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Abstract

Increasingly, the application of models in urban hydrology has undergone a shift toward integrated structures that recognise the interconnected nature of the urban landscape and both the natural and engineered water cycles. Improvements in computational processing during the past few decades have enabled the application of multiple, connected model structures that link previously disparate systems together, incorporating feedbacks and connections. Many applications of integrated models look to

assess the impacts of environmental change on physical dynamics and quality of landscapes. Whilst these integrated structures provide a more robust representation of natural dynamics, they often place considerable data requirements on the user, whereby data is required at contrasting spatial and temporal scales which can often transcend multiple disciplines. Concomitantly, our ability to observe complex, natural phenomena at contrasting scales has improved considerably with the advent of increasingly novel monitoring technologies. This has provided a pathway for reducing model uncertainty and improving our confidence in modelled outputs by implementing suitable monitoring regimes. This commentary assesses how component models of an exemplar integrated model have advanced over the past few decades, with a critical focus on the role of monitoring technologies that have enabled better identification of the key physical process. This reduces the uncertainty of processes at contrasting spatial and temporal scales, through a better characterisation of feedbacks which then enhances the utility of integrated model applications.

1. Introduction

Urbanisation is a global trend with more than 55% of the current world population living in urban areas. There are now 394 cities home to over 1 million inhabitants, an increase which represents a doubling of the urban population in a 50 year period[1,2]. As the urban population continues to grow and spread laterally into previously undeveloped areas, a significant stress is placed on natural resources and environmental quality, having a demonstrable impact on atmospheric and hydrological processes as well as the quality of the surrounding environments[3,4,5,6]. The presence of the urban heat island; alterations to air currents and increased particulate matter above urban space result in the emergence of microclimates, which impact on input precipitation and evapotranspiration rates. The creation of impervious surfaces alters dominant runoff-generating processes, flowpaths and infiltration, which can have a profound effect on the overall catchment water balance[7]. The input of freshwater supply and extraction of sewage via large piped networks provide artificial inputs and outputs to the catchment water balance, making it difficult to quantify the components of the water cycle[8].

Urban space also contributes to degradation of water quality, as contaminants and sediment are transferred into urban waterways[4]. Sources of urban pollution fall into two main groups: (i) those transported via the sewerage system and (ii) those such as roads, building sites and atmospheric deposition, which arrive in receiving water bodies via stormwater drains and diffuse pathways[6]. The second group poses greater concern to urban hydrologists and land managers as their pathways remain relatively unknown, hence monitoring and predicting these fluxes remains a considerable challenge[9]. With the emergence of new contaminants such as those found in personal care products, pharmaceuticals and industrial wastes, increasingly advanced monitoring regimes are required to map the spatio-temporal dynamics of their sources and pathways to formulate effective remediation strategies. Novel contaminants (e.g., microbial contaminants and pharmaceuticals) enter into sewer systems, where historically they would not have been targeted for treatment in wastewater treatment works, often resulting in their discharge back into the water environment untreated[10].

Here, we introduce and reprise the development of integrated modelling techniques to assess impacts of urban expansion on the aquatic environment, whilst discussing the central role that advances in monitoring technology (Table 1) has played in enabling their widespread application. We introduce an example of an integrated model structure to elucidate how such models are developed to tackle real-world problems and assess how the components, as well as the underpinning monitoring technologies have advanced in recent years. We conclude by addressing some of the remaining limitations and assess future priorities for the continued development of urban systems modelling.

2.Integrated Modelling and “big data”

2.1. Rationale, and definitions of integration

Historically, efforts to model the expansion and impacts of the urban environment on hydrological dynamics and aquatic integrity have relied on a fragmented network of independent model structures that simulate specific processes, often overlooking any connections or feedbacks between linked systems[11,12]. Increasingly, modelling of urban systems has started to adopt an integrated approach whereby complex systems are treated as interconnected rather than disparate, placing an emphasis on the interactions and feedbacks to enable sufficient understanding of the entire system[8,13,14]. Over the past 40 years, integrated modelling has grown concomitantly with improvements in computational power, which has traditionally been a barrier to high-level simulation[15]. In addition, expansive data requirements across multiple model structures have also traditionally precluded widespread application[15].

The role of modelling in urban hydrology has expanded with improvements in computational hardware to become a frequently utilised tool. Models of the urban water environment serve two primary functions: (i) elucidating some of the ‘black box’ processes that occur within the urban system, such as dominant flow pathways at both the surface and in pipe-systems and their connections, and (ii) predicting future environmental changes (both land-use expansion and climate change) and their subsequent impact on the hydrological cycle. Bach et al. highlight the 1982 study of the Glatt Valley in Switzerland, which sought a combined understanding of multiple components of an urban water system, as the pioneering study into integrated model applications in urban water studies[16]. This study also highlighted a key difficulty in early integrated modelling applications, as it required over 40 people to detect and track contaminants through the wastewater treatment system from rainfall input to the receiving water during a storm. It identified the impacts of stormflow on secondary wastewater treatment dynamics, whilst also tracking the temporal patterns of pollutants within the drainage system, treatment processes and receiving water bodies.

Integrated models are developed at contrasting scales, often to address very specific research questions. These range from assessing the impacts of development on receiving water bodies[17,18], to optimising the treatment steps and processes within a wastewater treatment plant[19,20], to large-scale modelling efforts that are used in decision and policy applications and utilise social and economic variables[21,22,23]. The differing scale and range of research questions posed of models can be viewed as defining the type and extent of “integration”. A comprehensive review of model integration in urban environments[12] identifies four tiers of integration which are described graphically (Figure 1) illustrating the increasing breadth of disciplinary coverage and spatial extent from component based models within individual sub-systems (Tier 1) to holistic integrated urban water systems (Tier 4). Historically, integrating activity has predominantly focussed on integrating components of drainage systems (Tier 2); examples of which enable multi-constituent modelling and the consideration of both acute and chronic impacts at a range of spatial scales.

Increasingly, integrated models have been applied as environmental decision support systems (EDSS), enabling officials and planners to assimilate sufficient information to assess the impacts of particular scenarios or policy implementations accordingly[24]. As with the integrated models themselves, examples of use of EDSS to assess impacts of urban areas cover a wide contextual spectrum, from the urban water cycle itself (UWOT)[25] to basin scale management (e.g. the Elbe-DSS)[26] to appraisal across wider domains as part of ecosystem services assessments (e.g. Envision)[27] As a consequence of the contrasting spatial and temporal scales, integrated models often operate as a balance between parsimony and pragmatism, whereby representation of the overall system is a key outcome, often achieved by representing complexity at local scales[12]. Increasingly, there is an array of 'off-the-shelf' model packages available (e.g., SIMBA[28]; Aquacycle[29] and MIKE toolbox), however these are often limited by their proprietary source-code, meaning users have little flexibility to modify how such models operate, though an increasing number of open-source integrated models are becoming available (e.g., CityDRAIN[30] and TyndallCities/ARCADIA USM[31]).

2.2. Integrated Modelling: a generic example and review of the development of component models

We consider an example whereby the role of urban growth on hydrological dynamics and water quality within a large basin is addressed using integrated modelling. Here, the overarching objective is to provide knowledge to inform future development policy that supports sustainable water resources (the POLLCURB project focussing on the Thames basin (southern UK): www.pollcurb.ceh.ac.uk). Such an endeavour requires research teams with diverse modelling and data collection skills. To achieve objectives coverage of the following domains with an array of interconnected models is essential: (i) Land-use change (ii) Urban drainage (iii) Rainfall-runoff (iv) Water quality. In this example modelling the impacts of climate change is also required and in addition, other features, notably population growth and wastewater treatment processes, must be represented. Pragmatism excluded explicit consideration of other relevant domains (e.g. urban air quality, aquatic ecology). The rationale involves identifying key relationships between land-use, flow and water quality derived at the local scale (< 100 km2) from case-study sub-catchments that have experienced rapid urbanisation in recent decades. By way of upscaling, these relationships would then be employed for predictive purposes at basin scale (> 5000 km2). Therefore, in terms of integration concepts (Figure 1) the overall model philosophy sits in Tier 4. Developments of modelling techniques covering the four key domains are charted below.

2.2.1. Land-Use Change Modelling (LUCM)

Urban growth modelling is primarily concerned with how growing populations and economic development lead to changes in land use. Classical urban models are based on equilibrium theory and the notion that land use will develop to an optimal spatial distribution of activities in terms of access to jobs, markets and labour. The Lowry model[32] is an early and highly influential model of this type whereby for a given employment in the basic sector it estimates the distribution of the population supported by these jobs, as well as the further retail and service jobs meeting local demand. The spatial distributions are based on distance-frequency relationships, known as gravity models. To date, equilibrium-based modelling has developed into two main classes of urban model: land-use transport interaction modelling

(LUTI) and Computational General Equilibrium modelling (CGE). CGE are used for macro-level economic modelling with strong theoretical rigor and minimal assumptions[33], at the cost of remaining relatively abstract in its conclusions. In contrast, LUTI models use a bottom-up approach, taking the behaviour of micro-economic agents as a starting point; these models typically represent the transport network in detail and consider the interactions between industrial sectors and socio-economic groups in spatially refined geographical regions. Being more pragmatic than CGE, LUTI models tend to include some ad-hoc assumptions at the interface between more-or-less established sectorial- or discipline-specific models[34]. In recent years the approaches have converged, with LUTI models adopting more theoretical rigour[35] and CGE models becoming more spatially explicit[36]. Both LUTI and CGE models are primarily economic models and whilst they are suited and used for integrated analysis, it is through the mechanisms of markets. The data required are typically collected by national statistical agencies and census bureaus (information on population and employment detailed by socio-economic groups, industrial section and, importantly, geographical regions). LUTI models in particular rely on fine resolution geographical detail, to realistically incorporate the role of the transport system.

In the late 1970s an important new paradigm was introduced to urban modelling as it was increasingly recognized that the development of urban systems is lagging behind the drivers of change, and urban systems are therefore chronically out of equilibrium. The growth of cities is chaotic, path-dependent and complex mathematically[37,38]. Early models based on this paradigm were no longer finding equilibrium solutions of urban configurations but were simulation models that explored possible trajectories of change[39]. A second development in urban growth modelling that occurred over the same time was the development of ‘cellular worlds’[40,41]. These abandoned the use of regions or zone-systems and the thereby implied assumptions about urban boundaries and structure in favour of dynamics based on neutral, fine scale, regular grids. These models simulated growth of urban systems as a bottom-up process, in a petri-dish kind of environment, very well aligned with mathematical[42], and economic[43] understanding of the macro-level manifestations of micro-level processes. These developments gave rise to the cellular automata land use change model[44,45] the dominant form of urban model today which has a more pronounced physical and geographical basis than LUTI or CGE. In addition to transport accessibility, factors determining where urban growth occurs include slope, soil and other factors of physical suitability, zoning (spatial planning) status, and the land uses found in the direct neighbourhood. As such the cellular automata models are highly appropriate for integration with other socio-economical and physical models[46,47].

2.2.2. Urban Drainage Modelling

The expansion of urban areas results in extension to subsurface urban drainage infrastructure alongside increased influence of flood storage areas and road drainage. A growing number of urban drainage model packages now exist, and with improvements in computational efficiency these incorporate increasing levels of spatial and temporal detail (Table 2). A common drawback in the early days was that urban models often operated in isolation from the surrounding landscape[14,48]. Urban drainage modelling has evolved markedly in recent decades in response to advances in computing power and an increasing interest from a planning, hazard and management perspective. Whilst underpinning mathematical functions have remained largely unchanged, modelling has been buoyed by the assimilation of GIS software, enabling rapid determination of complex drainage networks, runoff pathways and sewerage catchment areas[49]. The release of CHI’s PCSWMM program (an enhanced version of the US EPA SWMM model) combines GIS, hydraulic modelling and hydrological modelling in one platform. PCSWMM contains a GIS interface that supports Open Street Maps, Bing Maps, Google Maps (including a Google Earth interface) and ESRI OS® mapping products (Table 2) enabling easier identification of subcatchment boundaries, pipe networks and catchment characteristics.

In a perspicacious review, Bach et al., track the move towards integration in urban drainage modelling over the past three decades. Historically considered separately, Wastewater Treatment Plants (WWTPs) have increasingly become recognised as integral parts of urban water systems, particularly in relation to water quality and baseline chemistry[5]. Furthermore, if WWTPs receive large quantities of water from outside catchment boundaries, water balance is no longer explicable in terms of natural rainfall-runoff dynamics[50]. As a result, there has been increasing effort to link natural hydrological dynamics and urban drainage systems with WWTP processes. Sequential treatment processes within WWTPs are increasingly being represented intrinsically to give a 'plant-wide' model of both water fluxes and water quality[12]. Rates of urbanisation typically outstrip increase in WWTP capacity, often resulting in a reduction in the efficiency of some processes[51]. The 'plant-wide' approach to modelling WWTP simulations is successfully used to quantify impacts of urban growth on wastewater loads and thereby on effluent water quality.

2.2.3. Rainfall-runoff Modelling

Hydrological models of varying complexity are routinely used for predicting storm runoff volume and peak flow magnitude in urban areas. Several comprehensive reviews of urban runoff modelling have recently been published[52,53,54]. However, most of the recent advances are related to the hydraulic components, routing storm water across urban surfaces to and through sewer systems, and used to predict inundation[55,56]. These advances have capitalised on increasing processing power combined with the emergence of new spatial datasets allowing a more detailed hydraulic description of the urban geometry (e.g. high resolution topographical data such as LiDAR)[57]. When modelling infiltration in urban catchments, land is divided into two portions, one overlain by impervious surfaces (e.g. roads, roofs, pavements) the other characterised as pervious, suggesting soils covered by vegetation. The pervious areas comprise rural land outside the urban area or green-field sites within it, such as parks or gardens. Impervious areas are commonly subdivided into those directly connected to the man-made drainage system (also known as effective urban areas), and those lacking direct hydraulic connection[58]. Using this simplified representation, a number of hydrological models treat runoff generation as two separate systems, which when combined form the total runoff response[59,60,61,62].

A key consideration for urban impacts on runoff response is scale. For example, at very local level, particular features such as local storage ponds and the layout of the sewer system layout are likely to significantly affect runoff. However, further downstream the same urban area might only represent a small fraction of the total catchment, and the localised effects become relatively less important. In terms of land use data there is a strong case for using a scale-dependent level of detail. For example, at large scales a straightforward distinction between urban and non-urban may suffice, whereas in specific cases a detailed classification (e.g. suburban, peri-urban, effective impervious areas, or even down to roofs, roads, gardens, drive ways, pavements etc.) may be necessary. Several studies assess the impact of urban development on catchment runoff across scales, from small[63] to large[64,54]. A coherent view from the literature of the best means to represent urban areas at different scales to ensure appropriate representation of processes is lacking.

2.2.4. Urban Water Quality Modelling

A progression has been apparent in water-quality model development, from pathogens and DO[66,67,68]; to nutrients and eutrophication[69,70,71]; to industrial pollution legacies (e.g., toxic organics and metals)[72,73,74,75] and on to the present day where focus is increasingly on emerging contaminants (e.g., synthetic chemicals, pharmaceuticals and microbial contaminants) and nanoparticles, where models are at less advanced stages and remain active fields of research (Table 3). Many of the advances in water quality modelling have been pioneered by increasing access to high quality data, used for both model calibration and parameterisation. Monitoring of water quality is often fuelled by sporadic and seasonally motivated sampling, often driven by events or policies that address specific contaminant problems in a responsive manner. Consequently, datasets are rarely comprehensive, only providing a snapshot in many areas[76]. As novel contaminants continue to emerge, our monitoring ability must keep pace prior to simulating their dynamics in aquatic environments. In addition, the boundary conditions of water quality models for specific contaminants often remain understudied or poorly understood, where a dearth of observed data is available for calibration or parameterisation. For example, a better understanding of contaminant fluxes and interactions from landscapes or facilities (e.g., WWTPs) that input into urban areas is required, to support our understanding of how such dynamics will emerge in urban streams. As technology enhances the resolution and quality of data captured in the field, development of more robust and reliable models that provide a holistic representation of the urban water system become increasingly possible.

In summary, prerequisites for a successful water quality model stem from three key factors: (i) good water quality data for calibration and parameterisation of model structures, (ii) accurately capturing the hydrological dynamics and (iii) ability to determine the sources of pollutants from contrasting landscapes. As a result, efforts to model urban water quality are increasingly turning to an ensemble approach, whereby model structures are coupled with hydraulic, WWTP, surface and atmospheric models, to represent sources and sinks of sediment and contaminants. In terms of biological parameters, representation of autotroph photosynthesis and respiration is commonplace in water quality models as they are fundamental in influencing dissolved oxygen levels. Higher trophic levels, however, are rarely considered beyond their role in acting as a grazing control on autotrophs. Simulation of biological response has largely remained detached from integrated model systems.

2.3. Integrated Modelling and big data requirements: the monitoring context

To be effective, integrated models such as POLLCURB require a significant and diverse collection of (often high-resolution) data. In this case the key sources of information are long-term datasets of climate, land use and river water resources (hydrology and water quality). Our ability to access widespread sources of data has expanded considerably in recent decades. High-resolution data (both spatial and temporal) has become an invaluable tool for aiding our understanding and, accordingly, for reduction of predictive uncertainty. Whilst data availability and our scope to access it has improved greatly in recent years, a key barrier that remains to widespread implementation of integrated models is the considerable requirements that such “big-data” impose[15]. Furthermore, it is essential that datasets can be effective in identifying interactions and in the process of calibrating component models within the integrated framework. Consequently, capturing not only the physical processes but also their connections and feedbacks places considerable strain on modelling practitioners[12]. Taking each of the four key domains in turn, the remainder of this article will critically review how advances in monitoring techniques (summarised in Table 1) have improved integrated model development and applications (Section 3). In particular, we will explore how these advances enable feedbacks and interactions to be better represented. To conclude (in Section 4) we will reflect on how advances in monitoring frame the development of the POLLCURB model application, in particular pinpointing the main aspects needing strong integration.

3. A review of advances in monitoring to support integrated model components

Our example of quantifying impacts of urban growth illustrates unique demands imposed on integrated modelling by issues of interconnections, Urbanisation is driven by regional economic development, emergence of new industry, and housing accordingly; a linkage that has been long recognised dating back to Malthus[30]. Planning policy combined with economic growth models can yield detailed insight into how urban areas will expand into the future, both spatially and temporally. Calibration of such growth (to enable future predictions) requires considerable data from multiple sources including economic indices, population metrics, planning policy documents and remotely sensed urban imagery to track urban growth at contrasting spatial and temporal scales[11]. Furthermore, given the iterative nature of development in urban areas within developed countries, changes of land use typically occur across multiple years and in small spatial zones, thereby a reliance on long-term, high-resolution monitoring (often limited in its availability and prohibitively expensive[15]) serves to restrict the skill and utility of land-use change models. As urban growth occurs, infrastructure such as road networks, public transport infrastructure and urban drainage grows concomitantly. Assessing the impacts of growth on rainfall-runoff and urban drainage dynamics requires access to detailed network maps of existing drainage to assess how surface drainage dynamics will change accordingly. Such data can be sensitive and difficult to acquire as water management authorities can often be resistant to disseminate such large spatial datasets. However, accurate representation of urban hydrological dynamics is dependent upon knowledge of water routing, and so acquisition of drainage infrastructure maps is critical in integrated urban modelling[11]. Within urban areas, wastewater treatment facilities contribute considerable artificial inputs to catchment water balances and water quality. Quantifying the role of these plants is crucial to the success of integrated model applications, and as such, acquiring volumetric discharge and effluent water quality is important. Finally, representing the dominant rainfall-runoff dynamics and associated water quality from contrasting landscapes requires high spatial- and temporal-resolution monitoring regimes to connect component models, thereby enabling a sufficiently robust basis for calibration and reduction of uncertainty.

3.1. Data Requirements and Monitoring Advances in LUCM

Data informing cellular automata models are often, and increasingly, based on remote sensing, i.e. satellite imagery. Current developments are prominently geared towards containing and unravelling the inherent complexity of cellular automata, through sensitivity analysis and validation[77], and the development of methods for estimation/calibration[78,79,80]. Calibrating land-use change models requires reasonable baseline data, against which model outputs can be trained. The advance in urban remote sensing technologies and post-capture processing computing has enabled the rapid proliferation of land-use data at increasingly high resolutions. The earliest urban remote sensing for quantifying impervious surfaces made use of aerial photography[81,82]. Archives of aerial photography stretch back over 60 years in some places, however their coverage is often temporally and spatially sparse. Despite this limitation, aerial photographs provide a valuable resource for quantifying urbanisation in selected areas[83]. The upgrade to high-resolution imagery enabled increased accuracy for detecting urban spread[84]. Commercial high-resolution sensors have traditionally been restricted owing to international security concerns. However, such limitations have since been relaxed and high-resolution satellite imagery (e.g., IKONOS-2, ORBIMAGE Inc and Quickbirk-2) are now widely available, providing scope to track real-time changes within urban areas[85, 86, 87,88].

Most urban applications of remote sensing (RS) have used optical data, although radar and LiDAR are becoming increasingly common. For urban hydrology there are three key types of data set derived from RS, specifically land cover classifications, impervious surface and topography. Land cover classifications provide maps of land cover that typically include at least one urban class. Examples include urban extent from land cover maps which are widely available at a range of scales from country-wide, such as the US National Land Cover Database (NLCD)[89] and the 2007 UK Land Cover Map (LCM2007)[90], to continental and global-level with the Moderate Resolution Imaging Spectroradiometer (MODIS)[91] and GLOBCOVER products. Currently a number of single-class (i.e., urban/non-urban) RS products are in production including the European Urban Atlas and Germany's Space Agency (DLR) global classification of human settlement from TanDEM-X, generating a global urban extent dataset with a spatial resolution of around 30m[92]. Impervious surface mapping contains a number of methods that have been developed for capturing the percentage impervious area from satellite data[87,93,94]. An impervious surface product derived from Landsat data is now part of the United States national land cover data based (NLCD) products[65], including a procedure to update it[96]. Finally, topographical information can be derived from both LiDAR and InSAR data and have an important role to play in urban flood modelling studies[97]. Less research has been conducted on deriving 'effective' or 'directly connected impervious area' (EIA/DCIA), and how impervious such surfaces are in reality[87]. Early efforts to derive EIA/DCIA focused upon using empirical hydrological data or field surveys[98] while more recent research has utilized technology such as RS and GIS to estimate the effective impervious area[99]. With modern urban developments increasingly containing sustainable drainage systems (SuDS) such as permeable paving (that are difficult to distinguish via remote sensing from impervious paving), there is a requirement for more detailed mapping of urban land-use to reflect the diversity of hydrologic-footprint affected by contemporary urban land-use, for which high-resolution sensors provide scope.

3.2. Urban Drainage Data and Observation

Given careful processing of remotely sensed imagery or land-use data, GIS techniques permit connections to be made between changes in land-use and extension of drainage networks. In the United Kingdom, following the Pitt review into the 2007 flooding[100], most local authorities and water companies developed digital records of urban drainage infrastructure, enabling an increasingly accurate system-based representation of city drainage dynamics. Therefore, as charted in Section 2.2.2 the increasing widespread availability of digitised drainage data has enabled the development of detailed and powerful urban drainage models.

However, the use of such models has often been hindered by the lack of available observed data for calibration. Information pertaining to the design and operation of various key elements to urban drainage infrastructure are often being subject to administrative and political barriers to access. In this respect, specific information on diversion structures, treatment units, combined sewer overflows and depression storage areas, important for successful applications of complex process-oriented models are not always readily accessible.

These limitations of data accessibility are perhaps most severely compounded by uncertainty surrounding the integrity of such networks, where defective pipe systems can result in considerable influx or loss of water. Leaking infrastructure has been shown to contribute significantly to groundwater recharge across Europe. For example, Yang et al., estimated that 70% of groundwater recharge from the UK city of Nottingham could be traced to leaking infrastructure[101]. A report by the European Environment Agency demonstrated that leaking infrastructure can range from 3% in Germany to over 50% in Bulgaria[102]. Lerner identified recharge rates of 30% in Lima (Peru), and also attributed 50% of total recharge in Hong Kong to leakage contributions with values ranging from 260mm to 2950mm/yr[103]. Contributions from leaking infrastructure to soil water, and thus groundwater recharge, have long been recognised but quantifying their impacts has proven problematic, often resulting in adoption of environmental tracer techniques and model applications[104,105,106]. Whilst both approaches are valid, these estimates are often time consuming and uncertain. A direct method for measuring loss has thus far proved elusive.

Leakage detection systems (LDS) have advanced substantially in the past few decades as utility companies and governments recognise the economic and resource benefits of identifying and treating inefficient infrastructure[107]. Labour-intensive, costly traditional methods of detection are based on regular schedules or specific failure events[108,109]. Technological advances in Wireless Sensory Networks (WiSN) have resulted in the emergence of portable, inexpensive equipment of low-power requirement for remote real-time monitoring of pipe flow[110]. WiSNs monitor pressure, flow velocity and vibrations (via acoustic sensors) in supply- or sewage-pipe networks, where a change in the respective variable is recorded between sensor nodes and can be integrated into a GIS network, where wireless relays can highlight emergent problems[111]. Experimental applications of WiSN highlight potential to utilise this technology as a method for deriving volumes of water loss from underground transfer systems[112,113]. For example, Li et al., applied a WiSN to the water supply system in Beijing and estimated that it helped detect pipe defects and inform maintenance that reduced leakage volumes by $4*10^6$ m^3 of drinking water between 2007 and 2009, representing 80% of China's cumulative water losses ($5*10^6$ m^3)[114]. WiSN networks can provide data that quantifies water losses from infrastructure and allows utility providers to repair defective pipelines. However widespread application of such networks has not yet been realised.

3.3. Data Supporting Rainfall-Runoff Modelling

3.3.1. Rainfall Monitoring

Conventional rainfall gauging techniques have increasingly been supplemented by widespread radar coverage, which provides greater scope for predictive interpretation of evolving rainfall patterns, although it is accompanied by significant uncertainties. An extensive overview of rainfall-radar methods is beyond the scope of this review but is provided by numerous authors[115,116,117]. Radar has obvious advantages over standard gauges, providing areal averages as opposed to point measurements. Advances in radar-rainfall estimations have been two-fold: (i) in radar sensory systems and (ii) the processing algorithms that correct against known sources of error[118]. However, it should be stressed that prior to use in hydrological modelling, difficulties of handing large datasets and complications concerned with the calibration of radar data still remain and require pre-processing[119,120]. The recent emergence of X-band technology scales spatial resolution by a factor of 10, resulting in the ability to monitor precipitation data at the hectometre scale, rather than kilometre[121]. As a result, X-band radar has the potential for important applications in the urban environment, providing local authorities with smaller and more affordable infrastructure that has the capacity to generate forecasts and early warnings at the very local scale (e.g. EU funded project, RainGain (http://www.raingain.eu/en/raingain)). The development of X-band polarimetric radar instruments is an encouraging step, as they are capable of detecting ground clutter and enhance the reflection to rainfall relationship through the incorporation of polarimetric parameters and correction for attenuation[5]. The shift towards radar has resulted in increasingly high spatial and temporal resolution rainfall data, which is useful in modelling applications, particularly for the prediction and management of flood events[122,123].

New radar developed in UK measures humidity via a refractivity index, which is then calibrated via synoptic weather stations on the ground[124] This identifies areas where the outbreak of convective storms is likely to occur, thus potentially improving the ability to forecast pluvial flooding in urban areas. Additionally, the measurement of the emission from attenuating objects can now be determined and used to correct signal uncertainty[125] This technique is currently being implemented into the UK radar network and will result in the reduction of real-time estimation uncertainties. Increasingly novel methods are being applied to determine rainfall estimates across urban areas, which could greatly aid in calibration of more conventional monitoring techniques (Table 1). Zinevich et al., (2010) cite a relatively successful means of detecting rainfall fields using microwave tomography from commercial mobile phone networks[126]. This approach relies on the attenuation of microwaves during precipitation events and as terrestrial microwave links are normally within 10m of the ground surface, such a method provides a good basis for estimation of precipitation at the near-surface area[127]. Recent research by Bianchi et al., demonstrated the possibility of combining this information with rain gauges and radar to derive increasingly accurate precipitation estimates[128].

3.3.2. Evapotranspiration

Historically, evapotranspiration in urban areas has been the focus of urban climatologists interested in capturing the urban energy budget and quantifying the urban heat island[13,129]. Large buildings and street canyons create local variations in airflow and thermal regimes, which result in the transfer of energy and highly variable eddy currents, affecting the dynamics of evapotranspiration[130]. Furthermore, the patchwork presence of impervious surfaces and vegetated areas create a complex pattern of evapotranspiration, where empirical measurement can often be difficult. However, the varying height of the urban roughness sublayer and inaccurate spatial and temporal scaling laws was demonstrated to create major uncertainties in this method in urban areas[131]. Empirical studies of urban evapotranspiration initially utilised eddy correlation methods, which measure the vertical velocity and moisture content of air parcels[132]. Promising advances in technology, such as eddy covariance[133] and the application of scintillometers[134], are increasingly being employed to derive accurate and reliable measurements of urban heat fluxes and ET.

3.3.3. Infiltration Monitoring and Estimation

It is long established that the primary impacts of increase in impervious surface are reduced infiltration rates and travel times[96]. Recently, evidence relating these impacts to increased surface runoff, and reduced baseflow, has been extensive[5,135,136,137,138]. However, incorporation of new knowledge in hydrological models has been slow. In the past some modelling approaches have made the simplified assumption that 100% of the rainfall is converted into runoff transported overland and through stormwater drainage systems in impervious areas, yet this has often been found to be unrealistic. For example, Hollis and Ovenden found that the percentage runoff from a road was 50%[139]. Similarly, Ramier et al., found that between 30%-40% of rainfall on two roads was accounted for by evaporation and infiltration[140]. In a study of runoff from roofs, Ragab et al., found that, depending on pitch and aspect relative to dominant wind-direction, the percentage runoff varied from 61% to 91%[141]. Clearly, the term 'impervious' can be misleading, and runoff from these areas should not automatically be assumed 100% of rainfall. Similar generalisations are made for pervious areas, in which soil infiltration is typically assumed to occur ‘naturally’. In reality, compaction and mixing of soil horizons during construction results in reduction in porosity, where saturation excess is reached more quickly than in natural areas, and consequently runoff contributions from pervious areas are often underestimated in model applications.

Assumptions concerning infiltration dynamics in urban space can remain a substantial source of model error. However, increasingly advanced field and laboratory techniques to determine infiltration and percent runoff estimates have been developed and can be widely applied, and thereby be potentially very beneficial for modelling, reducing the reliance on generalised assumptions. Field and laboratory based permeameters provide a hydraulic head to drive water into soil surfaces and directly derive saturated hydraulic conductivity[142,143]. Mini disk portable tension infiltrometers are easily transported and deployed in the field, enabling multiple spatial estimates. However, some of these ‘direct’ methods remain invariably time consuming and costly; require certain conditions to facilitate their use (e.g., flat surface), and are not always representative of soil infiltration dynamics across wider areas. Electrical resistivity techniques are potentially cost-effective for determining in-situ estimates of soil infiltration capacity[132]. Furthermore, hyperspectral remote sensing techniques can determine soil moisture at the near surface zone with a statistically significant correlation ($R^2 = 0.7$) to simultaneous observations[144]. In urban areas, the applicability of these techniques is limited to pervious zones that permit measurement, often negating the inputs from infrastructure (e.g., septic tanks and ineffective infrastructure) sealed beneath urban surfaces[145]. This can be compounded by traditional assumptions that all rainfall in impervious areas is immediately routed to runoff and that infiltration defaults to zero. Determining the infiltration rates of pervious urban areas remains an important and active area of research which will provide insights on runoff generating mechanisms. It will allow more accurate coupling in models between pervious and connected impervious areas. Furthermore, a concerted effort is required to validate values of infiltration rate used for impervious zones, which currently may be impairing model performance.

3.3.4. Urban Runoff Observation

Conventional velocity-area measurements remain the most common technique (c. 90% of gauging sites worldwide)[145] for determining discharge, whereby water velocity is measured at selected intervals across a known channel cross-section. The 2010 World Meteorological Organization (WMO) Manual on Stream

Gauging[145] outlines the advances made in hydrometry since the first version was published in 1980[146]. The emergence of novel techniques such as Acoustic Doppler Current Profilers (ADCPs); Acoustic Doppler Velocity Meters (ADVMs); laser Doppler velocimetry (LDV); electromagnetic channel-width coil and radar gauges have improved the resolution of automated monitoring. Measurements are now possible in uniquely urban locations such as sewer pipes and storm drains that are difficult to access. In such environments monitoring has previously been absent hindering calibration of urban drainage models[12]. Furthermore, the implementation of such technology onto remotely operated platforms (e.g., ARC-Boat®, HR Wallingford) permits monitoring in particularly dangerous or contaminated environments without posing risk to the user. Monitoring flow and bathymetry via remote control[147] enables characterisation of hydrological events, making for better calibration of models. Finally, flow data capture, storage and transfer has been greatly advanced through application of satellite and radio based telemetry methods[148], giving rapid access to data that can be used to provide real-time river level information for flood warnings (e.g. UK Environment Agency, NOAA). For example, Fulton and Ostrowski applied hand-held radar to determine flow velocities from which real-time streamflow were derived and input to existing hydraulic routing models[149]. Furthermore, real-time hydrological data provides early diagnosis of equipment error, thus maintaining continuity and reducing data loss[148].

3.4. Water Quality Monitoring

Pollutants found in urban environments can be divided broadly into two main types: (i) those arising from natural processes as a consequence of overstimulation of production and decomposition, and (ii) toxic chemicals originating from manufactured products, including metals and persistent organic compounds. The group arising from natural processes is small and includes macronutrients (N and P), organic waste (usually described in terms of its impact on oxygen levels: Biochemical Oxygen Demand) and pathogens. The problems of these substances are historically well-documented and substantial although the diversity of their sources and transport pathways in the urban environment make prediction of their impacts challenging. With regards to toxic substances, little is known in terms of the detailed composition in the various pathways of diffuse urban pollution. They are often poorly retained in wastewater treatment works, especially the less soluble compounds, and their biotic accumulation and storage in sludge are becoming of increasing concern[150,151]. Increasingly, emerging pollutants such as nanoparticles[152]; pharmaceuticals and endocrine disruptors[153] and microbial bacteria and pathogens[154,155] have resulted in a shift in emphasis in water quality analyses, resulting in increasingly complex monitoring strategies and technologies. The following sections assess the advances in monitoring physical, chemical and emerging priority pollutants in urban river systems.

3.4.1. Geomorphic Erosion and Sediment Fluxes

Since the 1960s, substantial advances in understanding of how urbanisation affects geomorphic processes[156] have been made, yet three major challenges remain: first, there is a need to quantify the aggregate impact of urbanisation on sediment transport at spatial and temporal scales greater than those of individual studies[157,158]; second, there is a need to improve understanding of how changes to the downstream river environment may subsequently alter in-stream processes and channel morphology[159,160,161]; and third there is a growing recognition that impacts of urbanisation on aquatic ecology can be understood properly only when the effects of urbanisation on flow and sediment delivery are also taken into account[162]. Establishing the origin of sediment mobilised from urban areas is particularly important. However, the dynamic nature of sediment transport results in many well-recognised difficulties for field measurements in river systems. Sediment transport is typically divided into three categories, based on size classifications. Of primary concern regarding fluxes of contaminants from urban areas:

(i) Bedload sediment - represents larger material, including gravels and pebbles which will only be mobilised under elevated discharges or steep bed gradients, making for problematic monitoring conditions, and often confined to more upland areas[163]. Conventional measurements of bedload can be separated into four categories: box/basket (sediment traps), trays (or pans), pressure difference samplers (e.g., Helley-Smith) or pit samplers. In addition, grab samples and cores for laboratory analyses can provide insight into the physical and chemical compositions of bedload layers. Measurement entails a range of problems, including capturing of suspended sediment loads in addition to bedload[164].

(ii) Suspended sediment –Acoustic Doppler Current Profilers (ADCPs) are increasingly being explored as a surrogate method for determining suspended sediment concentrations[165]. Guerrero et al., (2011) assessed a dual-frequency ADCP in the laboratory[166] and concluded that acoustic backscatter (ABS) techniques are a robust methodology for determining grain size distribution for suspended sediment columns. Additionally, measures of optical turbidity provide an indirect method of estimating SSC, where strong correlations with discharge are apparent. State-of-the-art turbidity meters are increasingly deployed in conjunction with electrical conductivity probes, as dissolved solids can significantly affect the conductance rates[167]. As suspended sediments provide a viable mechanism for transporting chemical and biological contaminants, physical sampling for post-capture characterisation in the lab is desirable.

3.4.2. Chemical and Biological Water Quality Metrics

Chemical and biological indicators remain the dominant focus of policy. Legacies of intensive agriculture and industry resulted in elevated levels of a vast suite of contaminants in freshwater environments including phosphorus, heavy metals and nitrates. As the human footprint continues to expand and agricultural practices continue to intensify, chemical and biological contaminants still remain a dominant threat to water quality from both point and non-point sources.

Autosampling techniques were gradually introduced to enable more frequent and comprehensive sampling without the need for excessive manpower (e.g., programmable ISCO 3700 or flow-triggered Endress and Hauser Liquiport 2000EX samplers). Despite their undoubted utility (particularly for more complex chemical and biological water quality indicators that still require laboratory analysis) autosampling systems are often expensive, particularly where monitoring is spread across multiple areas (Table 1). The emergence of real-time remote monitoring (RTRM) systems have been widely utilised across many sectors providing cheap, accurate and dynamic data capture[168]. RTRM systems incorporate an array of sensors that capture, store and relay meteorological, hydrological and water quality data (Table 1). For water quality, multi-parameter sensors are attached to a bundled device (e.g., YSI 6600ED and RSHydro Manta 2 sondes) and deployed into water bodies, remotely recording and relaying data via radio or satellite telemetry systems. Continual monitoring networks provide an avenue to obtain accurate and long-term data, which can be used to determine spatial and temporal changes of water quality (including event response), drive policy decisions and inform modelling studies[168]. Furthermore, the introduction of alert systems provides a management tool for local and environmental authorities for responsive action at sites where toxic spills occur or large contaminant concentrations arise. In terms of biological monitoring, chlorophyll (a surrogate for phytoplankton biomass) and cyanobacteria are nowadays routinely captured using fluorescence sensors in RTRM systems, and are also extensively surveyed using airborne hyperspectral remote sensing[169]. When used in conjunction with cell-based characterisation through flow cytometry[170] valuable information is attained on functional composition. In sharp contrast, the highly constraining expense involved in monitoring other biological groups such as macro-invertebrates and fish restricts data availability to, at best, surveys at a seasonal-resolution.

3.4.3. Emerging Priority Contaminants

Increasingly complex contaminants, such as pharmaceuticals, personal care products, endocrine disruptors and nanoparticles are emerging and require novel monitoring techniques. The ecological threats from these contaminants (collectively referred to as Organic Wastewater Contaminants or OWCs) in surface water sources have long been recognised, despite their status as 'emerging priority contaminants'[171]. Their presence in wastewater is almost ubiquitous, presenting a major challenge to treatment plants[172].

Conventional methods for detecting OWCs rely on grab sampling, restricted by both cost and availability of manpower, and also by post-capture processing in the laboratory which requires tandem gas- and liquid chromatography mass spectrometry (GC-MS/MS and LC-MS/MS respectively)[172,173]. Where endocrine disrupting contaminants and pharmaceuticals and personal care products (PPCPs) are present only at trace levels, large volumes of water are required to facilitate sufficient detection, making regular transport of samples increasingly difficult. Some of these limitations can be overcome by autosampling technology, though as discussed in the previous section, this is often expensive and impractical, and such systems are not widely used in extensive sampling regimes[175]. Passive sampling techniques have been successfully applied in environmental monitoring for the best part of four decades for a range of pollutants[174,175]. Continuous submergence of samplers beneath the surface allows time-integrated flow of analyte molecules to pass through a receiving membrane until chemical equilibrium is reached, conceptually replicating absorption rates in aquatic ecology in a given location. For example, the Polar Organic Chemical Integrative Sampler (POCIS) is a highly sensitive device specifically tailored to detect PPCPs and pesticides/herbicides in freshwater systems. The POCIS system samples in the dissolved phase, which enables contaminant bio-availability to be estimated[174] and can be specifically configured to detect drug residues[176]. Passive sampling represents the state-of-the-art means to detect OWCs in water systems although extensive laboratory calibration and analysis is still required to accurately determine trace concentration levels (Table 1). To conclude, an in-situ method for detection that bypasses the need for extensive laboratory analysis and is suitable for real-time monitoring and alert systems remains lacking.

Overall, the ability to monitor contaminants in the urban environment has improved dramatically over the past few decades, greatly aided by the continual improvement in sensor- and communication technology. RTRM technology has enabled a method for identifying and managing high concentrations of contaminants and spills via an alert system. As increasingly complex contaminants such as microbial pollutants, PPCPs, endocrine disruptors and nanoparticles emerge, new challenges for monitoring fuels further research[177].

4. Discussion and Conclusion

4.1 Summary

Despite the advances made in urban hydrology over the past few decades, a holistic understanding of the complexities of the urban environment remains lacking, and is an active area of research. The evolution of monitoring and modelling techniques has advanced our capacity to obtain observations and predictions at increasingly high-resolution spatio-temporal intervals resulting in a paradigm shift toward integrated model applications. As new technologies continue to emerge, the scope for advancing our understanding of urban hydrology increases considerably. Here, we summarise findings from the preceding sections and identify how these advances in monitoring and modelling technologies can be used to drive the future of urban water research.

- Improvements in remote sensing platforms have greatly aided our ability to identify urban landscape at increasing spatial and temporal resolution. As sensor technology, coverage and post-capture analysis techniques continue to improve, there is real scope for remote sensing technologies to drive hydrological research in urban and rural areas alike.
- Whilst performance mismatch is still warranting much attention, the development of novel methods for detecting the spatial and temporal patterns of rainfall (e.g., radar and microwave tomography) have advanced our ability to predict and manage rainfall in urban areas. This is crucial for flood management and drainage design. The move towards now-casting and real-time application remains a current research priority.
- Quantifying evapotranspiration, which is historically the domain of urban meteorologists, has increasingly been recognised by urban hydrologists as crucial to closing the water balance. This remains an active area of research but technology such as eddy flux chambers and scintillometers demonstrate potential for widespread application in networks designed to measure ET for water balance assessment across contrasting urban space.
- Achieving realistic measures of infiltration in pervious and impervious areas alike has been identified as a significant knowledge gap. Overcoming the assumptions that impervious areas prevent any infiltration, whereas parkland areas behave ‘as natural’ are crucial to understanding rainfall-runoff dynamics. Additionally, the development of wireless sensory networks (WiSN) has greatly aided our ability to quantify leakage from infrastructure and determine infiltration into- and out of pipes. So far, limited applications have highlighted success at reducing water losses. Widespread implementation could reduce urban water loss considerably, whilst providing urban hydrologists with suitable data for parameterising recharge in urban hydrological models.
- The development of novel flow measuring devices has increased our ability to monitor in arduous conditions (e.g., flood events) and challenging locations (e.g., pipes), providing further insights into river bathymetry and velocity mapping. As recording devices become increasingly portable and inexpensive, widespread deployment across urban catchments allows us to understand flows and associated fluxes in increasingly disparate parts of the system, enabling us to trace sources and pathways for water and contaminants.

- A move towards RTRM technologies has greatly improved understanding of water quality dynamics in urban areas. Rapid detection of a suite of water quality contaminants and live telemetric transfer can aid in water quality management during storm events, or detection of spillages from overflows or industrial effluent outputs.
- Efforts to model the urban water balance and water quality have advanced through the move towards an integrated approach, which considers multiple connected systems (e.g., WWTP, surface runoff, pipe flow) as boundary inputs, creating a holistic representation of the urban area. As monitoring data continues to improve and become more readily available, models are becoming increasingly complex as we transition toward spatially distributed, continuous simulation structures in a systems dynamics approach. This is perhaps the most pragmatic way of representing the urban water cycle and more widespread application will improve our ability to represent the system in a more holistic manner, providing a robust baseline for future projection in response to changes in land use, urban expansion and climate.
- The advent of GIS techniques to support for example (i) increasingly high resolution topographic data, (ii) characterisation of urban infrastructure, more detailed land-use observations which better identify contiguous impervious areas has enabled the implementation of more powerful modelling techniques to relate land-use change to water resources.

4.2 Remaining Challenges

Remote sensing provides an unparalleled tool for tracking urban growth but as new developments continue to implement sustainable urban drainage features such as permeable paving and road surfaces, and green rooftops, increasingly high-resolution sensors are required to differentiate them from well-understood and conventional land classes. Despite its importance to the urban water balance, empirical derivation of evapotranspiration remains limited and our ability to satisfactorily quantify ET rates in urban areas is lacking. As urban planners continue to restore evapotranspiration as a sustainable management technique for stormwater, an improvement to current methodology is required for deriving ET estimates in urban areas. Although the emergence of radar and microwave tomography has advanced our ability to predict spatio-temporal patterns of rainfall, an important research priority is a move towards shorter-duration prediction and 'now-casting' techniques that will greatly aid in urban flood management. Whilst integrated models provide a mechanism for distilling the wealth of data that can now be attained through monitoring regimes as well as elucidating the potential impacts of environmental and land-use change there are still limitations in their scope, particularly in terms of ecological impacts. The links between water quality and ecology are not well understood. Channel geomorphology, which has been observed to undergo systematic change during urbanisation[156], in particular in terms of artificial modification, is known to influence in-stream biological communities, whereby reductions in in-stream sediment budgets reduce breeding habitats for fish and macroinvertebrates[4,178]. In this regard, taking steps to restore habitats is recognised as being of fundamental importance[179]. In terms of modelling, ecological response is typically represented by empirical statistically-based approaches[180], in part a consequence of the practical restrictions imposed by data collection methods. Any mechanistic representation of ecology is largely absent from integrated models of urban systems, and should be an aspiration for future research.

In spite of these shortcomings, in order to assess urbanisation at the river basin scale, at which water resources are managed for pollution control and for mitigation against extreme events, it is necessary to distil the impacts of urbanisation into simple yet robust formulations that capture the complex dynamics represented in process-based models. As a step towards achieving this, the overarching objective of the POLLCURB project, we consider that quantification of various essential features must be made, notably the extent of urban and suburban land-use, soil infiltration and rainfall regime, and key thresholds above which sewer infrastructure capacity will be breached. For this reason, we propose that at basin level description of natural water bodies is geographically detailed (an ICBM on Figure 1). Integration at the IUDM level is achieved with meta-modelling approaches, whereby simplified representations of urban infrastructure capture the main features of more-detailed models. In contrast, at the IUWSM level detail must not be compromised in the provision of land-use and climate change drivers

The improvements to monitoring technology and modelling capability present us with renewed ability to assess impacts of urbanisation. Availability of high-resolution remote sensing platforms allows identification of subtle spatial and temporal changes in urban and suburban land cover, hence detailed networks of pervious and impervious surfaces can be supplied to hydrological models. This can be combined with empirical measurements of infiltration capacity and high-resolution rainfall to better determine rainfall-runoff dynamics at increasingly fine scale. In addition, digitised networks of stormwater drainage infrastructure improve the accuracy of runoff routing and simulation of flows to urban channels. RTRM networks can be used in conjunction with rainfall-runoff from specific land uses to determine pollutant loads. This combination of high-resolution monitoring systems and integrated model structures can be used to identify overflow events and quantify their potential water quality and flood implications. As highlighted in Section 2.2, POLLCURB seeks to build on these advances by incorporating high-resolution data into an integrated model to be tested and used to make future projections. It will provide scope for assessing both continuous and event-based impacts of hydrology and water quality under changing land-use, population and climate scenarios. The implementation of robust data from advanced, high-resolution monitoring techniques creates scope to intrinsically link landscapes to their dynamics and assess change over space and time and contrasting scales.

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Figure captions

Figure 1: A typology of integration for urban water models (a concept adopted from Bach et al. (2014)[12]: Figure 2) comprising four tiers of increasingly broad integration: 1. Integrated component based models 2. Integrated urban drainage models/Integrated water supply models 3. Integrated urban water cycle models 4. Integrated urban water system models. The scope of the POLLCURB case-study model is indicated on the diagram with grey ellipses; and the position across these tiers of some of the models/EDSS cited in the text are also indicated in italics

Tables

Table 1: Chronological evolution of monitoring techniques used within urban hydrological studies

Variable	Decade	Monitoring Advances	Examples/Specification	References
Land Cover/Land Use	1910s	Aerial photography	Sub-meter resolution maybe manually or automatically analysed – costly as obtained to request with no regular coverage	82
	1970s	Landsat 1-8 (including ETM$^+$)	Broad land cover and change detection up to 15m panchromatic (pan) resolution	181 & 182
	2000s	QUICKBIRD-2	High resolution (up to 0.64m pan resolution) land cover/change to internal urban scale	183
	2000s	WORLDVIEW-2	Highest resolution commercially available (0.5m pan resolution) land cover/change	184 & 185
Topography	Historical	Surveys and Elevation Maps		
	1980s	Airborne LiDAR	e.g., GeoDigital TerraPoint	185
Impervious Fraction (%)	1970s	Landsat 1-8 (including ETM$^+$)	% urban pixel for ~30m resolution	181; 182
	1980s	Optical and LIDAR		
	2000s	Interferometric SAR (InSAR)	e.g., TerraSar-X	186
Rainfall	Historical	Collection bucket raingauges	Lumped volumetric measurement	

Point Measurements	1960s	Tipping bucket gauges	e.g., ASOS	
	2000s	Optical and acoustic gauges	e.g. HYDREON RG-11	
Areal averaging	1950s	Radar	Advances from C to X band	187
	2010s	Microwave tomography	Signal attenuation – mobile networks	126

Table 1 (Cont): Evolution of monitoring techniques applied within hydroclimatic studies in the urban environment

Runoff	Historical	Velocity-area discharge	Most common technique globally (c.90%)	135
	2000s	Ultrasonic and Radar flow gauges	e.g., ADCP and Vegapuls	146
	2010s	Remote ADCP monitoring	ArcBOAT® - remote controlled ADCP logger-mounted	188
Water Quality	Historical	Manual sampling and laboratory analysis	Manual Sampling and targeted laboratory analysis	
	1960s	Autosampling chambers	Auto-samplers (e.g., ISCO 4700)	135
	2000s	Passive sampling	Passive water quality samplers (e.g., POCIS)	174 & 189
	2000s	Real-time remote monitoring (RTRM)	Multiparameter sondes (e.g., YSI 6600; DataSonde IV and HydroLab) and remote telemetry	146 & 148

Table 2: Example models that are commonly applied in urban runoff analyses, including both urban drainage and rainfall-runoff models

Model	**Model Type**	**Description**	**References**
SCS	Simple, empirical urban rainfall-runoff model	Statistical method for calculating runoff based on soil type, antecedent soil moisture conditions and land use for a given area.	190
DR_3M	Distributed hydrologic and hydraulic model	Watershed model that routes water through branched sewer networks and into channels, incorporating soil moisture, overland flow and storages.	191
HSPF (aka Stanford Watershed Model)	Physically based process model	Uses continuous rainfall and associated meteorological metrics to calculate streamflow hydrographs and water quality with urban parameters.	192
Dynamic TOPMODEL	Distributed land-surface and hydrologic model	Newest version of TOPMODEL, implemented into R, encompassing land-use processes, rainfall-runoff at distributed scale.	193
USEPA SWMM5	Semi-distributed hydrologic and hydraulic model	Dynamic rainfall-runoff model that operates under continuous or event-based simulation of quantity and quality in both surface and pipe network systems.	194
MIKE-URBAN	Semi-distributed hydrologic and hydraulic model	GIS-based integrated model that incorporates SWMM and EPANET. Includes pipe-flow, rainfall-runoff and water quality transformation and transportation dynamics.	195
CITYDRAIN	Integrated hydrologic and WWTP model system	CITYDRAIN is a systems dynamic model that represents subsystems of the urban landscape, including sewers, receiving waters and WWTP processes.	28
SIMBA®6	Integrated hydrologic and WWTP model system	Complex model that simulates interactions between runoff, WWTP, receiving water quality – includes detailed representation of WWTP.	26
PCSWMM	Semi-distributed hydrologic and hydraulic model	Couples the powerful USEPA SWMM5 model with GIS software to enable streamlined GUI operation of hydrological, hydraulic and water quality simulation.	196

Table 3: Chronological evolution of water quality models commonly applied in the urban landscape

Model	**Model Type**	**Description**	**References**
Streeter-Phelps Model	BOD/DO	1D model for simulation of untreated effluent in riverine and estuarine environments. Owing to non-computer operation, the S-P model relied on linear kinetics and was applied to steady-state receiving waters.	197
Thomann's Expanded Streeter-Phelps Model	BOD/DO	Expansion of Streeter-Phelps model to incorporate multi-segment river systems with multiple contributions to oxygen development and use, including photosynthesis and sludge aeration.	198
DRAINMOD	In-stream Nutrient (N) Transport	An early nutrient model that sought to simulate transport and transformation of nitrogen in both shallow soils and in-stream environments.	199
WASP7	Dynamic in-stream flow and water quality model	Up to 3-D modelling of a suite of contaminants in the river, including metals, nutrients, VOCs, PCBs and sediment.	200
SHE	In-stream and groundwater flow and water quality	A physically-based, distributed catchment scale model which functions as a valuable decision support tool for a suite of hydrological contaminants.	201
QUESTOR	In-stream flow and water quality model	A flexible framework that can represent both flow and water quality within river reaches, incorporating multiple point sources and tributaries within a stream network.	202
QUAL2K	In-stream flow and water quality model	Building on the earlier QUAL2E model, this is a 1D model that simulates nitrogenous and carbonaceous BOD speciation, pH, sediment fluxes and pathogen dynamics.	203

Related Articles

DOI	**Article title**
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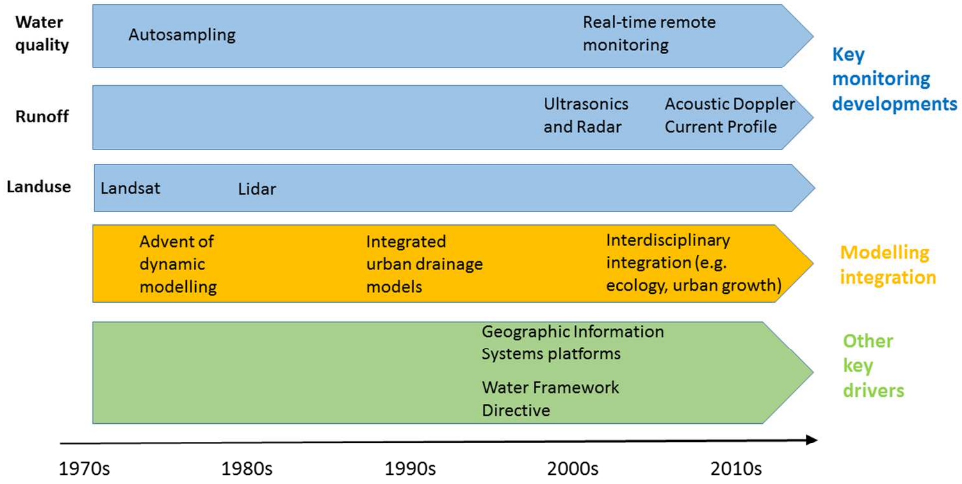

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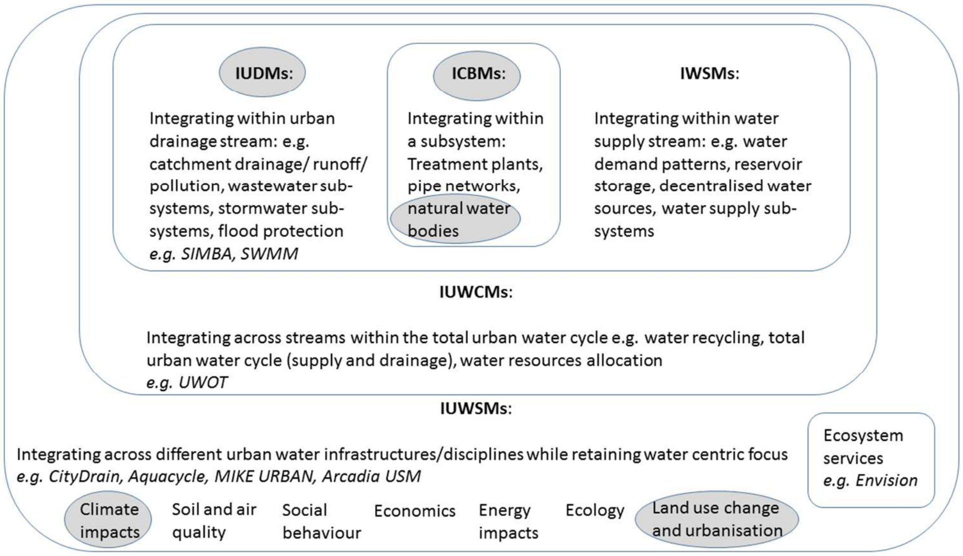

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