement ASM1. Commercial packages for integrated modelling of selected components of the integrated urban drainage system, such as SIMBA (IFAK, 2007), WEST (Vanhooren et al., 2003), MIKE toolbox (DHI, 2009), MUSIC (eWater, 2011a) and Aquacycle (Mitchell, 2005) gained popularity and found adoption in practice.

Research then broadened its integration scope to fields of sustainable stormwater management and inclusion of economics (e.g. Hauger et al., 2002). With a broader scope came an increased need to understand, quantify and qualify uncertainties, not only within integrated models of urban water systems but also within the broader environment (Walker et al., 2003; Refsgaard et al., 2007; Freni et al., 2008; Matott et al., 2009). The feasibility to implement integrated urban drainage models in RTC, given major advancements in the last decade, also became a more active area of interest (Rauch and Harremoës, 1999; Meirlaen et al., 2002; Seggelke and Rosenwinkel, 2002; Erbe and Schütze, 2005). More recent studies present integrated modelling of the fate of heavy metals and organic micropollutants through urban catchments (for which there are only few measurements despite increasingly stricter regulations) and interactions with surface water, groundwater and even the atmosphere (Lindblom et al., 2006; De Keyser et al., 2010; Vezzaro et al., 2012).

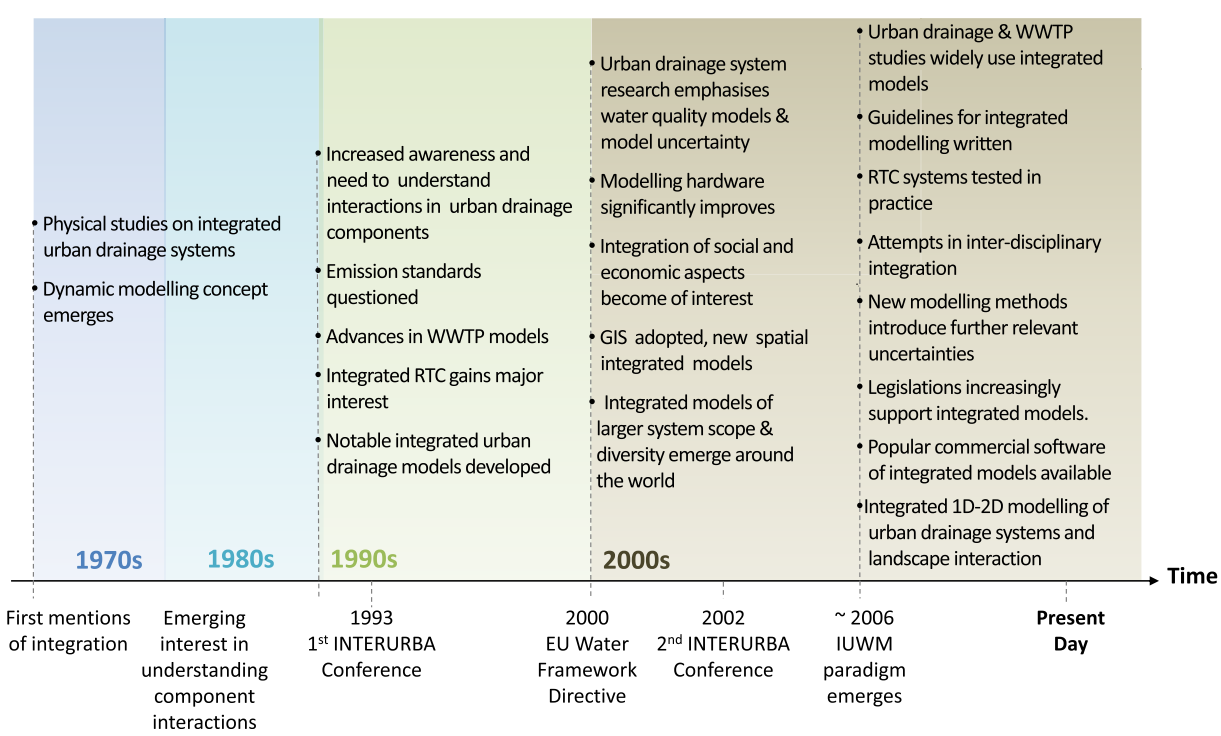

Fig. 1. Historical evolution of integrated modelling research from an urban drainage perspective.

There has been little mention of integration in the water supply literature. Studies in Luxemburg have focussed on RTC strategies to protect an important lake for water supply from combined sewer overflows (Henry et al., 2007; Regneri et al., 2010), but do not explicitly focus on the water supply system. We believe that two possibilities could explain this lack of mention in the literature: (1) perhaps due to the comparably more controllable nature of water supply and distribution systems, which for decades have been satisfactorily performing and are less influenced by natural phenomena in contrast to urban drainage systems (with exception of the reservoir, the storage volume of which fluctuates, but is not greatly impacted unless by severe climate extremes) and (2) different sub-systems of the water supply infrastructure are typically modelled at different temporal scales (e.g. reservoirs for inter-annual variability, distribution networks at sub-daily time-steps), which would imply that integrating these may not yield as many perceived benefits as is seen in the urban drainage sector. Recent water supply modelling does, however, feature integration aspects, due to a growing need to model alternative, often decentralised, water supply systems (such as, wastewater recycling and stormwater/rainwater harvesting) (e.g. Mitchell et al., 2001; Last, 2010) and to develop more complex and efficient water resource allocation strategies (e.g. Ducrot et al., 2011). This is, however, at the borderline of integrated river basin models, which is no longer limited to urban water systems and is beyond the scope of this review.

With new drivers for development such as rapid urban growth and needs for infrastructure rehabilitation and climate change adaptation, and stimulated by improvements in computational efficiency, integrated urban water modelling is currently expanding in new directions. Integrated 1D-2D modelling of the interactions between urban drainage systems and urban landscapes (i.e. streets, parks, etc.) during large pluvial flooding incidents has become possible in the recent decade (e.g. Bamford et al., 2008; Chen et al., 2010; Domingo et al., 2010), and modelling of more than the strictly technical and biophysical domain of the water system (e.g. social and economic domains) is increasingly attracting interest in recent years (e.g. Rauch et al., 2012; Ward et al., 2012).

3. Classifying Integrated Urban Water Models

3.1. Defining integration in urban water modelling

A single concise definition of integration in urban water systems modelling is difficult to formulate due to its complexity and broad scope. In fact, Parker et al. (2002) state that there is no generally accepted definition of "integration" in the broader environmental decision support literature. It is, however, possible to identify key aspects from experience in urban drainage modelling as well as recent and broader literature that are applicable to all aspects of the integrated urban water system.

A formal definition of integrated urban drainage modelling was suggested by Rauch et al. (2002a) as the "modelling of interactions between two or more physical systems". Although this definition stresses consideration of multiple components, their interactions and is fairly non-specific and mostly valid, it has become outdated, lacks other crucial aspects and is constrained to only the physical system, which has since been surpassed (Parker et al., 2002; Mcintosh et al., 2007; Moglia et al., 2010). Integration provides the ability to focus on understanding the behaviour of parts of the system with respect to the broader picture (Beck, 1976), considering relevant processes within each of the components (i.e. individual component complexity) and impacts they have on one another (i.e. relationships) (Marsalek et al., 1993; Erbe and Schütze, 2005). Modellers can consequently discover new emergent and quantifiable attributes and more efficient solutions, not attainable if examining systems individually as independent hydraulic services (Mitchell et al., 2007). Furthermore, integration also combines potential assessment of short-term (acute) and long-term (chronic and delayed) impact of processes (Rauch et al., 2002a), as well as water quantity and quality (point and diffuse source) aspects (Candela et al., 2012) in different parts of the system. This diversity of features implies highly specific modelling aims (potentially more specific than what classical modelling normally prescribes), thereby supporting their preferred use of integrated models as 'epistemological devices' and 'holistic tools' (for cost-effective research, management decision support, planning and policy making) contrary to hard engineering design and assessment (Mcintosh et al., 2007; Moglia et al., 2010). Seppelt et al. (2009), coming from ecology, adds further justification to this by listing the four major applications of integrated modelling: (1) exploratory modelling, (2) theory building, (3) scenario testing for participatory planning and (4) information provision for strategic planning and management. The fourth option, in particular, appears most prevalent in urban water systems (e.g. Nguyen et al., 2007; Vojinovic and Seyoum, 2008) and reflects how results are typically presented in much of the integrated urban water modelling literature.

Modelling case studies often show how integration achieve greater benefits than the traditional counterpart, whether in flood investigations (Bamford et al., 2008), receiving water body impacts (Beck and Reda, 1994; Bertrand-Krajewski et al., 1995; Bauwens et al., 1996), optimisation of processes or development of control strategies within a treatment plant (Vanrolleghem et al., 1996a; Grau et al., 2007a; Jeppsson et al., 2007) or much larger scale applications that involve urban planning and socio-technical disciplines (Biegel et al., 2005; Doglioni et al., 2009; Voinov and Cerco, 2010). An integrated urban drainage modelling case study, for instance, shows how reduction in CSO discharges does not necessarily guarantee ecological quality of a receiving water body (Reußner et al., 2009) due to the interactions between secondary clarifier at the WWTP and in-sewer/CSO storages. Conclusions frequently comprise qualitative roadmaps for strategic management as opposed to quantitative predictions.

It is thus possible to summarise the integration principle in urban water systems modelling as three key points:

(I) modelling of a multitude of components (biophysical, economic and beyond) and interactions between components,
(II) consideration of acute, chronic and delayed impacts of water quantity and quality processes in a long time period of simulation and
(III) ability to see both local processes and the global 'big picture perspective' to better inform decision-making, policies or scientific knowledge.

These aspects are believed to be applicable across the entire urban water system and not limited to one specific scope.

3.2. Proposed typology for classifying integrated urban water models

Despite apparent consensus on defining 'integration', there is, nevertheless a major diversity of unique integrated modelling approaches in the growing body of literature, which makes it challenging to perceive what integration in urban water systems actually encompasses (as already discussed in the Introduction). 'Integration' is an ambiguous term as it evokes a human perception of 'completeness' that is somewhat nonsensical as models are only partial representations of reality (Bertrand-Krajewski, 2007). The principle of parsimony stressed throughout the literature further

cautions against this perceived aim of 'completeness'. We generally *do not*, and with our lack of 'complete' understanding *cannot*, model our systems in their entirety. An effective typology for classifying these models can help researchers understand the diversity of integration approaches in the literature and provide some order and consistency to guide future research.

As an alternative to simply using 'integration' (also circumventing this misleading perception of 'completeness'), different 'degrees of integration' were initially suggested by Rauch et al. (2005). Several studies have framed these 'degrees' into typologies for integrated models (e.g. Rauch et al., 2005; Olsson and Jeppsson, 2006; Candela et al., 2012). The proposed typologies differ in the physical, disciplinary and administrative boundaries, as well as spatial and temporal scales involved. These were reviewed for their respective strengths and weaknesses. Rauch et al. (2005) expand integration beyond the urban drainage system and modelling to include public involvement. This is an incomplete view, not only because it is urban drainage centric, but also because it only distinguishes between types of integration based on scope and not on other aspects like model development techniques, calibration, validation, uncertainty and the contexts in which the models can be used. Olsson and Jeppsson (2006) introduce two terms in what they refer to as the field of 'integrated urban wastewater systems' that characterise integrated models as plant-wide (integration of processes within a wastewater treatment plant where water and sludge lines are considered) or system-wide (integration of several sub-systems, e.g. catchment, sewer system, wastewater treatment plant and receiving water body). Despite originating from wastewater treatment literature, these concepts are transferable to other urban water sectors. 'System-wide' has, however, become a vague description of a significant collection of integrated models of varying scopes that have emerged in the last decade and should preferably be refined. The typology provided by Candela et al. (2012) emphasises a more detailed representation of either point, non-point source pollution or both in the integrated model. Three categories, *urban-integrated models* (focus on point pollution, non-point pollution modelled in a simplified way), *river basin-integrated models* (focus on non-point pollution, point pollution modelled in a simplified way) and *fully integrated modelling approaches* (equal importance placed on representing both types of pollution) are suggested. Although this is widely applicable in the European context where treatment plant and combined sewer overflow discharges are of major concerns, the typology is, however, not very descriptive when water quality issues are not a primary concern in integrated modelling (e.g. when assessing available quantities of water for reuse as part of the total urban water cycle or levels of inundation in assessment of flooding). In short, existing typologies are not ontologically complete. Although they appear to be generic, they either lack the required level of detail to classify integrated models in the urban water system or the flexibility to expand upon each category and accommodate new innovations in integrated model development research.

To deal with the shortcomings of existing classifications, the authors propose a typology of integrated urban water models (Fig. 2) that is based on (i) physical and institutional system delineation, (ii) model complexity, and (iii) differences between their development philosophies (an extension of work by Rauch et al. (2005) but all-encompassing of previous efforts). At the lowest level, we focus on *components* of urban water *sub-systems* (e.g. an activated sludge tank of a WWTP) and *sub-systems* of the urban water *stream* (e.g. sewer network of an urban drainage stream/sector). These *streams* are part of the *total urban water cycle* and encompass the physical urban water *infrastructure*. When combined with other urban water-related *infrastructures* (institutional or drawing from other disciplines in relation to urban water e.g. climate impacts or ecology), these are integrated to the level of the *urban water system*. Based on these key terms, four different integration levels are defined:

(I) *Integrated Component-based Models* (ICBMs) represent the lowest level of integration and focuses on the integration of *components within* the local urban water *sub-system* (e.g. coupling several treatment processes within a wastewater treatment plant). These are analogous to *plant-wide* models, cf. the discussion above, but we generalise the term to also refer to other urban water *sub-systems* (e.g. pipe network or a natural water body).

(II) *Integrated Urban Drainage Models* (IUDMs) or *Integrated Water Supply Models* (IWSMs) that integrate *sub-systems* of either the urban drainage (wastewater and/or stormwater depending on location in the world) or water supply *streams*, particularly treatment and transport processes. These and higher categories are best described based on the existing literature as a finer breakdown of *system-wide* models.

(III) *Integrated Urban Water Cycle Models* (IUWCMs) that links these two *streams* (IUDMs and IWSMs) into a common framework to encompass what literature has referred to as the *total urban water cycle*.

(IV) *Integrated Urban Water System Models* (IUWSMs) is the highest level of integration that combines the different urban water *infrastructures* (institutional or physical) and *disciplines* (e.g. climate, economics, actor behaviour, etc.) of the *total urban water cycle*. Each of these will exert different impact on the *urban water systems*. The focus, however, remains explicitly on urban water.

From categories I to IV, the system scope becomes broader, the spatial scale becomes greater and the managerial problem of a higher level and more inter-disciplinary. The whole typology of Integrated Urban Water Models sits within the broader field of *Environmental Decision Support Systems* (EDSSs) alongside the greater river basin models. At the EDSS level, focus shifts from urban water systems to the broader environment, which can encompass a variety of aspects (e.g. ecology, climate, economics, etc.). This typology offers an enhanced nested continuum to classify integrated models accordingly. ICBMs, for example, are integrated models of single urban water *sub-systems*. These are nested within IUDMs and IWSMs, which means that these same sub-systems may also be appear and be integrated with other sub-systems in these larger *streams*, but not necessarily represented to the same level of detail as is done at the ICBM level. The typology frames the models of the integrated urban water system in their broader environmental decision-support context and is by no means intended to replace existing ways of describing integration (e.g. the broader idea reported by Parker et al., 2002). Consideration of combined water quantity and quality assessment and/or level of stakeholder and public involvement should be independent of category as these can be part of any of the four suggested types of integrated urban water models. However, facilitating stakeholder involvement across different integration levels is an 'integrative' task itself as the diversity of perspectives, interests, values and needs of different actors increases with complexity of the problem.

A model should generally be classified according to its highest level of integration (since lower levels are nested within this). For example, if a model can simulate rainwater/stormwater harvesting and reuse within an urban catchment, it would be classed as an IUWCM. A water supply model comprising reservoirs, distribution networks and resources allocation dynamics based on social and economic sub-models would be classed as an IUWSM. Removing

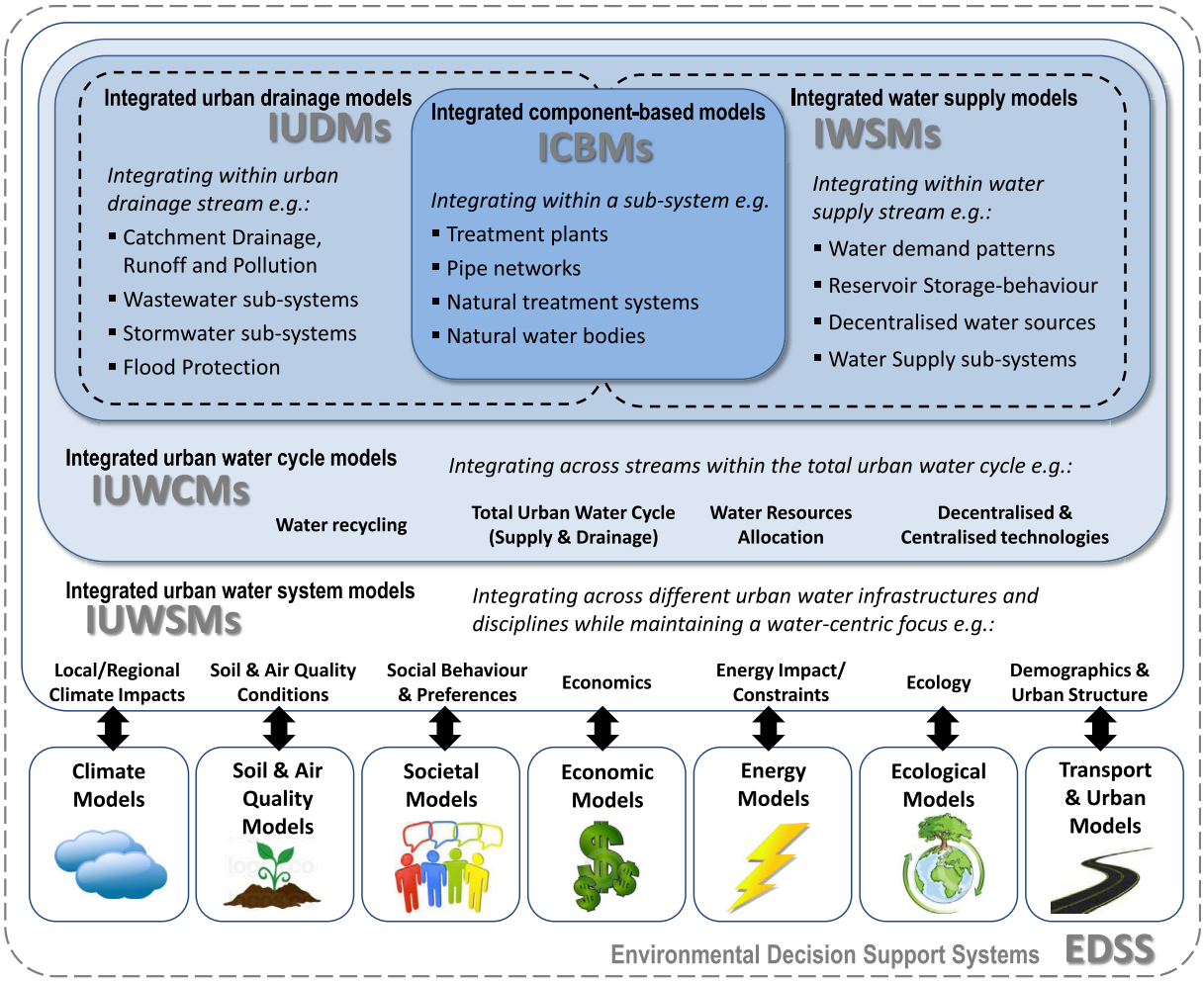

Fig. 2. Suggested typology for classifying integrated urban water models in the broader context of environmental decision support systems.

the social and economic components would, however, reduce it to an IWSM. It is also important to note coherence across models when classifying these. There should be consistency of complexity across interacting models in terms of the representation of the phenomena they are modelling. The issue of balancing complexity is also discussed later in the context of model development.

3.3. *Exemplification of the typology for existing software packages and documented case studies*

There are two kinds of approaches that studies on integrated modelling feature: (1) use of existing integrated modelling software packages and (2) combination of different existing models into one larger integrated assessment tool. The proposed typology for integrated urban water models can be applied in both cases and is demonstrated for both study types in Tables 1 and 2, respectively. Table 1 applies the typology to existing software packages using a transparent model review framework inspired by Elliott and Trowsdale (2007), and Table 2 applies the typology to a range of case studies documented in the literature.

3.3.1. *Integrated Component-Based Models (ICBMs)*

ICBMs encompass "whole-of-sub-system" or "plant-wide" integration. The idea stems from Vanrolleghem et al. (1996b), who discussed the idea of a more holistic approach to treatment plant design in light of changing legislation that would eventually follow from the first INTERURBA conference (Tyson et al., 1993). More details on achieving 'plant-wide' control followed a decade later (Olsson and Jeppsson, 2006). Integrating the water and sludge lines of a wastewater treatment plant and considering all processes from pre- to post-treatment would constitute an ICBM and has been extensively investigated before (Grau et al., 2007b; Jeppsson et al., 2007). Benchmark Simulation model No. 2 (BSM2, Tables 1 and 2) is a key example of an ICBM that refers to itself as an integrated model developed for simulating and optimising an entire wastewater treatment plant's performance. On the water supply side, the Stimela software (Van Der Helm and Rietveld, 2002; Rietveld et al., 2010) is an ICBM that considers different drinking water treatment processes (e.g. ozonation, pellet softening, etc.). Rietveld et al. (2008) demonstrate how Stimela can be used to optimise plant performance through scenario analysis.

BSM2 (Grau et al., 2007b), for example, considers processes such as activated sludge, clarifiers and anaerobic digesters. Each has its own well developed sub-model: Activated Sludge Model No. 1 (Henze et al., 1993), the popular secondary settler model (Takacs et al., 1991) and Anaerobic digester model no. 1 ADM1 (Batstone et al., 2002). They are linked in a common framework to provide specific information about each process within a wastewater treatment plant, which is then used for decision-making based on short-term and long-term impacts. Combining these models involves consideration of their interactions and feedbacks (e.g. hydraulic disturbance on biochemical processes), which is key to integrated models (Grau et al., 2007a). It has also been stressed by researchers that further work on such "plant-wide models" is

Table 1
Applying the new typology to different software packages used for integrated modelling.

Software	Reference	Type	Urban Water Processes									Urban Water Components					Types of Model Applications							
			Hydrology	Hydraulics	Pollution	Treatment	Downstream Impact	Storage-Behaviour	Water Consumption	Groundwater Interaction	Flooding	Water Transportation Network	Treatment Plants	Decentralised Technologies	Receiving Water Bodies	Built Environment	Assessment of Water Recycling	Operation & Control	Risk & Impact Assessment	Social Implications	Economics	Ecological Implications	Conceptual Design	Strategic Planning
BSM2*	Grau et al., 2007b	ICBM				■		■					■					■			■		■	■
EPANET	Rossman, 2000	ICBM	■	■				■	■	■		■						■	■				■	
Stimela	Rietveld et al., 2010	ICBM				■							■					■			■		■	■
InfoWorks CS	MWH Soft, 2010	IUDM	■	■	■	■		■			■	■	■					■	■				■	
SIMBA*	IFAK, 2007	IUDM	■	■	■	■	■	■				■	■		■			■	■		■		■	
SWMM	Rossman, 2004	IUDM	■	■	■						■	■		■				■	■				■	
SYNOPSIS	Schütze et al., 1999	IUDM	■	■	■	■		■				■	■					■	■				■	
WEST*	Vanhooren et al., 2003	IUDM	■		■	■	■	■				■	■		■			■			■	■	■	
CALVIN*	Lund et al., 2009	IWSM	■	■				■	■	■		■				■		■	■	■	■			■
CityDrain3	Burger et al., 2010	IUWCM	■		■	■	■	■					■	■	■		■	■					■	■
Aquacycle	Mitchell et al., 2001	IUWCM	■					■	■	■				■			■							■
City Water Balance	Last, 2010	IUWCM	■		■	■	■	■	■	■				■		■	■						■	■
MUSIC*	CRC-CH, 2005b	IUWCM	■		■	■	■	■	■	■				■			■		■		■		■	■
MIKE URBAN	DHI, 2009	IUWCM	■	■			■	■	■		■	■		■		■		■	■				■	■
UVQ	Mitchell and Diaper, 2005	IUWCM	■		■	■	■	■	■	■				■			■							■
UrbanCycle	Hardy et al., 2005	IUWCM	■					■	■	■				■			■							■
UrbanDeveloper	eWater, 2011	IUWCM	■				■	■	■			■		■			■		■				■	■
DAnCE4Water	Rauch et al., 2012	IUWSM	■	■	■	■	■	■	■		■	■	■	■		■	■		■	■	■			■
ReVisions	Ward et al., 2012	IUWSM	■					■	■					■		■	■			■	■			■
VIBe	Sitzenfrei et al., 2010	IUWSM	■	■				■	■			■				■		■	■					■

*BSM2, SIMBA, WEST, MUSIC and CALVIN are examples, which can perform or incorporate simple economic assessments (operational costs or life cycle costing), but are classed at lower integration levels because the economic algorithms are not highly integral to their operation. WEST additionally has capabilities to model ecosystem impact

Table 2
Applying the typology to different integrated modelling case studies in the literature.

Case study	Model (integration)	Description	Class
Jeppsson et al., 2007	BSM2	12 control case studies of a heavily loaded WWTP, showed many strategies sub-optimal, better optimum through integration	ICBM
Harremoës and Rauch, 1996	SAMBA, RUMBA, FOXTROT	Shows the possibilities of integrated control in controlling accumulative nitrogen pollution of receiving water bodies.	IUDM
Bauwens et al., 1996; Vanrolleghem et al., 1996a	KOSIM, ASM (IAWQ No. 1), SALMON-Q	Both studies investigate various design and operation alternatives for an urban wastewater system to minimise immission impacts.	IUDM
Holzer and Krebs, 1998	AQUASIM, MOUSE	Assessed effect of different retention tank technologies on peak ammonium discharges to receiving water bodies.	IUDM
Milina et al., 1999	SIMBA	Integrated model to investigate effects of upgrading local WWTP, number of alternative strategies tested to assess how extensive upgrade should be	IUDM
Seggelke and Rosenwinkel, 2002	KOSIM, ASM2d	Integrated control case study at a pilot plant, showed how extra capacity of the system could be mobilised during increased stormwater flows	IUDM
Butler and Schütze, 2005	KOSIM, ASM1, DUFLOW (SYNOPSIS)	Demonstrated benefits of integrated RTC over local RTC for improving river water quality through control of diurnal patterns of WWTP and CSO discharges	IUDM
Solvi et al., 2008	KOSIM, ASM2d, RWQM1	Conducted scenario analysis of emission-immission-based water quality management strategies in a semi-rural catchment	IUDM
Domingo et al., 2010	MIKE FLOOD, MIKE SHE, MOUSE	Coupled catchment hydrology with detailed drainage network model for urban flood analysis to account for wider array of factors causing floods	IUDM
Schellart et al., 2010	STORMPAC, MIKE NAM, InfoWorks CS, ICM	Assessed quantity and quality issues of urban drainage system, performed novel method of uncertainty and sensitivity analysis with the model	IUDM
Schindler et al., 2010	SIMBA-Sewer, EPA SWMM	Simulated sewer, WWTP and river body over 10 years to assess extreme statistics of peak-over threshold for water quality.	IUDM
Hardy et al., 2005	UrbanCycle	Investigated impact of residential allotment runoff in Western Sydney, modelled four different mitigation scenarios	IUWCM
Mitchell, 2006	Aquacycle	Used Aquacycle to indirectly assess micro-climate benefits of various decentralised strategies in a residential development	IUWCM
Doglioni et al., 2009	Agent-based method, SWMM, ASM1	Decision-support on the effects of land development on wastewater infrastructure, different land cover change scenarios modelled	IUWSM
Fagan et al., 2010	A number of different models, life cycle assessment and kinetics in a dynamic framework	Integrates numerous sub-models from water supply, stormwater, energy, greenhouse gas emissions, economics and environmental impacts assessment to model a large sustainability project in a Melbourne suburb	IUWSM
Urich et al., 2013	VIBe, SWMM, UrbanSim	Assessed flooding in a number of virtual alpine cities given different urban development and climate change scenarios	IUWSM

envisioned to expand into the greater urban drainage system (Olsson and Jeppsson, 2006; Breinholdt et al., 2008).

On the water supply side, two examples that fit this category are EPANET2 (Rossman, 2000) and AQUIS (Schneider Electric, 2012), outlined in Table 1. Being mechanistic models, their parameters have a sound physical basis, allowing them to be successfully calibrated and satisfactorily fulfil their objective of water supply system simulation. EPANET2 is more focussed on the specific state of the network for a given situation, but includes numerous additional features such as water quality assessment tools. AQUIS has already been extensively adopted in model-based real-time management applications in numerous cities across the globe (Schneider Electric, 2012).

3.3.2. Integrated Urban Drainage Models (IUDMs) and Integrated Water Supply Models (IWSMs)

IUDMs have been one of the most well-known and recognised forms of integrated models (especially in Europe, cf. the description in Section 2), and many current integration principles are the result of the plethora of IUDM studies. These typically comprise studying upgrade options for a local WWTP, assessing ways of reducing CSO emissions or showing the combined impact of different parts of the drainage stream on receiving waters (e.g. Bertrand-Krajewski et al., 1997; Meirlaen et al., 2001; Rauch et al., 2002b; Dempsey et al., 2008; Schindler et al., 2010). They recognise both combined and separate drainage systems. Studies (such as those in Table 2) have taken this experience further in developing real-time control (RTC) strategies (e.g. Bauwens et al., 1996; Seggelke and Rosenwinkel, 2002; Butler and Schütze, 2005; Tränckner et al., 2007) and even testing RTC pilot systems (e.g. Schilling et al., 1996; Wiese et al., 2005, Seggelke et al., 2005). Achieving truly integrated RTC, however, is still a reasonable challenge as it involves measurements and actions across different sub-systems and therefore necessitates parallel simulation of different sub-system models (Vanrolleghem et al., 2005a). Pleau et al. (2005), in fact, suggest that despite a rich theoretical basis, reasons for not readily adopting model-based global control in urban drainage could include limited computational speed and accuracy of hydrological and hydraulic models among other hardware reasons. The integration of these models can nevertheless be achieved either through building one common platform or through connected softwares (using frameworks e.g. OpenMI). OpenMI (Gregersen et al., 2007; OpenMI Association, 2010) is not an integrated model, but rather a standard for data exchange between models and modelling components. Its platform has been used for the development of highly integrated models such as FluidEarth (OpenWebAdm, 2013) and other studies (Bamford et al., 2008; Reußner et al., 2008; Fotopoulos et al., 2010). Advancements made in IUDM, due to institutional driving forces have led to the development of several key software packages (listed in Table 1). Whilst most of these packages originate from the integrated urban drainage field, it is not surprising to find that some are starting to encompass larger scopes of integration, such as the incorporation of decentralised technologies in SWMM (Rossman, 2004) or an ecosystem model in WEST (De Laender et al., 2008).

The typology should therefore not be regarded as a strict categorisation of models, but rather a hierarchy, which models can move through. A software package that begins as an IUDM can thus eventually become an IUWCM if it gains new features that fall under this category.

To ensure that the typology is ontologically complete and makes a clear distinction between supply and drainage streams, *Integrated Water Supply Models* (IWSMs) were coined and are of the same complexity level as IUDMs. As a fundamental difference, water distribution models, in general, neglect temporal dynamics, i.e. they compute flow and pressure in the network only for a given situation. Temporal dynamics are computed stepwise, referred to as quasi-dynamic simulations. Integration across water supply components builds knowledge on specific situations (e.g. influence of maximum daily demands on distribution network performance) and long-term dynamics (e.g. fluctuation in reservoir levels over several months or years, changes to pumping requirements as a result of seasonal demands). Results can be used to develop or revise resource management strategies. Although this kind of modelling already exists in practice and is often not referred to as integrated modelling (but rather simply as *models, decision-support systems* or *software packages*), underlying methods align with integration principles as already seen.

Examples of IWSMs in the literature are scarce (as supply-side integration has not been popular, see Section 2), but this distinct category can provide a place in the typology for organising future research. CALVIN (Jenkins et al., 2004; Lund et al., 2009), although described as a large integrated decision-support system for the California river basin, is one of the only models found that can be classified as an IWSM since it integrates models of surface and groundwater reservoirs, major aqueducts, rivers, water demands and pumping and power plants. In the water supply resources literature, it is also described as a 'hydro-economic model' – integrated models that represent water resource systems, infrastructure, management options and economic values (Harou et al., 2009). As mentioned previously, these models are at the borderline of integrated river basin models and beyond the scope of this review. For more information, Harou et al. (2009) have conducted an extensive review of over 80 such models with similar integration philosophies as described here.

3.3.3. Integrated Urban Water Cycle Models (IUWCMs)

The *urban water infrastructure* refers to the merging of supply and drainage streams of urban water management. This combination often results in models for assessing water recycling strategies using combinations of central- and decentralised technologies or more complex water resources allocation. Integration principles in Australia have been primarily applied in this context, giving rise to packages like Aquacycle (Mitchell et al., 2001), MUSIC (eWater, 2011a), Urban Developer (eWater, 2011b) and UrbanCycle (Hardy et al., 2005).

Despite their Australian-centric development (i.e. integration and model development at the time was driven by widespread support for water recycling practices due to an ongoing drought), knowledge gained from developing and using Aquacycle (Mitchell et al., 2001) and UVQ (Mitchell and Diaper, 2005) has been adopted in a recent model, City Water Balance (Last, 2010), which was applied to the city of Birmingham to investigate the potential of decentralised water management by means of water sensitive urban design. This model (presented in Table 1) was able to illustrate the impact that a wide range of strategies had on the urban water system (e.g. medium scale rainwater harvesting and borehole abstraction on centralised supply) and is recommended for use as a scoping tool for strategic management options.

At this level of integration, uses of these models have typically become more focussed on strategic planning and conceptual testing of sustainable solutions in urban water management rather than devising detailed control strategies. This is not to say that all models that go beyond drainage or supply streams have to be used in this manner. Furthermore, modelling aims and local context in which these models are developed may differ, but it appears that knowledge on how to accomplish integrated modelling at this level is transferable.

3.3.4. Integrated Urban Water System Models (IUWSMs)

The highest level of integration in the urban water system draws links with other environmental, social and economic models with a focus on water-related issues. IUWSMs emphasise the role of water in the urban environment, but adopt an inter- and multidisciplinary perspective in describing this problem. Social, economic, atmospheric, energy and other aspects are considered where relevant. To the authors' knowledge, until today, there are only a few published studies that truly integrate at this level. The models developed by Fagan et al. (2010), the VIBe framework (Sitzenfrei et al., 2010), ReVISIONS framework (Ward et al., 2012), UrbanBEATS (Bach, 2012) and DAnCE4Water (Rauch et al., 2012) are considered IUWSMs as they combine a number of other different aspects (e.g. economics, social, urban form, energy and a number of sustainability indicators) to urban water modelling. Additionally, a number of major international projects are currently working at this level of integration, the EU-FP7 projects PREPARED (Prepared, 2010) and CORFU (CORFU – FP7, 2012; Verbeek et al., 2012) being two examples.

Beyond this, the urban water system finds itself within much broader river basin models, ecological models and within the field of EDSSs that do not necessarily place the urban water system under direct focus, as already discussed.

4. Developing Integrated Urban Water Models

Having proposed a typology for classifying integrated urban water models, we now address several key topics related to model development including key features, development steps, calibration, validation, optimisation and uncertainty.

4.1. Key model features – the quest for parsimony and pragmatism

The nature and structure of integrated models developed over the past decades are diverse (reflected in different integration levels presented previously). There are diverging opinions as to which features an integrated model should comprise. A lot of these features are initially chosen based on modelling aims and objectives, ranging from model structure to purpose, data requirements and availability, computational issues, spatial, temporal and process nature issues (Table 3). They are usually iteratively refined at various stages of model development by striking a balance between study objectives, data requirements and available models to integrate (Rauch, 1995; Schütze and Alex, 2004).

Whilst an important consideration of any modelling exercise is the issue of parsimony (or 'Occam's razor') (Wagener et al., 2004), one of the highly emphasised aspects in integrated model development in addition to this is the need for pragmatism (Harremoës, 2002; Butler and Schütze, 2005; Olsson and Jeppsson, 2006; Bertrand-Krajewski, 2007). In fact, the term 'pragmatism' is stressed more frequently in the integrated modelling literature than in classical modelling; e.g. in Rauch et al. (2002a), Willems (2006), Vojinovic and Seyoum (2008) and Schellart et al. (2010). It reminds us that focus should be on simulating the 'big picture'

Table 3
Key model features requiring consideration during integrated model development.

Category	Modelling options	Description
Data requirements & availability	• Quantitative • Qualitative	Types of data that have to be obtained based on their purpose and contribution (e.g. calibration, validation, reality checking) to the study
	• Spatial resolution • Temporal resolution	Amount of data required both in space and time. Determines logistics and intensity of measurement campaign
	• Accuracy • Reliability	Ensuring understanding of the collected data set, potential errors and uncertainties therein to avoid surprises during modelling later on.
Model structure	• Empirical • Conceptual • Mechanistic	The extent of simplification of real-world process representation – from laws of physics to analogous representation to fully data-driven representations
	• Primary • Surrogate	Simpler/surrogate algorithm may be chosen to act as substitutes to their primary counterparts to reduce computational burden of model (e.g. in RTC)
	• Deterministic • Stochastic	One input always provides the same output, or an attempt to account for natural variability by introducing randomness
Simulation configuration	• Sequential • Parallel	Each process in the model runs for whole time series one after another or whole model simulates all processes a time step at a time, then moves to the next
	• Online (nowcasting) • Offline (fore-/backcasting)	Results required immediately to control operation or try to fill in historical gaps or predict possible futures
Spatial detailing	• Branched • Looped	Information flow is uni-directional (one-way interaction) or bi-directional (two-way interaction)
	• Distributed • Lumped	Processes consider variability across system or assume system is homogeneous.
Temporal detailing	• Continuous simulation • Discontinuous simulation	Simulating processes over an extended time-series or only using individual key events
	• Uniform time step • Variable time step	Same time step across entire simulation period or alternative time steps for e.g. dry and wet weather period
Process nature	**Quantity**: Hydrology Hydraulics	Quantity is frequently considered, extent of detail required is normally considered depending on scope and aims
	Quality: None Physical Chemical Biological	Quality is sometimes considered. Physical, chemical and biological processes taken into account, determine transformations of pollutants in the system. Combination of quality processes depend on modelled component.
Computational power	• Single-core processing • Multi-core processing	Calculations carried out on one single computer processor core or a network/grid of multiple cores
	• Single-run • Optimisation • Scenario analysis	Computational power devoted to a single model run, finding a possible optimum or to exploring possible outputs to get statistics
Software development	• Supermodel • Interface • Hybrid	All algorithms coded into common language and placed in one software package or wrapped by a container and made compliant with each other. Both can also occur.

instead of each individual process in great detail (Harremoës and Rauch, 1996).

Despite similar meaning, 'parsimony' and 'pragmatism' are complementary. Parsimony advocates the simplest solution that reliably answers the question and sits within a highly technical dimension. Pragmatism, on the other hand, should be seen in the human dimension, dependent on the modeller's judgement. Rather than the perceived idea of achieving 'completeness' implied by 'integration' (as discussed earlier), it can be suggested that integrated model development is a quest for complexity (in this case the amount of "causes" or factors/sub-models of which interactions need to be considered to explain the "effect" or outcome rather than intricacy of each individual algorithm). Nguyen et al. (2007) suggest, that while model complexity needs to be manageable, a model that is too simple may provide useless (perceived here as 'highly uncertain' or 'insufficient') information to decision-makers. A good modeller should therefore play along Occam's Razor by making pragmatic decisions on when complexity suffices in model development and not be mislead by the need to achieve 'completeness'.

4.2. Key development steps

We distinguish between building the model (model structure, equations, data requirements, etc.) and testing the model (in a well-defined calibrated exercise, a scenario assessment or Monte Carlo approach). The whole development process from building integrated urban water models to testing them (in preparation for their application) should typically follow a systems approach (Bertalanffy, 1971), similar to classical modelling philosophy. Clearly defined aims and objectives and an initial assessment of data availability are required before making a selection of potential model features (listed in Table 3). They also influence the likely methodology for model application. Issues of model structure (e.g. empirical/conceptual/mechanistic, deterministic/stochastic) and process nature (hydrodynamic/hydrologic, physical/chemical/biological) need to be considered primarily for model building. However, simulation configuration (sequential/parallel, online/offline), and computational power (single-core/multi-core, single-run/optimisation/scenario analysis) are more heavily influenced by how the model will be tested and ultimately applied. Data collection (quantitative/qualitative, spatial/temporal resolution, accuracy/reliability) is prevalent throughout the entire process, comprising data about key processes for model building and data on diverse case studies for calibrating/validating/testing the model and for its subsequent application. Overall, an iterative process is unavoidable, as revisiting the choice of model features may occur throughout various development steps. Several publications have aimed to outline a transparent methodology for building integrated models, e.g. Marsalek et al. (1993), Beck (1997), Poch et al. (2004) and HSGSim (2008). Many are similar, but vary in level of detail. A generally agreed upon methodology at INTERURBA I (Lijklema et al., 1993) was later updated by publications in the late 90s (Schilling et al., 1997; Rauch et al., 1998; Vanrolleghem et al., 1999)

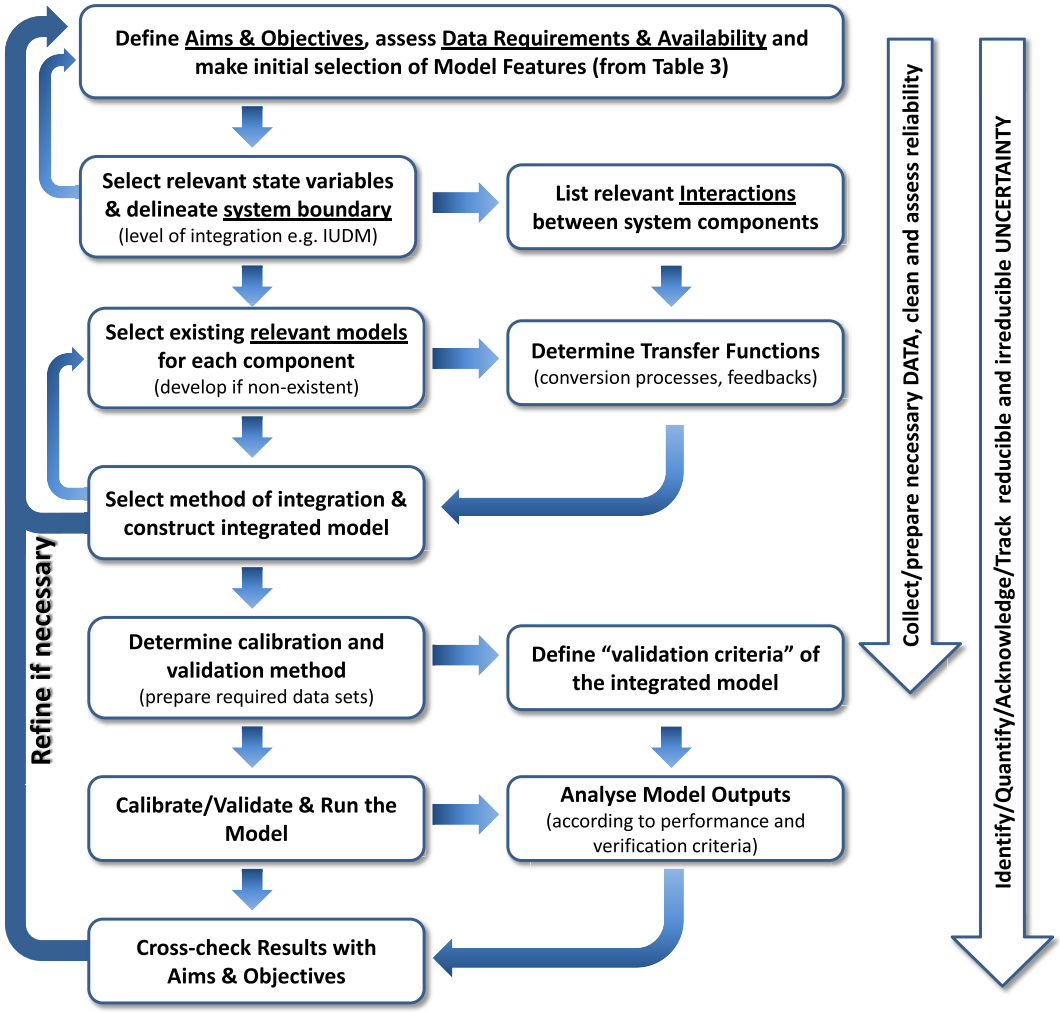

Fig. 3. Suggested universal approach to developing integrated models of the urban water system (from model building to application) collated from recommendations in the literature.

with an emphasis on data requirements. The Central European Simulation Research Group (HSG) furthered this work (Muschalla et al., 2008) and have published an extensive guide. Key steps are similar across all publications (a generalisation of which is suggested in Fig. 3) and should be applicable to all the model categories.

It is not the aim of this paper to explain each of the steps in great detail (please see the above-mentioned publications), but to emphasise key considerations. Defining aims and objectives is an intrinsic feature of classical modelling (Box and Draper, 1987; CRC-CH, 2005; Bertrand-Krajewski, 2007) and should be no different with integration. These will, however, determine and be partly determined by whether online or offline simulation is required and whether the ultimate goal involves searching for a global optimum (e.g. in design) or exploring possibilities of model outputs (e.g. strategic planning). Model structure considerations, spatial detailing and the nature of processes to include in the model are investigated when selecting relevant state variables and existing relevant models. Muschalla et al. (2008) suggest undertaking a *deficit analysis* (identifying negative factors to optimise the system against) to better accomplish these steps.

The degree of spatial detail is particularly challenging (Mitchell et al., 2001; Schmitt and Huber, 2006; Vojinovic and Seyoum, 2008; Heusch et al., 2010) and is dependent on which interactions are considered (a step between choosing variables and existing models). Flow of information will involve either the branched (one-way interaction, e.g. catchment runoff impacts sewers, which impact WWTP processes and then the receiving water body) or looped options (two-way interaction, e.g. alternative water sources, which alter the potable water demand and therefore impact, all the way back through the system, on the supply reservoirs). The latter is often preferred for information flows in the model when real time control is involved. To manage complexity when choosing sub-models, two issues are important: (1) compatibility at interfaces (e.g. the WWTP model components simulate the same pollutants) and (2) balanced complexity across structure of sub-models (Cierkens et al., 2012) (i.e. there is little benefit in using a highly complex and accurate upstream model when followed by a simpler, less accurate downstream model, Erbe et al., 2002).

Integration of models is a simulation and software question. Sub-models either run for the entire time-period before information is conveyed on to the next component (sequential) or models are run alongside each other (parallel) (Erbe et al., 2002; Schmitt and Huber, 2006). Limitations of sequential simulation have been recognised, especially where incorporating feedback in the system is necessary (Erbe et al., 2002; Rauch et al., 2002a) such as the evaluation of strategies for RTC examples (Vanrolleghem et al., 2005a). Sub-models can either be coded in a common language

and environment (supermodel approach) or by establishing links between individual software (interface approach) (Bauwens et al., 1996; Meirlaen et al., 2002; Schütze and Alex, 2004; Bamford et al., 2008; Reußner et al., 2008).

Despite this adequate understanding of what is involved in integration, it is still a major challenge (Fronteau et al., 1997; Erbe et al., 2002; Rauch et al., 2002a; Vanrolleghem et al., 2005b; Grau et al., 2007a; Reußner et al., 2009). This may be due the following: (1) variables change their "meaning" from one component to another, or simply do not exist in all sub-models and (2) conversion processes at the interfaces are tedious and error-prone (chemical processes in particular, where the chemical mass balance cannot be easily solved). These issues also primarily relate to water quality rather than quantity. There have been advancements in addressing the conversion issue with the use of 'Petersen matrices' (recently referred to as 'Gujer matrices') as a reference for the model (Vanrolleghem et al., 2005b). The problem of state variables and their "meaning" is one of communication rather than science – therefore advocating pragmatism while still maintaining scientific rigour (e.g. adhering to mass continuity).

A final consideration that is made throughout the different development stages involves computational hardware requirements. Rapid advancements in technology have allowed for exploration of possibilities for increasing computational efficiency through single-core and multi-core options on individual or clusters of computers. The subject is investigated in the context of integrated modelling in some studies (e.g. Barreto et al., 2008; Benedetti et al., 2008; Burger et al., 2010). Nevertheless, modellers must remain parsimonious as well as pragmatic and work within the scope of modelling aims, using this newfound computational power to improve the efficiency of entire modelling process shown in Fig. 3 rather than the model software itself.

4.3. Integrated model calibration, validation and optimisation

The calibration and optimisation of integrated urban water models share similar difficulties, which have been widely acknowledged (Beck, 1997; Vanrolleghem et al., 1999; Rauch et al., 2002a; Vojinovic and Seyoum, 2008). Three possible approaches are recommended by literature (Erbe et al., 2002; Olsson and Jeppsson, 2006; Freni et al., 2008): (1) calibrate/optimise the whole integrated model at once (often a very difficult and sometimes impossible task), (2) begin calibrating/optimising only the upstream models (water quantity followed by quality) and gradually add the next immediate downstream component and (3) calibrate/optimise individual sub-models separately before integrating them. The third approach, however, has been argued as detrimental for the overall outcome by both the water sector (Rauch et al., 2002a; Voinov and Cerco, 2010) and the broader modelling community (Hodges and Dewar, 1992), since it is not known how the remaining errors present in the sub-models will behave upon integration. This third sub-optimisation approach has also been shown to falter at the smallest integration level, the ICBMs (Jeppsson et al., 2007).

The problem of integrated model calibration is further exacerbated by two key issues. Firstly, highly parameterised, complex and "unwieldy integrated models" (Ashley et al., 1999) require vast amounts of data for every urban water component represented. Secondly, usefulness of such models usually lies in conducting long-term simulations (more than often at short time steps) in order to transcend short-lived and seasonal effects (Rauch et al., 2002a; Hardy et al., 2005; Mitchell and Diaper, 2005). Modellers are thus not only faced with an intellectual but also logistical challenge to obtain the required data.

Advancements in computational technology have led several researchers to adopt more intensive calibration and optimisation procedures (which were earlier typically used for the calibration of simpler models) in an integrated context. These range from simple Monte Carlo methods (Freni et al., 2008) to controlled random search (Butler and Schütze, 2005) and evolutionary algorithms (Muschalla, 2008). New and existing methods applied in a multi-objective calibration and optimisation context such as Pareto preference ordering (Khu and Madsen, 2005), Genetic Algorithms (Rauch and Harremoës, 1999; Schütze et al., 2001) and Multi-Objective Complex Evolution Method (MOCOM) among others (see Wagener et al., 2004 for additional examples) are further examples of a range of methods being investigated in the integrated modelling literature. Such methods applied to integrated models are, however, still in their early phases of research and an overwhelming number of modelling exercises that have been presented throughout this review still rely on experienced modellers and manual trial-and-error approaches to model calibration. Furthermore, data availability and model complexity (e.g. ICBMs, IUDMs and IWSMs) remain limiting factors that threaten the feasibility of these newer automated techniques. Consequently, high levels of integration simply do not feasibly permit highly detailed and extensive calibration procedures in return for the type of broader-scale output they provide. However, as we advance our knowledge and progress towards using more highly integrated models for more detailed quantitative predictions, we will need to begin incorporating integration concepts into monitoring programs to ensure that the required data to improve calibration of these models is collected. Additionally, error and uncertainty propagation through models have been found to be more pronounced with increasing complexity (Freni et al., 2008; Reußner et al., 2009). For example, errors made by upstream modules can be compensated by downstream modules, while ill-posed modules (with strongly correlated coefficients) can add another level of uncertainty (Beck, 1997; Beven, 2006; Bertrand-Krajewski, 2007; Freni et al., 2008; Dotto et al., 2012b). The multi-objective nature of the phenomena questions the existence of a true global optimum (Bertrand-Krajewski, 2007). Khu and Madsen (2005) and Muschalla (2008) effectively illustrate that integrated model calibration/optimisation procedures are evolving into decision-making exercises where most optimal solutions of one objective function are no longer sought, but rather a choice of trade-offs along a Pareto front between key criteria (e.g. several different objective functions or more practical variables like investment decision and water quality indicators).

Beck (1997) touched upon possibly revising "model calibration" more than a decade ago, and alternatives to conventional calibration have also been investigated. Partially fictitious case studies have often been conducted when data on the real system is scarce. This was done for comparative reasons to develop a better understanding of system dynamics (Schulz et al., 2005). To reduce model complexity (in the form of overparametrisation) while improving efficiency, Meirlaen et al. (2001) adopted Ashley et al.'s (1999) suggestion to integrate simpler meta-models or *surrogates*. These are calibrated to their complex mechanistic counterparts and retain their ability to describe system dynamics at the relevant integration level without major loss of predictive accuracy.

To summarise, conventional calibration and optimisation approaches are at their limits when faced with overpowering model integration/complexity. It seems evident that sound techniques for integrated model calibration (and validation) have yet to be fully realised (Nguyen et al., 2007). We nevertheless understand that our calibration and optimisation methods are becoming more multi-objective and acknowledge their inherent difficulty.

4.4. Uncertainty in integrated models

The topic of uncertainty in integrated models itself warrants an entire paper due to its highly disputed and broad nature. There is confusion as to how uncertainty is perceived across different water sectors, an extensive discussion of which was reported in Vanrolleghem et al. (2011). As a matter of fact, a recent study indicates that uncertainty literature (as pointed out here for the integrated modelling literature) likewise lacks clarity and coherent approaches for assessment (Deletic et al., 2012). Our purpose is not an extensive review of this topic, but rather a brief look at key aspects of uncertainty in integrated modelling. Although its importance is fully recognised (Lijklema et al., 1993), we are only slowly gaining a better understanding of the types and nature of uncertainties, their sources and methods of reducing or quantifying them if they are irreducible (Walker et al., 2003; Refsgaard et al., 2007; Matott et al., 2009; Deletic et al., 2012).

Refsgaard et al. (2007) list four key levels of uncertainty: (1) statistical (quantifiable), (2) scenario (explorable), (3) qualitative uncertainty (neither quantifiable nor explorable) and (4) recognised ignorance (Van der Keur et al., 2008). Statistical uncertainty has received much attention due to its quantifiable nature and pervasiveness in integrated as well as classical modelling. Furthermore, the nature of this uncertainty is classed as either reducible 'epistemic' uncertainty – due to imperfect knowledge – or irreducible 'aleatoric' (also known as 'stochastic' or 'ontological') uncertainty – due to intrinsic natural variability (Walker et al., 2003; Refsgaard et al., 2007). Much effort has been spent on quantifying statistical uncertainty, and it has been established that overparametrisation resulting from consideration of too many mutually dependent factors sometimes lead to magnifying statistical uncertainty (Rauch et al., 1998; Bertrand-Krajewski, 2007; Freni et al., 2008; Deletic et al., 2012; Dotto et al., 2012b). The problem is further exacerbated in the model integration process, where trade-offs between overparametrisation and model simplification have to be made. Overparametrisation is often an unavoidable, yet common consequence of integration (Freni et al., 2008) resulting from the imbalance between model complexity and data availability. Without enough data to properly calibrate each sub-process, we consequently face poor predictive performance when using a model with too many uncalibrated/unvalidated parameters against a small data set. Improving predictive performance would require extensive data collection campaigns and time-consuming calibration/validation processes (often not feasible). Using simplified sub-models, however, can alleviate this issue, but at the expense of accuracy and possibly not fully address modelling aims.

Uncertainty Analysis (UA) (Ham et al., 2007), probability boxes (Sun et al., 2012) and many other techniques reviewed extensively by Matott et al. (2009) to quantify statistical uncertainties (as well as distinguish between their epistemic and aleatoric natures) can improve confidence in model results, but impose additional computational burden. Schellart et al. (2010) have recently demonstrated an innovative and efficient method of quantifying uncertainty in integrated models that combined prior expert elicitation with traditional statistical methods and parameter sensitivity information contained in a database format (to avoid constantly re-running the integrated model). Instead of Uncertainty Analysis, however, many have advocated using simpler sub-models with well-known limits (Rodríguez et al., 2010) and a lower number of parameters (Freni et al., 2008). This again faces the outlined problem that uncertainty propagation upon integration either amplifies or reduces total uncertainty (this will depend on the processes being modelled and how their sub-models are being integrated). It is therefore very unlikely that the total uncertainty is a simple sum of sub-model uncertainties. Only recently have Kuczera et al. (2006) and Deletic et al. (2012) provided new insights and proposed a new framework for assessing total uncertainties in integrated urban drainage models; the method has been currently tested on integrated rainfall/runoff and pollution generation models (Kleidorfer et al., 2009; Dotto et al., 2011). Identifiability of integrated models has also been an interest (Freni et al., 2011; Kleidorfer et al., 2011) in hope of improving the efficiency of the integrated modelling process.

Scenario uncertainty (where outcomes are known but cannot be assigned a quantitative probability) and qualitative uncertainty (where not all possible outcomes are known and quantifiable) (Brown, 2004) have been given less attention. Urich et al. (2013) have recently made an effort to grasp scenario and qualitative uncertainty and transplant it into a statistically quantifiable form. However, Nguyen et al. (2007) suggest that integrated models themselves should not depend on classical uncertainty tests, but instead on numerous emerging methods that either need to be developed or have been rejected in the past due to their subjective nature (e.g. peer review).

In addition to the levels and nature of uncertainty, we also need to acknowledge different sources of uncertainties in our modelling exercise, ranging from Refsgaard et al. (2007): (1) framing of system boundaries, (2) inputs into the model to represent external influencing factors, (3) model structure, (4) model parameters and their representation of the system and (5) model technicalities (including the computational implementation, choice of spatial and temporal approximations). Effective data provision for alleviating input uncertainties and model calibration/validation has been frequently explored in the literature (see e.g. Vanrolleghem et al., 1999; Meirlaen et al., 2001; Bertrand-Krajewski, 2007; Freni et al., 2009; Schellart et al., 2010; Voinov and Cerco, 2010).

It is inevitable that we cannot evade the uncertainty issue in integrated modelling. However, it is worth exploring suitable methods that can be prioritised at different integration levels (e.g. greater focus towards understanding scenario uncertainty at the IUWSM level). Such efforts are now being invested in the development of various large-scale integrated models like VIBe (Sitzenfrei et al., 2010), DAnCE4Water (Rauch et al., 2012), the ReVISIONS framework (Ward et al., 2012) and UrbanBEATS (Bach, 2012).

5. Adoption of Integrated Urban Water Models

Lijklema et al. (1993) listed 18 barriers against integrated planning at the first INTERURBA. Although this list is not directly transferable to integrated modelling, the influence of some obstacles (e.g. data requirements, administrative fragmentation and interests, inconsistent indicators or targets to plan for and lack of perceived benefits of integration) can be seen in integrated modelling. Even though this list is hailed as a significant contribution to the field in the last twenty years (and has also contributed to the motivation for conducting this review), a lot of obstacles have changed since then. It is not our intention to solely base our discussion in this section on Lijklema et al. (1993)'s list, but rather revise this list based on our review of the more recent body of literature.

The adoption of integrated models in current practice seems to be progressing more readily than in previous decades e.g. new commercial software packages (Table 1). These technological advancements were possible thanks to (Rauch et al., 2002a; Argent, 2004; Devesa et al., 2009): (1) improvement of geographic information systems (GIS), (2) emergence of new software and models (irrespective of their 'look and feel') and (3) greater uptake of artificial intelligence (AI) tools that allow incorporation of expert

knowledge into decision-support. The third point is not as prominent, but still plays a significant role. The use of AI in companion modelling approaches has produced useful integrated models that can actively engage stakeholders, particularly in the water resources management sector (see e.g. Farolfi et al., 2010; Ducrot et al., 2011). Cortes et al. (2000) conducted an insightful review of the benefits of AI techniques in environmental decision support systems with some examples from the urban water sector. Despite this progress, we must not become complacent as there are still many technical as well as societal challenges that can affect future adoption (Tyson et al., 1993; Schütze and Alex, 2004; Rauch et al., 2005; Mcintosh et al., 2007).

5.1. Drivers for adoption

Three crucial factors, which have promoted uptake of integrated models, were found: (1) the integrated urban water management (IUWM) paradigm, (2) changes to management objectives and legislation and (3) increased innovation in the use of integrated models.

5.1.1. The IUWM paradigm

Considering all parts and interactions of the urban water system simultaneously (Harremoës, 2002; Mitchell, 2004; Gabe et al., 2009) has redefined conventional management and systems thinking in the urban water sector. The change has acknowledged the complexity of the urban water system and has promoted bottom-up management with stakeholder involvement (including broader community). Conventional modelling practices were in turn also affected by this shift toward integration, reflected in the progress made in our understanding of the interactions between different urban water sectors.

5.1.2. Changes to management objectives and legislation

Climate impacts and the paradigm shift in water management have influenced recent changes in management objectives and legislation across the globe such as the transition from emissions to ambient water quality standards at the turn of the century or an emphasis on recycled water systems. Such changes are reflected in e.g. EU Water Framework Directive (EU, 2000), US Clean Water Act (United States Environmental Protection Agency, 2002), UK Urban Pollution Management Manual (FWR, 1998) and Australian National Water Quality Management Strategy (Australian Government, 2000). Additionally, the European Water Initiative (European Commission, 2012) and United Nations Millennium Development Goals (United Nations, 2010) prescribe compliance targets that urge more sustainable and integrated water resource management. Although these standards do not explicitly force integration upon urban water managers, they encourage a holistic view and broader system boundary for assessment (Vanrolleghem et al., 2005b; Vojinovic and Seyoum, 2008). Integrated modelling thus becomes a more attractive and beneficial undertaking.

5.1.3. Increased innovation in integrated model use

Urban water systems have been recognised as 'complex socio-technical systems' (Brown et al., 2009; Wong and Brown, 2009). In fact, Hardy et al. (2005) and Bamford et al. (2008) both regard integrated models as essential in increasing our understanding of the wider urban water systems and their interactions with society. However, in order to capture this in our models, research needs to expand in this non-technical direction. Innovations have emerged that are making integrated modelling more versatile and are addressing this complexity. Impact analysis and mitigation studies range from assessment of external factors (e.g. Fu et al., 2009; Dong et al., 2012), to intrinsic system vulnerability (Möderl et al., 2009; Sitzenfrei et al., 2011), as well as their mitigation (Fu et al., 2010). One particular example, Fu et al. (2009), explored impacts of new residential developments on receiving water quality. Modelling of hypothetical or semi-hypothetical (i.e. fictitious) case studies (based partly on a real location) allow reduction of aims and objectives by reforming the problem statement. These have helped decision-makers gain some insights despite the absence of good, reliable calibration data; e.g. Harremoës and Rauch (1996), Meirlaen et al. (2002), Butler and Schütze (2005), Doglioni et al. (2009), Diaz-Granados et al. (2010) and Urich et al. (2013). Other new innovations include multi-objective control (Fu et al., 2008) to illustrate cost-performance trade-offs in decision-making, increased use of surrogate modelling e.g. Kriging approach known as DACE (Fu et al., 2009), artificial neural networks and other artificial intelligence algorithms are observed in real-time control research (Vojinovic et al., 2003; Schulze et al., 2005). Such multi-objective assessment tools have also evolved to become broader technology optioneering tools (e.g. Makropoulos et al., 2008; Dotto et al., 2012a; Rauch et al., 2012; Ward et al., 2012; Bach et al., 2013) for the design of infrastructure management and adaptation strategies. Economic valuation, energy considerations and social dynamics are gradually being considered in such modelling exercises (e.g. Cutlac et al., 2006; Fagan et al., 2010; Moglia et al., 2010; Ducrot et al., 2011; Kragt et al., 2011; Rauch et al., 2012). The social dimension appears in the form of both agent-based models and direct expert involvement in the model setup, scenario development and discussion of results. Overall, these studies cover a wide spectrum of modelling aims (Seppelt et al., 2009), including: (a) understanding of system response to different pressures, (b) designing control strategies, (c) optimisation of system performance and (d) development of decision-support frameworks.

5.2. Barriers against adoption

Changes in urban water management (e.g. increased positive awareness, coordination among stakeholders and evolving legislation) have significantly reduced Lijklema et al. (1993)'s list of 18 barriers. However, several key barriers remain, both in the biophysical/technical and social/managerial realm. A summary is provided in Table 4.These can be discussed in the context of four key aspects: (1) model complexity (inclusive of model building, application and data requirements), (2) user friendliness (relating to computational and logistical resources for using integrated models), (3) administrative fragmentation, and (4) communication. Direct reference is also made to obstacles that were listed by Lijklema et al. (1993), which the authors feel are still prominent and has been picked up specifically by modellers in the literature.

5.2.1. Model complexity

The effort and cost required to use integrated models (data requirements, model setup, calibration, simulations) is often perceived (at least in practice) to outweigh the value of the output gained as opposed to conducting multiple smaller modelling exercises (Ahyerre et al., 1998; Erbe et al., 2002; Peters et al., 2006). As such, literature claims that complex models are frequently seen as research tools, while simpler models are more appropriate management tools (Ashley et al., 1999). An important exception is MIKE URBAN, a detailed deterministic hydrodynamic integrated model, which is extensively used in practice (DHI, 2009). Through advancements in hardware and model development research, it is likely that larger, more detailed and highly integrated models will eventually find practical uses. It is therefore important that practice takes active steps to embrace model complexity. Achieving this, however, may also require formulation of effective decision

Table 4
Key Social and Biophysical Barriers against Integrated Model Adoption relating to issues of model complexity, user friendliness, administrative fragmentation and communication.

Topic	Type	Barrier
Model Complexity[a]	Social/Managerial	• Aversion to highly complex models and/or unwillingness to deal with risk, which results in unrecognised potential of these models. • Not enough understanding of the ways in which integrated models can be used (facets of decision support for strategic planning as opposed to detailed design)
	Biophysical/Technical	• Reasonably low level of model accuracy due to simplified algorithms that try to represent a complex system in many spatial and time scales • Huge data requirements for calibration of model due to large project size, calibration process slow, costly, uncertain and computationally more intensive than traditional models. Calibration sometimes neglected because of this. • Assessments plagued by uncertainties on model structure, input data, calibration data and not as easily reduced due to the size of the integrated model • Lack of highly interlinkable and robust sub-models that can be easily used, integrated with each other and adapted to suit modelling aims.
User friendliness	Social/Managerial	• Highly skilled personnel (in model setup, calibration and application) often required, intensive training involves high financial costs and time to transfer knowledge.
	Biophysical/Technical	• Limitations in computational hardware prevent widespread adoption of more complex decision support systems. • Bad design of commercial packages, fail to assist users who become frustrated with trial and error and only hypothetical optimum solutions.
Administrative fragmentation[b]	Social/Managerial	• Many existing regulations do not necessarily require use of integrated models (only recent ones in some countries). • Fragmented responsibilities for different water streams, not interaction between managers of different systems • 'Silo-thinking' induced by too specialised education in civil, environmental engineering or other disciplines
Communication[c]	Social/Managerial	• Lack of realisation of economic benefits of integrated approach in preventing 'over-design' or ability to better understand environmental economics of a problem through these models. • Lack of stakeholder involvement in the modelling process, participatory modelling is advocated but not carried out • Lack of managed or controlled vocabularies • Lack of consistent representation/expression of a models ontology

[a] Related to Obstacles #4, 5, 8 and 9 in Lijklema et al. (1993)'s list.
[b] Related to Obstacles #2 and 12 in Lijklema et al. (1993)'s list.
[c] Related to Obstacles #1, 13, 14 in Lijklema et al. (1993)'s list.

protocols used to manage complex systems and relates to administrative issues discussed later.

Even if complex models are more actively sought after, data availability (both potential before and actual after measurement campaigns), remains a challenge and determines the level of detail that can be adopted in the model building exercise as well as the quality and uncertainty of model output to address modelling aims. Uncertainty and sensitivity analysis incorporated into integrated modelling can help determine data requirements and understand part of the complexity (Schellart et al., 2010). Work from the Kallisto Project at Eindhoven (The Netherlands), for example, conducted global sensitivity and uncertainty analysis to assess the robustness of different RTC scenarios in mitigating ecosystem impacts from a performance and cost perspective (Benedetti et al., 2012; Langeveld et al., 2013). The project identified suitable cost effective measures for the modelled catchment for meeting EU Water Framework Directive requirements using an integrated approach. Closer collaboration between researchers of different fields (a slowly emerging trend) can lead to highly interlinkable and robust sub-models, ASM1(Henze et al., 1993) and RWQM1 (Reichert et al., 2001; Shanahan et al., 2001; Vanrolleghem et al., 2001) being the most common examples in this case. These are, however, ICBM-level models linked to form IUDMs. Future work is needed to achieve such harmonisation (not only in terms of model development but also to address data availability issues) at higher integration levels.

5.2.2. User friendliness

Complexity can often be attributed to a bad user interface, inhibiting a model's adoption/use (Marsalek et al., 1993; Mitchell et al., 2007). Several well-known packages like MUSIC, MIKE and SWMM have effectively overcome this issue. Additionally, the field of Human–Computer Interaction (HCI) research, especially in EDSSs (Poch et al., 2004; Mcintosh et al., 2007) instills much confidence in addressing this barrier. However, these advancements and improved user friendliness can also produce overconfidence in models with a lesser perceived need to understand their internal structure. Heusch et al. (2010) recommend building models that are applicable to people with an intermediate level of understanding, for example, using easy-to-determine parameters or a highly customisable and user friendly interface.

5.2.3. Administrative fragmentation

Split responsibilities in management of urban water systems has been a common reason for lack of integration (Rauch et al., 2002a) and remains an issue today, albeit less of a barrier than before. Social research in recent years has been addressing the issue of intra- and inter-organisational capacity (Brown, 2008; Van De Meene, 2008; Harvey et al., 2009). The problem is much deeper, defined as 'institutional wickedness' (Harvey et al., 2009), where capacity building of institutions is faced with complex impediments. Both Brown (2008) and Van De Meene (2008) suggest some measures (e.g. political institutionalisation, reform, policy, capacity development) as means for evolving current organisational structure and 'breaking disciplinary barriers'. Harvey et al. (2009) refer to this as the 'Art of being undisciplined'. Building capacity, however, should already begin at the early stages of education in civil, environmental or other disciplines to forge a mindset that is not 'siloed' into a single discipline.

5.2.4. Communication

Communication challenges are one of the strongest barriers against the uptake of integrated models. Despite long-standing efforts to encourage effective research reporting (Beck, 1976), the literature remains conservative. The reality is that many publications present single case studies, where some form of integration has been applied and is shown to provide quantifiable benefits that fall short of becoming transferable. Very few studies proceed to explore greater scopes of integration and instead choose to remain with two or three sub-systems, a point with which Schmitt and Huber (2006) agree. To fix this flaw, more effective communication must be promoted and the authors can suggest a systematic approach: (1) broader scope, but still focussed, (2) objectivity, (3) comprehensiveness in reporting, and (4) transparency of approach, results and conclusions. Accomplishing effective communication is, however, easier said than done. Some instances of communication challenges can be illustrated. For example, modellers may choose to acknowledge 'wicked problems' (Pahl-Wostl, 2007; Harvey et al., 2009), but effort needs to also be invested in understanding and reporting on these, particularly issues that shroud integrated models in uncertainty. Integration helps address increasingly complex problems, yet uncertainty plaguing integrated models is seen as dangerous by decision makers (Lijklema et al., 1993). It is, however, not only important for modelling research to better understand how to deal with uncertainties, but for decision makers to also evolve their processes to address uncertainties in order to not slow down the adoption process. Use of well-defined performance metrics to quantify modelling objectives (e.g. cost or sustainability) and comparison of outputs is increasingly recognised in integrated modelling studies as well as beneficial to objectivity and transparency in communication. As such, adoption of models in policy design and practical decision making can become more achievable (Fagan et al., 2010). Useful performance metrics can, however, only be ascertained if scientists and decision makers actively coordinate (Liu et al., 2008). It is inevitable that greater stakeholder involvement is required if integrated modelling is to progress down a fruitful path (Beck, 1997; Peters et al., 2006; Refsgaard et al., 2007; Liu et al., 2008).

6. Future outlook of Integrated Urban Water Models

We have presented numerous innovations in the literature in previous sections. Many of these are likely to be nurtured in the near future. As a general outlook for the future, however, three key emerging issues can be identified as: (1) availability of online tools for integrated management of the urban water system (Butler and Schütze, 2005; Diaz-Granados et al., 2010), (2) a more transparent process when using integrated models (Pahl-Wostl, 2007; Liu et al., 2008) and (3) more inter-disciplinary work on integrated modelling research (Rauch et al., 2005; Pahl-Wostl, 2007; Refsgaard et al., 2007; Seppelt et al., 2009; Urich et al., 2013). Achieving these will require improvements in handling and understanding spatial and temporal issues in such models (Mitchell et al., 2007), devoting attention to achieving longer time series simulation (Rauch et al., 2002a; Willems, 2006) and overcoming the communication barrier. Notably, the data requirement issue has been an ongoing topic (Lijklema et al., 1993; Vanrolleghem et al., 1999; Erbe et al., 2002; Rauch et al., 2005; Bertrand-Krajewski, 2007; Fagan et al., 2010) and will continue to be equally important, especially when considering higher integration levels.

From a hardware perspective, it has been emphasised that software, which does not take advantage of parallel computing will not only stagnate, but is also likely to degrade on future devices (Sutter, 2005). Whilst Moore's law states that transistor count doubles roughly every two years (Moore, 1965), physical limits in semi-conductor technology prevent the advancement of single processor performance that persistently achieving this law has paved the way for the multi-core era. Integrated urban water modelling has begun exploring parallel computing with efforts to improve performance of existing software such as SWMM (Burger et al., 2014) or CityDrain3 (Burger et al., 2010) for multi-core systems or model development to run on clusters of computers (Barreto et al., 2008; Benedetti et al., 2008). Burger et al. (2014), in particular, show that multi-core technology is particularly suited to more complex systems (e.g. networks with greater number of pipes benefit from higher speedups in computation time), highlighting that these advancements can help facilitate the use of large, complex, integrated models. Despite these advancements, the future of parallel computing is envisioned to be more complex with the emergence of not only multi-core systems, but also *heterogeneous core systems* and *elastic compute cloud core systems* (Sutter, 2011) requiring researchers to be adaptive to the evolving computational technology.

The likely future for research will focus on addressing the broader range of uncertainties in integrated models (Deletic et al., 2012). These not only include those encountered in engineering practice, but also in non-physical and non-technical systems. Dynamics of anthropogenic development and socio-economic attitudes will need to be embraced, thereby evolving integrated models into "virtual playgrounds", which researchers will use to test new theories and gain new understanding. Despite greater inter-sector communication, practice remains fragmented. Current legislation relies on simple assessments. Setting realistic and more context-sensitive management targets should be encouraged. The participatory approach is likely to gain in popularity. Integrated urban water models can encourage this, whether between plant manager and engineer at the ICBM level, public utilities and local municipality at the IUDM/IWSM level, urban water managers and the general public at the IUWCM level or among any and all of these managers and stakeholders at the IUWSM level. Further promoting the benefits of integrated models will facilitate this. Advances in model development to harness multi-core technology (on single desktop or through cloud computing) will provide support for future research and is being actively embraced. Finally, integrated models are likely to improve in terms of software design and implementation (Laniak et al., 2013) and find a stable footing in the software market.

7. Conclusion

This critical reflection was undertaken following more than 30 years of experience in integrated modelling. Aiming to enhance structure and clarity in the field, we outlined key events in the three decades of history, developed a clear and new typology for classifying integrated models, defined key model features and development steps alongside uncertainty issues (a rapidly growing field and thus not addressed in detail here). We investigated emerging applications and adoption issues by revisiting critique from 20 years ago. This was followed by a future outlook. Five key messages that have emerged from this review can be summarised as follows:

- Historically, much of the motivation for integrated of urban water modelling originated from the urban drainage field (which comprised the majority of literature found on the topic) until the late 1990s where it attracted broader attention from other urban water sectors.
- The term 'integration' is ambiguous, especially in this diverse and growing field. Therefore 'degrees of integration' were defined in a typology of integrated urban water models comprising four model integration levels: Integrated

Component-Based Models (ICBMs), Integrated Urban Drainage or Integrated Water Supply Models (IUDMs/IWSMs), Integrated Urban Water Cycle Models (IUWCMs) and Integrated Urban Water System Models (IUWSMs) that sit within the sphere of environmental decision support systems.

- Integrated urban water model development principles are similar to that of classical models, but significantly more purpose-driven and lying between completeness and parsimony, guided by subjective judgement of pragmatism. Data considerations (for understanding sub-processes and model calibration/validation) are prevalent throughout the entire process.
- Adoption of integrated urban water models in practice has been driven by new emerging paradigms and changing legislation. Uptake of integrated urban water models in practice has improved in the recent decade and could be further strengthened through emphasis on better data collection and availability, effective education to embrace model complexity and overcome administrative fragmentation, user friendlier software and transparent communication, in particular, a consistent vocabulary.
- With rapidly improving software design and computational efficiency, integrated models are likely to play an important future role as both effective communication platforms as well as "virtual playgrounds" for exploring new knowledge and uncertainties of both technical and non-technical aspects of the urban water system. The novel typology presented in this review will aid in providing organisation and clarity to this rapidly growing field.

In summary, we judged the rapidly growing body of literature as confusing and intimidating to new researchers in this field due to linguistic uncertainty and localised schools of thought. This is partly due to the diversity of research that has emerged in recent decades on the topics, the historical context in different parts of the world (e.g. water recycling in Australia vs. ecological management in Europe) and the participation in this research field by a variety of academic disciplines. The review also identified a gradually increasing uptake of integrated models in practice with several 20-year-old barriers against adoption, however, still prevailing. Despite these barriers, we are nevertheless hopeful that this critical review has set up a checkpoint to guide future research and practice.

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