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Impacts of urbanisation on hydrological and water quality dynamics, and urban water management: a review

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ABSTRACT

As urban space continues to expand to accommodate a growing global population, there remains a real need to quantify and qualify the impacts of urban space on natural processes. The expansion of global urban areas has resulted in marked alterations to natural processes, environmental quality and natural resource consumption. The urban landscape influences infiltration and evapotranspiration, complicating our capacity to quantify their dynamics across a heterogeneous landscape at contrasting scales. Impervious surfaces exacerbate runoff processes, whereas runoff from pervious areas remains uncertain owing to variable infiltration dynamics. Increasingly, the link between the natural hydrological cycle and engineered water cycle has been made, realising the contributions from leaky infrastructure to recharge and runoff rates. Urban landscapes are host to a suite of contaminants that impact on water quality, where novel contaminants continue to pose new challenges to monitoring and treatment regimes. This review seeks to assess the major advances and remaining challenges that remain within the growing field of urban hydrology.

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1 Introduction

In March 2012 the global population exceeded 7 billion people for the first time, representing a doubling of the global population in less than 50 years (United States Census Bureau, 2012). It is estimated that more than 55% of the global population live in cities and that 394 of the world's cities have a population that exceeds 1 million inhabitants (UN 2011). Furthermore, it is anticipated that 83% of the developed world and 53% of the developing world will live in urban areas by 2030 (Cohen 2004). As the global population continues to grow at a rapid rate, the expansion of urban areas continues to pose a significant threat to natural dynamics, resource availability and environmental quality, and advancing our knowledge of urban hydrological processes remains a priority within the field of hydrological science (Niemczynowicz 1999, Vörösmarty *et al.* 2000).

The discipline of urban hydrology remains relatively young and has become increasingly relevant in a world that has experienced a marked, rapid growth in population in the past few decades, with varying dynamics of urban growth across the world (Jacobson 2011). Traditional research sought to assess a catchment scale response to urban development, seeking to identify the impacts of upstream urban development on downstream hydrological and water quality dynamics. In developing countries, urban growth continues to occur across large spatial scales, often with entire cities being constructed in short times (e.g. Binhai New Area, China; Li *et al.* 2015). By contrast, urban development in developed countries occurs at local scales, with individual buildings or small housing estates being typical, aided in part by advances in monitoring technologies (such as high-resolution remote-sensing platforms) that provide insight into changing dynamics within the urban environment (Ragab *et al.* 2003, Blocken *et al.* 2013). A universal metric for measuring urban expansion remains elusive and the terminology of "urban" remains frustratingly disparate, impacting our scope for comparative analysis (MacGregor-Fors 2011). Multiple studies have addressed urban expansion using several metrics ranging from total population and population density to total or effective impervious area as a driver for hydrological dynamics, though a comprehensive metric of urban space remains elusive.

The urban landscape has a demonstrable impact on meteorological and hydrological dynamics alike. The artificial thermal properties and increased particulate matter from urban areas impact the way rainfall is generated and enhance downwind precipitation and may enhance the generation of convective summer thunderstorms (Jin and Shepherd 2005). Expansion of urban space results in an increase of impervious landscape and expansion of artificial drainage networks that can facilitate dramatic changes to the magnitude, pathways and timing of runoff at a range of scales, from individual buildings to larger developments (Walsh *et al.* 2005, Fox *et al.* 2012, Dams *et al.* 2013). The fabric of individual buildings can alter the way rainfall is translated into runoff and the interconnected nature of pervious and impervious surfaces impact the effectiveness

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of surface drainage during rainfall events (Yang *et al.* 1999). Furthermore, the presence of water supply and sewage treatment infrastructure help to transfer vast quantities of water and wastewater across urban areas. Traditionally, urban hydrology has sought to separate infrastructure flows from natural hydrological analysis, though an increasing awareness that inefficient and defective networks can lead to an additional influx of water and contaminants to natural systems has resulted in a step toward integration of the two cycles (Grimmond *et al.* 1986, Leung and Jiao 2006).

Urban hydrologists have increasingly focused on the water-quality implications of the expanding urban area and have sought to find ways of mitigating the risk of degradation to water bodies and their in-stream habitats (Walsh *et al.* 2005, O'Driscoll *et al.* 2010, Fletcher *et al.* 2013). The generation of runoff from urban surfaces can carry a suite of contaminants including heavy metals, major nutrients (e.g. sodium, nitrate and phosphorus), litter and rubber residue from roads (Tong and Chen 2002). More recent efforts have addressed the sources, fluxes and fate of more complex pollutants such as microbial contaminants (Tetzlaff *et al.* 2010, McGrane *et al.* 2014), synthetic chemicals (Heim and Dietrich 2007, Sullivan *et al.* 2007), pesticides (Varca 2012, Anderson *et al.* 2013) and pharmaceuticals (Jones *et al.* 2005, Burkholder *et al.* 2007). In developing countries, there remains a considerable threat from untreated wastewater discharging directly into natural streams, exhibiting a profound impact on aquatic integrity (Srinivasan and Reddy 2009). In developed countries, there is a growing movement to treat stormwater as a renewable resource as opposed to just a nuisance or hazard. Sustainable management practices and increasingly water-sensitive urban design strategies are being implemented to reduce the impacts of pluvial events in urban areas and help create areas that mimic "pre-development" dynamics and encourage ecosystem development, whilst providing an amenity to urban residents. Such strategies are widely implemented in new urban developments, but are also increasingly being applied at the individual building scale.

This paper seeks to review the role of scale and spatial density of urban areas in terms of both their impacts and management in the field of urban hydrology. It seeks to review how urban development at contrasting scales impacts on hydrological and water quality dynamics, whilst assessing how management of water in the urban environment is occurring at increasingly local scales. A characterisation of urban areas, including a disparity in the terminology and metrics of urban expanse are discussed in Section 2 and impacts of urban areas on hydrological dynamics and water quality are discussed in Sections 3 and 4 respectively. Management of water in urban areas is discussed in Section 5 and the paper concludes by highlighting some of the key research questions that require addressing to fully advance the science of urban hydrology.

2 Characterising the urban area

2.1 Urban definitions

A standard definition for an urban area remains frustratingly vague and lacking in the scientific literature and has been appropriated in various political, social and economic contexts. MacGregor-Fors (2011) highlights the multiple uses of urban areas relative to population density, overall population, and presence of specific structures such as housing/schools, impervious surfaces and percentage of non-agricultural economic activities, and the international disparity of the terminology is illustrated in Table 1. Boving and McCray (2007) highlighted the need for a more thoughtful definition of urban hydrology that includes a wider focus at the interface of physical and chemical hydrology and the urban environment. Understandably, a single definition of an urban area is incredibly difficult to implement in a meaningful manner that captures the diversity of global population distributions, economic practices or extent of impervious areas. Weeks (2010) argues that the most pragmatic way to undertake such a task is to abandon the concept of an "urban–rural" dichotomy and begin to think of these terms as ends of a continuum, with

Table 1. Example definitions of "urban" space from a range of countries, highlighting the inconsistency of the terminology.

Country	Definition	Source
Australia	Defined as "urban centres" where populations are >1000 and with densities above 200 people/km^2	Australian Bureau of Statistics
Brazil	Urban and suburban zones of administrative centres of municipalities and districts	UN Demographic Yearbook
Canada	Population of at least 1000 and no fewer than 400 persons/km^2	
Greenland	Localities of 200 or more inhabitants	UN Demographic Yearbook
India	A minimum population of 5000 people, at least 75% of the working population in non-agriculture, and minimum population density of 400 people/km^2	Census of India, 2011
Ivory Coast	Population of over 10 000 with 4000 being employed in non-agricultural industry	UN Demographic Yearbook
Japan	50 000 or more inhabitants with more than 60% of houses being within the main built up area. Alternatively, a shi having urban facilities and conditions as defined by the prefectural order is considered urban.	UN Demographic Yearbook
Peru	Populated centres with 100 or more dwellings	UN Demographic Yearbook
Republic of Ireland	Cities and towns including suburbs of 1500 or more inhabitants	UN Demographic Yearbook
South Africa	Places with some form of local authority	UN Demographic Yearbook
Spain	Localities of 2000 or more inhabitants	UN Demographic Yearbook
Turkey	Population of settlement places, 20 001 or over	UN Demographic Yearbook
United Kingdom	*England*: Population of over 10 000 people	ONS (2004)
	Wales: Population of over 10 000 people	ONS (2004)
	Scotland: At simplest definition, settlements of over 3000 people	Scottish Government (2012)
	N. Ireland: Derry and Belfast are the only "urban" areas differentiated from Large Towns	NISRA (2005)
Venezuela	Centres with a population of 1000 or more inhabitants	UN Demographic Yearbook
Zambia	Localities of 5000 or more inhabitants, the majority of whom all depend on non-agricultural activities	UN Demographic Yearbook

contrasting degrees of "urbanicity", incorporating economic, population, social and built environment indicators. Such an approach would provide a much more informative set of indices than simple population, percentage impervious area or density statistics, which alone do not provide a robust definition of contrasting urban dynamics. Although population is an important aspect of an urban environment, the urban area is a spatial concept that is only partly defined by the population within its boundaries (Weeks 2010). Wandl *et al.* (2014) recently proposed a new territorial classification technique, using a range of data that encompasses CORINE land-cover data, population and infrastructure statistics to separate urban and rural areas from "territories-in-between" that transcend urban and rural classifications, where "rural" areas exist within urban areas and vice-versa, which may serve as a pragmatic classification tool to distinguish between contrasting urban areas.

2.2 Metrics of measure for urban areas

The transformation from undeveloped spaces into urban environments results in marked alterations to the landscape, with spatial and temporal dynamics of change varying between developed and developing countries. Impacts are observed through the alteration of the topography and surfaces as a result of new construction, demolition and redevelopment and occur at a range of scales. Anthropogenic alteration of landscapes to expedite construction of buildings and infrastructure will impact on the dominant runoff-generating processes and key flowpaths, having a substantial impact on catchment boundaries and drainage pathways (Rodriguez *et al.* 2013). Modification to slopes, elevations, soils and vegetation coverage all impact on the way rainfall is captured, stored and released in hydrological systems. Conversely, the removal of natural gradients via the smoothing of surfaces (e.g. during the construction of roads and walkways) results in the development of simplified drainage structures to transfer water from urban surfaces as quickly as possible. Individual buildings alter the way water is captured, stored and transferred, where variables such as building material, infrastructure (e.g. drainage) and aspect are significant. In addition, major infrastructure projects (such as the development of road and rail networks) result in modifications to the natural landscape via the development of embankments and creation of sloped, impervious surfaces that are designed to streamline transportation and prevent the build-up of surface water, respectively. In flat landscapes, roads are often raised or realigned to prevent flooding and rising water tables during storm events, creating an artificial gradient and subsequent runoff pathway. The input of crossfalls (or cambers) is governed by road-building standards, where an angle (usually 3% for a paved road) is built into the carriageway design to aid water mobilisation off the road surface by the shortest path which consequently alters topography. The development of the Olympic Park at Stratford in London required excavation and removal of land to provide a suitable surface for the development of large buildings, sport arenas and open public access areas (Webster 2013). Much of the surrounding area of Stratford was desolate land that sat alongside degraded waterways, and development of the Olympic Village resulted in the generation of widespread areas of flat, impervious material and drainage networks, drastically altering the dominant hydrological pathways (Davis and Thornley 2010). More extreme examples occur in developing nations, where rapid urbanisation results in widespread clearing of lands to facilitate the development of entire new cities. For example, the development of the Lanzhou New Area in China where a 10 square-mile area of low montane topography is being flattened to accommodate a new development and has resulted in a significant transformation of the landscape (China Daily 2012). The Binhai New Area in Tianjin began development in the 1980s, covering an area of 3000 km^2, where considerable wetlands have gradually been replaced by widespread urban development (Li *et al.* 2015).

There remains considerable debate in the literature regarding the most pragmatic method for quantifying the spatial extent of urban areas and which metric is most suitable in urban impact analyses (OECD 2012). Historically, urban distribution was determined using population thresholds (see Section 2.1 and Table 1) and census data at a lumped scale, whilst more contemporary aerial photography and remote-sensing techniques have enabled rapid, high-resolution proliferation of data on urban extent and change. Research conducted at the city scale continues to utilise descriptive metrics such as total urban population and population density as proxies for urban areas. Often, such studies seek to consider either (i) the impact of urban growth on urban water infrastructure (e.g. Parkinson and Tayler 2003, Mikovits *et al.* 2014) or (ii) the impacts of population variance on water-quality dynamics (e.g. Xian *et al.* 2007, McGrane *et al.* 2014) over large spatial scales. From a hydrological perspective, population and density data provide limited scope for elucidating dominant rainfall–runoff dynamics at the more local scale, which is crucial for determining flood risk and localised areas of water quality concerns (Thomas *et al.* 2003). Rather, such dynamics are governed by physical surface characteristics and infrastructure contained within the urban landscape. In their perspicacious review of urbanisation impacts on hydrological and water quality dynamics in the Southern United States, O'Driscoll *et al.* (2010) highlighted a number of studies that utilise Total Impervious Area (TIA) as a reliable metric at the basin scale. Whilst lumped physical parameters such as TIA may serve as a practical method for determining the impacts of urban extent on natural dynamics at a large spatial scale, similar problems arise in the quantification or qualification of more local dynamics. The *Flood Estimation Handbook* in the United Kingdom (Institute of Hydrology 1999) adapted the CORINE Land Use maps into gridded formats that outline the portion of land in a given cell that is urban (URB_{EXT}) or suburban ($SUBURB_{EXT}$), enabling a more distributed classification of urban areas based on the density of a given land use within a cell (Bayliss 1999). The URB_{EXT} parameter is weighted by a factor of 0.5 to accommodate the role of gardens, parklands and grassland within the dense urban landscapes to produce the $SUBURB_{EXT}$ parameter. This enables a more distributed classification of the urban area and enables a more localised analysis of hydrological impacts at the city scale, providing scope to better determine dominant flood-generation processes (Miller *et al.* 2014). At much smaller scales, effective or directly connected impervious areas (EIA/DCIA, hereafter EIA) are used to differentiate impervious areas that are directly connected to urban streams via the presence of stormwater drainage networks that transfer water from the urban surface into adjacent channels (Roy and Shuster 2009). Whilst EIA arguably provides the highest-resolution measure of urban extent, with the spatial specificity to identify dominant runoff pathways, its derivation is often complex and data intensive and requires considerable effort, whilst presenting multiple challenges (Lee and Heaney 2003, Walsh *et al.* 2012). Derivation of EIA is further complicated by unknown runoff routing, a lack of available drainage data, and an increasing use of green infrastructure such as permeable paving, meaning field investigations are often necessary to complement GIS-based mapping analysis (Roy and Shuster 2009). Whilst this is feasible for sub-catchment or plot scale investigations, larger scale analyses at the city or catchment scale remain an arduous and often infeasible task, precluding its applications for larger spatial studies. As a result, the disparity in techniques for determining the urban area requires a pragmatic researcher to know what level of detail their particular study requires and an in-depth understanding of the available data and techniques for determination accordingly.

2.3 Urban soils

Human activity results in the compaction and sealing of natural soils (Duley 1939, Singer 2006, Scalenghe and Marsan 2009), mixing of materials and import of synthetic materials during the expansion of industry, commerce and residential land uses (Brown *et al.* 2009, Lorenz and Lal 2009). Green urban spaces facilitate many important functions in the urban environment including purification of urban air, carbon sequestration, social enhancement and ecosystem development. Such spaces are often assumed to behave "naturally", as the absence of the impervious sealing layer permits infiltration and recharge to occur. Often such spaces are constructed to enhance urban amenity (e.g. Central Park, New York and Olympic Forest Park, Beijing), provide leisure facilities and encourage ecosystem development. The development of urban parks often results in an artificial soil representative of imported material that is mixed and compacted during construction, as well as mechanically altered soils with disparate pore structures and organic content, rather than representing natural soils that occur over time as a result of geological processes (Solano 2013). Altered urban soils can preclude or retard natural processes such as infiltration and throughflow, resulting in increased ponding or surface runoff (Horton *et al.* 1994, Richard *et al.* 2001). Gregory *et al.* (2006) highlighted that compaction of soils from construction activities in northern Florida reduced infiltration rates from 70% to 99% in low-impact development (LID) areas. Conversely, green space that has evolved from private land (e.g. Hyde Park, London) is subject to development of surrounding areas, excavation for services and construction of paved areas, small buildings and associated infrastructure. In such instances, naturally occurring soils are impacted by compaction, mixing during excavation, removal of macropore structures and the addition of artificial content such as rocks and debris from adjacent construction, altering dynamics and hydraulic behaviour (Solano 2013). Furthermore, recreational green spaces are often underlain by artificial drainage structures to prevent saturation of the near-surface soil horizons and ponding of water. In either case, the hydrological behaviour of green space is usually markedly different from natural environments as the spatial and temporal dynamics of infiltration and resultant subsurface transfer are mere artefacts of anthropogenic modifications rather than undeveloped lands, as they are often considered.

3 Impact of urban areas on the urban water cycle

Over the past few decades, the field of hydrology has advanced to better understand some of the impacts of urban development on natural hydrological processes. Despite this, the impact of the built environment on natural hydrological dynamics is complex and our collective understanding remains limited (Niemczynowicz 1999, Fletcher *et al.* 2013). The urban water cycle is often differentiated from the "natural" hydrological cycle on simple geographical boundaries. The presence of engineered water systems, which include the import and export of water via piped networks and artificial routing of water into subsurface drainage networks have traditionally resulted in a separation of the two cycles. However, the realisation of interactions via inefficient infrastructure has resulted in a revisionist approach, increasingly treating urban hydrology as an integrative area of research, encompassing both natural and engineered water dynamics. Traditionally, assessing the impacts of urban characteristics on hydrological dynamics has occurred at the catchment scale, seeking to assess the wider impacts of substantial development on both quantity and quality dynamics of freshwater systems. However, there is an emerging recognition that small, local developments including individual buildings or neighbourhoods with contrasting materials, topography and infrastructure impact on the rate of transformation and flow pathways of water during its transition from atmosphere to the ground.

3.1 Urban-scale impacts on rainfall

Efforts to understand the dynamic relationship between the hydrosphere and landscape intrinsically begin with the input of precipitation. Niemczynowicz (1999) identified the study of precipitation as a "weak point" of urban hydrology, as the urban environment has a demonstrable impact on rainfall dynamics and efforts to understand urban rainfall remain an active field of study (Huff and Changnon 1972, Shepherd *et al.* 2002, Burian and Shepherd 2005, Ashley *et al.* 2012). The concentration of heat-absorbing materials, heat-generating processes and lack of cooling vegetation contribute to increased temperatures in urban areas (urban heat island (UHI) effect), impacting on rainfall proliferation in downwind areas (Oke 1982). This is further impacted by the presence of natural and anthropogenic aerosols, which contribute to thermal insulation and act as condensation nuclei for cloud-microphysical processes. These resultant changes to the surrounding atmosphere can have a profound impact on precipitation intensity and variability, not just within the locale of the city but also at a more regional scale, where atmospheric perturbations can result in changing precipitation dynamics downwind of urban areas compared to upwind observations (Burian and Shepherd 2005). Indeed, Shepherd *et al.* (2002) identified a 28% increase in warm-season, downwind precipitation around six cities in the southern United States, with a more modest increase in rainfall within the metropolitan areas (5.6%), highlighting the expansive influence beyond the local urban scale. Furthermore, a series of studies (Bornstein and Lin 2000, Shem and Shepherd 2009, Bentley *et al.* 2010, Ashley *et al.* 2012) identified the role of the UHI in the emergence of convective summer thunderstorms in Atlanta and a resultant increase in precipitation in downwind areas, again highlighting the scaling effects of micro-perturbations to regional-scale climate dynamics. In spite of a growing consensus, some studies continue to highlight our uncertainty of how urban areas impact rainfall dynamics. For example, despite identifying an average 8% increase in winter precipitation across cities in Europe, Trusilova *et al.* (2008) identified a 19% reduction in summer rainfall in urban and downwind areas, though disparity in this value is evident in contrasting geographical regions. Kaufmann *et al.* (2007) also identified a reduction in dry-season precipitation across the southern region of China, identifying an increase in aerosols contributing to atmospheric cooling and increase in condensation nuclei (Chen *et al.* 2006).

3.2 Local rainfall–runoff transformations

Increasingly, urban hydrologists and engineers are assessing the local responses of urban areas to precipitation, assessing the fate of rainfall at the building and street scale. The emergence of hygrothermal research has resulted in methodologies to assess how moisture and heat move through building surfaces. Wind-driven rain results in wetting of building facades where contrasting materials exert variable responses on the subsequent dynamics that occur (Blocken *et al.* 2013). For example, predominantly glass buildings create a smooth facade resulting in the rapid translation of water into runoff (Carmeliet *et al.* 2006). By contrast, buildings with predominantly brick or concrete compositions have porous spaces where water can seep into the building and be considered as a hydrological loss, particularly in older buildings with load-bearing and cavity walls. The impacts of these dynamics on the wider catchment water balance remain uncertain; however, localised pluvial flood risk can be exacerbated by buildings of particular material and inefficient supporting drainage infrastructure. Ragab *et al.* (2003) concluded that 30% of rainfall that lands on rooftops in the south of the UK is either intercepted or evaporated. There has been an increasing interest in modelling the volume of water that is translated into runoff from urban rooftops as rainwater harvesting seeks to translate rain into a sustainable resource, and volumetric understanding is crucial to designing harvesting storage tanks (Gash *et al.* 2008). During storm events, rooftops accentuate the rate of rainfall transformation, contributing to the acceleration of runoff-generating processes in urban environments (Shaw *et al.* 2010). The size, pitch, material and routing infrastructure on rooftops also impact on rainfall transformation processes. Pitched rooftops of impervious materials route rainfall into storm drains or storage vessels via gutter systems, resulting in a loss to the overall water balance. Rooftops experience similar losses to building facades as surface roughness can provide small storage spaces for rainwater to accumulate and remain until it is evaporated or transferred into porous spaces in the building structure (Blocken *et al.* 2013).

As building density increases and larger neighbourhood areas emerge, greater impervious surface area modifies the way rainfall is translated into runoff at the surface and near-surface levels (Miller *et al.* 2014). A recent study by Verbeiren *et al.* (2013) identified that a small increase in sealed surface area results in “considerably higher peak discharges”. This is particularly true in peri-urban catchments where EIAs collate to route runoff into subsurface drainage networks and ultimately into nearby stream channels. Increasingly, the implementation of sustainable urban drainage features in new housing developments seeks to break up impervious surfaces to mitigate against increasing runoff, though their efficiency is often hindered by poor maintenance regimes, where clogging and saturation results in them connecting flowpaths (Janke *et al.* 2011). The continued implementation of green infrastructure is needed to break up growing areas of EIA as urban density increases through population growth, though careful planning and management policies are required to upkeep efficiency, particularly in larger installations (Section 5).

3.3 Hydrological losses in the urban area

The presence of widespread impervious surfaces alters the dynamics of infiltration and results in contrasting impacts on baseflow behaviour at a range of scales (Walsh *et al.* 2005). Although some urban areas demonstrate a reduction in infiltration and recharge as a result of widespread soil sealing, some pervious areas within the urban landscape facilitate transfer of water from surface to subsurface. Small-scale development, including the sealing of private gardens to make way for driveways, reduces permeable spacing for water to percolate into. This is an increasing practice in developed countries, where greater vehicle security and reduced maintenance requirements of garden environments are seen as preferable by many home-owners (Warhurst *et al.* 2014). As this practice aggregates throughout neighbourhoods, overall infiltration and evapotranspiration reduces, resulting in a heightened urban flood risk, as Warhurst *et al.* (2014) demonstrated for the UK city of Southampton. This has resulted in local authorities in the UK providing information on the use of permeable paving when such planning applications are made (EA 2008). Similar thresholds are observed in developing countries, where Eshtawi *et al.* (2014) identified a 1% increase in urban area contributing to a 41% reduction in total infiltration in an experimental catchment in the Gaza Strip. The assumption that impervious surfaces result in zero infiltration was demonstrated to be incorrect, as Ragab *et al.* (2003) highlighted that nearly 10% of annual rainfall infiltrates into the road surface network for an experimental site in the south of the UK. This is further supported by Mansell and Rollet (2006), who explored the behaviour of water on contrasting paving surfaces, identifying markedly different infiltration and evaporation dynamics. For example, brickwork facilitates infiltration losses of 54% through the combined joints and pores. Contrastingly, asphalt and bitumen preclude any infiltration but facilitate high (44% and 64% respectively) evaporative losses. Implementation of sustainable urban drainage techniques, such as infiltration trenches, biofiltration swales, permeable paving and widespread plantation of trees and vegetation, can facilitate infiltration and recharge and these are being more widely implemented in new housing developments in peri-urban environments.

3.4 Surface runoff dynamics

The presence of urban landscapes significantly impacts on surface-runoff dynamics and runoff-generating process (Table 2). The conversion of landscapes from pervious to impervious surfaces has demonstrated increases in overall runoff volumes (Dunne and Leopold 1978, Arnold and Gibbons 1996); reduction in runoff lag time (Leopold 1968, ten Veldhuis and Olsen 2012, Konrad 2013); increasing flood return periods (Hirsch *et al.* 1990, Hollis 2010, Houston *et al.* 2011); and elevated peak discharges during storm events (Leopold 1968, Packman 1979, Konrad 2013). Increased flashiness of streamflow is also a commonly observed "symptom" of urban development and impervious surfaces, whereby rapid runoff generation transfers volumes into nearby stream systems via shortened flow pathways and without the need for saturation excess. Runoff is impacted by the nature of surface materials in the same way as infiltration dynamics. For example, brickwork converts 9% of received water into runoff, whereas concrete rapidly converts between 69% and 93% of water into runoff, depending on the inclination of the surface (Mansell and Rollet 2006). Rim *et al.* (2010) recently provided some experimental data that highlight a reduction in rainfall intensity required to generate runoff between cobble surfaces (a common characteristic of older UK towns) (>0.04 mm/min) and concrete surfaces (>0.02 mm/min). Whilst the local response of particular surfaces may dampen out at the larger scale, local pluvial flooding may occur in some instances where particular flat, impervious surfaces with low runoff ratios (such as bitumen) are present. In some instances, this requires engineering features being built into surface designs to create an artificial camber or gully that captures and routes surface water to a nearby drain. For example, construction of new road networks often includes gradients to route water into a particular pathway, creating the potential for alteration of catchment drainage pathways and subsequent catchment boundaries (DFID 2005). As discussed in Section 3.3, small-scale developments such as paving over garden areas can markedly alter local hydrological dynamics, including runoff. The impact of aggregation, where urban areas become increasingly dense, can drastically alter how rapidly urban surfaces are able to manage rainfall during pluvial events. In such instances, the issue of scale is particularly important as localised alterations to gradients and surfaces can impact the way water is detained

Table 2. Impact summary of urban areas on the natural water cycle compared to undeveloped catchments.

Variable	Urban impact	References
Quantitative impacts		
Rainfall	Increases downwind rainfall and enhances convective storms	Huff and Changnon (1972), Krajewski *et al.* (2010)
Infiltration	Reduction	Walsh *et al.* (2005), O'Driscoll *et al.* (2010)
Evapotranspiration	Reduction	O'Driscoll *et al.* (2010)
Overall discharge	Increases	Arnold and Gibbons (1996), Walsh *et al.* (2005)
Flood magnitude	Increases	Walsh *et al.* (2005), Hollis (2010)
Erosive flow frequency	Increases	Wolman (1967), Grimmond and Oke (1991), Konrad (2013)
Lag time to peak flow	Shorter lag to peak	Leopold (1968), Hood *et al.* (2007), ten Veldhuis and Olsen (2012)
Recession timing	Reduction	Dunne and Leopold (1978), Walsh *et al.* (2005)
Baseflow	Reduction	Klein (1979), Rose and Peters (2001), Kim *et al.* (2002), Hardison *et al.* (2009)
Qualitative impacts		
Channel width	Widening of channels	Walsh *et al.* (2005) and Hardison *et al.* (2009)
Stream depth (and pool development)	Both increase	Wolman (1967), Paul and Meyer (2001) and Walsh *et al.* (2005)
Macronutrients (N, P, K)	Increases	Paul and Meyer (2001)
Toxic contaminants:		
Heavy metals	Increases	Horowitz *et al.* (1999)
PAHs*	Increases	Garcia-Flores *et al.* (2013)
PCBs**	Increases	Yamamoto *et al.* (1997)
Pesticides	Increases	Brown *et al.* (2009)
Pharmaceuticals	Increases	Halling-Sorensen *et al.* (1998)
Debris loads	Increases	Walsh *et al.* (2005)
Temperature	Increases	Poole and Berman (2001)
Microbial contaminants	Increases	Gibson *et al.* (1998)
Aquatic ecosystems:		
Fish	Reduction in population	Yoder and Rankin (1996)
Macroinvertebrates	Tolerance dependent	Walsh *et al.* (2005)
Algae	Decreased species diversity	Chessman *et al.* (1999)
Macrophytes	Reduced diversity	Suren (2000)

* *polycyclic aromatic hydrocarbons, ** polychlorinated biphenyls*

and routed across the surface. Indeed, EIA is of critical importance in how urban areas translate rainfall to runoff, where high percentages of EIA contribute to rapid proliferation of stormwater runoff into adjacent channels, resulting in an elevated flood risk to urban areas (Miller *et al.* 2014). In the same study that assessed threshold impacts on infiltration, Eshtawi *et al.* (2014) identified that a 1% increase in urban area yielded up to a 100% increase in runoff.

Whilst the impact of impervious surfaces on runoff is relatively well understood, there remains a degree of uncertainty about the role of pervious areas within the urban environment. Modelling approaches to hydrological dynamics often treat pervious areas as "rural" landscapes. As discussed in Section 3, modifications to soils and underlying infrastructure can impact the way water behaves, and the "rural" representation of such areas can be a gross oversimplification. Furthermore, in pervious urban landscapes, hydrological connectivity to impervious areas is important in two senses: (1) runoff from impervious zones that passes over pervious land can rapidly increase the rate of saturation and result in quick attenuation of flows; and (2) saturation-derived flows that are adjacent to impervious areas have pathways of low resistance that can facilitate rapid transfer of large volumes of water. The role of these zones on overall water balance is uncertain and the subject of continued research, but such dynamics may have a significant impact on small-scale proliferation of stormwater and resultant local flood risk (Seo *et al.* 2013).

3.5 Subsurface flow dynamics

Price (2011) highlighted the complex scaling nature of the relationship between urbanisation and subsurface flow dynamics, identifying the combined role of the urban surface, presence of water management and drainage networks and wider catchment characteristics (e.g. geology, soil, vegetation, topography). Hardison *et al.* (2009) and O'Driscoll *et al.* (2010) identified reductions in baseflow when an increase in total impervious area is observed, whereas Lerner (2002) and Garcia-Fresca (2005) both highlighted an increase in recharge in urban areas. The presence of significant infrastructure beneath the urban surface can impact on subsurface dynamics, whereby exfiltration from both water supply and sewage infrastructure contribute to recharge of groundwater and reduce the self-cleansing capacity of groundwater aquifers (Jacobson 2011), whilst infiltration or inflow (I/I) into sewer networks can reduce water reaching groundwater zones. Heywood (1997) estimated that I/I can contribute 15% to 55% of total sewer flow. Lerner (1986) also demonstrated the significance of recharge in Lima (Peru) and Hong Kong, where leakage contributes 30% and 50% of total recharge, respectively. More recently, Ruban *et al.* (2007) observed a correlation between sewage-pipe baseflow and the water table for the city of Lyon. Indeed, rising water tables in urban areas are a common source of groundwater flooding to low-level properties such as basements and cellars, particularly in areas underlain by chalk bedrock or sand and gravel drift coverage (BGS 2010). Contributions to subsurface flow are dependent on two primary factors: (1) the spatial expanse of the urban infrastructure network; and (2) the age and integrity of the infrastructure, as older and increasingly worn infrastructure will be more likely to fail than newer systems that have been installed using more contemporary materials. In newer urban developments, where sustainable urban drainage installations are increasingly emerging, Newcomer *et al.* (2014) identified that SUDS (sustainable urban drainage systems) facilitate infiltration at an order of magnitude more than typical green spaces such as lawns and subsurface flows.

In many developing countries, subsurface aquifers are heavily utilised as sources of drinking water as a "cleaner" alternative to many surface-water bodies that are often impacted by contamination from intensive agricultural or industrial practices (Park *et al.* 2014). Over-pumping of groundwater is underpinned by three major issues: (i) an exponential increase in developing world populations; (ii) a finite available water resource; and (iii) a reduction in recharge owing to the widespread and rapid sealing of urban soils (Braadbaart and Braadbaart 1997). As a consequence of poor regulation and management of groundwater resources, many developing countries end up with particularly depleted groundwater tables, which has consequences for continued water resource utility; reduced self-cleansing capabilities and the emergence of geophysical hazards (Ozdemir 2015).

4 Qualitative impacts

There have been several extensive reviews that have sought to address the degradation of urban water quality and the physical, chemical and biological conditions of receiving waters (e.g. Paul and Meyer 2001, Meyer *et al.* 2005, Walsh *et al.* 2005, O'Driscoll *et al.* 2010, Fletcher *et al.* 2013). Despite the expanse and advance of research in this field, there remain many areas of uncertainty and considerable gaps in our knowledge, particularly as a result of emerging priority contaminants (Wenger *et al.* 2009). Understanding the spatial and temporal variation in urban water quality is an area that continues to fuel research, as the quest for sustainable ways to manage flow and water quality are sought (Mulliss *et al.* 1996). There remains a disparity in the primary drivers of urban water degradation in the developed and developing countries. Whilst point-source contaminants are increasingly well regulated in developed nations, the dumping of untreated wastewater into rivers and oceans remains a frequent practice, whereby an estimated 25% of urban residents in the developing world do not have access to adequate sanitation. By contrast, the developed world is primarily concerned with (i) diffuse contaminants from contrasting land surfaces and (ii) emerging priority contaminants (e.g. pharmaceuticals, nanoparticles and endocrine-disrupting chemicals) and how to effectively detect, trace and treat them (Walsh *et al.* 2005, Fletcher *et al.* 2013, Pal *et al.* 2014).

4.1 Physical/geomorphic impacts

The interconnected pervious and impervious surfaces contribute to multifaceted alterations to sediment budget and channel morphology. In his seminal 1967 study, Wolman identified an immediate increase in sediment yield during the development phase in urban environments as surfaces are stripped of their natural cover and their bare soils exposed, which is supported by more recent findings from Nelson and Booth (2002). As urban areas spread and mature, the supply of coarse sediment is gradually reduced as soils are sealed and impervious surfaces preclude interaction with the bare material below. As a result, sediment reaching urban channels tends towards finer composition, inclusive of suspended sediment washed in from adjacent urban surfaces (Duncan 1999). A comprehensive review by Taylor and Owens (2009) highlights the complexity of sediment dynamics in urban areas, where contrasting land uses impact on the volume, dimensions and nature of urban sediment (Franz *et al.* 2014). In conjunction with the loss of coarse sediments, impervious surfaces and the stripping of bank vegetation result in increased stream power and flashier stormwater runoff response, leading to an increase in erosive flows (Konrad 2013). Previous research has demonstrated that urbanisation often results in enlargement of urban channel cross-sectional area (Hession *et al.* 2003, O'Driscoll *et al.* 2009), incision of the stream channel and separation from the riparian zone (Groffman *et al.* 2003, Richardson *et al.* 2011) and lateral channel migration (Hession *et al.* 2003, Leopold *et al.* 2005, Wolfert and Maas 2007). In developed countries, engineering solutions are often applied to mediate the impacts of erosion in urban streams, resulting in the artificial lining of channels with concrete, rock or geomembrane materials (e.g. LLDPE, reinforced polyethylene and XR-5). Extreme examples of this are streams that are entirely culverted in concrete channels, flowing beneath urban areas and reducing the risk of surface flooding, often entirely destroying urban river ecosystems, though a recent move toward stream restoration ("daylighting", where streams are returned to their natural states) has sought to restore in-stream urban ecosystems (Broadhead *et al.* 2013). In developing countries, urban growth is often so rapid that the unregulated development of large, unplanned settlements can directly result in an increased risk from fluvial–geomorphological hazards (e.g. Akpan *et al.* 2015). For example, during intense rainfall events, the underlying soils and steep slopes often destabilise and can result in devastating landslides with significant loss of life and economic cost (Kometa and Akoh 2012, Akpan *et al.* 2015, Laribi *et al.* 2015). The depletion of groundwater via over-pumping can also trigger the emergence of large sinkholes by removing the supporting buoyancy of groundwater, resulting in the emergence of underground cavities beneath densely populated urban areas (Al-Kouri *et al.* 2013). There is therefore a pressing need for more pragmatic landscape planning and groundwater management in developing countries, owing to the high geomorphological risk associated with a lack of regulation in these areas.

4.2 Chemical/water-quality impacts

The contributions from both point-source and non-point-source pollutants from urban surfaces can greatly degrade the chemical water quality of urban streams and other receiving waters, often transporting "dirty" water over vast areas into downstream, estuarine and coastal environments. The increase in impervious surfaces expedites the mobilisation of contaminants through increased surface runoff and hydraulic efficiency. Additionally, the connectivity of surface and sub-surface flowpaths increases the rate and magnitude of transference into receiving waters (Pringle 2001, Tetzlaff *et al.* 2007, Jackson and Pringle 2010). A vast suite of contaminants stem from urban areas, including increased nutrient loadings (Garnier *et al.* 2012, Carey *et al.* 2013), volatile organic compounds (VOCs) (Lopes and Bender 1998, Mahbub *et al.* 2011), heavy metals (Sorme and Lagerkvist 2002, Pastor and Hernández 2012) and thermal pollution (Wang *et al.* 2008), where water temperature has become an important proxy for in-stream aquatic integrity (Chang and Psaris 2013). In developing countries, contaminants are often pumped directly into water courses from industry, agriculture and untreated domestic wastewater, where limited access to widespread sanitation and poor regulatory policies have contributed to significant degradation of aquatic bodies. By contrast, in the developed world, legislative powers from state, national and international authorities have addressed their role in impacting water quality and set out to control their fluxes and limit their inputs from both point and non-point sources (EUWFD 2001, USEPA 2001). However, in recent years, there has been increasing focus on "emerging priority pollutants" such as herbicides (Caux *et al.* 1998), microbial contaminants (Kay *et al.* 2007, Tetzlaff *et al.* 2010, McGrane *et al.* 2014); pharmaceuticals (Heberer *et al.* 2002) and polycyclic aromatic hydrocarbons (PAHs) from vehicular emissions (Van Metre *et al.* 2000). Microbial contaminants have been shown to increase with population, where concentrations are higher in urban areas (McGrane *et al.* 2014). Additionally, pharmaceutical concentrations and PAHs experience higher loads in urban areas but dominant sources and pathways remain uncertain. As increasingly complex pollutants have emerged (and continue to emerge from the increase in nanoparticles and antimicrobial resistant pathogens), efforts to sample and analyse concentrations become harder, relying on increasingly sophisticated techniques, software and model structures. The effort to predict pollutant concentrations in urban streams has been described as one of the greatest challenges for urban hydrologists in the past 20 years (Fletcher *et al.* 2013). Whilst a substantial volume of research has focused on the impacts of urban space on water quality, our understanding of the dominant sources, pathways and dynamics of pollutants remain limited and a priority area for continued research.

4.3 Ecological impacts

Determining the impacts of urban areas on in-stream ecological communities has been the focus of many excellent detailed reviews (for example, see Paul and Meyer 2001, Walsh *et al.* 2005, O'Driscoll *et al.* 2010). Aquatic ecosystems are impacted by the degradation of urban streams through geomorphological and chemical alterations to surface water bodies (Table 3). Most research outputs have reported a loss of assemblage diversity, richness and biotic integrity, where more sensitive species disappear and increasingly tolerant species become more abundant (Wenger *et al.* 2009). The Ohio EPA identified a threshold

Table 3. Potential impacts of engineering infrastructure on the urban water cycle (UWC).

Urban engineered feature	Impact on UWC	Reference
Quantitative Impacts		
Water supply network		
• Public water supply • Industrial water abstraction • Irrigation	Leaky infrastructure increases recharge. Supply pumped from external sources alters water balance equation.	Grimmond *et al.* (1986), Price (1986), Ashton and Hope (2001), Leung and Jiao (2006)
Sewage treatment		
• Sewer networks • CSOs • WWTPs	• Leakage, aquifer recharge • Reduce surface runoff • Inconsistent volumetric response	Reynolds and Barrett (2003), USEPA (1999)
Stormwater management		
• Surface drainage • Rainwater harvesting • Stormwater retention • Stormwater infiltration • Bioretention systems	• Reduce surface runoff • Reduce runoff and infiltration • Increase lag-to-peak and increased losses (ET/harvesting) • Increased infiltration/reduced runoff volume • Reduced runoff volume/velocity	Burian *et al.* (2000), Domenech and Sauri (2011) USEPA (1999) Mikkelsen *et al.* (1996), Siriwardene *et al.* (2007) and Denman *et al.* (2012) USEPA (1999), Hatt *et al.* (2009) and Hunt *et al.* (2008)
Qualitative impacts		
Water supply network		
• Public Water Supply • Industrial Water Abstraction • Irrigation	• No known impact • Reduction in flow/impact on aquatic ecosystems • No known impact	Nilsson and Renofalt (2008)
Sewage treatment		
• Sewer networks • CSOs • WWTPs	• Leakage contamination of groundwater resources • Overflows carry untreated domestic and industrial waste • Discharges can carry a suite of chemical and biological contaminants	Held *et al.* (2006), Wolf *et al.* (2012) Hall *et al.* (1998) and USEPA (2004) Gust *et al.* (2010) and Oberholster *et al.* (2013)
Stormwater management		
• Surface drainage • Rainwater harvesting • Stormwater retention • Stormwater infiltration • Bioretention systems	• Routes contaminants away from receiving waters • No known impact • Reduces sediment load and slows erosive flow velocity • Settles out contaminants and sediment, slows erosive flows • Removal of bacteria, contaminants and sediment	Semadeni-Davies *et al.* (2008) UESPA (1999), Koskiaho (2003) and Todeschini *et al.* (2013) Dechesne *et al.* (2004) and Hatt *et al.* (2009) Trowsdale and Simcock (2011) and Kim *et al.* (2012)

response of fish abundance to urban areas, where different proportions of urban land use have demonstrated a reduction of fish fauna populations (Yoder and Rankin 1996). However, there remain many gaps in our understanding of fish populations in urban streams, including the mechanisms that alter fish abundance, production rates, mobility or behavioural ecology in urban areas (Wenger *et al.* 2009). Although increased nutrient loads can result in an increase in algal biomass (e.g. Hatt *et al.* 2004, Walsh *et al.* 2005) algae also tend to a reduction in species diversity, often attributed to alterations in the water chemistry and changes to bed conditions that limit productivity and accumulation respectively. Macrophytes are less well studied in urban environments and remain an area of uncertainty, though Paul and Meyer (2001) highlighted results that demonstrate a reduction in diversity owing to changes in nutrient enrichment and changes in bed sediment (Suren 2000). Macroinvertebrate assemblages are the most widely studied area of urban water ecosystems, where a reduction in sensitive species and rise in tolerant species taxa is commonly identified (Walsh *et al.* 2005). The ecological impacts of urbanisation are profoundly felt across large spatial areas, from localised impacts within the urban environment (where streams are often culverted, artificially lined or re-directed) to downstream impacts into the estuarine and coastal waters where economically important shellfish populations are resident (McGrane *et al.* 2014).

5 Management of water in the urban environment

Historically, stormwater was viewed as a hazard in urban areas with complex networks of drainage infrastructure implemented to remove surface water and transport it away from the urban area. Additionally, open channels were culverted and sealed beneath the urban surface, limiting the risk from flooding but also reducing the potential for ecosystem development within urban streams. Increasingly, localised responses are being implemented across urban areas to manage stormwater at source and reduce the adverse impacts urban runoff can have on surrounding environments. A shift toward sustainable urban drainage has resulted in urban planning policies incorporating consideration of aqueous environments and ecosystem habitats increasingly being implemented at a range of spatial scales (Wong 2007). Whilst extensive networks of urban drainage systems remain a pragmatic component of managing urban water, increasingly small- and medium-scale SUDS strategies are being implemented by local and national authorities, enabling individual houses and businesses to capture water and use it for greywater applications. The application of SUDS has demonstrated success in North America, Europe and Australia but remains untested in developing countries. Instead, many developing countries rely on conventional stormwater drainage systems and, in some extreme instances, rely on gravity and open sewers with no proper sewerage systems in place, having a major impact on environmental quality and public health (Parkinson *et al.* 2007).

5.1 Local management of pluvial flood risk

Large-scale conventional stormwater drainage systems are constrained by a design capacity, where pluvial events that exceed these design thresholds result in inundation of drainage networks. Upgrading of large infrastructure is both expensive and disruptive, requiring large-scale excavation of surface areas including main road networks, so implementation of increasingly sustainable methods that capture stormwater runoff are favoured (Houston *et al.* 2011). Individual buildings or new development complexes increasingly incorporate local stormwater management techniques to reduce the volumes of rainfall being converted into runoff during pluvial events. Rainwater harvesting systems are increasingly being adopted to provide a supplementary water supply to mains supplies (Domenech and Sauri 2011). Rainwater harvesting captures rainfall that has directly fallen onto a relevant surface where it is subsequently transferred to storage tanks or routed into drainage networks. Such systems reduce the localised impacts of pluvial events by removing water from the wider urban cycle. The most common mechanism of collecting rainfall is the establishment of "roof catchments", which collect rainfall into conventional gutters that is then piped into storage tanks near buildings (Singh *et al.* 2013). Increasingly, incentives are being developed at regional and national scales worldwide, with many new developments being equipped with rainwater harvesting technology (Herrmann and Schmida 1999, Domenech and Sauri 2011). Such systems not only provide a clean alternative in water-scarce areas (particularly in developing countries), but also translate water from being viewed as a risk into a resource. In addition, vegetated rooftops provide a multifunctional method of reducing the environmental impact of the built environment by reducing roof surface temperature, increasing urban biodiversity and retention of stormwater during pluvial events (Carter and Keeler 2007). They capture, retain and evapotranspire rainfall back into the atmosphere, thus reducing the volume that is converted into runoff. Effectiveness of vegetated roof structures is a product of antecedent conditions, temperature and moisture retention capacity of vegetation retention, and water reduction rates of 34% and 69% have been reported (Teemusk and Mander 2007, Simmons *et al.* 2008, Gregoire and Clausen 2011). Shuster *et al.* (2013) assessed the impacts of installing 174 rain barrels and 85 rain gardens at the individual property (or parcel) scale, noting that these added detention capacity at even small scales, impacting overall runoff peak and the rising limb dynamics.

Another strategy increasingly being applied to reduce stormwater runoff at the parcel scale is the planting of trees and vegetation boxes. The presence of trees in urban settings can aid in infiltration of rainfall, resulting in evapotranspiration losses as well as both throughfall and stemflow, facilitating the transfer of water into the root structures and soils (Denman *et al.* 2012). For example, Milwaukee, Wisconsin (−22%) and Austin, Texas (−28%) (MacDonald 1996) demonstrated success in sustainable reduction of stormwater, and a more widespread implementation of planting strategies has subsequently been adopted across other cities and states in the United States (e.g. Peper *et al.* 2008, Seitz and Escobedo 2008, San Francisco Planning Department 2010). Whilst trees serve as an effective method for allowing infiltration into the subsurface, there is a caveat that their expanding roots beneath the urban surface often result in damage to paving and subterranean infrastructure, often resulting in costly repairs (Mullaney *et al.* 2015).

5.2 Larger-scale stormwater retention and infiltration techniques

At the larger scale of housing developments and industrial parks, ground-based retention techniques (e.g. ponds, wetlands and bioretention systems) are commonly applied to reduce and treat stormwater runoff (Hirschman *et al.* 2008). Retention basins collect stormwater to prevent flooding and reduce downstream erosion, whilst removing loads of sediment and contaminants through sedimentation, flocculation, ionic exchange, agglomeration and biological uptake (Urbonas and Stahre 1993). In addition, they provide amenity for urban residences and promote biodiversity for both animal and plant populations, which will colonise such wetland areas (SEPA 2013). Efficiency of these structures is a product of the storage volume of the pond, the catchment area it serves and the hydraulic residence time (HRT), the last of which determines the effectiveness of treating stormwater quality (USEPA 1999). Retention ponds and wetlands require regular maintenance or build-up of sediment will significantly reduce the HRT, thus reducing the amount of water that can be retained during any given storm event. For example, Verstraeten and Poesen (1999) assessed the efficacy of retention ponds in Belgium during storm events, concluding that sediment deposition during storm events resulted in a considerable economic cost to maintain regular dredging schedules to ensure their continued utility and that, owing to rapid filling during storm events, they may not serve as best management practice (BMP) for stormwater management. Bioretention systems are the most commonly used stormwater treatment techniques in the United States and are increasingly being incorporated globally in developed and developing nations alike (e.g. Fujita 1997, Wong 2007, Davis *et al.* 2009, Trowsdale and Simcock 2011). Such systems combine grass buffer strips, sand filter beds, impoundment areas, an organic layer and biota (Davis *et al.* 2009). Bioretention ponds show contrasting results for removing load reductions of suspended sediments and heavy metals (Davis 2008, Li and Davis 2008, Hatt *et al.* 2009), though there are cases where they have demonstrated success in reducing peak flows ranging between 14% and 99% at the sub-catchment scale (Hunt *et al.* 2008, Hatt *et al.* 2009).

Infiltration systems are designed to collect stormwater from adjacent impervious areas and provide a pathway for water to infiltrate into the soil and subsurface areas, providing a natural recharge to groundwater systems (Butler and Parkinson 1997). These are constructed as excavations, which are lined with a specific media filter such as sand, gravel or crushed stones and are sometimes wrapped in a geotextile, and are increasingly being installed in new housing developments to alleviate runoff risk (Siriwardene *et al.* 2007). Such systems capture stormwater in detention and gradually filter it back into surrounding soils and deeper groundwater storages, purporting to restore hydrological behaviour to its pre-development state by breaking up large areas of EIA (Mikkelsen *et al*. 1996). Another method increasingly being applied to break up EIA is the installation of permeable paving surfaces that enable rainwater to infiltrate through the surface, into the soil structure below. Brattebo and Booth (2003) highlight the success of permeable pavement in reducing surface runoff when 121 mm of rainfall during a single storm resulted in only 4 mm (3%) of runoff. The same study observed a significant reduction in both copper and zinc concentrations in the water that permeated through the pavement filters and also a marked reduction in the presence of oil from motors, highlighting their utility in improving water quality. The increasing popularity of permeable paving in car parks, roads and pavements has resulted in an increasing distribution across the United Kingdom (Newman *et al*. 2013). As many urban extent parameters such as TIA, EIA and URB_{EXT} are derived using remote-sensing imagery, permeable paving presents a new problem for land-use classification, as it is indiscernible from normal paving from aerial photographs. As such, modelling-based approaches to determine the hydrological response of urban landscapes has an added emphasis on site visits, as the treatment of permeable paving as "impervious" would result in a gross over-estimation of stormwater runoff from the plot to the sub-catchment scale, inhibiting BMP for stormwater runoff (Jacobson 2011).

6 Remaining challenges

Despite our many advances, there remains a great degree of uncertainty surrounding the urban water cycle and the impact of urban development on natural hydrological processes, water quality and ecosystem processes. New monitoring technologies and modelling strategies have advanced the way we capture, analyse and model data in the urban environment, but scaling our detailed understanding of hydrological processes at the plot or sub-catchment scale to the wider catchment scale remains exceedingly difficult. Some critical areas of study remain pertinent to advance our understanding of the urban water cycle, which may be summarised as follows:

- Quantifying the impact of urban areas on climate dynamics is crucial for the prediction of precipitation forecasting at increasingly short temporal scale. This requires continued collaborative research between meteorologists and hydrologists and is crucial for understanding the role of the UHI in contrasting climatic environments (Arnfield 2003).
- An empirical method to derive evapotranspiration rates as a result of urbanisation is needed to help in fully determining the urban water balance. Evapotranspiration is increasingly being introduced as a sustainable technique for managing stormwater runoff and quantifying fluxes of ET in urban areas is crucial for design of such systems.
- Rates of infiltration across urban areas remain poorly determined and work has highlighted that many existing assumptions are invalid. A research priority is therefore to determine infiltration rates across impervious and pervious urban areas to quantify losses from the overall urban water cycle.
- Contributions from the engineered water cycle remain poorly determined, though estimations from field studies have suggested up to 40% of water from leaky infrastructure could enhance urban recharge. Novel technologies that can be deployed along pipe systems can provide new insight for monitoring leakage and infiltration, though these lack widespread testing in urban areas. Wider deployment is required to elucidate the contributions from city networks to the natural water cycle and also help local authorities and utility companies reduce volumes lost from the systems.
- The complex patchwork of impervious and pervious areas and their interconnected dynamics results in a complex response to the rainfall–runoff transformation that remains poorly understood. Whilst the hydraulic runoff dynamics from pervious areas are relatively predictable, the dynamics of pervious space and the role of EIA remain undetermined, and a pressing area of further research.
- Urban surfaces are sources for a suite of contaminants that are harmful to humans and aquatic ecosystems alike and understanding the spatial and temporal patterns of contaminant fluxes into urban rivers remains an important task. In particular, tracing the sources of contaminants remains a primary requirement for the management of stormwater pollutant loads, as this will provide a mechanism for differentiating the types of pollutants that are associated with particular urban land uses.
- As more sustainable methods for implementing urban drainage are sought, new developments are increasingly incorporating sustainable drainage systems to both reduce and treat stormwater. The extent to which this is successful remains uncertain and further work is required to assess the contrasting scales at which these have a demonstrable effect on the overall urban water cycle.
- The effects of climate change on urban hydrology will result in changes to the magnitude and frequency of rainfall events. Therefore, there is a pressing need to understand these likely changes and the scales at which they will occur. Furthermore, there is a need to understand the impacts that such events will have on stormwater infrastructure, flood risk and water quality. In addition, as populations continue to grow within urban boundaries, the impacts of climate change on water resources also remain uncertain,

where increasingly sustainable management of rainfall and stormwater runoff present an encouraging yield of renewable water.

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