School of Geography and the Environment

University of Oxford

Green and Grey Drainage Infrastructure: Costs and Benefits of Reducing Surface Water Flood Risk

Name: Cherona Marie Chapman

College: St Edmund Hall

Degree: DPhil Geography and the Environment

Supervisor: Professor Jim W. Hall

Word Count: 40,323

Abstract

It is now estimated that in the UK alone 3.2 million properties are at risk of surface water flooding – an increase of almost half a million from ten years ago – and it is expected that this problem will increase further under current climatic changes and urbanisation. Sustainable Drainage Systems (SuDS) seek to reduce flooding from surface water without relying on conventional piped sewer networks by restoring the pre-development hydrological conditions of an area through mimicking natural drainage processes. As their behaviour is more complex compared to their traditional, greyer counterparts, there is still incomplete understanding of their performance during intense rainfall. Research to-date has focused on the optimisation of their design at an infrastructure-scale for achieving hydrological benefits, and a growing number of case studies into their inclusion in small, neighbourhood developments. However, an understanding of the influence of external factors on SuDS behaviours and the additional range of co-benefits SuDS may provide are also important for the design of effective systems, whilst an appreciation of their potential role at greater scales will allow a more informed consideration of drainage alternatives in larger-scale developments. Thus, this thesis investigated how built form influences SuDS' performance and how the inclusion of SuDS in regional-scale developments may contribute to wider environmental goals.

To analyse the effect of urban built form, a range of 1 hectare urban tiles were developed to represent different housing typologies, urban densities and SuDS implementations under current design principles drawn from *The SuDS* Manual (CIRIA 2015). The rainfall-runoff model Stormwater Management Model (SWMM) was used to simulate storm events of varying magnitudes and the

resultant hydrographs analysed. These tiles were then applied to a proposed regional development spanning five counties in south-east England, the Oxford-Cambridge Arc, under eight different scenarios of urban development, and the length of pipes required to connect such developments estimated. Finally, a methodology was developed to further assess these regional-scale urban development designs for their potential contributions to green infrastructure (GI) networks. The designs were assessed against four goals for GI provision: 1. Ecosystem Services; 2. Ecological Status; 3. Ecological Connectivity; 4. Proximity to the Population. Each of these goals was assessed using existing approaches which utilised readily available datasets to allow for widespread application of the methodology.

It was found that the differences in impermeable surface areas as a result of different built form designs influenced peak and total runoff volumes from a storm event, both with and without the inclusion of SuDS, although to what extent was dependent upon the SuDS infrastructure(s) employed and their overall implementation. Notably, in some urban designs, a lower proportional implementation of a SuDS infrastructure at a higher development density saw greater reductions in peak and total runoff volumes than a higher proportional implementation at a lower development density. These proportions were of available surface type for SuDS (e.g. roof area for green roofs). More dense urban configurations provide greater potential surface area for their construction. The spatial arrangement of these built form elements, however, also proved an important consideration due to the spatial variation of external landscape characteristics (such as soil type and slope) which also impact runoff dynamics.

The spatial arrangement of these built form elements, however, also proved an important consideration due to the spatial variation of external landscape characteristics (such as soil type and slope) which also impact runoff dynamics. In such a way, the developed approach proves particularly useful, as by combining a tile approach for designing urban developments with rainfall-runoff modelling, the methodology allowed for these landscape and built form elements to be readily varied and scrutinised at both local and regional scales.

Investigation of pipe requirements found that for all housing typologies the use of SuDS could reduce the minimum required pipe diameter, although not consistently for all SuDS designs. Different spatial development approaches also resulted in different required pipe network lengths. Given that current guidelines permit high cost as a justification for not constructing SuDS in developments, such findings suggest that financial savings could be found elsewhere with a well-designed SuDS system.

When considering co-benefit provision, the inclusion of SuDS consistently saw greater GI provision scores, although it is worth noting that urban spaces presented opportunities for GI provision even without. When considering individual GI elements, this is particularly clear. For ecosystem services, very few SuDS designs were able to score higher than the pre-developed state, and these occurred only where existing land cover was poorly-scoring.

Once again, the specific SuDS infrastructure(s) employed played a strong role in determining which co-benefits were provided, and to what extent. By their nature, infrastructure-based SUDS, for example, require less free space in a development, and as such can help minimise loss of undeveloped land in an urban area (or provide more room for compact development to help reduce overall sprawl). Whilst this indicates that SuDS choice is an important component in achieving the specific aims of a development project, interactions between different SuDS infrastructures and/or elements of a development design highlights the need for trade-offs to be understood and adequately balanced in resultant designs.

From this research, four key considerations for urban planners arise when designing new development involving SuDS infrastructure. First, the location and layout of the development; second, the choice of housing typology; third, the comparison of multiple SuDS infrastructure and combinations; fourth, the opportunities posed by urban space in providing GI (even without SuDS).

Table of Contents

List of Figures

List of Tables

1. Introduction

It is estimated that approximately 3.3 million properties in the UK (62% of all properties at risk of flooding) are at risk of surface water flooding (Department for Environment, Food & Rural Affairs, 2021) – a figure which has risen by half a million since the National Flood Risk Assessment of just over ten years ago (Environment Agency, 2009). Surface water flooding occurs when infiltration rates and/or capacities in an area are exceeded, leading to the ponding of water on the land surface (Priest et al., 2011).

Flooding is a natural part of the water cycle and occurs in both natural and artificial landscapes when the rates of water inflow (from processes such as overland flow or precipitation) are greater than those of outflow (through processes such as evapotranspiration or infiltration), which generates an accumulation of water in that area (Ashley et al. 2020). The greatest risk from these events occurs in urban environments as these possess the larger population densities, and risk is characterised by population exposure (Kaźmierczak & Cavan 2011). As larger proportions of their surface area are impermeable, leading to greater runoff volumes, these spaces also experience surface water flooding in lower intensity rainfall events which occur on a more frequent basis. Bevan (2018) also argues that many of those in these populations are unaware of the risks of surface water flooding, particularly compared to those in riverside or coastal communities who may logically expect a risk of flooding, and are thus also poorly equipped to respond to a flood event.

As well as the greater risk, urban spaces also suffer the greater socioeconomic cost from a flood of a given magnitude as they have higher densities of properties, businesses and critical infrastructure (such as electricity substations and transport hubs) (Guo et al. 2021). Flood impacts can therefore long outlast the duration of flood inundation and have further indirect consequences, such as disrupted supply chains and loss of productivity (Bernet et al. 2017). It is therefore important to manage and mitigate the risk and impacts of these flood events and the risk they pose, both for local and indirectly affected communities.

Furthermore, challenges around flood risk management and forecasting, particularly in respect to surface water flooding, are expected to increase in the future. Largely, this is due to climatic changes which are expected to result in more frequent and more intense rainfall events in the UK (Kreibich et al., 2015; Yazdanfar & Sharma 2015). Additional catchment changes, however, such as from urban sprawl or land use change, will also impact surface water flood risk (Xu et al. 2020). It is important to note that these projected impacts are uncertain, due to limitations and uncertainties in both climate projections and hydrological models, but there is a wide consensus that pressures on drainage infrastructure will increase even without wide-scale changes in population distribution and densities (Arnbjerg-Nielsen et al., 2013).

This requirement for increased drainage capacity brings into question our current approach to managing urban drainage (see section 1.3), as system capacities become insufficient with time. Instead, we need to learn to incorporate and work with water in the urban system, rather than attempting to remove increasing volumes for management elsewhere (Lashford et al. 2019). The UK is not alone in facing this problem, either – for example, in China it is urban surface water flooding that is the main causal factor for risk (Yin et al. 2015) and in

Canada, these events are considered the most economically costly natural disaster (Oubennaceur et al. 2019).

Unlike fluvial or coastal flooding, one of the greatest challenges we face at present in regards to surface water flooding is the modelling of such phenomena for forecasting tools. This is because urban areas often have a less natural topography as a result of urban development. Additionally, there are constructed subsurface drainage networks whose constituent infrastructure create artificial drainage pathways, and capture flows from areas that may not correspond to those pre-development (Guo et al. 2021). This challenge is often further exacerbated through the difficulty in accessing elements for data collection, as often land, pipes or other built elements are privately owned and operated (Noh et al. 2016). Thus, the hydrological and hydromorphological processes in the urban environment present interactions which are less well understood and more problematic to model. Nonetheless, development in recent years has seen models of increasing quality and complexity introduced into regular practice within the field (Song et al. 2014).

1.1: Hydrology & Surface Water Flooding

Surface water flood events typically arise from pluvial flooding (where rainfall volumes and/or intensities are responsible for infiltration exceedance) and occur on a relatively local scale, which adds to their forecasting difficulty (Kaźmierczak & Cavan 2011). However, Parker, Priest and McCarthy (2011) highlight that other mechanisms can contribute to their occurrence, including sewer or groundwater flooding, and thus the two terms cannot be considered synonymous.

Nevertheless, they also acknowledge the interconnected nature of the two phenomena which makes distinct separation of the two difficult.

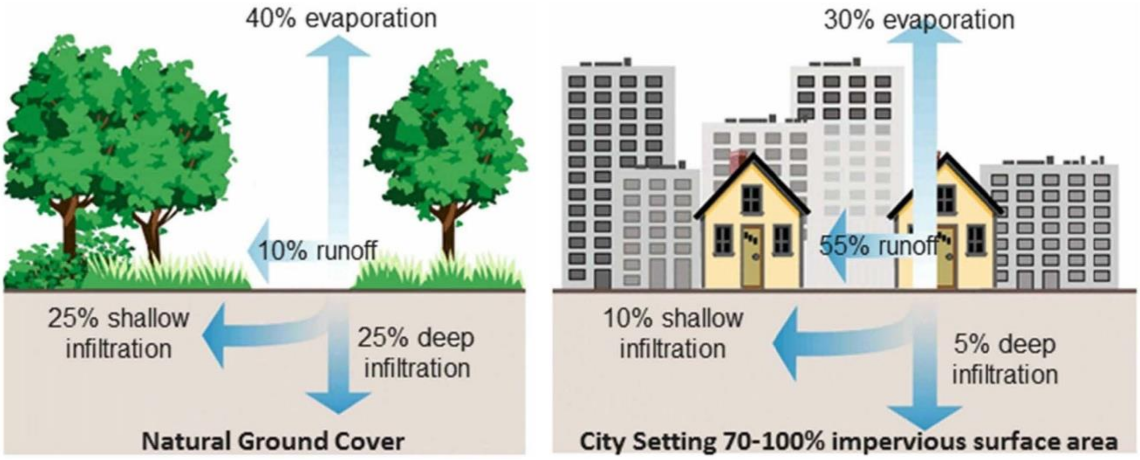

Figure 1.1: A comparison of drainage processes in rural and urban landscapes (US Department of Agriculture, 2014)

A comparison of the different hydrological processes and their dominance in the urban and rural settings can be seen in Figure 1.1. Both urban and rural settings are subject to precipitation events, yet differences in their surface compositions generate different hydrological responses. With a greater proportion of impermeable and sealed surfaces, there is considerably less infiltration in urban environments, leading to greater volumes of water being present on the surface (Leigh & Lee 2019). Where topology and/or artificial drainage systems allow, this water is transported as runoff – either surface or subsurface. However, when these systems are overwhelmed, working inefficiently due to poor maintenance, or when surface runoff rates are lower than precipitation rates, this water collects, creating areas of inundation (Carter et al. 2015).

This is not a rare or new problem – it is, in fact, quite common to see large puddles on pavements and roads during, and following, rainfall events. Whilst often these can be seen as nothing more than a minor inconvenience, all it takes is a particularly heavy rainfall event or several successive events in a short time period to exacerbate conditions, flooding basements and disrupting subsurface infrastructure – occurrences we are likely to see more of with predicted climatic changes (Yazdanfar & Sharma 2015). In addition, increasing urbanisation required to accommodate current population dynamics will lead to this problem becoming even more widespread. It is thus imperative that our cities have sufficient capacity to cope with intense rainfall, whether through methods that enhance the processes of infiltration, evaporation, safe runoff or a combination (Lashford et al. 2019).

1.2: Urban Drainage History

An urban drainage network is defined as a means of water conveyance and collection responsible for transporting runoff from an urban environment (Yazdanfar & Sharma 2015). In the UK, this water transfer became common following the industrial revolution as a form of urban flood management, and usually saw the urban runoff conveyed to a nearby watercourse (Lashford et al. 2019). The enclosing of these systems, however, was not widespread until London's sanitary revolution in the 19th century, as the connections between poor drainage and poor health were better understood (Ashley et al. 2015).

The traditional form of an urban drainage network, both in the UK and other developed nations, uses a centralised network of pipes into which the runoff is drained, which then convey the water to 'safe' outlets (Ashley et al. 2020). Combined systems also utilise these pipes for sewage and wastewater disposal, with the outlets being treatment plants (except during exceedance events when combined sewage overflows lead to serious pollution events). In many locations, however, these have seen little modification since their first construction, despite increased pressures from urbanisation and climate change. London's network, for example, which was originally built in the 19th century, was designed to overflow into the Thames (on average) 4 times each year, but now does so more than 50 times a year (Dolowitz et al. 2018). In response, the Thames Tideway Tunnel (an infrastructure megaproject) is now under construction, which aims to intercept the overflow from 34 outlets and store and transport it to a sewage treatment works, before this then clean water is released into the river (Tideway 2021).

Sustainable urban drainage systems (SuDS) first began to emerge as formal elements of drainage infrastructure in the 1970s, when the impacts of these combined sewer overflows (in London and the US) became a focal point of environmental campaigning (Dolowitz et al. 2018). Half a century later, they are still not particularly widespread when compared to traditional piped drainage, although there has been a steady increase in their uptake globally, particularly as modelling, optimisation and design guidance tools are improved (Eckart et al. 2017).

We face challenges in our design approaches to both piped drainage and SuDS. Often, the design of an urban drainage network occurs once the plans for the development's primary purpose (e.g. housing or retail) has been produced, and is thus already constrained by other infrastructure (Ashley et al. 2020). Also, the underground nature of many elements in traditional drainage can lead to poor asset management due to their hidden, 'out of sight, out of mind' nature (Ashley et al. 2020).

1.3: Sustainable Urban Drainage Systems (SuDS)

SuDS are infrastructure used in drainage that seek to replicate natural drainage processes, restoring the hydrological conditions of the area as close to those pre-development as possible (Anim et al. 2019). They can broadly be divided into two types, based upon their operational principles, with some offering increased permeability to promote infiltration (such as porous pavements), whilst others offer increased storage potentials for surface runoff (such as detention basins) (Liao, Deng & Tan 2017). SuDS can also be constructed upon other infrastructure, such as with green roofs, and can thus have an important role in dense urban environments where there is little undeveloped land remaining.

A key advantage of SuDS over traditional drainage infrastructure is the co-benefits they can provide beyond surface water flood alleviation (Ellis & Lundy 2016). SuDS can also be beneficial for water supply purposes, with infiltration and slow subsurface flow allowing increased recharge of groundwater, as well as water quality improvements (Drake, Bradford & Marsalek 2013). Beyond hydrology, these co-benefits can include the provision of recreational spaces, biodiversity improvements, and mental health benefits among others. In this way, the inclusion of SuDS in developments can also help meet additional environmental targets, such as increased greenspace – an idea which is expanded upon in section 1.4. They also add increased resilience to the system as they are not designed for a single purpose (Leigh & Lee 2019).

In this way, SuDS (a type of green infrastructure) are often considered the more progressive and sustainable option when compared to traditional, piped systems (grey infrastructure) (Dolowitz et al. 2018). It is worth noting, however, that each location has its own opportunities, limitations and hydrological dynamics, and so the most suitable and sustainable drainage solution is a place-specific concept. In some instances, this may mean grey drainage infrastructure is a better approach than green (Ashley et al. 2020).

Four types of SuDS infrastructure are specifically addressed and modelled within the thesis – bioretention, detention basins, green roofs and permeable surfaces. The following sub-sections offer a brief introduction to the current design and operation principles of each.

1.3.1: Bioretention

Bioretention zones (also commonly referred to as rain gardens) are dips in the topography, usually artificially created, lined with vegetation to collect, store and infiltrate stormwater runoff. As a depressed topographical point, runoff is preferably channelled to the bioretention area, allowing storage and infiltration through the permeable surface, decreasing the effective impervious area of the urban landscape (Carter et al., 2015). Careful design of the multi-layered system often sees them utilised for water quality improvements, but there has been a recent growing acknowledgement of their considerable water quantity reduction potentials, too (Yang & Chui 2018).

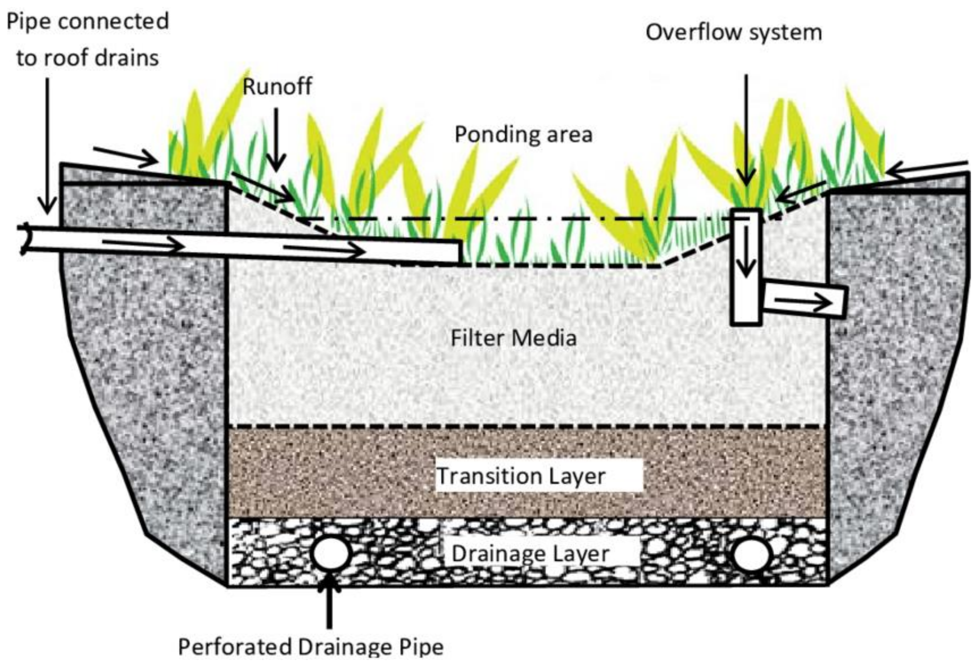

Figure 1.2: Schematic of the cross-section of a typical bioretention system (Rahman et al. 2016)

As illustrated in Figure 1.2, a standard bioretention system consists of four separate layers – one surface layer (vegetation), and three subsurface layers (filter, transitional and drainage). As with many other SuDS, the presence of the vegetation promotes interception and evapotranspiration from the system whilst also reducing the velocity of surface water (Tahvonen 2018).

The filter layer is designed to promote subsurface infiltration, slow percolation and provide water quality improvements (Osman et al. 2019). Traditionally, natural soils with high permeability, such as loam or loamy sand, were used, and whilst these are still common in many developments, the high clay content of some loams can lead to system failure, and thus there has been a steady move towards soil/media mixes (Davis et al. 2009). The added presence of organic matter has also proved beneficial for moisture retention and supporting vegetative growth (Osman et al. 2019).

The drainage layer is separated from the filter layer by the transition layer, typically made of a coarse sand. This helps prevent material from the filter layer being lost through drainage, as well as offering additional water retention benefits (Osman et al. 2019). The drainage layer encourages the percolation of water into underlying soils, an underdrain, storage facility, or other constructed drainage system, extending the length of time for which the system can operate before saturation occurs (Tahvonen 2018). As such, coarser soils or media are used, such as gravel or coarse sand (Roy-Poirier et al. 2010).

1.3.2: Detention Basins

Similar to bioretention areas, detention basins are areas of low topology (usually constructed), which encourage the flow of surface water into them (Travis & Mays 2008). However, in contrast to bioretention areas, they lack filtration layers (CIRIA 2015). They typically also include an outlet, which can be used to drain the feature post-rainfall or to prevent exceedance events (see Figure 1.3). This makes them useful in urban locations, where basin overflow could exacerbate flooding in the immediate vicinity (Jovanovic 2007). Once collected, water percolates into the surrounding soil through the sides and base of the basin, and also evaporates. During non-rainfall events, detention basins are designed to not keep a permanent store of water – this is what makes them distinct from retention basins. To achieve this, the basin is constructed such that its base lies at a higher level than the natural water table, with an outlet located near to the basin floor (Fang et al. 2017).

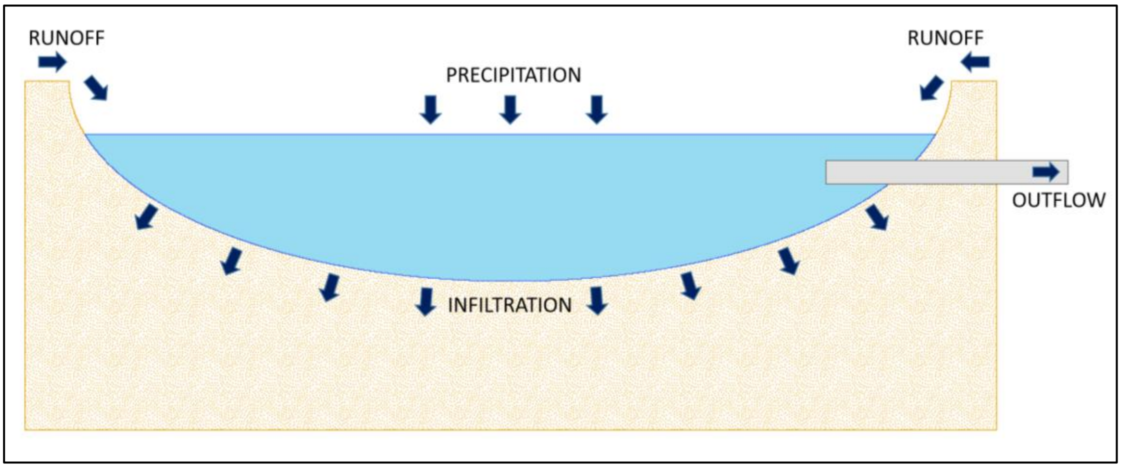

Figure 1.3: A schematic drawing of the operation of a detention basin

1.3.3: Green Roofs

A roof, intentionally covered by vegetation, a growing medium and a structural membrane (preventing damage to the building), is referred to as a green roof (Haowen et al. 2020). These features are not a new innovation, particularly in Scandinavia, although designs have been optimised to improve their urban drainage potential – traditionally, for example, these used a thick layer of soil to minimise potential damage to the building, whereas contemporary designs use a protective membrane for this purpose (Dietz, 2007). By providing a greater vegetated surface area within the city, these surfaces increase interception of rainfall, reducing runoff volumes, and offer a permeable area that reduces the speed through which water reaches the sewer network (Mentens et al., 2006).

Figure 1.4: A schematic of the layers in an example green roof (Brachet et al. 2019)

Green roofs are designed with multiple layers that each have a distinct role to play – vegetation, retention, transition, drainage, and protection (see Figure 1.4). The uppermost – vegetation – mimics the ground surface of an undeveloped area, reducing surface water velocities, and promoting both evapotranspiration and infiltration (Mentens et al. 2006). The substrate below is responsible for supporting this plant life whilst also encouraging infiltration and water retention. It is therefore not uncommon for the substrate to be formed of multiple layered soil types, encouraging infiltration and percolation in the uppermost, and promoting water retention further down (Savi et al. 2013).

As with bioretention, a transitional layer divides this from the drainage layer to reduce erosion. A high density polyethylene (HDPE) membrane is often used for drainage, storing water in cups whilst allowing drainage through small perforations, from where it is transported from the roof using pipes (Green Roof Guide, 2021). This layer is underlain by an additional layer to reduce damage to the building, which is typically composed of a roof barrier and waterproof material (Brachet et al. 2019).

There are two main types of green roof, characterised by the height and variety of vegetation – intensive and extensive. Extensive roofs are the most common due to their lower relative maintenance and roof load requirements. High quality extensive green roofs have a biodiverse mix of sedums and wildflowers and can be resistant to drought, but cheap sedum mats offer little plant diversity and are less resilient to droughts (Li & Yeung 2014). This is heavily dependent on their design, however, as discussed in the Green Roof Organisation Code (2021). Conversely, intensive roofs have a wider vegetative diversity, including shrubs and bushes, but require a greater substrate depth to adequately support such growth, and without efficient hydrologic operation can cause stress to the roof supports (Tabatabaee et al. 2019). This additional substrate depth means they have greater drought resilience. They also typically require greater maintenance, including watering of the vegetation (Paithankar & Taji 2020).

1.3.4: Permeable Paving

Permeable (or porous) pavements aim to increase the permeability of the urban landscape by providing a surface through which water can flow or filter, and in some cases offer a potential for storage. Although referred to as ‘pavements’, these surfaces have been applied to many other locations within the urban environment, including roads, driveways and parking lots (although differing functions of these spaces has led to modifications in design to improve the lifecycle of the paving) (Ahiablame & Shakya 2016). Replacing traditional impermeable surfaces, this infrastructure reduces the volumes of surface runoff during a rainfall event, reducing the peak discharge, and increasing the time for this water to reach the drainage system (Carter et al. 2015).

Current permeable pavement designs can be categorised as continuous surfaces (using pervious asphalt or pervious concrete) or grids (using concrete or plastic) (Ball & Rankin 2010). Figure 1.5 illustrates a schematic for both of these systems, and it can be seen that in the subsurface, they utilise the same operational principles. Firstly, the surface layer, constructed of materials such as pervious asphalt or interlocking paving slabs, creates small discontinuations in the surface to allow water to infiltrate into the underlying layers (Mullaney & Lucke 2013). Beneath is a layer, or series of layers, of coarse soil or sediment, which then overlay a storage facility (Ball & Rankin 2010). Water is then drained from the system, both through percolation into underlying soils (where possible and appropriate) and through a piped underdrain (Mullaney & Lucke 2013).

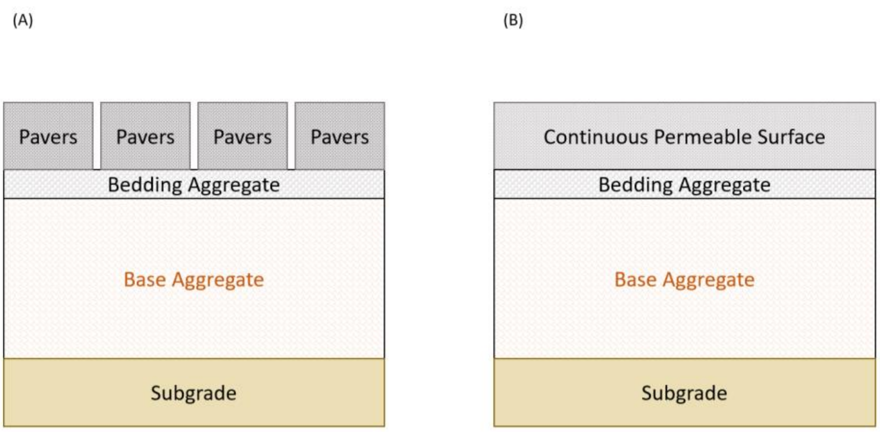

Figure 1.5: A schematic drawing of the two dominant permeable paving designs – grids (A) and continuous surface (B) (adapted from Mullaney & Lucke 2013)

1.3.5: SuDS Trains

A development is not constrained to the use of just one SuDS type or infrastructure, and it is in fact current best practice for multiple SuDS to be used in series, creating an inter-connected, multi-SuDS system referred to as a SuDS train (CIRIA 2015). Not only does this allow for a greater storage capacity and/or increased permeable surface area (reducing runoff at the source), but also a more diverse range of co-benefits to be introduced to the system (Huang et al. 2020). Channelling runoff between elements can also increase the extent of a given benefit, too – water quality may be improved by a SuDS infrastructure, for example, but as this is passed through several successive SuDS infrastructures, the quality can progressively increase further (such that in some cases it may be high enough for discharge into a local, natural watercourse) (Maqbool & Wood 2022).

1.4: Environmental Policy

SuDS can also work in conjunction with other tools and infrastructure to achieve wider environmental goals. They are an example of nature-based solutions (NBS) – actions and management within the natural environment and ecosystems which address challenges whilst providing benefits for humans and biodiversity (Lafortezza et al. 2018) – and can provide many ecosystem services – benefits from the environment which support and enhance human life (Dolowitz et al. 2018). In such a way, they can contribute to wider environmental networks, such as Nature Recovery Networks (NRNs) and Green Infrastructure (GI) corridors (Rodríguez-Espinosa et al. 2020).

Movements to increase ecosystem service provision and nature connectivity are growing worldwide, and many targets for this are being formalised in local, national and international policies. The sub-sections below outline a few key policies of particular relevance to this thesis and its findings.

1.4.1: Natura 2000

The Natura 2000 network is a comprehensive area of sites across Europe, forming the largest global coordinated system of protected areas (Algador et al. 2012). The composite sites were selected under a range of scientific criteria to help preserve examples of Europe's varied biogeographic regions, protect as great a range of its endangered species as possible, and cover both land and marine sites (Gurrutxaga et al. 2010). They are protected by legislation and conservation targets set out by the European Commission, making them distinct from national parks or nature reserves, but are still open to the public for non-intrusive activities such as hiking (Algador et al. 2012). As a global example of ecological preservation and a project for increasing nature connectivity, Natura 2000 has been the focus of many studies outlining tools and methodologies for enhancing and assessing other natural spaces and their connectivity.

1.4.2: The 25-Year Environment Plan

The 25 Year Environment Plan is England's national plan, focusing on maintaining and improving environmental health, both of land and marine spaces. Announced in 2018, the plan outlines ten main goals for the next 25 years (Department for Food, Environment and Rural Affairs 2018):

1. Achieve clean air
2. Achieve clean water
3. Ensure plants and wildlife thrive
4. Reduce risk from environmental hazards
5. Ensure sustainable use of natural resources
6. Enhance engagement with the natural environment
7. Mitigate impacts of climate change
8. Minimise waste
9. Enhance biosecurity
10. Manage environmental exposure to chemicals

Rather than considering the environment in isolation, the plan looks at the interactions between humans and the natural world in a drive to make impacting changes that are long-lasting. As part of this, natural capital is proposed as a decision-making tool for locating urban developments, enabling connectivity of natural spaces to be enhanced and promoted, as well as conserving biologically-important sites (McKinley et al. 2019). The mainstreaming of sustainable development options is also a priority, aiming to make these commonplace where appropriate, whether from existing solutions or new innovations (Department for Food, Environment and Rural Affairs 2018).

1.4.3: Natural England's Conservation 21

In 2016, Natural England set out its conservation strategy, *Conservation 21*, to inform future projects in line with the government's goals and ambitions for the environment. Criticising our current approach, which sees conservation and other land uses as mutually exclusive, the program looks to place a greater emphasis on interactions between people and the environment, much as with the 25 Year Environment Plan. To do so, it outlines three key principles for its work going forwards (Natural England, 2016):

1. Put people at the heart of the environment
2. Increase natural capital
3. Create resilient landscapes

1.4.4: The Oxfordshire Plan 2050

The Oxfordshire Plan 2050 is a regional plan for the county of Oxfordshire, developed in consultation with a range of stakeholders in the area, including local residents, government officials, and developers. It seeks to achieve a balance between the preservation of local heritage and environment, whilst also creating opportunities for economic growth and providing sufficient housing and other infrastructure (Oxfordshire County Council 2019). In such a way, the program is seen as a sustainable future initiative, promoting environmental and cultural conservation whilst avoiding socioeconomic stagnation. The plan also sits within Oxfordshire's larger Strategic Vision, which looks to establish the county as a champion of sustainable practices and aims to achieve carbon neutrality county-wide by 2050 (Oxfordshire Growth Board 2021).

1.5: Use of SuDS in the UK

Whilst surface water flooding has been acknowledged as a matter requiring urgent attention in the UK (Bevan 2018), and SuDS have been widely cited as an opportunity for assisting in new build and retrofit scenarios (see Ashley et al. 2020; Dolowitz et al. 2018; Huang et al., 2020), there has not been widespread implementation of these systems in the UK. Much of this stems from three core reasons: first, the perceived economic cost; second, an absence of design and construction guidance for planners and developers; and third, uncertainty arising from the perceived lack of knowledge on SuDS operation and performance beyond the infrastructure- or neighbourhood-scale (Ellis & Lundy 2016; Melville-Shreeve et al. 2018).

Cotterill & Bracken (2020) argue that much of this first challenge – perceived economic cost – can be attributed to the undervaluing of the co-benefits SuDS offer. Many of these co-benefits have no obvious economic value and there are challenges in appropriately capturing the advantages they do provide in a monetary form. Furthermore, as intangible and complex concepts, quite often they even prove difficult to reduce into a numerical figure, which may allow other means of comparison, and are thus not included in measures of infrastructure values and/or costs (Ossa-Moreno et al. 2017).

The policy dearth acknowledged by the second challenge varies across the UK by nation. Wales remains the only constituent country with statutory design guidance for the construction of SuDS, although the other three all offer some level of non-statutory guidelines (Woods-Ballard et al 2017). Additionally, in Scotland, SuDS are required in new developments for the management of surface water drainage (Melville-Shreeve et al. 2018), although Vilcan & Potter (2020) indicate the range and simplicity of conditions by which exemptions to this can be granted. This lack of consistency, which also extends to the question of who will maintain the systems after construction, is yet another reason cited for limited SuDS implementation in the UK (Ashley et al. 2015).

The third challenge (understanding of impacts at a wider scale) is one this thesis aims to contribute to. To-date, much work in relation to SuDS has been on optimising their design criteria for water quality improvements and/or quantity reductions (see Yang & Chui, 2018; Chatzimentor et al. 2020). Whilst crucial work for improving the efficiency of their operation, a consequence is that the majority of SuDS studies have focused on an infrastructure- or neighbourhood-scale, and thus interactions, performances and potential challenges at wider scales have not been assessed or identified. Furthermore, this optimisation is typically inward-focused – questioning how design elements of different infrastructures may affect their performance – and so, as yet, little work has looked at how external elements of urban environments (such as development layout) may influence their operation.

Thus, the thesis looks to address the following questions:

- how do external characteristics of built form influence the operation of SuDS?

- what benefits may the inclusion of SuDS offer in regional-scale urban development?
- how can SuDS contribute to environmentally-based goals and movements, such as green infrastructure?

1.6: Thesis Outline

In order to answer the proposed questions, a combination of rainfall-runoff modelling and geospatial analysis was used. The Stormwater Management Model (SWMM) was used to simulate rainfall events of different intensities under a range of urban form designs, both with and without the inclusion of SuDS infrastructure. This dynamic rainfall-runoff model, originally developed by the US Environmental Protection Agency in 1971, has been used widely in urban runoff modelling (see Fletcher et al. 2013; Fu et al. 2019; Randall et al. 2019), including SuDS performances with the more recent development of tailored modules (see Baek et al. 2020; Haowen et al. 2020). Further details on its design and previous applications can be found in Chapters 2 and 4.

Urban tiles were used to capture heterogeneity in the urban form, based on previous work by Hargreaves (2015). A range of 1-hectare tiles were drawn up to represent different housing typologies, SuDS infrastructures and urban densities, with the designs and layouts of features informed by current national guidance. These tiles then allowed simulation and analysis to occur at both the individual lot scale and at larger scales through the combination of multiple tiles. The final designs used for each element of the study can be found in chapters 3-5.

Finally, a methodology was developed to assess these different urban designs under four key criteria of green infrastructure provision (ecosystem services, ecological status, ecological connectivity and proximity to the population). Four simple scoring metrics, which were then normalised to allow cross-comparison and compilation, were developed – one for each criteria – drawing on established assessment tools with readily available datasets. It is hoped that such a design will allow for easy replicability of such an approach in future, both as a tool for planners in the UK and for illustrating conditions in other countries.

The remainder of the thesis is therefore structured as follows. Chapter 2, the literature review, builds on the ideas initially outlined in this chapter, illustrating previous and current research within the field to provide the theoretical background in which this work is set. From this, Chapters 3-5 take the form of three research papers, each addressing one of the questions identified above. Chapter 6 then draws out prominent results and conclusions from across the research, as well as identifying opportunities for furthering the work from this thesis.

2. Literature Review

Prior to the widespread flooding in Britain in 2007, minimal attention had been paid in research and policy to surface water flooding and its potential impacts outside of those related to river or coastal flooding (Parker, Priest and McCarthy 2011). This was, in part, due to the relatively weak understanding of the phenomenon in comparison to fluvial or coastal flooding (Environment Agency 2009), and led to no organisation being responsible for the monitoring or management of surface water flood events. However, following the Pitt Review (2008), increased attention, analysis and monitoring of the phenomenon were proposed, with the Environment Agency taking on responsibility for the strategic overview of surface water flood risk in the UK (Environment Agency 2009). That is not to say that it was a new phenomenon, however, simply that it had not previously been widely formally recognised in British policy and planning.

It is not just in Britain, either, that the surface water phenomenon has become more widely recognised in recent years. Graham et al. (2012), for example, identify how the Pitt Review has influenced Canadian policy on surface water flood risk, whilst Thieken et al. (2016) discuss the impact the 2013 surface water floods in Germany have had on government policy there. Despite vastly different physical and human environments globally, however, common challenges can be identified in the literature surrounding the monitoring and management of surface water flooding.

Firstly, it is widely acknowledged that forecasting and the provision of warnings for surface water flooding are particularly difficult due to the complex and compound factors that interact to cause such an event (Parker, Priest & McCarthy 2011; Lane, Landström & Whatmore 2011; Dale et al. 2014), and the localised scale of the resultant flooding (Ahiablame & Shakya 2016; Kreibich et al. 2015). Secondly, challenges remain in introducing and retrofitting management solutions (typically additional drainage infrastructure), as the most at-risk areas are highly developed urban entities that generally face a shortage of available space for such development (Graham et al. 2012; Haaland & van den Bosch 2015).

2.1: Flood Risk Management

Flood risk is a dual-aspect term which seeks to encompass (a) the likelihood of a flooding event occurring and (b) the magnitude of impacts upon existing infrastructure or populations of such an event (Emanuelsson et al. 2014). Gathering data, assessing this risk, and then implementing and reviewing appropriate responses is known as flood risk management (Hall et al. 2003). The first aspect of flood risk – event likelihood – is expressed by event return periods which represent the average time a flood of a given magnitude occurs (Lane, Landström & Whatmore 2011). For example, a 1-in-50 year flood will, on average, occur once every 50 years. However, as Priest et al. (2011) highlight, this can often be misleading for those outside of the industry, such as the general public, as it can easily be misconstrued as an event which will only occur once in the given time frame. Thus, they propose an alternative method should be used instead, such as the corresponding probability (2% in any given year for the above example).

The second aspect of flood risk – scale of impacts – is composed of two elements: (1) exposure and (2) vulnerability. The former of these appreciates the number of people or things people that would be affected by an event, and the latter the likelihood of these people or things being harmed (Kaźmierczak & Cavan 2011). High exposure or vulnerability does not necessarily mean the other element will be high too, however – a large community, for example, may present high exposure for an event, but if suitable policies and effective defences are in place, vulnerability will be low (Kreibich et al. 2015). Furthermore, whilst not strictly an element of flood risk, flood resilience has emerged as an important concept when managing flood risk (Potter & Vilcan 2020). It is defined as the capacity of a society to absorb the adverse impacts of a flood event and can both improve, and be improved by, effective flood risk management (Disse et al. 2020).

2.1.1: Approaches to Flood Risk Management

Current approaches to flood risk management are largely divided into two categories – structural and non-structural – with the former encompassing physical infrastructure projects, such as levees or flood barriers, whilst the latter includes policy-based solutions, such as flood risk zoning (Alexander et al. 2016). Preference towards a given approach, however, varies greatly on a national scale, with mixed opinions as to the effectiveness of each. Flood prevention strategies in France and the UK, for example, have primarily focused on restricting development on floodplains, including the removal of housing stock following the 2010 floods in France (Lumbroso and Vinet 2011), whereas the Netherlands places a primary focus on embankment construction in vulnerable regions (Bubeck et al. 2015).

Since the turn of the century, there has also been a marked divide within the structural approaches to flood risk management. Traditional approaches which focus on piped infrastructure, including sewers and wastewater treatment plants, are referred to as grey solutions, and usually constitute part of a centralised network of management infrastructure (Ashley, Gersonius and Horton 2020). Conversely, green infrastructure, which is also referred to as green-blue infrastructure, uses soil, vegetation, water features and ecosystem functions to mimic natural drainage processes, and is more commonly found in isolated or localised networks within the urban setting (Ellis 2013). Examples include swales, green roofs and constructed wetlands. However, more frequently, these are being developed into larger networks to create sustainable and environmentally-efficient drainage networks called Sustainable Drainage Systems (SuDS), Low-Impact Developments (LIDs) or Water Sensitive Urban Designs (WSUD) (Fletcher et al. 2015).

Studies into such green infrastructure have primarily focused on their optimisation for water quality improvements (e.g. Drake, Bradford and Marsalek 2013) and water quantity reductions (e.g. Ellis and Viavattene 2013). This research has included comparisons into both different forms of the same infrastructure under different designs and/or conditions (see Drake, Bradford and Marsalek 2013; Winston et al. 2016), and between different infrastructure types in the same environment (see Hoang and Fenner, 2016; Liu et al., 2016). They have also employed both case study-based work (such as Vineyard et al. 2015) and modelling approaches (such as Randall et al. 2019).

2.1.2: Integrated Flood Risk Management

The challenge of retrofitting drainage networks (as creating effective SuDS trains in many existing cities requires large-scale renovation and reshaping of the urban landscape (Gill et al. 2007)) has sparked a shift in approaches globally towards an integrated flood risk management (Bubeck et al. 2015). These look to incorporate elements of structural and non-structural management approaches, including both green and grey infrastructures, into a comprehensive network, complementing and supporting the existing urban drainage system rather than redeveloping significant elements of the urban form (Alexander et al. 2016). Despite considerable uptake of this approach in western Europe, however, Kreibich et al. (2015) highlight that current attempts prove insignificant in both the UK and France, where development pressures on drainage infrastructure continue to grow, placing an increasing number of people, services and buildings at risk.

In order to achieve the most effective integration, it is important to understand not only the respective strengths and weaknesses of each drainage element under varied environmental conditions, but also their relative impacts on one another (Halbe et al. 2013). Results from efficiency studies, even into the same type of infrastructure, have varied greatly – studies in the US into green roof performance, for example, have cited precipitation retention efficiencies varying between 38% and 71% (Dietz 2007). Furthermore, much of this research has focused on green infrastructure rather than traditional grey, non-structural or hybrid drainage solutions. Stovin et al. (2012) in particular noted the benefits these could bring in retrofitting projects as they found the introduction of green drainage systems alone to be an economically-prohibitive venture.

2.1.3: Contemporary Pressures on Flood Risk Management

It is clear that addressing the challenge of surface water flood risk is not an easy task, and that whist SuDS provide a promising option for future drainage infrastructure, work is still needed to understand how such systems may best be implemented. However, this task is further complicated by the pressures of two key global changes – urbanisation and climate change. Whilst climatic modelling predicts average annual precipitation in the UK will be lower by the end of the century than at present, the temporal distribution of this during the year is expected to change, leading to more frequent and more intense rainfall events during the winter months (Charlesworth 2010; Arnbjerg-Nielsen et al. 2013). It is important to note that these projected impacts are uncertain, due to limitations and uncertainties in both climate projections and hydrological models, but there is a widespread consensus that pressures on drainage infrastructure will increase (Yazdanfar & Sharma 2015).

Urbanisation is increasing worldwide, such that now more people live in urban locations than rural, and it is estimated that this trend is to keep growing (Haaland & van den Bosch 2015). By 2030, three times the area of urban spaces as in 2000 will be globally required to adequately support the predicted volumes of urban population (Felappi et al. 2020). The development of these environments, with features such as increased surface sealing, can lead to the occurrence of undesirable phenomena (e.g. urban flooding and heat stress) without appropriate design and management (Alves et al. 2019). Urban developments take different forms, and each has its own unique impacts on the subsequent flows and systems within the settlement. In such a vein, many European cities (and indeed worldwide) have a dedicated plan to inform the spatiality of new developments – see FSOD (2012) and MRD (2015) – but this is not the case in the UK (Williams 2014). Whilst these plans all differ due to the influence of political ideals and desired outcomes, there is a common acknowledgement of the need for physical infrastructure to support new and versatile flows of goods and people as demanded by the twenty-first century (Ashley et al. 2013).

2.2: Urban Flooding

Flood events occur when the inflow of water exceeds the outflow, resulting in the build-up of water in a location (Anim et al. 2019). Broadly speaking, this rainfall to runoff process can be conceptualised into two stages – first, losses occur due to infiltration, interception, evapotranspiration and depression storage; second, the then effective rainfall becomes surface flow (Granata et al. 2016). It is this surface flow, if not adequately reduced, channelled and/or managed, which then becomes the floodwater as it collects and traverses the land (or urban) surface.

Elements of the urban form interact with these processes, contributing to or limiting them. Areas of vegetation and surface water are often reduced, for example, decreasing losses by evapotranspiration and limiting potentials for storage, whilst increased surface sealing decreases infiltration losses and natural sub-surface flow pathways (Lashford et al. 2019). Urban drainage systems can modify elements and processes in this hydrological cycle, too – losses, for example, can be increased further through artificial “infiltration” into subsurface channels (Arnbjerg-Nielsen et al. 2013). Appreciating the dynamics of urban hydrology, therefore, (especially in regards to stormwater) requires both elements of the natural hydrological cycle and interactions of manmade infrastructure to be accounted for (Barbosa, Fernandes & David 2012).

2.3: Modelling Urban Drainage & Flooding

Urban drainage models are designed to replicate the system on the ground, allowing users to assess the impacts of changes in design and/or storm magnitudes in the given area (Fletcher et al. 2013). Modelling of urban drainage can be considered to have begun as early as 1850 with Mulvaney’s rational method which offers much of the basic functionality of today’s rainfall runoff models, although computerised modelling, as we usually perceive the term to mean, didn’t emerge until the 1970s, and even then development was initially slow due to the limitations in computing power (Beven 2012). However, as these improvements were introduced and models consequently refined, urban drainage modelling has become a crucial part of drainage engineering, allowing decision-making and designs to be more informed and better reflect potential future conditions (Fletcher et al. 2013).

At their core, urban drainage models are based on current understandings of urban hydrology, representing how both losses and surface flow occur in the defined scenario, and any influences an urban drainage system may have. To do so, the principles of runoff generation, overland flow, natural sub-surface flow and pipe flow (that in closed channels, typically underground), as well as basic hydrological processes (e.g. evapotranspiration), must be adequately appreciated within the model (Barbosa, Fernandes & David 2012). Models in the field can be sub-divided into three dominant types - (1) empirical, (2) physically-based, and (3) conceptual – although recently there has also been increased interest in those utilising artificial neural networks and support vector machines, which stem from work in artificial intelligence (Granata et al. 2016). In addition, models can be stochastic or deterministic, with the former incorporating elements of randomness whilst the latter does not.

Many of the standard models used in drainage modelling today are deterministic, physically-based models (Bach et al. 2014). It has become increasingly recognised, however, that a single modelling approach may be insufficient to accurately appreciate all elements and interactions of the system, and thus coupling and integration of existing models is becoming more widely used (see Baek et al. 2020; Ellis & Viavattene 2014; She & You 2019). These allow all components of the urban drainage system to be modelled, including potential involvements and interactions with other systems (Bach et al. 2014).

By their definition, all models involve some element of simplification, yet without suitable acknowledgement of resultant uncertainties or limitations, questions over the accuracy and utility of the model are raised (Tscheikner-Gratl et al. 2016). Using high-quality input data, calibrating models (where applicable) and careful choice of the drainage model to match the aims of a study are key steps to increasing the confidence and accuracy of model outputs (Sitzenfrei & Rauch 2014). Whilst good practise on paper, however, critics argue that such opportunities are often limited due to model and/or data availability and access

(Langeveld et al. 2013) – a problem that is an even greater challenge for coupled and integrated models that require even more, diverse datasets for calibration (Oberascher, Rauch & Sitzenfrei 2022).

2.3.1: SuDS in Urban Drainage Modelling

Increased popularity of SuDS infrastructure has seen the development of modules to enable the inclusion of green drainage elements within the frameworks of several urban drainage models, such as STORM and the Stormwater Management Model (SWMM). The modelling of SuDS introduces further complexities into the existing challenge of urban drainage modelling, as there is greater spatiality to the performance of different infrastructures with local environmental contexts informing parameter values (Palla & Gnecco 2015), and they can require the representation of three-dimensional flow processes in the more complex modelling approaches (Harris et al. 2016). Furthermore, they each present their own unique interactions with both the hydrological processes and flows in the urban system, as well as the existing urban drainage network, where they are introduced (or modelled) as elements of the existing urban wastewater network (Sitzenfrei & Rauch 2014).

It thus follows that significant focus in SuDS modelling has been placed on the distribution of SuDS in the urban form, identifying challenges and best practices, as well as developing interconnected, green drainage "trains" which feature multiple infrastructure types. These have led to the development of spatial allocation optimisation tools (SAOTs), which look to identify both which SuDS infrastructures are best suited to a region (based on the external environmental characteristics), and the best locations in the region for them to be placed. A comprehensive overview and comparison of different types is presented by Zhang & Chui (2018), who conclude that whilst there are many SAOTs, each of which have their own objectives and select set of infrastructures that can be considered, the spatial scales at which they can be applied are limited, and the appreciation of both subsurface hydrology and ecological parameters are generally poor. This echoes findings from urban flood management modelling as a whole, where approaches to infrastructure selection strategies often rely on intensive modelling and ‘expert’ opinions, although more recent frameworks have looked to address this, such as Webber et al. (2017).

Ahiablame, Engel & Chaubey (2012), Jayasooriya & Ng (2014) and Eckart et al. (2017) offer comprehensive reviews of different models that currently allow appreciation of SuDS hydrology, with Jayasorriya & Ng (2014) including models that provide economic calculation for SuDS too. Whilst they all identify that the specific design of each model brings about unique restrictions, there are still several overarching limitations that need to be considered when using such models – most notably, the scales of required input datasets which restrict the model’s suitability to the planning-scale as opposed to a finer-scale design, and the use of in-built databases which restrict certain parameters, making application only suitable in the climatic/topographic region the model was originally designed for.

Nevertheless, with careful consideration of limitations and model choice, such modules have proved sufficiently accurate in studies with a range of locations, scales and chosen SuDS designs (Baek et al. 2015). In fact, Eckart et al. (2017) argue that modelling is the best approach for optimising SuDS design at all scales, although they stress the importance of accurate and realistic parameters. Previous research using SuDS modelling techniques has addressed many elements of their use and potential, including Hernes et al. (2020) who compared the performance of several different infrastructures in reducing combined sewer overflows, Stovin et al. (2012) who assessed the impacts of retrofitting SuDS infrastructure into existing urban environments, and Singh et al. (2019) who optimised the design of a specific bioretention system. Water quality elements and co-benefit provision have also been addressed by SuDS modelling approaches, yet much development is still required, as these present even greater challenges in calibration, process simulation and quantification of co-benefits (Cotterill & Bracken 2020).

2.4: Appreciating Co-Benefits of SuDS

It is widely acknowledged that in addition to hydrological advantages, SuDS can result in other benefits, often referred to as co-benefits. Many studies have looked to identify these, which include air quality improvements, aesthetic value, biodiversity, carbon storage, urban cooling, food provision, leisure opportunities, and mental health benefits among others, and quantify their impacts (see Alves et al. 2019; Jose, Wade & Jefferies 2015; Mayrand & Clergeau 2018). Often the focus, however, has been on a single benefit or type of benefit (e.g. ecological). Furthermore, many studies report difficulty in measuring certain benefits, such as mental health benefits, and achieving an approach that enables a direct comparison between different benefits (Hoang, Fenner & Skenderian 2017). In an analysis of the One Health method, which aims to measure animal health, human health and environmental factors in an equal and comparable methodology, Felappi et al. (2020) identified that studies using this approach tended to focus on the first two for a health perspective as the third factor required considerably different data collection and analysis techniques.

Several studies have tried to address this latter challenge through the monetisation of benefits, including Gómez-Baggethun & Barton (2013) and Johnson & Geisendorf (2019). Whilst this allows cross-comparison and is a format easily understood by many stakeholder groups, Alves et al. (2019) raise concerns over such an approach, echoing similar sentiments by Thomas & Littlewood (2010) and Salomaa et al. (2016). They argue that such an emphasis on monetisation encourages active management of green infrastructure to reap maximum gains from benefits that can be readily monetised, placing bias on sites with a high proportion of provisioning ecosystem services (those providing the output of physical goods, e.g. wood or food), and overlooking variation in the sociocultural values of different land functions by different social groups.

Another limitation in co-benefit consideration is the small-scale focus of such research. Typically, studies analyse the benefits offered by each type of infrastructure, whereas Hansen & Pauleit (2014) argue that benefits vary significantly from location to location and are influenced by other SuDS infrastructure in the area, too. These interactions can be synergistic, supporting one another to increase the benefit provision further, or act more like a trade-off, and thus it is important to consider the wider-scale context when assessing benefits (Haase et al. 2012). This proves even more important when SuDS are being designed to deliver particular benefits in a region, e.g. the use of green roofs to encourage insect mobility and diversity in urban locations (Benvenuti 2014). Felappi et al. (2020) also highlight how these trade-offs can occur within, as well as between, features – widespread grass or grass-like surfaces, for example, are good for some species but not others.

2.5: Barriers to SuDS Adoption

Regardless of physical or sociocultural context, the uptake of SuDS faces many barriers (both real and perceived), and Ashley, Gersonius & Horton (2020) highlight that current uptake rates and investment (in both green and grey drainage infrastructures) are insufficient to match the rate of changing demands. Nevertheless, whilst there is a consensus that the UK has a fragmented implementation, explanations of this spatial distribution vary. Hoang & Fenner (2016) cite regionalised planning bodies and poor communication between government bodies and construction companies, whilst Cotterill & Bracken (2020) highlight the role of community engagement as well as industrial stakeholders. Li et al. (2020), however, point to challenges of cost perceptions – one of particular importance to address in England as current policies allow new developments to not use SuDS where they are considered to not be economically proportionate (Ellis & Lundy 2016).

At a more global scale, one of the main challenges identified throughout the literature has been the limitations of scale, which predominantly focus on infrastructure-specific or small-scale cases. Ahern (2013) argues that this is an underlying flaw in urban planning, where visions are slow to appear and development is limited through a reserved approach that tends away from innovative practices beyond aesthetic design. Instead, they argue that a safe-to-fail mindset should be adopted, building on practices of adaptive management from the fields of resource and wildlife management. With this, bold innovations in the urban form can be implemented and adapted, with the lessons learned from "failed" projects – those that don't meet the ambitious goals typically given to projects trying to go against the flow – given value, and appreciation granted to the benefits received by the area (even if they are lower than originally intended). This approach is currently being implemented in Philadelphia, USA, where city planners aim to green nearly 10,000 acres of currently impermeable surfaces in the next 25 years (Dolowitz et al., 2018). Langeveld et al. (2022), meanwhile, emphasise the importance of managing the drainage infrastructure after construction, as not only can poor stewardship exacerbate water quality/quantity issues, but such failures can damage public perceptions of the infrastructure too.

2.6: Urban Sustainability

2.6.1: Sustainable Urban Forms

The question of form for a sustainable city has seen much attention, forming the focus of debates in British planning in the 1990s, and remaining a key focal point in urban form research in recent years. Echenique et al. (2012) proposed that ecological principles are used to inform green technological development, around which traditional urban form is then constructed, and Hansen & Pauleit (2014) support this, arguing that it is the flexibility of these systems that is crucial for defining their future sustainability. However, neither identified a spatial form this ultimate development might take.

Manesh, Tadi & Zanni (2012), Bach et al. (2013) and Eckhart, McPhee & Bolisetti (2017) have all looked to investigate the impacts of urban form on sustainable urban development, although with a variety of focal points – Manesh, Tadi & Zanni (2012), for example, centres the debate on implications for the energy sector, whilst Bach et al. (2013) looks more at the effects for surface water and wastewater systems. Results from across these three studies, however, identify the importance of urban density, diversity and proximity on sustainable systems, although there is a lack of quantification of a threshold value or range above which sustainability can or cannot be achieved.

2.6.1.1: The Compact City Model

In the British planning context, the Compact City model is currently viewed as a highly successful urban form, offering high building density in mixed-use environments, mirroring the drive for urban spaces to be multi-functional (Williams 2014). However, Echenique et al. (2012) suggest there is no long-term benefit for energy or resource consumption in a traditional compact city, identifying that urban density dynamics have a significant role in informing the success and efficiency of urban flow networks, particularly in a sustainable context. Keenleyside et al. (2009) echo this impact of the compact on sustainable systems and future development, drawing examples from Germany and Australia where compact city development has given way to poly-centric development in new sustainable centres.

Nevertheless, there are many key design aspects from the Compact City model which can be considered relevant to the sustainable city form debate. The resultant high densities of compact developments are often considered sustainable as they limit encroachment into the rural and greenspace environments surrounding existing city cores, and create multi-functional spaces that reduce the reliance on private transport for travelling significant distances (Artmann et al. 2019). Popularity of the Compact City model is thus seeing a resurgence driven by other planning movements, such as ideas of walkability (Echenique et al. 2012), and several principles from the approach are promoted in the UN's New Urban Agenda for sustainable urban development (Felappi et al. 2020). On the other hand, however, high-density environments offer less space-per-capita in which to provide the urban services and infrastructure required, and demanded, by the area's residents, which can lead to ex-urban migration in poorly designed cities (Haaland & van den Bosch 2015).

2.6.1.2: The Green City Movement

Elements of the Green City movement are also frequently cited in debates on sustainable city form. These champion the regular inclusion of greenspace elements in the urban form, helping to address localised challenges of urban form (such as the fragmentation and widespread destruction of ecosystems) and providing a greater connection between urban residents and the natural environment (Steele, Davidson & Reed 2020). However, these urban locales typically lead to less dense development, resulting in a greater urban footprint for a given population size (Haaland & van den Bosch 2015). With less dense development, the variety of functions offered within a given area is also typically lower, and thus a greater requirement for travel of significant distances exists, as evidenced by the increased per-capita consumption of CO_2 in less dense cities (Albino, Berardi & Dangelico 2015).

2.6.1.3: The Smart-Compact-Green City

Aware of the relative benefits and limitations of each, Artmann et al. (2019) propose a third basis for considering urban form development, the smart-compact-green city, which looks to provide a balance between favourable elements from both. They are not the first to make such a consideration, but prior studies proved largely superficial and theoretical, whilst Artmann et al. (2019) propose specific actions which could help achieve this balance. Generalised city models, however, with their broad aims and design features, are what Newman (2014) sees as part of the problem in developing a sustainable city form, arguing that the context and existing urban forms and problems in a region or country should shape how future developments respond to the sustainability challenges faced by, and relevant to, said region.

This importance of context is also identified by Yang & Li (2010), who report that sociocultural dynamics, as well as the influence of physical forms, dictate and influence the sustainability of a region. They highlight the role of this human factor through the comparison of both a conventional and a SuDS drainage system in a new town development, where several neighbourhoods revert from SuDS to traditional drainage options through socially-driven demands. This finding has been echoed across many other studies, too, with varying geographical foci, including Williams et al. (2012) (England), Bach et al. (2013) (Australia) and Martin-Mikle et al. (2015) (USA).

2.6.2: Urban Greenspace & Biodiversity

The role of urban greenspace has been increasingly recognised in the literature from many fields, including hydrology (Hoover & Hopton 2019), human health (Felappi et al. 2020), leisure (Akpınar 2019), and ecology (Aronson et al. 2017). Through the provision of ecosystem services, urban greenspace can not only support urban ecosystem integrity, but also help reduce urban heat island impacts, attenuate noise, offer food provision, create spaces for sport and recreation, and improve visual aesthetics (Wolch, Byrne & Newell 2014). With such a growing recognition of the important and varied benefits these spaces can bring, especially in the light of the recent pandemic (da Schio et al. 2021), calls for urban greenspaces are increasing, too, even outside of sustainable city debates (Felappi et al. 2020). However, with the preservation of this natural (or semi-natural) space in the urban form, it is clear they contribute to such sustainable forms, too (Haaland & Van den Bosch 2015).

One of the most widely recognised benefits of urban greenspace is the biodiversity it can provide and support (Aronson et al. 2017). These spaces exist as natural, artificial and semi-natural environments throughout the urban form and with vastly different areas (Derkzen et al. 2017). In fact, Wolch, Byrne & Newell (2014) suggest that urban locales can lead to unique combinations of environmental conditions, creating habitats not widely observed in the ‘natural’ world. However, poor management and urbanisation are leading to the degradation and loss of such spaces (Aronson et al. 2017) – a challenge Russo & Cirella (2018) suggest we are ill-prepared to address as our appreciation of the range and magnitude of benefits (and particularly how we can measure and compare across different urban spaces) remains limited.

Beyond the urban sphere, concerns about the increasingly fragmented nature of greenspaces are growing too, both in Europe and worldwide, which has led to several countries developing guiding policies and legislation to encourage the conservation and reconnection of these spaces through the use of green infrastructure (GI) (Algador et al. 2012) and prudent urban planning approaches (Pauleit et al. 2020). One such document was released by the European Commission in 2013 – “Green Infrastructure – Enhancing Europe’s Natural Capital” – which is focused on the promotion of GI in planning considerations with an emphasis on the development of ecological corridors (Hansen & Pauleit 2014). Several studies have since analysed how such connections can be achieved in various European regions, including Algador et al. (2012) and Rusche, Reimer & Stichmann (2019).

As a subset of GI, SuDS are a regularly named example of a potential intervention (Pauleit et al. 2020). They are considered of particular value, too, as (when implemented) they are concentrated within and across the urban form (Russo & Cirella 2018) and can be retrofitted with greater ease than some other GI infrastructure (such as new urban greenspaces) (Rusche, Reimer & Stichmann 2019). Furthermore, as previously identified, they are also associated with many other benefits above and beyond biodiversity, and can thus (with careful planning) help to address and enhance many different conditions simultaneously. Dolowitz et al. (2018) exemplified this in Philadelphia, USA, using a co-benefits study to inform the local drainage plan. A heavy focus on green infrastructure development (and in particular SuDS) emerged due to these multiple and wide-ranging side benefits. Rodríguez-Espinosa, Aguilera-Benavente & Gómez-Delgado (2020), however, identify that many co-benefit assessment tools currently have a distinct biodiversity focus and that considering a wider range of benefits may create different networks which ultimately prove more beneficial in a greater variety of spheres.

2.7: Research Objectives & Contributions

As previously identified, urban hydrology and SuDS operation are influenced by their context. However, there has been limited assessment to-date focused on how urban characteristics affect the performance of SuDS. Bach et al. (2013)

assessed the influence of existing soil types (and their associated hydrologic parameters) on the performance of SuDS infrastructures in Melbourne, Australia, whilst Hoang, Fenner & Skenderian (2017) undertook a similar performance study in Portland, USA, instead focusing on site-specific climatic factors. Brody et al. (2013) applied a wider-scale focus, analysing settlement clustering in Mexico and the southern USA, and the consequent impacts on runoff and SuDS efficiencies. More recently, Rodriguez et al. (2021) analysed how the resilience of SuDS were impacted by their spatial location in urban environments.

Beyond these, however, there has been little work to identify and quantify how characteristics in the urban built form, such as urban density and building footprints of different housing typologies, influence the performance of SuDS. The first objective of this thesis, therefore, looks to address this gap.

The second objective then aims to extend this to a wider regional context, assessing the impact of development design principles whilst also introducing an element of spatial variation present in real-world catchments.

Beyond hydrologic performance, whilst it is acknowledged that SuDS infrastructures offer a diverse range of co-benefits, communicating these effectively to the wide variety of stakeholders involved in urban developments can be a challenge, particularly when considering how to balance trade-offs and losses in a multi-infrastructure project (Hansen & Pauleit 2014). Furthermore, quantifying (and often monetising) these benefits can be difficult, particularly when they have no intrinsic economic value, such as mental health benefits

(Alves et al. 2019). It is also the case that often SuDS infrastructures can play a vital role as GI elements, helping to reduce the fragmented nature of greenspaces and support benefit provision in urban locales. Prior work in optimising the location of GI networks, however, has typically employed a strictly ecological focus and/or utilised “expert knowledge”, which neither encompasses the diversity in co-benefits that could be provided nor provides an easily replicable methodology for developers or planners.

Objective three of this thesis, therefore, seeks to develop an accessible approach that addresses multiple elements of GI benefit provision, and use this to identify the extent and spatiality of benefits SuDS infrastructure can provide in regional scale development projects.

3. The Influence of Built Form and Area on the Performance of Sustainable Drainage Systems (SuDS)

3.1: Abstract

In the face of increased housing demand and climatic change, sustainable urban drainage systems (SuDS) are often viewed as an alternative to traditional piped drainage networks, offering multiple benefits. However, whilst design guidelines for sustainable drainage systems (SuDS) exist, there is little systematic understanding of how SuDS perform for different urban forms at a neighbourhood scale. This paper, therefore, explores the response of a one hectare urban area to rainfall events of varying magnitude under a range of different scenarios for the built environment (development density, SuDS type, residence type and SuDS deployment extent), using the Stormwater Management Model (SWMM). It finds that whilst increased development density leads to an increased peak runoff rate, the reduction of this rate under SuDS implementation is greater in the higher density scenarios, to the extent that in some cases lower SuDS deployment in higher density scenarios leads to lower runoff rates than high deployment in a lower development density. It is important to note, however, that these implementations reflect proportions of available surface type and so absolute areas may not be lower. The type of SuDS also has a considerable impact on runoff dynamics, with those constructed on existing infrastructure offering greater proportional reductions in runoff rates under higher development densities than those constructed on previously undeveloped land.

3.2: Introduction

There is a longstanding debate about the relationship between the density of urban development and cities' sustainability. One aspect of this debate has concerned energy use (see Rode et al. 2014; Stevenson & Gleeson 2018), with more compact cities minimising energy use for transport up to a point after which the energy intensity of the most dense cities apparently increases. The debate is also reflected in discussion of the 'liveability' of cities, which idealise a walkable city environment and reduced urban sprawl, promoting a compact, dense city form, whilst calls for increased urban green space and the maintenance of nature networks seemingly demand the opposite (Artmann et al. 2019). When it comes to considering sustainable urban development, high and low density solutions present their own strengths and weaknesses, and thus a delicate balancing act is required in the development of urban spaces to create the best of both worlds (Lehmann 2016).

This balance is further complicated by the additional challenges of climatic change, which will have resultant impacts on urban conditions (e.g. surface runoffs, urban heat island effects) that will need to be managed through urban design (Caparros-Midwood, Barr & Dawson 2017). Storm events are expected to become more frequent and bring higher volumes of precipitation (Zuniga-Teran et al. 2020), and thus stormwater management in urban contexts will be increasingly important. As well as developing appropriate methods and technologies to cope with these changes, the spatiality of these infrastructures and their integrated nature into the built environment are equally important considerations (Yazdanfar & Sharma 2015).

Irrespective of whether we densify existing cities or construct new settlements, urban development sees a proportional loss of permeable surfaces for impermeable (in traditional developments), which leads to the loss of natural drainage pathways (Miller & Hess 2017). This leads to a resultant increase in the peak surface runoff volume and rate from a catchment during a rainfall event, which can result in, or exacerbate, flooding. Traditional drainage networks use built infrastructure (usually underground) to capture and transport this water out of the urban area. Connecting new/infill developments to these existing networks can overload the existing network, requiring costly capacity expansion (Yazdanfar & Sharma 2015). Lennon, Scott & O'Neill (2014) argue that traditional hard engineering techniques will become increasingly inappropriate in the face of urbanisation and climatic changes, and thus promote the inclusion of green infrastructure techniques (such as SuDS) in urban design as a move towards mitigation and adaptation.

SuDS are an alternative to traditional drainage, mimicking natural drainage processes (Anim et al. 2019). They can also create habitats for nature, opportunities for water reuse, and offer water quality improvements (Ellis & Lundy 2016). Some types of SuDS offer storage of surface runoff, whilst others focus on increased drainage of surface water through increased permeability (Liao, Deng & Tan 2017). In this research, we focus primarily on the latter, although post-infiltration all three modelled systems (bioretention, green roofs and permeable surfaces) have the potential for storage. We also distinguish between two categories of SuDS in our study - infrastructure-based SuDS, which alter impermeable surfaces to increase their permeability, and freespace SuDS, which are constructed on undeveloped surfaces and can boost the permeability of an already-permeable area through altering soil or vegetative conditions. However, whilst some SuDS (such as green roofs) can only fall into one of these categories, others (such as swales) can be constructed on developed or undeveloped surfaces. For this research, bioretention areas were considered freespace SuDS, whilst permeable surfaces and green roofs were infrastructure-based SuDS.

Since the end of the last millennium, there has been an increase in the use of SuDS (Fletcher, Andrieu & Hamel 2013), with well-publicised examples of uptake in China, Scandinavia and Australia (Fu et al. 2019; Yazdanfar & Sharma 2015; Zuniga-Turan et al. 2020). The United Kingdom has seen a regionally-divided uptake, largely attributed to the differing planning policies and statutory guidance issued in its constituent nations (Vilcan & Potter 2020). The use of SuDS in new developments is mandatory in Scotland and, under certain conditions (e.g. development size), in England and Wales, however, Ellis & Lundy (2016) note that policy loopholes mean that no real impact on uptake can be seen, particularly in England, which also lacks any statutory standards for SuDS (Vilcan & Potter 2020).

The increased prevalence of SuDS schemes has led to a simultaneous increase in SuDS-based research, particularly concerning the influence of the SuDS design on its efficiency in regards to water quality and quantity. There has been less focus, however, on how different urban layouts influence potential options for SuDS schemes. Bach et al. (2013) identified how different densities and soil types found in Melbourne, Australia impact infiltration and runoff, but didn't consider other conditions, such as slope or housing typology. Similarly, Hargreaves (2015) employed a tile-based approach to illustrate the impacts of different housing typologies alone on density, and how this may more generally impact upon potentials for ‘green’ technologies and decentralised supply/management systems. However, beyond this, there remains the need to better understand the interplay between urban design and SuDS efficiencies – a need previously determined by the Pitt Review (2008) which, having identified nature-based solutions as key for resilient urban developments, called for increased understanding of the role of urban design in this risk reduction.

As a result of the close interaction between surfaces in the urban development form, the effect of SuDS in a development is dependent on the features and design of this form. For example, since by their definition infrastructure-based SuDS are constructed on infrastructure, denser settlements, which provide a greater proportion of these surfaces (e.g. roads, buildings) within a given area, hold a greater potential for infrastructure-based SuDS over freespace SuDS (those built on permeable surfaces). If only freespace SuDS types are being used in a development catchment, this means that not only is there a reduced area for SuDS interventions, but increased runoff into the system from the increased surface area of impermeable infrastructure.

Many other characteristics of the catchment can also influence the available surface areas for different SuDS types, and affect other aspects of SuDS designs (e.g. slope). Hargreaves (2015) illustrated that many of these are inextricably linked to housing typology, with the different densities that can be offered in a given-sized space resulting in different proportions of roof area, roads and paving, and remaining green space. This local-scale focus is also best placed to understand green space provision benefits for residents (Bach et al. 2013).

Building on Hargreaves (2015) tile-based approach to urban catchment analysis, this study looks to better understand this urban design and SuDS provision relationship, through addressing the following questions:

- How do the responses of different SuDS infrastructure to a rainfall event vary under different urban density scenarios?
- How do hydrological characteristics of the urban environment (e.g. antecedent soil moisture, slope, soil type) influence the SuDS response to a rainfall event?
- For a given urban design, is there a density threshold that can be identified, achieving a balance for meeting housing demand (whilst limiting impacts on natural capital and urban footprint growth) and offering space for SuDS to reduce flooding impacts?

3.3: Methodology

3.3.1: Urban Hydrology

Appreciating the dynamics of the hydrological cycle in the urban domain requires consideration of both the natural water cycle and the manmade elements which interact with it, such as those for storage and conveyance (Barbosa, Fernandes & David 2012). Not only does this lead to more complex pathways through the cycle due to the increased number of elements involved, but creates challenges for data collection as often these manmade elements are owned by private companies, leading to uncertainties and difficulty in accessing information on channel (pipe) size and locations (Noh et al. 2016).

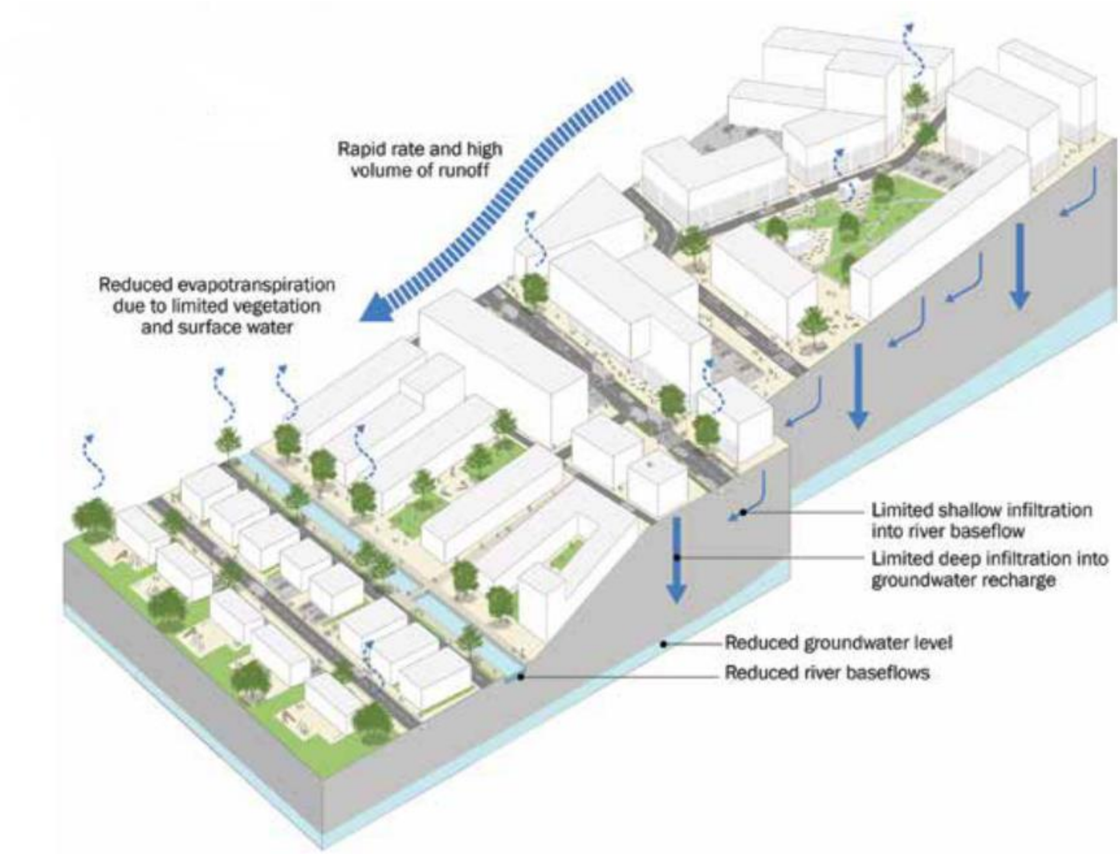

Figure 3.1: The impacts on the hydrological cycle of urban development (CIRIA 2015)

In addition, the processes and storages of the natural water cycle are also altered in the urban domain, with potential for some being reduced (see Figure 3.1). For example, the increased impermeable surface area relative to an undeveloped parcel leads to reduced infiltration and evapotranspiration, with a resultant increase in surface runoff (Anim et al. 2019). Reduced infiltration into permeable, undeveloped land also leads to reduced groundwater level and reduces the resilience of the land to prolonged dry periods.

Management in the urban form usually results in attempt to control where the water is located, too, through the channelling of water into drains and pipes. This is then conveyed out of the populated area. Consequently, when the inflow is greater than the outflow (during intense and/or persistent rainfall), problems are exacerbated as there is little storage capacity and water collects in these areas.

As a result, the spatial and temporal scale of the sub-processes in the hydrological cycle are much smaller in the urban domain than the rural. This led Niemczynowicz (1999) (and later Paz et al. 2019) to argue in their review of the field that data collection would ideally occur at this smaller scale to improve accuracy in the modelling and monitoring of these processes. The impracticality of this, however, means that many of our contemporary models operate using data from much larger spatial scales.

3.3.2: SuDS & their Impacts on Urban Hydrology

SuDS work to mimic processes of natural water cycle in urban setting, addressing one or more of the changes discussed above. Table 3.1 illustrates which of the five changes to the natural water cycle in an urban location (as identified by CIRIA 2015) is addressed by which of the common SuDS methods. Three different SuDS methods were chosen for modelling in this study, covering the full breadth of the five influences, and offering infrastructure-based and freespace alternatives – bioretention areas, green roofs and permeable paving.

Table 3.1 – Common SuDS infrastructure and their influence on urban hydrology

	Increased shallow infiltration	Increased deep infiltration	Reduced surface runoff rate and high volumes	Increased evapo-transpiration	Increased groundwater flows/higher groundwater levels
Bioretention areas	✓	✓	✓	✓	✓
Constructed wetlands	✓	✓	✓	✓	✓
Detention basins	✓	✓	✓		
Drainpipe disconnection	✓	✓			✓
Green roofs			✓	✓	
Permeable paving	✓	✓	✓		✓
Rainwater harvesting			✓		
Retention basins			✓	✓	
Swales	✓		✓	✓	

Bioretention areas are landscaped regions designed with engineered soils and vegetation to promote infiltration – both from the surface into the bioretention area, and also from the bioretention area into the existing underlying soil. In so doing, surface runoff is reduced, increased evapotranspiration promoted and the recharge of underlying groundwater supported (Eckart, McPhee & Bolisetti 2017). Bioretention areas also typically contain a storage potential, allowing water to be retained in the catchment from rainfall events and released slowly, reducing peak runoff rates and volumes, and helping to maintain groundwater and soil water during drier periods (Liao, Deng & Tan 2017). As a freespace infrastructure, the maximum potentials of bioretention areas are limited by available undeveloped land in the urban locale, but it can simultaneously act as an urban greenspace (Filazzola, Shrestha & MacIvor 2019).

Green roofs are vegetated areas constructed on building roofs. Adapting a traditionally impermeable surface to become more permeable, they reduce surface runoff volumes and rates, and instead promote the infiltration of water into their systems (Haowen et al. 2020). Compared to bioretention areas, the storage potentials associated with green roofs are often much smaller, decreasing peak flows in the system and increasing the time to peak for runoff, rather than decreasing water volumes draining from the catchment. Green roofs are often combined with other SuDS infrastructures, such as disconnected drainpipes or rainwater harvesting, but in this study they are modelled in isolation to examine their individual role.

Permeable paving is an alternative to traditional impermeable surfaces in the constructed urban environment, being designed to enable and promote infiltration. This water is then typically filtered (offering water quality benefits) before being stored and/or conveyed out of the catchment (Eckart, McPhee & Bolisetti 2017). In promoting infiltration, surface runoff rates and peak volumes are reduced, whilst storage potentials allow for deeper infiltration into the underlying soil profile, with benefits for groundwater and base flows/heights of local natural water bodies. Traditionally, these materials are used as permeable surfaces for pavements and driveways, but recent research and case study sites have identified their potential for a wider use, such as in low-duty, residential roads (see Weiss et al. 2017).

3.3.3: Model Design

The Storm Water Management Model (SWMM) is a dynamic rainfall-runoff model originally developed by the US Environmental Protection Agency in 1971. It uses continuous precipitation data to model runoff, primarily in urban locations (Haowen et al. 2020). Conceptually, the model visualises the drainage network as four systems – atmosphere, land, groundwater and transport (an existing, constructed drainage network) – with their own internal operations and potential interactions. Surface runoff is estimated for a subcatchment using a non-linear reservoir model (EPA 2016).

Table 3.2 indicates the main processes in the hydrological cycle, and the equations used by SWMM to represent these. It is important to note that the parameters in the evapotranspiration equation are unaffected by changes in vegetation, and so relative contributions of vegetation types and coverage extent to evapotranspiration losses will not be observed. Coefficient and parameter values used can be found in Appendices 1 and 2. In this study, only the atmospheric and land systems are represented - since the model is representing a new-build scenario, it is assumed there is no pre-existing grey drainage infrastructure, whilst groundwater interactions are not considered by the research.

Table 3.2: The equations used by SWMM to represent various processes of the hydrological cycle

Process	Equation
Evapotranspiration, *ET (mm/day)*	$ET = 0.0023\left(\frac{R_a}{\lambda}\right)T_r^{0.5}(T_a + 17.8)$ where R_a = extraterrestrial radiation ($MJm^{-2}d^{-1}$), T_r = maximum temperature (°C), T_a = daily mean temperature(°C), λ = latent heat of vaporisation ($MJkg^{-1}$)
Infiltration, *I (mm/hr)*	$I = k\frac{1 + \Psi(\Phi - \theta)}{F}$ where k = hydraulic conductivity (mm/hr), Ψ = suction head (m), $(\Phi - \theta)$ = proportional change in moisture content, F = cumulative infiltration (mm/hr)
Percolation, *P (mm/hr)*	$P = K(\theta)\left(1 + \frac{\Psi(\theta)}{D}\right)$ where K = hydraulic conductivity (mm/hr), Ψ = capillary tension (mm), D = depth of soil layer (m)
Runoff, *Q (m^3/s)*	$Q = B\ \frac{1}{n}\ S^{\frac{1}{2}}\ (y - y_d)^{\frac{5}{3}}$ where B = catchment breadth (m), n = Manning's roughness coefficient ($s/m^{\frac{1}{3}}$), S = catchment slope (%), y = surface water height above catchment surface (m), y_d = surface depression storage (mm)

SWMM has been used regularly in urban hydrology studies related to surface water runoff, dynamics and water quality impacts (see Fu et al. 2019; Hamouz & Muthanna 2019; Krebs et al. 2013). These studies have been independent of the original developers (United States' Environment Protection Agency) and spanned a range of scales, climates, geologies and urban extents. For example, Chow, Yusop & Toriman (2012) consider urban runoff quality and quantity under tropical climates, whilst Hamouz & Muthanna's (2019) analysis is focused on a cold climate, and Krebs et al. (2013) focus on high-resolution analysis in a boreal zone. Furthermore, the introduction of new modules within the model has led to the inclusion of SuDS developments in recent simulations (see Arjekani et al. 2020; Peng

& Stovin 2017; Rosa, Clausen & Dietz 2015). Similarly, Cipolla, Maglionico & Stojkov's (2016) study is concerned with a unit-scale analysing the long-term performance of a green roof feature, whilst Fu et al. (2019) apply the model district-wide in China's Yizhuang district.

Results from these studies frequently identify the strengths of the model and take a positive outlook on its performance. Krebs et al. (2013) highlight the good performance statistics offered, whilst Fu et al. (2019) note that even without observed runoff data, SWMM provides credible results for large-scale urban rainfall runoff simulation, and in comparison with measured data, shows low error values. In addition, in a comparison of urbanisation conditions, Jang et al. (2007) illustrated the potential of the model in successfully simulating urban runoff at varying scales and urbanisation conditions, and with the resultant varying discharge characteristics. When considering the LID-module, Gülbaz & Kazezyılmaz-Alhan (2017) investigated the bioretention system under different design parameters, and concluded that the performance of SWMM for such modelling was more than suitable for its purpose.

Furthermore, some studies have offered comparison with other similar modelling environments. Yazdi et al. (2019), for example, compared SWMM with the Hydrologic Simulation Program-Fortran (HSPF). They noted that whilst both performed adequately in simulating runoffs, SWMM proved more sensitive to imperviousness and offered slightly higher correlation coefficients during extreme events – two characteristics that are important for this study.

However, there are still elements of SWMM that have faced scrutiny and criticism in the literature. The LID-module has seen mixed feedback from studies, in-part due to its new and relatively undeveloped nature in comparison to the base model and other modules. Campisano, Catania & Modica (2017), for example, in an explicit evaluation of the rain barrel element within the LID-module, point to overestimation in systems smaller than $2m^3$ (which are typically those seen at the household level). Peng & Stovin (2017) question the ability of the model to predict evapotranspiration to a sufficiently accurate degree – a finding later supported by Zhang & Valeo (2022) who also identify the limited appreciation of soil layer dynamics in such modules - but when using adjusted evapotranspiration values, Peng & Stovin (2017) record a Nash-Sutcliffe coefficient of over 0.9 (suggesting high model accuracy). Nevertheless, with consideration of the identified weaknesses and comparison to other runoff models, SWMM was deemed to be an appropriate model for the aims of this study.

Each subcatchment is conceptualised by SWMM as a single surface, orientated as a sloping plain in the direction perpendicular to the flow. This direction is determined by the location of input and output nodes (EPA 2016). In this research therefore, a single subcatchment in SWMM is used to represent each modelled square sloping tile, which is then sub-divided into permeable and impermeable surfaces. Surface water (standing or as runoff) can infiltrate into the soil profile in only the permeable surfaces, with a rate described by the infiltration expression. All our simulations used the Modified Green-Ampt equation to model infiltration (I in Table 3.2), and was chosen as it offers a more nuanced control of moisture depletion in soil during low intensity rainfall, producing more realistic infiltration behaviour when modelling such events (EPA 2015). Furthermore, each subcatchment was considered to have uniform slope and soil conditions as these are underlying assumptions of the model. To represent variation in the study area, multiple subcatchments would need to be used.

SuDS infrastructure was modelled by increasing the proportion of permeable to impermeable surfaces in the scenarios featuring SuDS, and was represented using SWMM's separate LID module. Here, the SuDS infrastructure was treated as a third surface type, with infiltration rates defined using the same equation as the subcatchment, but separately defined soil conditions (see Appendix 1). Figure 3.2, below, illustrates the three conceptual diagrams provided by SWMM as to the inflows/outflows of the SuDS infrastructure used in our scenarios, and their units of structure within which parameters can be independently defined.

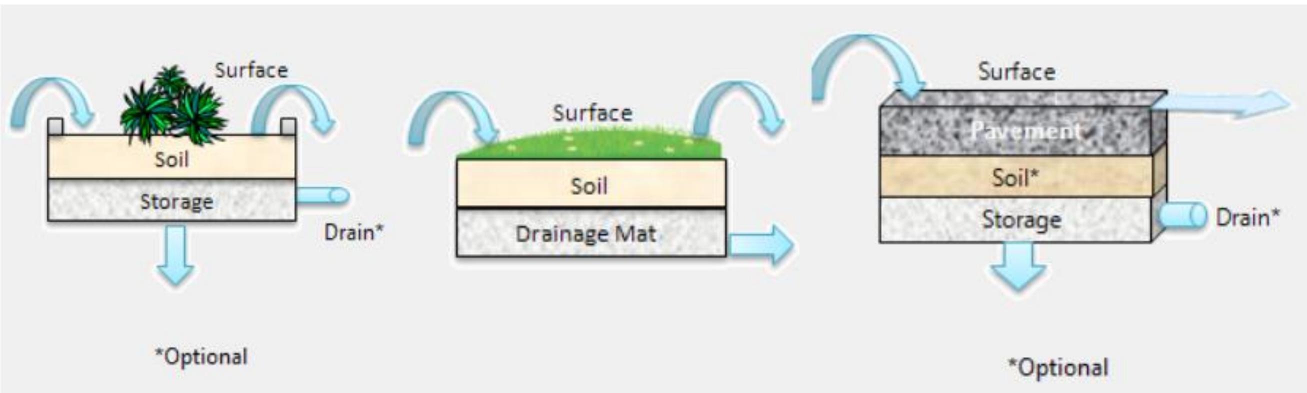

Figure 3.2: SWMM conceptualisations of flows and component layers of three SuDS infrastructures (left to right: bioretention, green roofs, permeable paving) (EPA 2015)

Several types of SuDS offer a storage component, such as bioretention and permeable roads and paving. In the model, outflow from this is treated as infiltration from the SuDS infrastructure into the underlying soil through the base of the infrastructure. When correctly designed, this infiltration rate will be lower than the infiltration into the surface of the LID, causing a build-up of water in the SuDS feature, which will continue to gradually infiltrate out after the rainfall event. Water may also be retained in the storage when soil moisture in the external environment is saturated. SWMM also has the option to add a piped drainage feature to this storage layer at a chosen height. This allows excess water to be drained from the feature and prevents a backing up and saturating of the SuDS infrastructure. The equation determining this rate is described as “outflow” in Table 3.1. This optional drainage, however, was not utilised in this research and so upon saturation rainfall was converted to runoff.

In this research, one hectare urban tiles were created to visualise a range of urban conditions in order to address the identified questions. Three distinctive housing types were chosen and their minimum footprints identified from the national Technical Housing Standards (2015) – detached (74 m^2), terraced (51 m^2) and apartments (68 m^2). These were then each used to create three density scenarios with a homogenous housing type at 20-, 30- and 40- residences per hectare. Apartments were designed two-per-floor with a maximum of three floors – whilst floor numbers can far exceed this, high values are typically seen in city centres whilst suburban locations see lower values (Bibri, Krogstie & Karrholm

2020). Road networks were added to connect the houses, with figures for minimum widths and component spacing drawn from the UK Manual for Streets (DfT 2007). To reduce runoff impacts from settlement layout, a uniform design approach was then applied. That is, a main central road was identified through the tile, and additional side roads added individually only when required. For simplification and due to the great variety in design and need for such features from neighbourhood to neighbourhood, other elements of neighbourhood design such as garages, external car parking and sheds were not considered. The resulting nine tiles can be seen in Figure 3.3, and land use footprints are listed in Table 3.3.

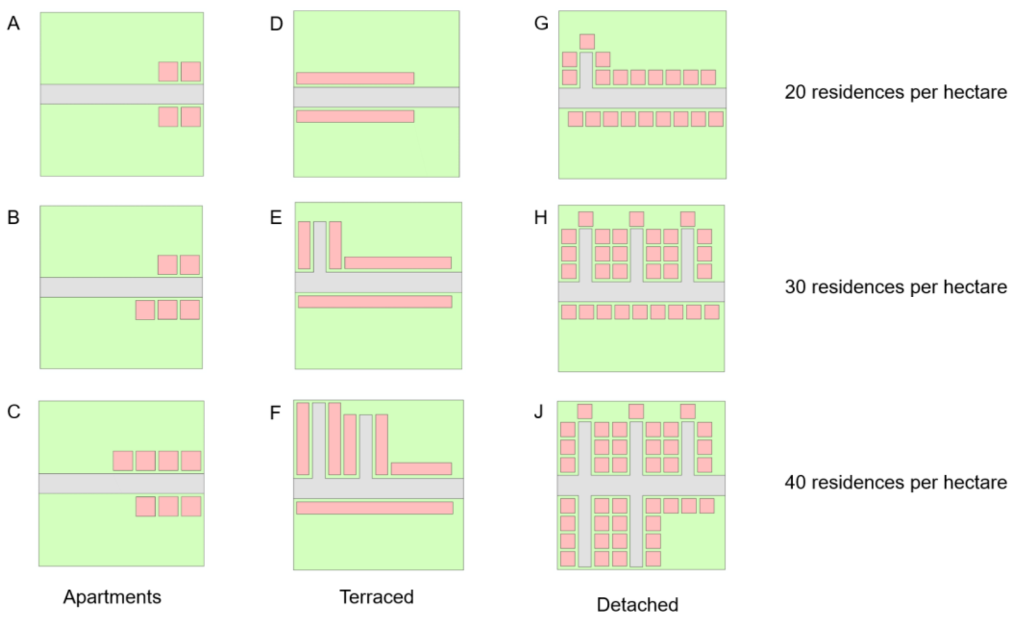

Figure 3.3: The nine 1-hectare scenarios illustrating varied housing type and development density. (When slope is present, each scenario slopes from top to bottom.)

Table 3.3: Areas of each land use in the tile scenarios (m^2)

Scenario	Building	Road/Pavement	Greenspace
A	544.0	1200.0	8256.0
B	680.0	1200.0	8120.0
C	952.0	1200.0	7848.0
D	1004.2	1200.0	7795.8
E	1506.3	1427.0	7066.7
F	2008.4	1814.0	6177.6
G	1480.0	1359.0	7161.0
H	2220.0	1916.0	5864.0
J	2960.0	2553.0	4487.0

Simulations were undertaken using SWMM for each of the tiles, under a variety of different urban environmental conditions. These were achieved by independently varying the following parameters: antecedent soil moisture, soil type, tile slope. These housing designs and urban environmental conditions together created different scenarios. Undeveloped land was assumed to be short grass in all scenarios to allow comparability, but this should be varied in future work as vegetation types influence interception, evapotranspiration and infiltration. Given this assumption about uniform vegetation, it was reasonable to use constant evapotranspiration rates and groundwater conditions to assist comparison.

SuDS infrastructures (bioretention, green roofs, and permeable paving) were also modelled within each of these designs, and the area covered by this SuDS type varied. That is, the potential space for each type of SuDS was calculated in each scenario (total undeveloped space for bioretention, total roof area for green roofs, total road area for permeable roads and paving), and then a simulation undertaken with the SuDS constructed on between 0-100% of this area at 10% intervals. Each infrastructure was considered independently from the others, with no scenario involving more than one type of SuDS.

All urban designs, with and without SuDS implementations, were independently simulated under three rainfall events to simulate the response to different sized storm events. Distribution of the rainfall during the event was considered to be uniform, both temporally and spatially, and were representations of the present not potential future magnitudes as a result of climate change. Return periods up to a 1-in-10 year event were chosen as this is typically the magnitude of event used when designing greywater systems to prevent system overbuild. Greater return periods, such as 1-in-50 year or 1-in-100 year, represent more extreme rainstorm events when urban flooding is likely, and thus whilst they should be considered when modelling severe urban flood events, such magnitudes were not considered necessary within the remit of our study.

Rainfall magnitudes of 13.6, 17.3 and 20.2 mm/hr were used to represent a 1-in-2, 1-in-5 and 1-in-10 year event respectively. These values were obtained for the Oxfordshire region from the Runoff Calculator (CivilWeb 2017) which uses the Modified Rational Method for preset rainfall data for 250 urban locations across the UK. This approach estimates rainfall intensities for a given return period from the M5-60 (expected rainfall volume for a given location) and r Ratio (type of expected rainfall) values. A 2-hour storm duration was used to reflect a duration at the high end of the modal duration for these events.

3.4: Results

The resultant hydrograph responses of the different scenarios were analysed to ascertain the impact of different soils and topographies, housing design and SuDS design in the tile.

3.4.1: Urban Hydrology

Irrespective of building or SuDS design, alterations to the urban hydrology illustrated the same patterns across the scenarios. Lower antecedent moisture conditions saw an increased time to peak and reduced peak runoff rate as soils had a greater capacity to infiltrate and store more water before saturation occurred. Post-saturation, additional water to the system became surface runoff, which was then directly discharged. An increase in slope led to a reduction in the time to peak and an increase in peak runoff, as water on the surface is transported downslope faster by gravity, reducing the opportunities for infiltration and evapotranspiration, and thus decreasing overall transport times through the urban tile. Different soil types result in different runoff rates, too, due to their varied textures and consequent hydrologic properties. The sandy soil type saw the lowest peak runoff and longest time to peak, which is likely due to its increased pore volume, boosting permeability, whilst the clay soil saw the opposite.

3.4.2: Housing Design

Both housing density and housing type present impacts on the hydrograph response to the design rainfall events. With increased housing density, we see an increase in both total and peak runoff, as illustrated in Figure 3.4. This occurs across all three housing types, albeit to varying magnitudes, with apartments seeing the smallest difference and detached housing the greatest, and is a result of increased surface sealing. That is, a greater volume of housing in the tile (giving it the greater density) means a greater proportion of the area has an impermeable surface, resulting in less infiltration and greater surface runoff. The variation between housing types comes from the different impermeable surface areas for the housing types under the different densities.

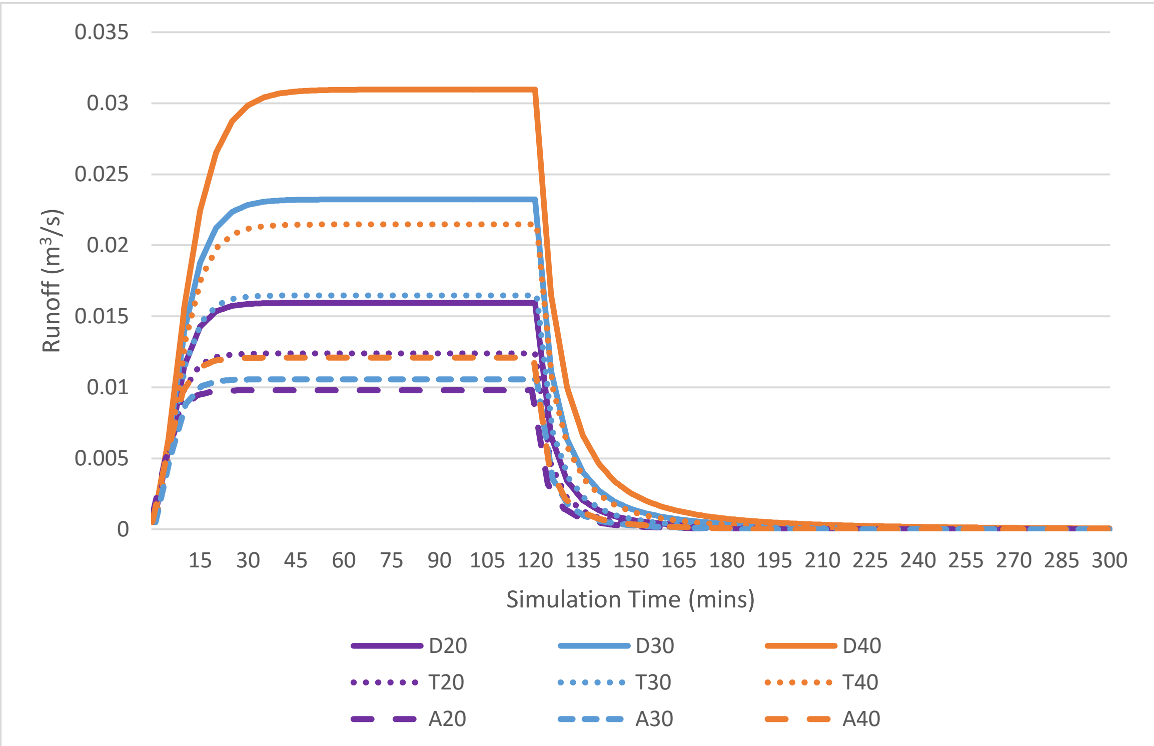

Figure 3.4: Influence of residence type (detached - D, terraced - T, apartments - A) and residence density (20-, 30-, 40-houses per hectare) on the hydrograph of a one hectare site without SuDS during a 2-hour duration, 1-in-10 year rainfall event

Nevertheless, the hydrographs for all nine scenarios without SuDS interventions show a similar shape, as seen in Figure 3.5. With limited greenspace to promote infiltration and evapotranspiration into subsurface

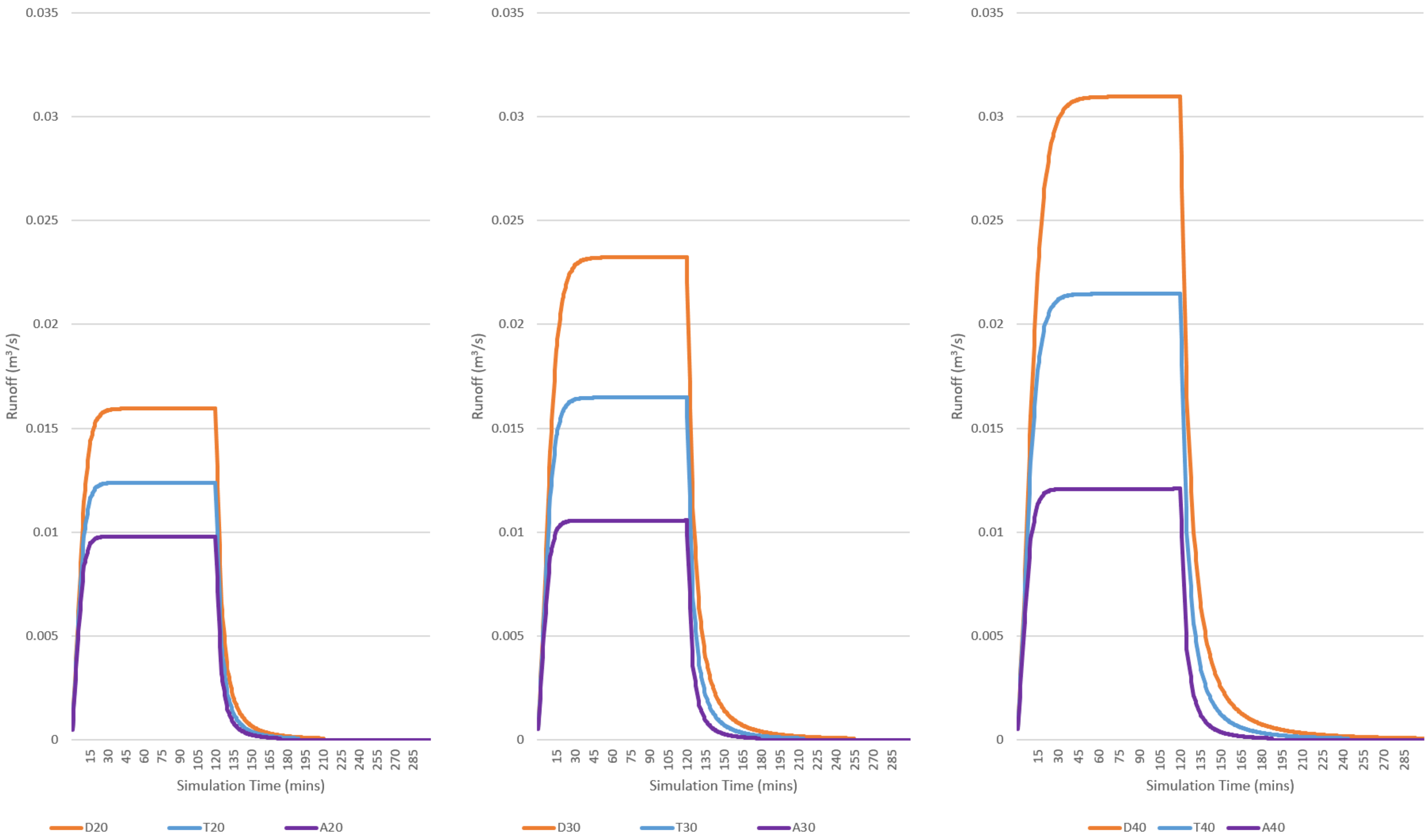

Figure 3.5: Influence of the housing type (apartments – A, detached – D, terraced – T) on the runoff hydrograph of a 1-hectare tile under a 2-hour duration, 1-in-10 year rainfall event, under 3 density scenarios (20-, 30-, 40-houses-per-hectare)

soils (and thus slowing the movement of runoff through the urban tile), and without infrastructure to store runoff, all scenarios see runoff occurring from the beginning of the storm event, with a greater increase in runoff rates in those with less greenspace (e.g. denser scenarios and detached housing typologies). After a period of time, a plateau is reached, illustrating a continuous peak runoff rate and indicating that the ground has become saturated. As previously identified, the magnitude of this is related to the housing type. Finally, following the end of the storm event, runoff rates then decline rapidly, before returning to pre-event levels, as no additional water is being added to the system.

A higher density of certain housing types can show lower total and peak runoff than lower density scenarios of other housing types. An example would be the terraced houses at 40 houses per hectare, which offers a lower peak runoff (0.0215 m^3/s) than 30 houses per hectare in the detached form (0.0232 m^3/s). This is partly a result of the reduced footprint of a terraced house compared to a detached house, meaning less surface sealing occurs per terraced house than per detached house. It is also, in part, due to the additional infrastructure required in the scenarios. In order to be defined as detached houses, space is required between dwellings, meaning that properties are more dispersed across the tile and so require greater road surface to connect all the houses than in a terraced setting. This supports the findings of Jia et al. (2019) who identified changes in runoff dynamics due to varying spatial layouts of urban neighbourhoods, attributed to changing proportional requirements of road, greenspace and other infrastructure.

The apartment scenarios show a much lower peak runoff than the other two housing types. Apartments have the greatest floor area per storey. However, these by their nature have multiple dwellings vertically as well as horizontally, reducing their overall footprint for the same number of residences. Regardless of storeys in a terraced or detached house, they are still considered a single dwelling.

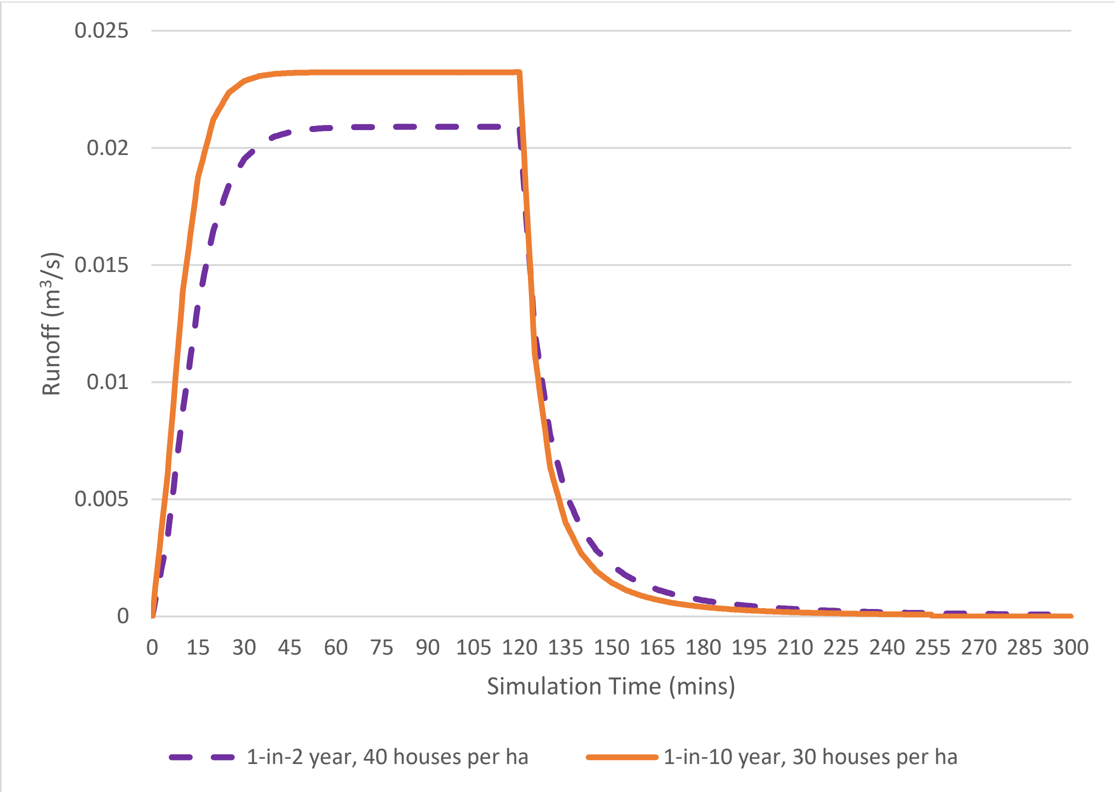

Figure 3.6: Influence of rainfall intensity and housing density on the hydrograph of a 1-hecatre tile of detached housing under a 2-hour duration rainfall event

These patterns are also seen across all three design storm sizes, with peak runoffs increasing and time to peaks decreasing with an increased storm magnitude. This is to be expected from the increased rainfall intensities, and consequent volumes of water, that must be managed. However, some of the denser housing scenarios show less peak runoff in more frequent storm events than less dense scenarios under less frequent (i.e. higher intensity) events. For example, the 40-houses per hectare density in a 1-in-2 year event has a lower peak runoff rate (0.0209 m^3/s) than the 30-houses per hectare in a 1-in-10 year event (0.0232 m^3/s), as shown in Figure 3.6. This emphasises the importance in appreciating a range of storm return periods for a development, and then balancing frequency with magnitude when creating a planning design approach.

3.4.3: SuDS Design

The implementation of SuDS can offer its own influences to the scenario hydrographs, regardless of the other features. Figure 3.7 represents three urban tiles, and illustrates the impact of the different SuDS infrastructures in each. Compared to the baseline (non-SuDS) scenario, all SuDS implementations see a reduction in the peak runoff rate and an increase in the time to peak. Post-rainfall, there are typically greater runoff rates in the SuDS scenarios than the baseline, due to the slowed rate at which the water passes through the tile with increased infiltration, percolation and throughflow processes. This slowed rate and consequent low runoff periods, however, have important biodiversity and soil/groundwater recharge properties (Martin-Mikkle et al. 2015). Green roofs are the exception to this, not offering a reduction in the peak runoff rate, but an increased time to peak and a longer low flow period post-event can still be seen.

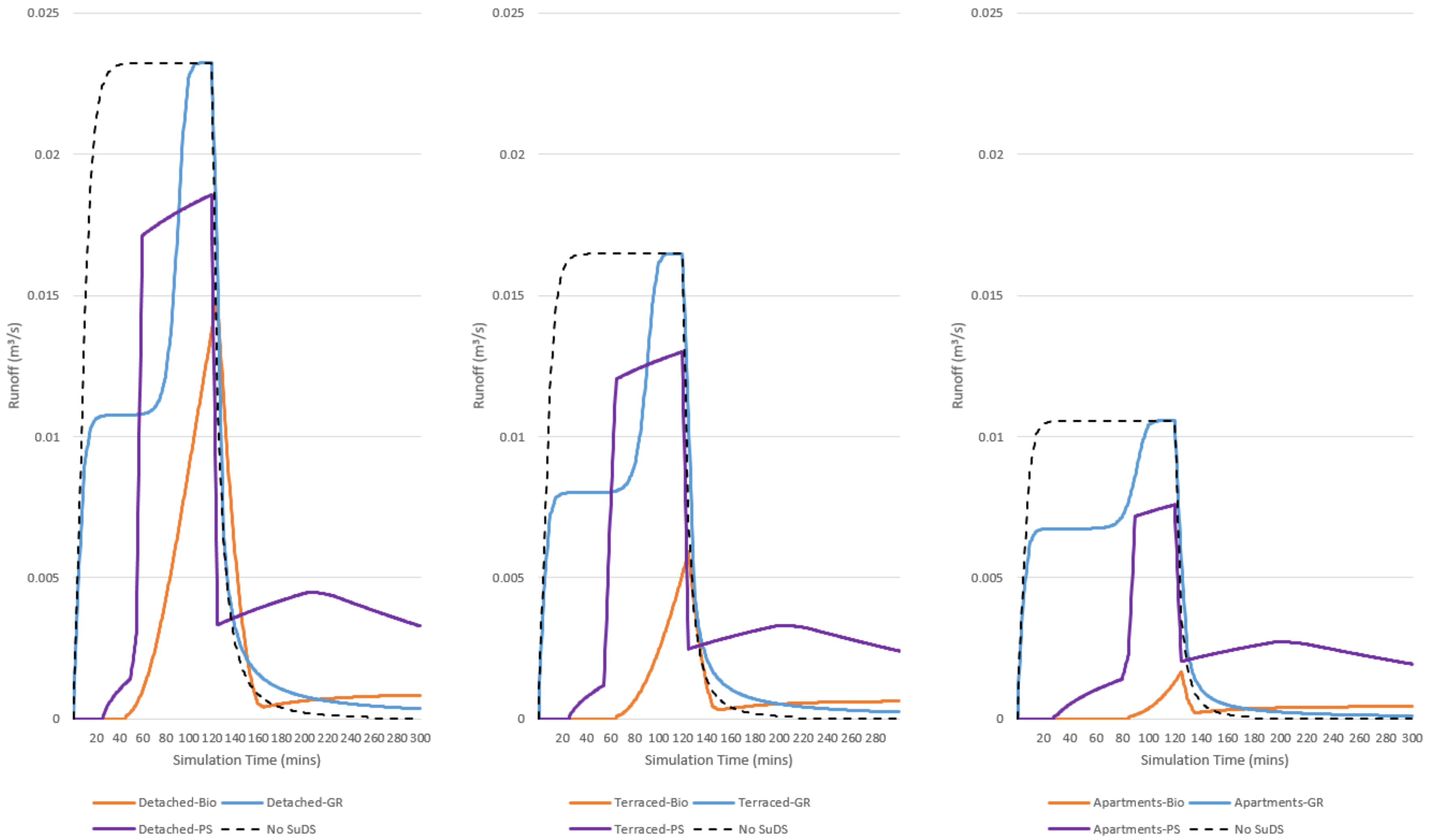

Figure 3.7: Influence of different SuDS infrastructure (bioretention, green roofs, permeable paving) at 50% implementation on the hydrograph of a 1-hectare tile under a 2-hour duration 1-in-10 year rainfall event, dependent on housing type

The specific type of SuDS deployed also shows a notable difference in the overall hydrograph shape, both in terms of peak runoff and time to peak, as well as the runoff rates at other points during the event. This variation acts as an indication of the effects of different storage, drainage and infiltration properties of the different SuDS infrastructures. In all of the scenarios, however, the peak runoff rate is achieved at 120 minutes, which marks the end of the rainfall event. As with the baseline scenarios, the green roofs see a plateau at this peak rate due to saturation of the tile. This response is due to the design of the rainfall events in the simulation, which see rainfall distribution across the 2-hour duration of the storm as uniform. This rainfall then ceases abruptly after the 120th minute of simulation, and without additional water being added to the system, runoff volumes decrease.

Bioretention responses see a simple curve, much as with a basic hydrograph, featuring a rising limb to the peak runoff and then a falling limb. After this sharp fall in the runoff rates, there continues to be a low runoff from the tile, with a much smaller second peak, as runoff slowed by the LID continues to drain from the tile post-event. As the implementation of bioretention is increased, as seen in Figure 3.8, the peak runoff rate is reduced and the time to peak increased – a response seen across the housing types and densities. This is because bioretention promotes infiltration, and so with more bioretention area, more water can be infiltrated, reducing surface water runoff. In the low extent implementations, a peak runoff plateau is created, much as with the baseline scenario, and the peak runoff rate (caused by saturated ground and SuDS infrastructure, and the consequent surface runoff) persists until the rainfall event finishes.

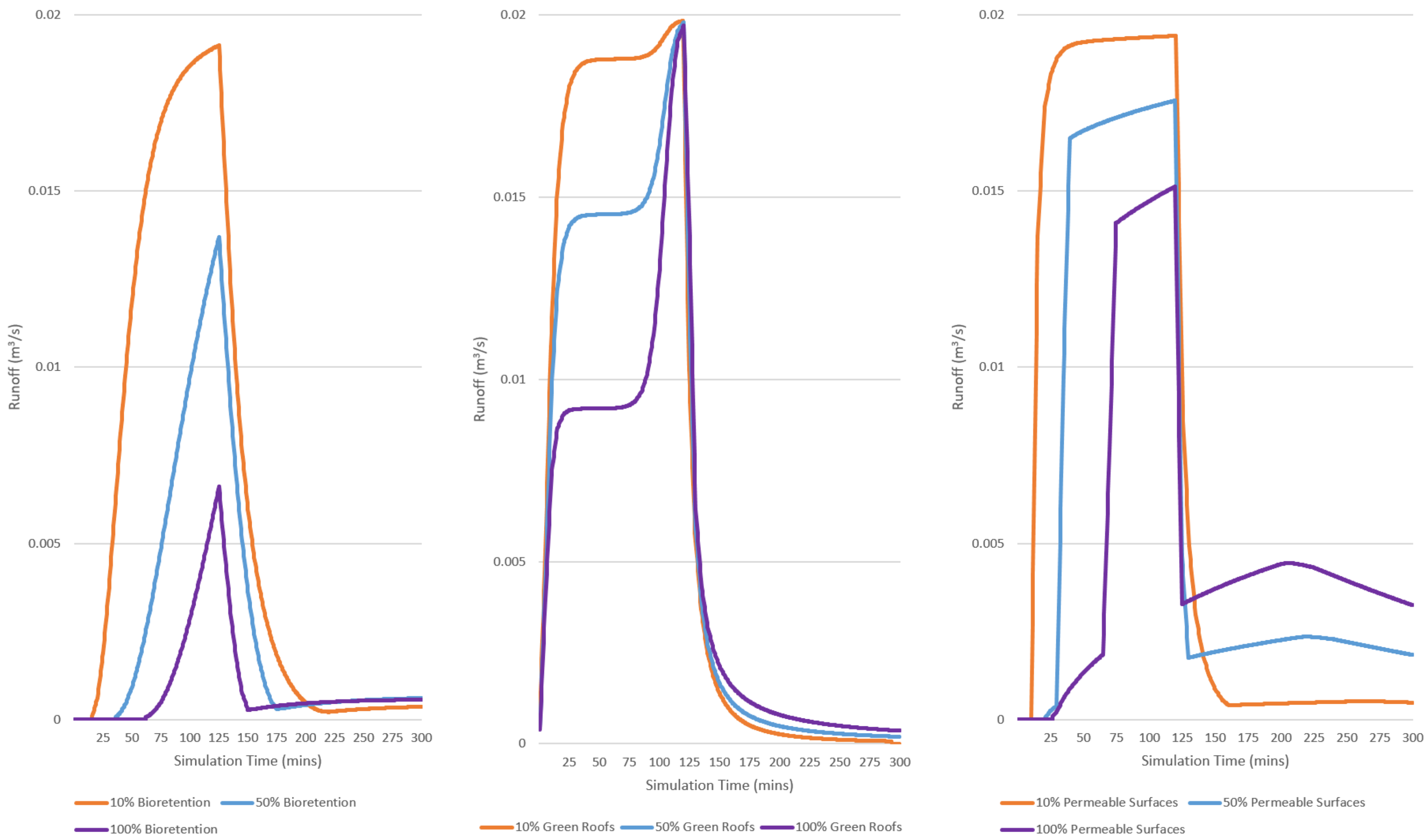

Figure 3.8: Influence on the runoff hydrograph of the extent coverage of a SuDS infrastructure (i.e. the proportion of maximum potential area covered) on the runoff hydrograph of a 1-in-10 year, 2-hour duration rainfall event on a 1-hectare tile at 30-detached-houses-per-hectare

In the green roofs response, there is an initial rise in runoff rates before a plateau is reached. This is as not all rain in the scenario will land on green roofs (and is not channeled from non-green roofs to the green roofs), and so this initial rate represents runoff from the non-green roof areas. From the first plateau, this is followed by a second increase in runoff rate to a second plateau. As the green roof LIDs become saturated, runoff is generated from these too, which causes the second increase in runoff rates, before levelling out at the second plateau – the peak runoff rate which represents saturated conditions and surface runoff.

Permeable paving also sees a two-stage increase in runoff rates, with a dramatic increase between the first and the second. This reflects the different layers of the permeable paving system, and is sensitive to their relative permeability which leads to saturation occurring at different rates. Then, following the rainfall event, runoff rates decrease sharply, followed by a secondary peak in the runoff. Whilst peak runoff rates decrease and time to peaks increase with increased permeable surface paving, this secondary peak increases and occurs sooner.

3.4.4: SuDS & Housing Design

When we consider the influence of housing type and density, bioretention consistently offers the greatest peak runoff reduction, followed by permeable paving and green roofs. Between the apartment scenarios, bioretention in particular offers a noticeable difference between its peak runoff reduction and those of the other SuDS. This is a result of the significantly larger area bioretention can occupy in the urban tile, as the roof and road areas (the base areas for green roofs and permeable surfaces) are much smaller than the undeveloped land area (the base area for bioretention). However, bioretention also makes a noticeable peak runoff reduction in terraced scenarios and, to a much lesser extent, in the detached scenarios, also as a result of the relative areas of “undeveloped” land to roads/buildings. It is important to note, however, that in reality there will be additional demands for use of this “undeveloped” area.

Similarly, as the area of bioretention (at a given extent implementation) is decreased as housing density increases (see Figure 3.8), peak runoff rates are increased and time to peaks reduced due to the relative decrease in available SuDS infrastructure for infiltration, percolation and storage. Less water is also lost through evapotranspiration and interception by the vegetation. The same can be seen in the first plateau for the green roof scenarios. However, the second plateau remains unaffected, because it represents a saturated environment, with the roofs generating surface runoff, and so the same total runoff rate will be reached as this is dependent on the magnitude of the rainfall event.

Additionally, with permeable paving, we also see differences in the times to peak between the different housing densities. Whilst across the permeable paving scenarios runoff rates begin to increase around the 20-minute mark (the point at which non-SuDS scenarios begin their peak runoff plateau), the start of the peak runoff plateau in all densities and housing types occurs between 55 and 96 minutes later when permeable paving is present. This is due to the greater proportion of permeable surfaces in the scenario (whether natural or SuDS) that encourage infiltration (and resultant

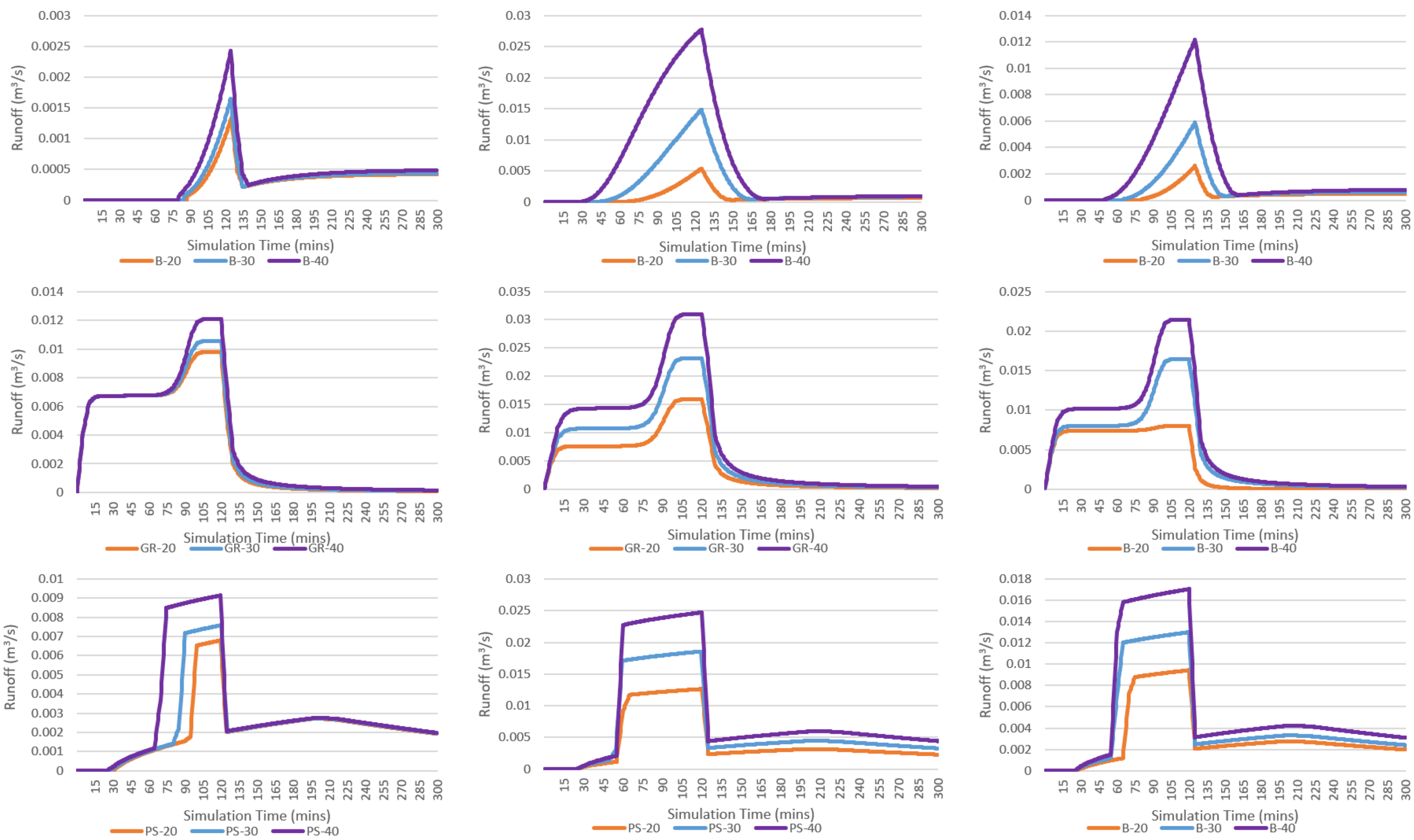

Figure 3.9: The impact of housing density (20-,30-,40-houses-per-hectare) and type (left to right: apartments, detached, terraced) on the hydrographs of a 1-hectare tile under a 2-hour duration, 1-in-10 year storm event with the maximum potential implementation of different SuDS infrastructure (top to bottom: bioretention, green roofs, permeable surfaces) (NB: scales on the vertical axes are not consistent throughout)

percolation), throughflow and groundwater recharge. These processes slow the transport of water through the tile, increasing the time taken for significant volumes of runoff from the tile to occur.

The two-stage response of green roofs and permeable surfaces raises a point of note in some scenarios. In both the terraced and detached housing types, the permeable surfaces reach their second peak plateau before that of the green roofs, which could create challenges on the ground if used in combination with green roof runoff, exacerbating surface-level runoff conditions. Undesired build-up or runoff of surface water could occur at an earlier timeframe in the permeable surface scenario than the green roofs, despite the absolute peak runoff rate being reduced to a greater extent in the former. This goes to reinforce the findings of Sörensen et al. (2016), who argue that the dynamics of a catchment response need to be considered when designing flood management responses, not just overall figures of peak and total runoffs from large-scale events.

Equally, we find that in some scenario designs, a lower implementation of one SuDS has a greater impact on reducing peak runoff than a higher implementation of another. This can be seen, for example, with the 50% bioretention extent, which boasts a lower peak runoff than the 100% extent of both permeable surfaces and green roofs in the same scenario. This can have important implications for the design and planning of green infrastructure interventions, where less can actually do more.

3.5: Conclusion

The management of stormwater in cities is becoming increasingly important as dual pressures from urbanisation and climate change look to exacerbate existing drainage infrastructure limitations (Yazdanfar & Sharma 2015). Whilst several significant stormwater flooding events worldwide have raised formal calls for increased appreciation of surface water management in urban spaces, such as the UK's Pitt Review (2008), there has been little research into how features and design of the urban environment impact upon the potentials for, and responses of, SuDS elements. Using an urban-tile approach, different urban designs were simulated using SWMM to identify the impacts of different features on the hydrograph response. These included changes to urban design, housing design and the design of the SuDS.

It was found that runoff from the tiles was affected by both the type and the extent of implementation of SuDS elements, which were in turn influenced by the type of development and its density. That is to say that in denser housing scenarios, infrastructure-based SuDS (i.e. on roads and roofs) delivered a greater reduction in peak runoff rates and increased extension of the time to peak, as the maximum areal extent for the implementation was increased. It is also the case that the inverse is true for freespace SuDS (in this research, bioretention areas), which see a decreased maximum potential extent with increased housing density.

All three SuDS elements delivered an increase in the time to peak compared to a baseline (non-SuDS) scenario, and this increased with increased extent of implementation. Bioretention and permeable paving also led to an overall reduction in the peak runoff rate but, due to saturation, this was not achieved with the green roofs intervention. However, no overflow drain was considered in any of our infrastructure, and so saturation occurred faster and post-saturation flows became surface runoff. With the implementation of an overflow drain, some water could be channelled away, delaying the saturation of the area and reducing surface runoff volumes.

Bioretention consistently offered the greatest reduction in peak runoff for a given scenario, due to the greater surface areas that could be covered, with the greatest magnitudes in peak runoff reduction seen in the terraced and detached scenarios as the higher housing densities resulted in a larger change in the permeable-impermeable surface ratio. In reality, however, it is important to note that greenspace is required for other purposes (e.g. allotments/recreation) as well as other development (e.g. car parking) and so such areas would not be attainable. Nevertheless, with the type and footprint area of both houses and SuDS elements influencing the shape and magnitude of the response to rainfall events, it is clear that how we design and build our urban environments is as important a consideration as what we build when we consider the influence on urban hydrology.

Whilst up to a 100% implementation of a SuDS infrastructure was modelled in the study, it is not realistic to make such an assumption for the uptake in reality. Realistic potentials would vary on a case-by-case basis for both the type of SuDS and the catchment itself, and therefore are development-specific. For good practice, different SuDS elements should be combined in management trains, so future modelling studies should look to understand how multiple SuDS may interact under differing urban conditions. The investigation of post-event dynamics is important too, as illustrated in this work, and should therefore also feature as an important part of drainage system modelling.

Furthermore, as has already been illustrated through a vast range of work (see Fletcher, Andrieu & Hamel 2013; Liao, Deng & Tan 2017; Weiss et al. 2017), the design of specific SuDS elements has a significant impact on their ability to infiltrate, evapotranspirate, store and filter rainfall events. Whilst figures for this research were drawn from recommendations and best practice guidelines, such as CIRIA (2015) and Department for Transport (2007), altering element design will have its own impacts on catchment response that will be important to consider in the drainage network design. There were also limitations in the representation of contributions to evapotranspiration by vegetation. SWMM has the potential to incorporate user-provided values, so future work could employ this approach (using an alternative evapotranspiration equation or experiment-provided data) to better appreciate these losses.

There is also the need for future work to consider greater storm sizes. A range of design storms were chosen in this study up to the 1-in-10 year magnitude, since this is typically the size used to design greywater systems in order to avoid huge infrastructure dimensions and potential system overbuild. However, an appreciation of how a system may react to larger events is important for additional response and planning considerations (Sörensen et al. 2016).

Developing this approach further, there is a need to identify what proportion of the runoff is surface, and what is subsurface, as this divide will have important consequences for flood management. From this, more detailed spatial analysis could also help identify whether the surface runoff is uniform across the catchment, or whether particular design approaches cause concentrated areas of surface water flooding.

4. The Potential for Sustainable Drainage Systems (SuDS) in a Regional Urbanisation Project

4.1: Abstract

Large-scale urban development is required to support and sustain growing urban populations, which are expected to reach 5 billion by 2030. At the same time, city planners are facing the pressures of climatic changes, which forecast more intense rainfall events, further exacerbating the existing challenge of surface water flooding in urban locations. Sustainable drainage systems (SuDS) are one proposed solution to help alleviate such problems, yet much still remains to be known about their operation, performance and potential benefit provision beyond the neighbourhood scale, or within a mixed-form development. Using a case study of the Cambridge to Oxford Arc (a region of England earmarked for extensive urbanisation), development patterns of different extents and spatial layout were modelled and the required pipe lengths to connect such developments estimated. The Stormwater Management Model (SWMM) was then used to simulate surface water runoff conditions in these developments during a 1-in-10-year rainfall event, and minimum pipe diameters calculated based on commercially-available sizes to identify reductions SuDS could bring in hybrid (green and grey drainage) systems. Whilst denser scenarios typically led to greater peak runoff rates and total runoff volumes, this was not always the case under some SuDS designs as the denser scenarios provided the opportunity for more infrastructure-based SuDS provision. The proportion of different surface cover types (permeable and impermeable areas and different types of SuDS provision) had a strong influence on runoff volumes and rates, and since the different housing typologies offered different proportions under each development scenario,

there was no single typology that showed the lowest or highest runoff volume across all scenarios. The findings of this study highlight the importance in a planning context of considering multiple housing and SuDS typologies and their footprints to maximise the potential of the development design in achieving the development's goals.

4.2: Introduction

Urban planners have always faced the challenge of meeting multiple objectives in development plans - a challenge which is being intensified by pressures for urbanisation, sustainability, and climate resilience (Xu et al. 2020). Global urban populations are expected to reach 5 billion by 2030, and to support such growth under current densities and designs, the total urban area must be triple that of 2000 (Felappi et al. 2020). The dilemma of balancing the need for residences and non-domestic buildings whilst minimising negative environmental and social impacts is reflected in global discussions on suitable urban form designs (McPhearson et al. 2016).

The compact city, referred to as 'smart growth' in the North American context, promotes high density, the use of brownfield sites, and infill development, and can support some elements of contemporary urban movements, such as walkable cities (Artmann et al. 2019). However, through maximising the use of space for urban developments, compact cities often see a loss of existing urban greenspace and limited green elements in the finished development (Bibri, Krogstie & Karrholm 2020). The resultant increased soil sealing leads to decreased infiltration of rainwater and increased surface water runoff (which has implications for surface water flooding), as well as ecological and social consequence of greenspace and habitat loss (Boulton et al., 2020). It is anticipated these runoff dynamics will also be exacerbated in the future as climatic change leads to alterations in rainfall intensities and duration, with a general tendency towards more convective downpours in a warmer climate (Lee et al. 2018).

Conversely, the green city approach prioritises urban greenspace and, more recently, the connectivity of these in ecosystem corridors, which typically leads to less dense settlements and a greater urban footprint (Artmann, Inostroza & Fan 2019). Thus, whilst offering benefits of urban greenspace within the city, it can be seen to have a greater sprawl than the compact alternative, infringing on greenbelt and other previously undeveloped areas (Echenique et al. 2012). Concerns have been raised over this, however, as areas affected by sprawl are typically large-scale greenspaces. Whilst the introduction of urban greenspace can help reduce any net greenspace loss, they are typically of a smaller area and cannot provide some of the ecosystem services offered by well-established, large-scale greenspace (Algador et al. 2012).

It is widely acknowledged that neither compact nor green cities offer the perfect solution for sustainable development (Echenique et al. 2012; Mouratidis 2019; Boulton et al. 2020). Recent dialogues in planning, therefore, have looked to find a compromise between these two approaches, identifying how elements of the green city can be introduced to a compact form. For example, rather than being viewed as two contrasting and opposing approaches, Artmann, Inostroza & Fan (2019) argue that green and compact city movements can complement one another, and through careful balance can be fused to create a stronger approach to urban development – the smart-compact-green city. They highlight that space-efficient urban forms and green infrastructure are not mutually exclusive, with technologies such as green walls and green roofs being key examples of how such a fusion could work and offering assistance in tackling other problems such as surface water flooding. Nevertheless, as Algador et al. (2012) highlight, small-scale greenspaces alone are insufficient, and thus optimising the design and location of cities and their greenspace elements is integral for achieving a range of ecosystem benefits at different scales (Davies et al. 2015).

In order to minimise the expansion of urban footprints, multi-functionality has widely been recognised as an important factor, with the more services provided by a given infrastructure, the fewer additional infrastructures required to provide the same range of services (Hansen et al. 2019). In such a vein, sustainable drainage systems (SuDS) are regularly considered as a good example of infrastructure that can provide essential functions (e.g. the removal of excess stormwater) as well as urban greenspace, which provides recreation, ecological and aesthetic benefits, whilst also integrating with active travel (walking and cycling) routes (see Jose, Wade & Jefferies 2015; Fenner 2017; Hunter et al. 2019). However, whilst there have been many studies that have looked to identify and quantify the co-benefits (such as greenspace provision) that these infrastructures provide, much of the focus has been on individual infrastructures and/or in a theoretical context (e.g. Alves et al. 2019). Whilst these offer beneficial insights into the range and extent of benefits that could be provided by a given infrastructure, little consideration is paid to the impacts one may have on the operation of another in a scheme that utilises multiple infrastructure types, or how context-specific conditions may affect their operation. Zuniga-Teran et al. (2020) argue that these are two considerations fundamental to effective design and implementation of green infrastructure, with Haase et al. (2012) offering critical insight into how important an understanding of these synergies and trade-offs can be. Furthermore, where case studies have been employed, these often focus on small-scale developments, such as a single neighbourhood, whereas there is also a need to better understand the wider interactions and cumulative impacts for larger scale developments (McPhearson et al. 2016).

To explore these tensions, this study compares a range of urban designs that each utilise multiple SuDS infrastructures, and identifies the different implications these may have on runoff characteristics (and hence surface water flood generation) in a regional-scale development. In doing so, the benefits that large-scale SuDS implementation can bring in an area of new-build development are quantified, in order to address the following questions:

- how do different proposed densities and spatial development patterns affect the potential areas and performance of different SuDS interventions?
- how does the variation of regional characteristics (such as slope and underlying soil conditions) affect the relative performance and benefits of different SuDS infrastructure?
- what implications does this have for planners in designing regional-scale developments involving SuDS?

4.3: Methodology

The multi-scale methodology involves (1) identification of possible large-scale patterns of urban development, given targets for total housing provision, using an urban development model (OpenUDM); (2) use of ‘urban tiles’ to represent how such development could look at the street-scale, whilst achieving target housing density values. These tile designs also included a range of sustainable drainage infrastructure (SuDS) interventions, tailored to the configuration of building development. (3) Rainfall-runoff modelling was then undertaken on the subsequent development design to assess surface water under the different development, urban design and SuDS provision scenarios.

4.3.1: Case Study Location

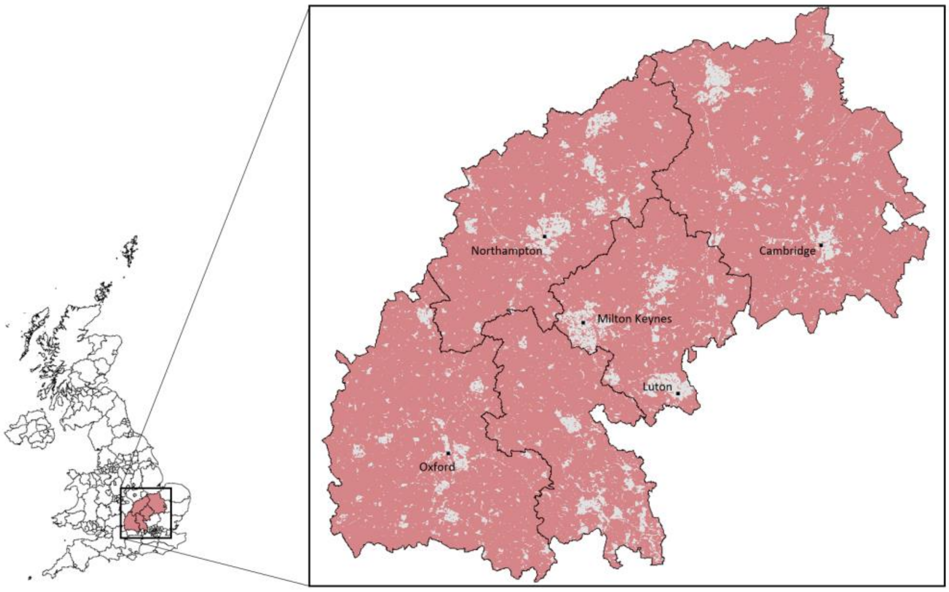

Figure 4.1: An outline map of the case study location showing existing urban development (grey)

The Cambridge to Oxford corridor, located in south-east England (see Figure 4.1), is an area to the northwest of London encompassing the existing cities of Cambridge, Milton Keynes and Oxford, and covering five counties (Berkshire, Buckinghamshire, Cambridgeshire, Northamptonshire, and Oxfordshire). It is recognised as an area of great economic potential, but these potentials are said to be facing constraints from existing poor infrastructure in the region, both in terms of transportation and housing (Infrastructure Transitions Research Consortium 2020). Development of the region is therefore proposed, with the goal to maximise both the social and economic potential whilst exemplifying and promoting sustainable development (National Infrastructure Commission 2019).

One of the fundamental findings of a regional report (NIC 2019) is the lack of suitable, affordable and sufficient housing, which is seen as a fundamental crux on which the success of the region relies. To counteract this and maximise economic potential, it is estimated 1 million new dwellings will be required by 2050, doubling current rates of development (NIC 2019). Research into where these new homes should be located, and in what form, is still ongoing, granting the opportunity for sustainability to become a key cornerstone for consideration in the development. This vision for sustainable development is also supported by the government's 25 Year Environment Plan, which champions such actions, promoting the regaining and retention of good environmental health and investment in a future that benefits both the environment and the economy (DEFRA 2018). Within the study area, these goals also align with those of localised plans and movements, such as the Oxfordshire Plan 2050 (Oxfordshire County Council 2019).

4.3.2: Urban Development Model

The urban development model utilised (OpenUDM) is a spatial optimization tool for the creation of high resolution scenarios of heterogenous urbanization, subject to spatial attractors and constraints. OpenUDM combines multi-criteria evaluation and cellular automata approaches, with the former assessing an area's suitability for development and the latter simulating conversion to urban land use based upon this (Ford et al. 2019). At a 100m grid scale, key features of the existing natural and built environment are identified, alongside the factors which will attract development and those which will constrain it. These include proximity to transport networks, the location of existing settlements, and sites protected for their historic or ecological importance. Housing density criteria are also calculated to represent different scenarios of the development that could occur, based upon proximity to urban centres and transport hubs (with higher densities close to these features). Sites for new development are then identified using target housing densities of the future scenarios (Mok et al. 2020). Outputs from the UDM represent the dominant type of development in each pixel for each scenario as 0 (no development), 1 (existing urban form) or 2 (new urban development), as well as quantifying dwelling densities for developed pixels.

Eight different future development scenarios for the Cambridge to Oxford corridor were simulated under UDM, and the outputs from these formed the basis of this study. The scenarios represented a rate of growth in the area of either 23,000 (23) or 30,000 (30) dwellings per year under a “green” (G) or “grey” (Y) set of development restrictions and following a new settlement (N) or existing settlement expansion (E) pattern. Specific development scenarios are hereafter referred to by a three-part abbreviation to indicate these parameter values – e.g. 23-G-E for the 23,000 dwellings per year expansion pattern under the green development restrictions (see Figure 4.2). “Grey” scenarios placed relatively more weight on the proximity of roads as a development attractor and relatively less weight to avoiding natural capital loss, whilst “green” scenarios placed more weight on proximity to railway stations and were additionally constrained by not developing in areas designated within a nature recovery network. New settlement scenarios focused development around proposed future railway stations, whilst expansion scenarios focused development near existing settlements and allowed some development on green belt land (Mok et al. 2020). The proposed rates of construction were based upon those required to meet target growth goals set out in the 2019 report on the corridor by the National Infrastructure Commission (NIC).

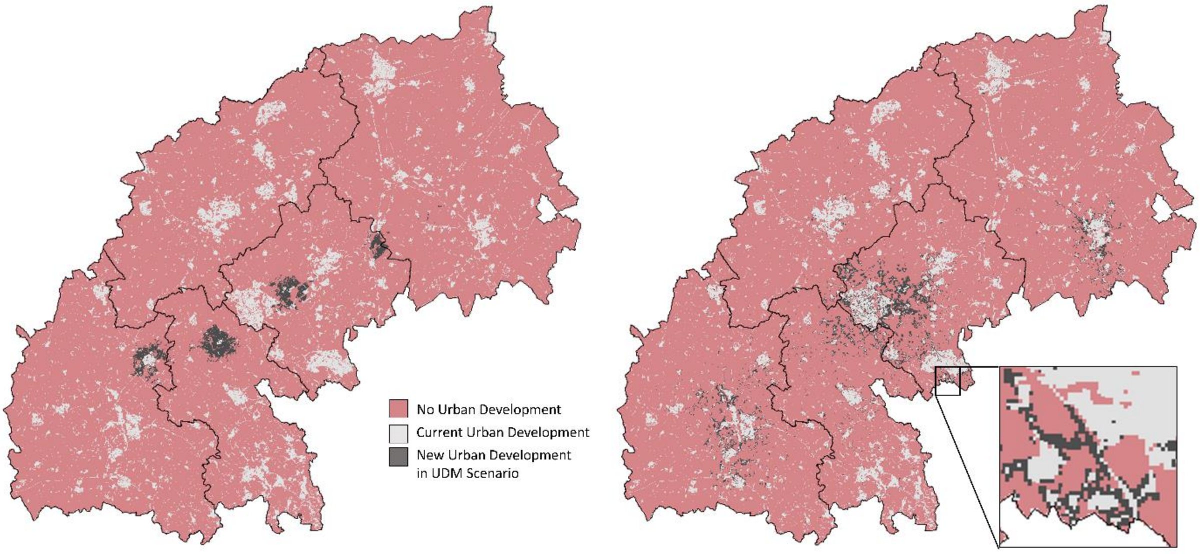

Figure 4.2: Two development scenarios investigated by the study: 23-G-E (left), 30-Y-N (right)

4.3.3: Urban Tiles

Whilst there has been increasing analysis of building stock and its impacts upon resource consumption (Kavgic et al. 2010), many modelling approaches have relied on existing urban maps (thus not considering potential future urban forms) and/or had a limited appreciation of building variability. To overcome these challenges, Hargreaves (2015) developed an urban tiling approach, representing land use and building footprints at a residential-lot scale, based upon analysis from the English House Condition Survery (EHCS). These resultant 1-hectare tiles allow average densities from urban development models to be down-scaled to the lot-scale, including variation in roof areas and garden size, which have key implications for the development of localised, decentralised infrastructure (Hargreaves 2015). As a result, this approach has been used in a variety of spatial urban modelling, including in consideration of alternative water supplies (see Hargreaves et al. 2019) and future energy scenarios (see Ahmadian et al. 2021; Hargreaves et al. 2017).

In order to better understand the land use changes and the potentials for SuDS interventions in the study scenarios, a range of urban tile designs were drawn up to spatially represent a theoretical layout for these urban developments. Urban environments are not homogenous spaces when it comes to urban form, and so to reflect this diversity, four combinations of different housing typologies were represented for each density. Each tile represented 1-hectare (for consistency with the pixel size of the UDM), and arranged the requisite houses, roads and pavements. The remaining space was assumed to be greenspace, and thus car parking or external builds (e.g. sheds) were not considered.

Footprints for the built form elements were based upon design guidance from the Manual for Streets (Department for Transport 2007) and the housing tiles developed by Hargreaves (2015). The latter also provided density thresholds for each housing typology – that is, the average number of dwellings for each typology in different density contexts. These figures then informed the number of dwellings for each typology present in the scenario for each of our four density levels (low, medium, high, very high). So for example, at a low density, Hargreaves (2015) calculated that there were, on average, 6 detached houses per hectare, and thus in our low density scenarios a hectare of detached housing was considered to consist of 6 dwellings. The dwelling numbers used in this study per density level for each typology are given in Table 4.1.

Table 4.1: Dwelling numbers per hectare for each housing typology used in each scenario

Scenario	Typology	Low Density	Medium Density	High Density	Very High Density
Apartments & Detached	Apartments	6	7	11	11
	Detached	3	6	11	15
Apartments & Terraced	Apartments	6	7	11	11
	Terraces	11	34	45	54
Detached & Terraced	Detached	3	6	11	15
	Terraces	11	34	45	54
Apartments, Detached & Terraced	Apartments	6	7	11	11
	Detached	3	6	11	15
	Terraces	11	34	45	54

The resultant 16 urban design tiles can be seen in Figure 4.3. These were further subject to a range of SuDS designs (see section 4.3.4), generating a total of 40 different potential tile designs, examples of which are shown in Figure 4.5. Table 4.2 indicates the proportional coverage of these built form elements for each typology and density.

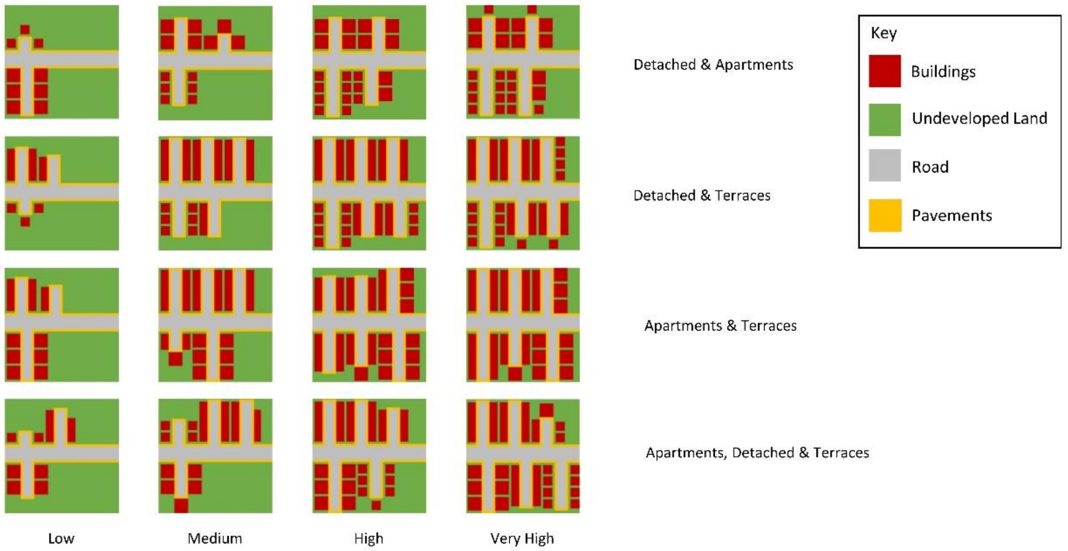

Figure 4.3: The urban tile layouts used in the modelling

Table 4.2: Proportional area of tile covered by each surface area type (%) before SuDS were added

Housing	Density	Buildings	Pavement	Roads	Undeveloped (Greenspace)
Apartments & Detached	Low	10.6	6.1	16.3	67.0
	Medium	13.9	7.1	18.2	60.8
	High	21.5	9.3	22.7	46.5
	Very High	24.0	9.8	23.5	42.7
Apartments & Terraced	Low	14.0	8.0	20.1	57.9
	Medium	27.1	11.3	26.6	35.0
	High	33.4	12.7	29.5	24.4
	Very High	36.4	13.6	31.1	18.9
Detached & Terraced	Low	7.3	6.8	17.5	68.4
	Medium	20.9	11.4	26.7	41.0
	High	28.0	13.0	30.1	28.9
	Very High	29.4	13.0	30.1	27.5
Apartments, Detached & Terraced	Low	10.5	7.0	18.1	64.4
	Medium	18.8	9.2	23.0	49.0
	High	26.8	11.5	26.9	34.8
	Very High	32.7	13.1	30.3	23.9

These urban tile designs were assigned to each pixel of the OpenUDM output for each scenario as follows. A blank tile (i.e. 100% greenspace) was assigned to each 0-value (no development) pixel. Housing densities (dwellings per pixel, each pixel being 1 hectare) for the 1-value (existing development) and 2-value (new development) tiles were then obtained from the OpenUDM outputs. These housing densities were amalgamated into groups of similar values (low, medium, high, very high), and a selection of potential tiles assigned to each group (dependent on the number of dwellings provided by each tile design). Tiles were then randomly assigned to each 1- and 2-value pixel, but only from the selection of tiles in that pixel's dwelling number group. This allowed the resultant tiles assigned to better reflect the dwelling density whilst offering a variation among tiles of the same dwelling density. Figure 4.4 provides a graphical representation of this process as an example. Counts of each tile type for the different development scenarios can be found Appendix 3.

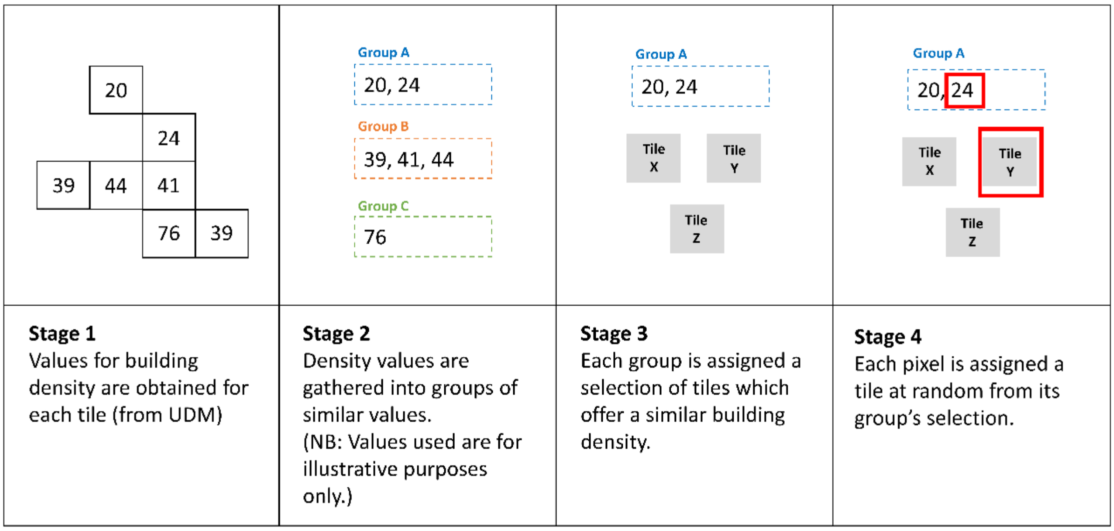

Figure 4.4: Allocation of urban tiles from UDM outputs

4.4.4: SuDS Designs

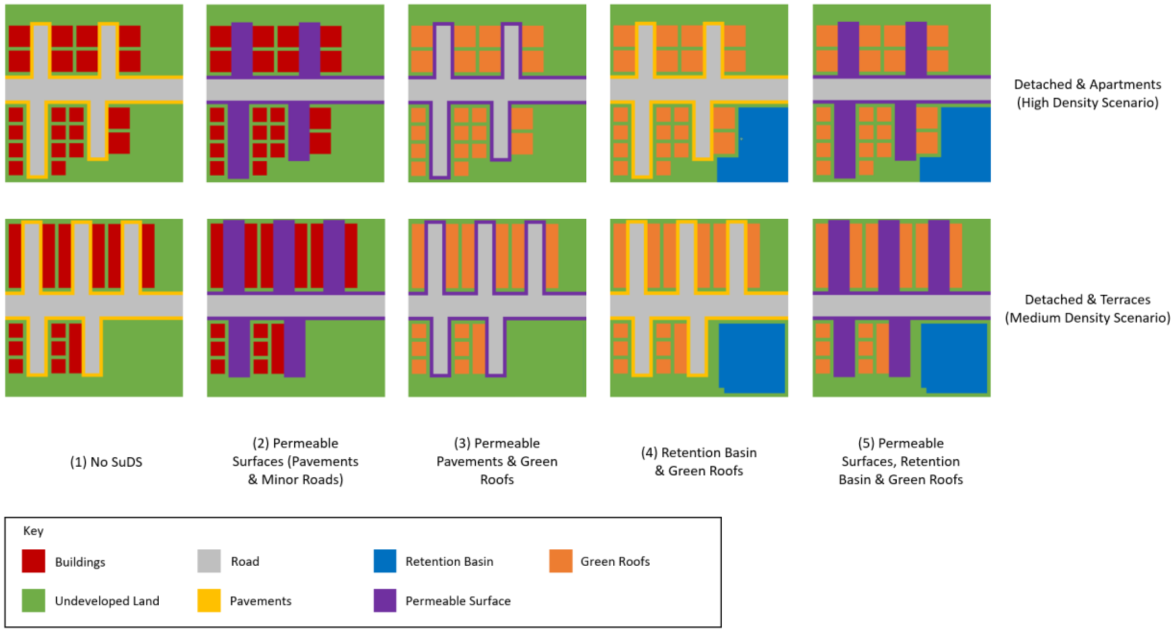

Figure 4.5: Example Tiles for the Five SuDS Scenarios

To better understand how the use of SuDS in these development scenarios may influence stormwater runoff in the study region, five different SuDS scenarios were modelled for each. The five scenarios were: (1) no SuDS; (2) permeable surfaces on pavements and minor roads (PS); (3) permeable surfaces on pavements, and green roofs on residential buildings (PS+GR); (4) lot-scale retention ponds and green roofs on residential buildings (RB+GR); (5) permeable surfaces on pavements and minor roads, lot-scale retention ponds and green roofs on residential buildings (PS,RB+GR). Figure 4.5 illustrates these different scenarios under a given tile design. Whilst these five designs were applied across all tile designs, it is worth noting that some SuDS types are better suited to some designs than others – for example, whilst typically having flat roofs, apartments are better suited to green roofs than houses which (at least in the British context) traditionally have sloping ones.

A soil map for the study region was obtained from the National Soil Centre (Cranfield University 2021), and used as a basemap to define the predominant soil type in each urban tile, which in turn defined drainage parameters for the tile (such as hydraulic conductivity). Each of the resultant scenarios (covering all combinations of development and SuDS scenarios) were modelled using the US Environment Protection Agency's Stormwater Management Model (SWMM) under a 2-hour duration, 1-in-10 year storm event. SWMM is a model noted by previous studies for its credibility in large-scale urban simulations even without observed data (Fu et al. 2019), and comparative sensitivity to imperviousness in relation to other rainfall-runoff models (Yazdi et al. 2019). For this study, the model was accessed using PCSWMM (a direct software implementation of the model) as this both provided ease-of-use through a graphical interface and facilitated the direct population of fields using imported GIS datasets (Hamouz, Møller-Pedersen & Muthanna 2020).

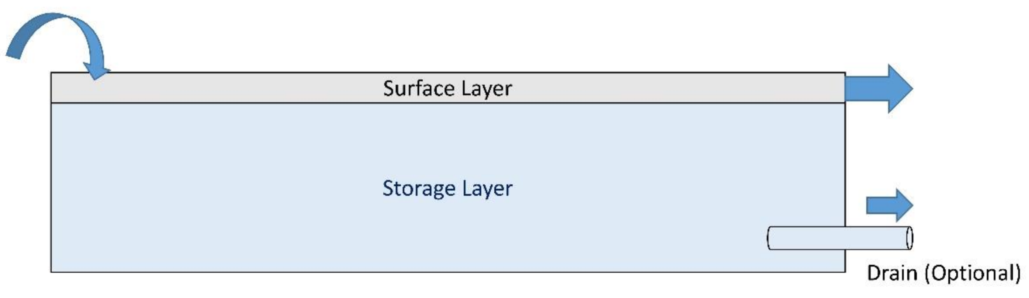

Figure 4.6: SWMM conceptualisation of a retention basin (adapted from EPA 2015)

Figure 4.6, above, illustrates a retention basin system in SWMM. Runoff from the study area is stored until the capacity is reached. The stored water is drained out via a drain, with the outflow rate varying dependent on stored water volume, pressure and outflow drain design (EPA 2015). This outflow continues even after a rainfall event in order to drain the storage unit – for this study, the drain was considered to be at the bottom of the basin to allow complete draining of the basin post-rainfall event, although the model allows this to be altered to simulate permanent pools or ponds. Basin areas were calculated based upon the area of otherwise undeveloped land in each tile and guidance from the CIRIA manual (2015) on sizing such features. As such, total basin area values from tile to tile as guidance indicates a maximum proportion of the available surface on which it is to be built (in our case, undeveloped land) which a basin should occupy.

As part of the modelling process, catchments for the study region were automatically delineated in PCSWMM, based upon a LIDAR composite digital terrain model (DTM) at 5m resolution, sourced from the Ordnance Survey (Ordnance Survey 2021). Figure 4.7, below, shows the delineated catchments in the study area. Catchment-scale results are discussed further in section 4.4.2, using the highlighted catchment (catchment A – 17,280 ha) as an example.

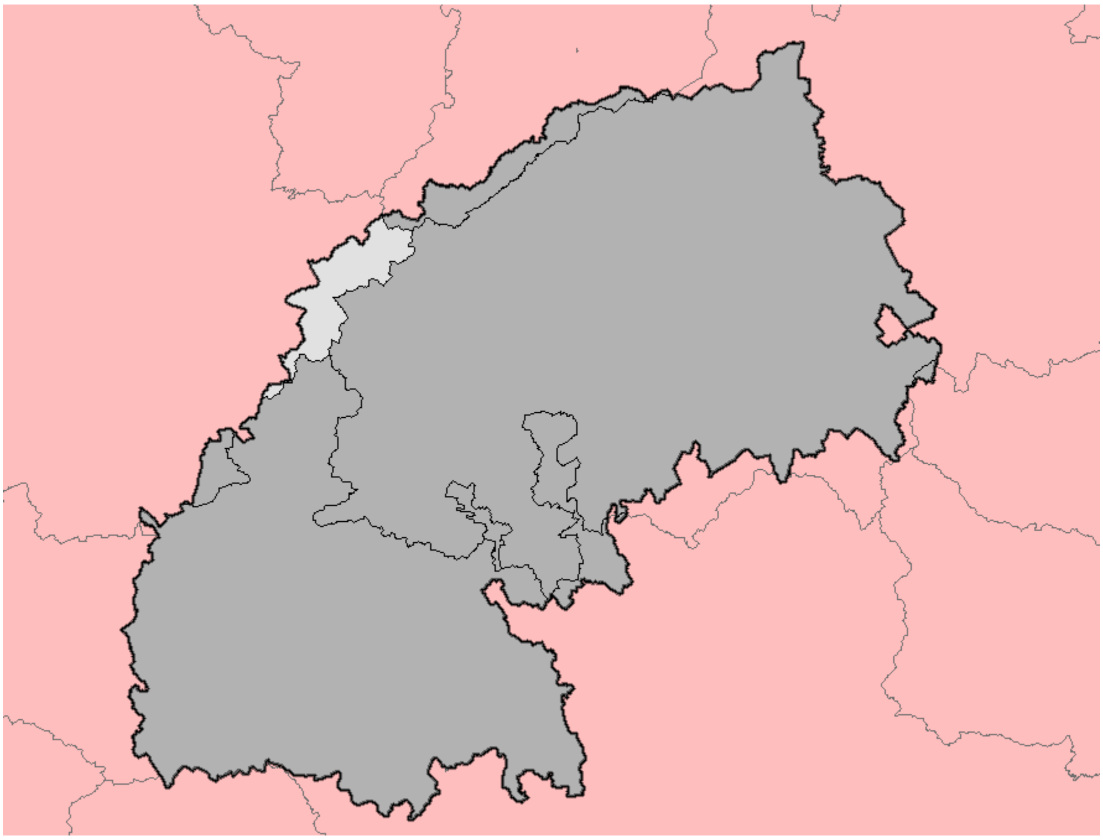

Figure 4.7: Map of watersheds delineated by the program – area outside the study area (pink), study area (grey), catchment A (light grey)

4.4.5: Piped Drainage Requirements

To allow for comparison between SuDS and conventional piped drainage, current design standards (Ministry of Housing, Communities & Local Government 2010) were used to helped identify layout design and junction locations for piped systems. For each tile, a main drainage pipe was located at the midpoint of each road, running the length of the road, and each building was connected to this via an additional pipe running perpendicular to the main pipe. An example tile with its proposed pipeline connections can be seen in Figure 4.8. To capture the required length across each development scenario, the breakdown of component tiles (i.e. the number of each tile design present in the scenario) was obtained, the total length for each design calculated (by multiplying the total number of each tile design by the length of pipe in the design) and these lengths summed.

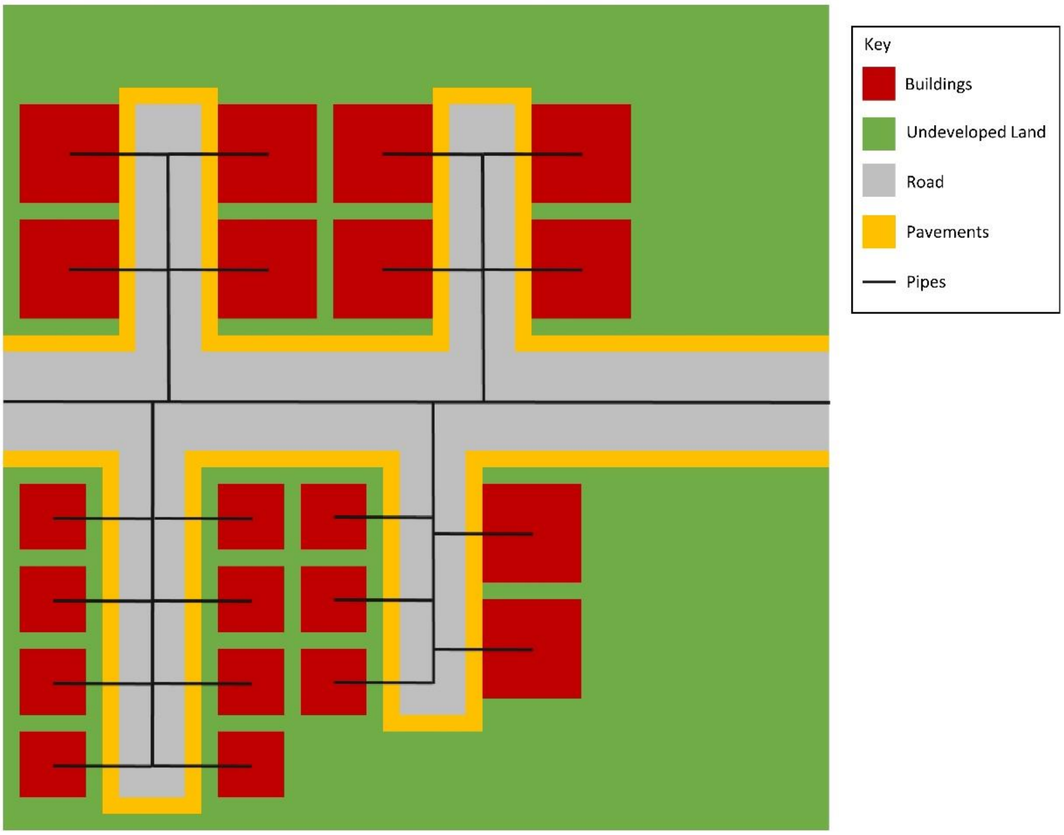

Figure 4.8: The simulation stormwater pipe network for a tile

The minimum pipe diameter was computed according to Whitesides (2012):

(Equation 1)

$$D_o = \sqrt{\frac{4Q}{v_o \pi}}$$

where D_o = pipe diameter (m), Q = flow rate (m^3/s), v_o = flow velocity (m/s). Peak flow rate for the scenario, taken from the results of the SWMM runoff modelling, was used as the flow rate, and the flow velocity was taken as 1.0 m/s, given that British drainage standards require a minimum flow rate of 1.0

m/s (Water UK, 2019). The equation only applies to Newtonian fluids, assumes the pipe is flowing full and that the velocity is continuous throughout the length of pipe (e.g. not significantly altered by pipe fittings, connectors or other additional features) (Whitesides 2012).

Since the research was focused on stormwater, this piped system was treated as separate from the sewer network. However, in practice, the volume of stormwater not captured by SuDS in a storm event will likely flow into the piped sewer network, which in Britain is usually a combined system. By reducing the volume flowing into the combined sewer system, SuDS can contribute to reduced frequency of sewer overflows.

4.4: Results

4.4.1: Tile-Scale

As would be expected, when modelled under the same rainfall conditions, the higher housing density tiles presented a greater peak runoff rate and total runoff volume, as seen in Figure 4.9. Within a given density group, the relative performance of each tile was dependent on housing type, and the consequent area of impermeable surfaces, with greater areas of impermeable surface generating a greater volume of surface runoff as less infiltration into the subsurface can occur. The relative runoff performance for each density scenario varied in line with their relative impermeable areas for each housing scenario.

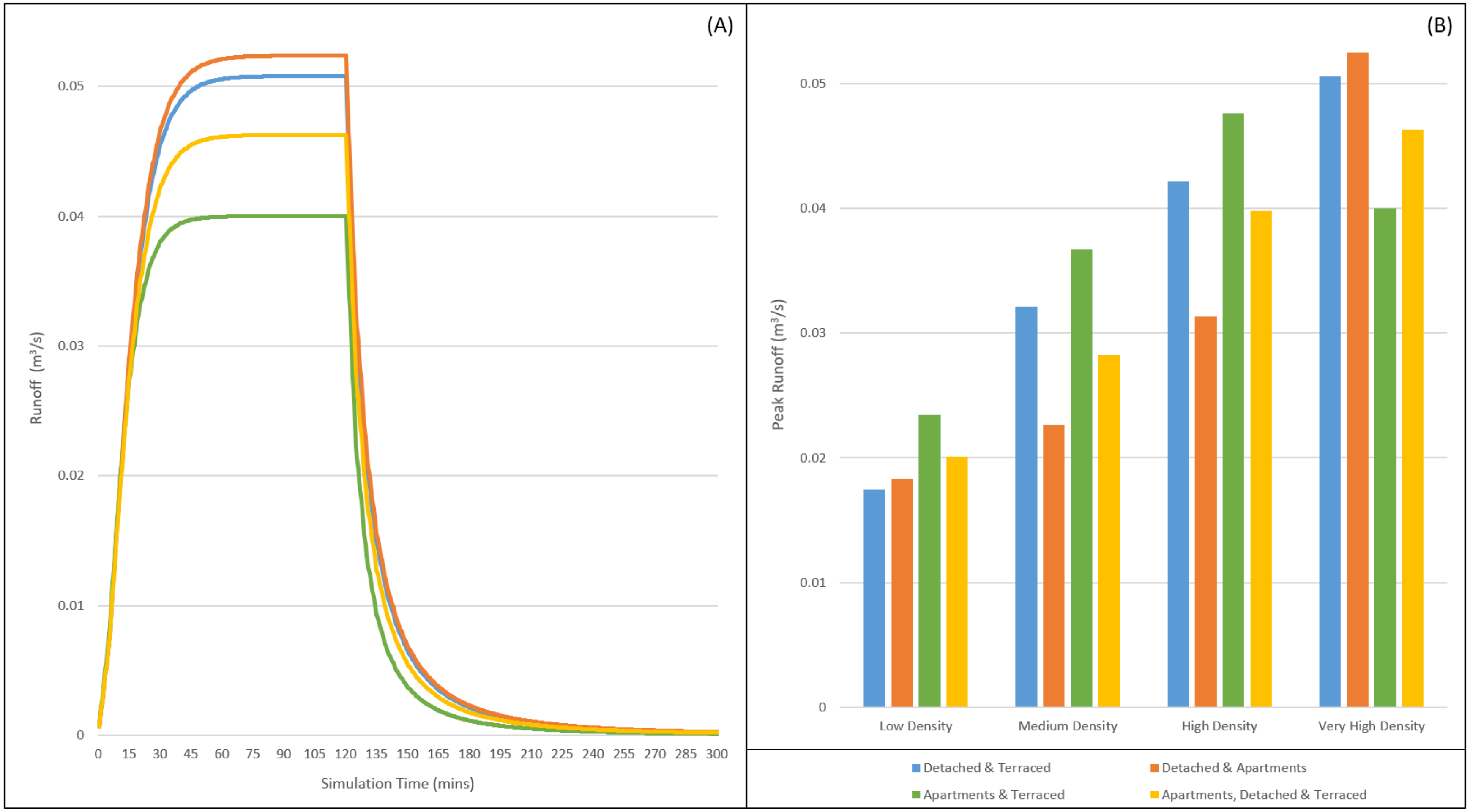

Figure 4.9: Runoff from a 1-in-10 year 2-hour duration rainfall event in each of the tile configurations without SuDS infrastructure: (A) hydrograph of the very high density scenario; (B) peak discharges for the different housing typologies under the different density conditions

This result highlights the importance of considering multiple housing typologies for a given project. As a concept, apartments are typically thought to be a compact solution, providing many residential units arranged vertically and thus reducing the footprint on the ground for such a number of dwellings. However, in each of the density groupings, apartments were present in the scenario with the greatest runoff volumes, and were not present in the scenario with the least runoff at low densities. Even for a specific typology, its relative performance cannot be assumed in relation to the other typologies – apartments & terraces, for example, generated the greatest runoff in the low, medium and high density cases, but the least runoff in the very high density case. This is because per building unit, apartments require the largest building footprint and area of other built elements (e.g. roads) of all the typologies. However, as each building unit can accommodate a greater population than the other housing typologies, as the population to be housed increases, fewer apartment units are required in comparison to the other typologies, offsetting the impact of this initially-large built form area requirement. As such, per dwelling, this typology has the lowest runoff (as demonstrated in Chapter 3), but when considering multi-form and comparable densities (as opposed to dwelling numbers), the picture is more complex.

These findings suggest that planners should carefully consider the housing typologies used in developments of different densities, especially in areas particularly susceptible to surface water flooding, and look to consider how different combinations of typologies may perform relative to one another in a range of metrics that represent and balance the goals of a development (e.g. built-form footprints, surface runoff generation, greenspace provision).

Some typologies have a relatively high building footprint, for example, suggesting green roofs will provide greater runoff reduction in these cases than in those with a smaller proportional building area. Furthermore, the impermeable area gained in a denser scenario can provide greater opportunity for infrastructure-based SuDS development – without roofs, for example, we cannot build green roofs, and if greater roof area is accompanied by a relatively small increase in other impermeable surfaces, it can offer greater runoff volume reduction (as seen with the Apartments & Terraced scenarios). Undeveloped land, as previously identified, also contributes to runoff reduction, however, and thus there is a careful balance to be struck between its loss and other gains.

These proportional densities, as shown in Table 4.2, vary with both housing typology and density, and thus case-by-case consideration is required so that different priorities can be balanced accordingly. Additional projects within the development may also impact this proportional surface type division, further affecting the suitability and impact of different urban designs. Creation of public or active (e.g. walking or cycling) transport networks, for example, reduce overall demands (and thus area required) for car parking (Mueller et al. 2020).

Table 4.3, below, illustrates the minimum commercially-available pipe diameter that would be required to fully capture the 1-in-10 year rainfall event under the different scenarios for each of the tiles. These sizes are taken from the commonly available commercial diameters for thin-wall surface water pipes made from HDPE as these are often used underneath paths and roads, even in heavy duty areas (British Standards Institution 2018).

Table 4.3: Minimum commercial pipe diameter (mm) required under each SuDS scenario for the development tiles to capture a 2 hour duration 1-in-10 year rainfall event

Housing	Density	PS	PP & GR	RB & GR	PS, RB & GR	No SuDS
Apartments & Detached	Low	375	450	375	300	450
	Medium	600	600	450	375	600
	High	600	600	600	450	600
	Very High	900	900	900	900	1050
Apartments & Terraced	Low	600	600	600	450	600
	Medium	600	600	600	600	600
	High	900	900	900	600	900
	Very High	900	900	900	900	900
Detached & Terraced	Low	450	450	450	450	450
	Medium	600	600	450	450	600
	High	600	900	600	600	900
	Very High	900	900	900	900	1050
Apartments, Detached & Terraced	Low	450	600	450	450	600
	Medium	600	600	600	450	600
	High	600	900	900	600	900
	Very High	900	900	900	600	900

As would be expected, this patterning is closely related to the different proportional areas of land cover presented in each scenario (see Table 4.2), which determines the size of different SuDS implementations. For example, the detached & terraced housing tiles consistently require less or equal additional piped drainage than the apartment & terraced tiles at a given density for the permeable surface scenarios due to their greater area of roads and pavements, and thus greater area of permeable surface. This should not be seen as a call for greater road and pavement areas, however, (as most undeveloped land covers will offer better drainage than such systems), but an acknowledgement that the contribution of such SuDS systems in different housing development approaches will be affected by the typology and (for infrastructure-based SuDS) the proportional variation of resultant additional infrastructure employed.

Development density should also be considered during design. At lower densities, the apartments & detached typology, for example, gives some of the lowest pipe diameters seen in any scenario (300mm for all SuDS and 375mm for permeable paving & green roofs), but at very high densities has some of the highest (1050mm for no SuDS and 900mm for all other scenarios).

Furthermore, whilst greater densities require greater or equal commercial piped drainage magnitudes due to increasing impermeable surface areas, these figures hide some of the peak runoff variation observed in the hydrographs. For example, as previously discussed, when using green roofs and a retention basin or green roofs and permeable pavements, the apartment & terraced housing typology saw its highest peak runoff under the high density scenario, but the minimum commercial pipe diameter required was equal for both high and very high densities. This is not to say that the peak runoff is not reduced by the implementation of SuDS, but that the reduction is insufficient to affect the required commercially available size.

There is no one typology identified by the study which consistently outperforms all others in reducing piped drainage requirements, highlighting the importance of considering both housing and SuDS typologies and their interplay when designing developments, yet it is important to note that dwelling numbers were not comparable across the typologies used (but instead used density threshold values based on Hargreaves (2015)), and so through such an approach, future work may show a clearer relationship between housing typology and performance.

4.4.2: Arc-Scale

Table 4.4: Estimated length of pipes (km) that would be required for the new developments across the Arc in each scenario

Scenario	Dwellings/year	Expansion	New Settlements
Green	**23,000**	68,979.17	70,500.37
	30,000	71,852.18	72,696.01
Grey	**23,000**	71,517.48	71,924.90
	30,000	72,706.20	74,350.50

Without any SuDS interventions, regardless of scenario, approximately 70,000 km of pipe will be required to manage stormwater runoff across the case study region (see Table 4.4). This figure only considers the new development areas and does not take into account lengths required to connect these new pipes to the existing network. By nature of the design, this unaccounted length is likely to be greater in new settlement scenarios than expansion. These figures are similar across the scenarios due to the design of the tiles, which see a length of pipe and road running across the centre of the tile regardless of how far across the tile development spreads. Whilst in design this allows neighbouring developed tiles to connect with one another infrastructure-wise, it means that estimated pipe requirements only differ dependent on the length of side roads and short building-to-road connections. In reality, the spatial organisation of developments, particularly in low density scenarios, mean that this central road and pipe is not always required (such as in cul-de-sac developments), which would show greater variation between low and high density developments.

The true cost and extent of this piped system will depend also on the volumetric capacity the system is designed to take, which will influence the diameter of pipes used. Even if SuDS interventions used in a development are not the sole solution for stormwater management, they will lessen the capacity required from the pipe network, and thus reduce costs by reducing required pipe size. Given previous research and existing drainage developments, however, it is not implausible to suggest that single developments or neighbourhoods could optimise their SuDS designs and thus not depend on a separate piped stormwater drainage network (e.g. Hodsons 2019) – indeed, this is a key principle of SuDS. If this were the case, managing an exceedance event during a large storm could be considered within a combined sewer approach, taking advantage of the pipes already required for the sewerage system. However, design for this would need to be very carefully planned as combined sewer overflows are currently causing excessive pollution.

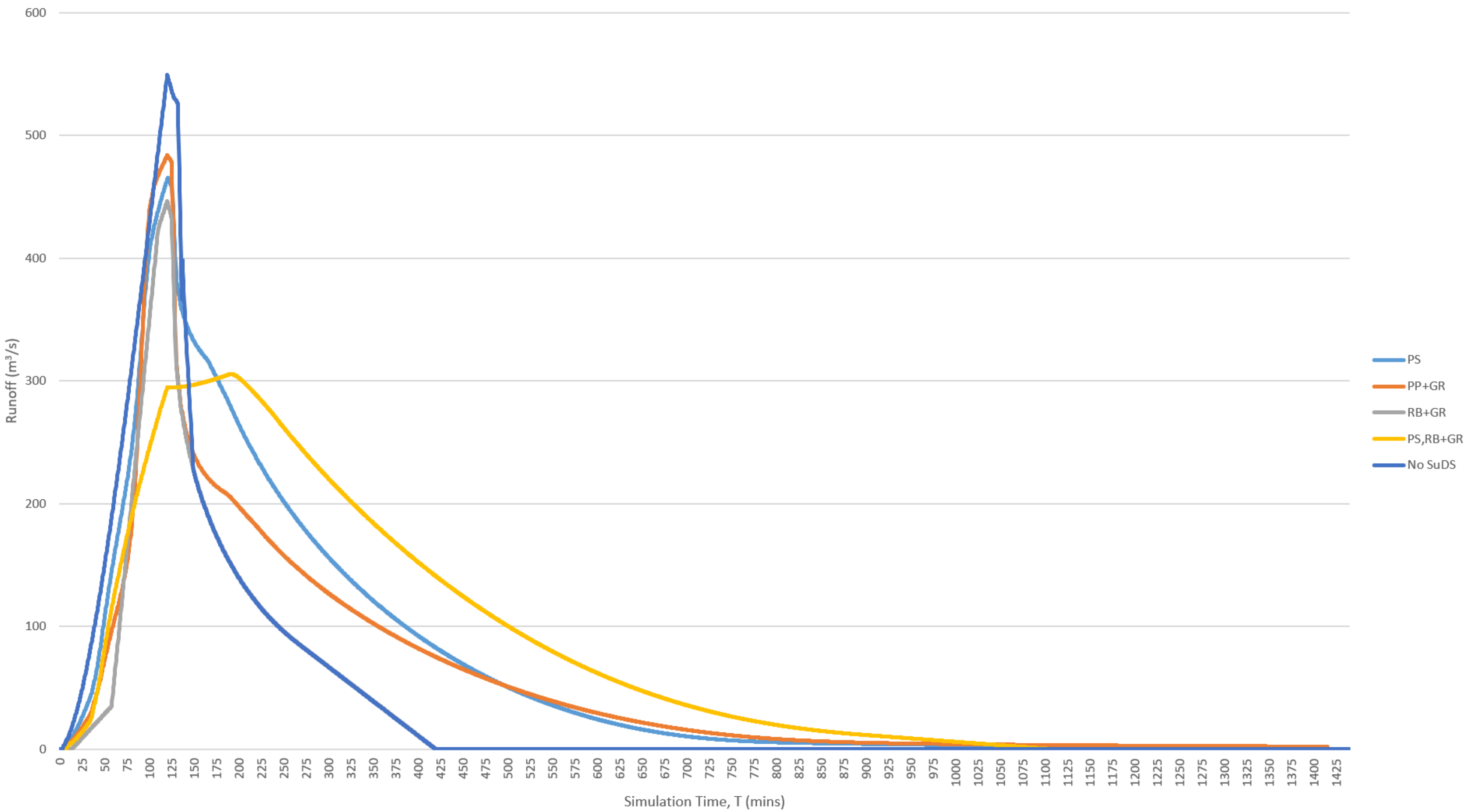

Figure 4.10: Runoff rate for a watershed (Catchment A) in the study region under the different SuDS scenarios

When we consider the different SuDS scenarios, each watershed delineated by the model shows a similar response pattern to a given SuDS scenario, albeit to a different extent dependent on watershed size and proportional developed area. Figure 4.10 shows the runoff profile from an example catchment (Catchment A, shown highlighted in Figure 4.7, with an area of 17,280 ha) under the five SuDS scenarios. Time to peak and peak runoff rates depended both on how urbanised the catchment is, and the design of urban tiles assigned to any areas of new development within the catchment. As would be expected, the greatest peak runoff occurs where no SuDS are present (549.08 m^3/s). In this scenario, all the developed urban surfaces (i.e. building, roads, and pavements) are impermeable and so generate surface runoff during a storm event. Similarly, the PS,RB+GR scenario offers maximum SuDS implementation (within this research) and so gives the lowest peak runoff rate (294.51 m^3/s). Among the other three scenarios, varying proportions of the surfaces offer infiltration and/or storage, and so the total runoff generated is reduced.

The scenarios which feature a retention basin (RB+GR and PS,RB+GR) offer a longer time to peak than those without. The basin offers storage for a given capacity of water which would else be runoff, and this additional time represents the time taken for it to reach capacity. Following this, the rate of increase in runoff rate accelerates as a greater area of the catchment is generating surface runoff. It is important to note that there is still runoff during this initial time to peak, however, as not all the catchment is drained by the retention basin feature. Simulations undertaken in this research show exceedance of the retention features within 85 minutes of the storm event starting. The design of these features, however, could be optimised to drain a smaller area of the catchment or offer a larger storage capacity, which could extend the time before exceedance. However, draining a smaller area of the catchment will not reduce the total volume, and thus requires runoff from the unaddressed area to be managed through several smaller basins, or else in another way or location.

The development approach also has an impact on the runoff response. As illustrated in Figure 4.11, the scenarios supporting a higher volume of housing exhibit a greater peak runoff rate, whilst there is little observable difference between the green and grey development scenarios (likely due to limited appreciation of vegetative conditions in those areas protected under the green restraints). This pattern can be seen across both the catchments and SuDS scenarios. With more houses comes more development and thus a greater spatial extent of impermeable surfaces, generating more surface runoff during a rainfall event. Furthermore, whilst the green and grey development restrictions create different spatial distributions for the development at the study area scale, many catchments see little change in the extent of development between the two, and thus show little difference in runoff responses. Where this is not the case, the scenario with the greater proportion of developed tiles shows a greater peak runoff rate.

This catchment-scale analysis also used the same rainfall intensity as the tile-scale (20.2 mm/hr). Whilst this is a reasonable intensity for a hectare, such an average intensity over a catchment-scale area (17,280 ha for Catchment A) simulates a rather intense rainfall event, and future studies

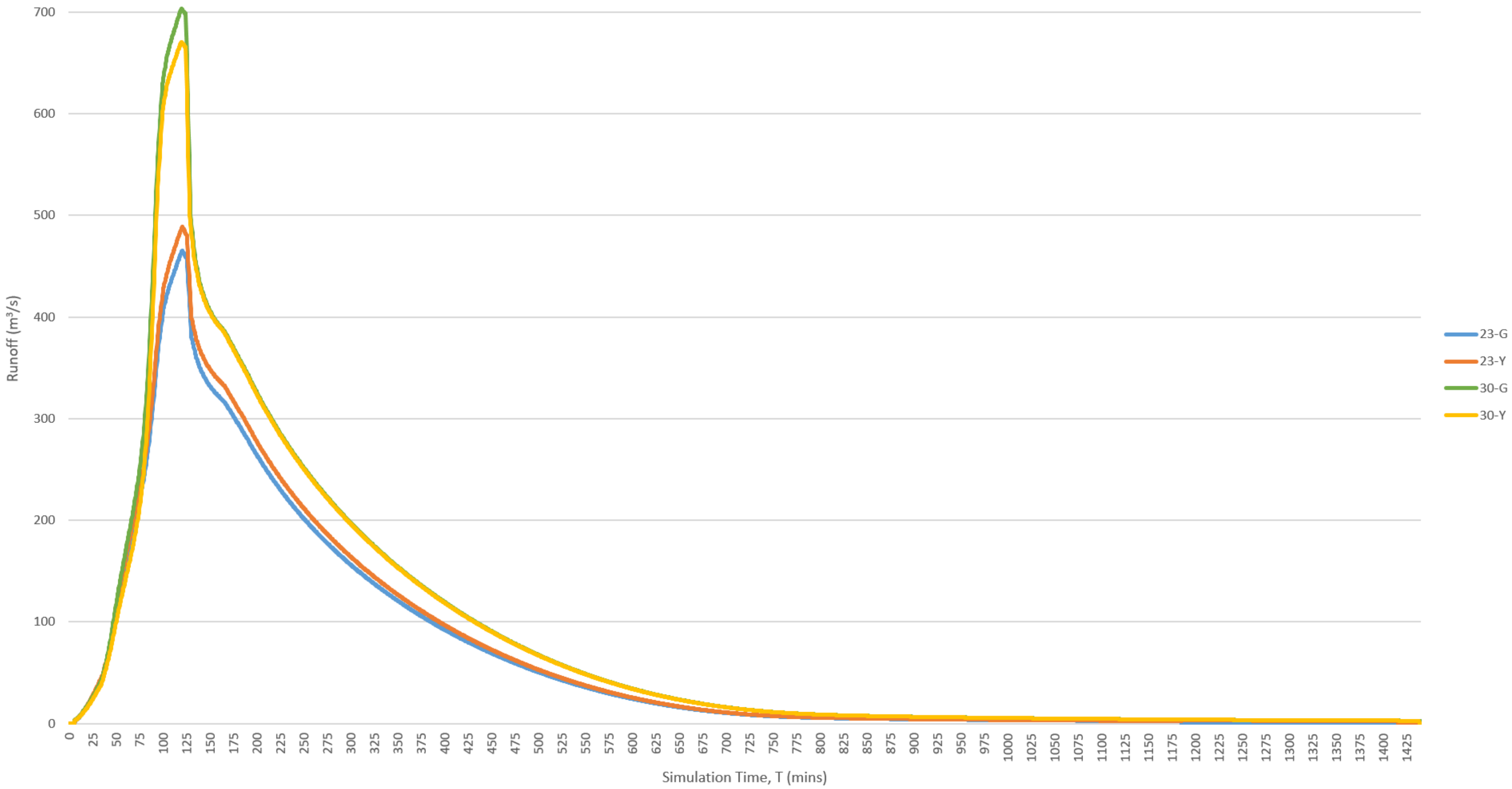

Figure 4.11: Runoff rate for Catchment A during the PS scenario under different development approaches

should look to model these catchment dynamics at a more frequent return period (i.e. less intense event).

4.5: Conclusion

Facing challenges of urbanisation and future climatic changes, the future of urban development is uncertain, with competing pressures favouring contrasting approaches. Urban population growth is expected to continue to rise in upcoming decades, and without a considered approach to managing and supporting the resultant urbanisation, development could result in severe environmental consequences, such as urban sprawl and car-dependency (Boulton et al. 2020). There is still much debated as to the best-suited built form, and whilst many acknowledge that neither the Compact City nor Green City models are ideal, uncertainties still remain surrounding a suitable hybrid approach (Artmann, Inostroza & Fan 2019). Similarly, when it comes to urban drainage, whilst SuDS are considered a sustainable tool for stormwater management, different combinations of infrastructure offer different efficiencies in stormwater runoff management as well as differing co-benefits.

Using a proposed regional development as a case study, this paper simulated a range of potential urban development scenarios based on the outputs of an urban development model. These futures considered different scales of development (a growth of 23,000 or 30,000 houses), different planning forms (new settlement construction or existing settlement expansion) and different spatial restrictions (a “green” and “grey” scenario), as well as the introduction of different combinations of SuDS. SWMM was then used to simulate a 1-in-10 year storm event.

Higher housing density scenarios are generally predicted to result in greater runoff volume and peak runoff rate, but this was also affected by the impermeable surface area. Different housing typologies offered different building footprints at different densities, and thus a given typology did not consistently present the highest or lowest runoff volume for a given density. In fact, whilst yielding the highest runoff volume at low, medium and high densities, apartments and terraces yielded the lowest runoff volume at a very high density as the relative loss of permeable surfaces required to achieve the density increase was low, highlighting the importance of considering multiple typologies and their relative footprints over favouring a single approach, although it is important to note that the total number of dwellings were not equivalent across the layouts. The surface types modelled were also a simplification of the urban form and thus future work could look to incorporate a wider variety (e.g. gardens or car parking) to offer a more realistic proportional area of the different types.

The commercially available pipe diameters required in each scenario to fully capture the simulated rainfall event was also calculated. This, too, showed variation between housing typology and density, but masked some of the patterns observed from the hydrographs. For example, the apartments & terraced typology with green roofs and retention basin required the same pipe size at high and very high densities, despite having a higher peak runoff rate in the high density scenario than very high.

At the wider scale, it was observed that the larger scale developments saw a greater peak runoff than their lower scale counterparts due to the increased surface sealing by impermeable development across the area, whilst the planning type and spatial restrictions had a more spatially-diverse impact, only creating a significant difference where the extent of development differed greatly within the catchment between the two scenarios. However, the model does not capture the differences in natural capital or other co-benefits provided by SuDS.

With the introduction of SuDS, all catchments in all development scenarios showed a reduction in the peak runoff rate as the infrastructure offered increased capacities for infiltration and storage. The extent of the reduction varied with SuDS combination and type, with the greatest impact in that which utilised all modelled infrastructure types.

The findings of this research highlight several key considerations when planning new developments, including (1) the impact of different typologies, densities and SuDS infrastructure on the proportional areas of different surface types, and the impacts/opportunities these may provide in terms of runoff; (2) the additional pipe infrastructure requirements to connect more spatially diverse developments, and the impacts this may have on runoff; (3) how potential areas for SuDS may help reduce (or eliminate) the need for a separate stormwater drainage system, reducing infrastructure costs.

<u>5. Designing Green Infrastructure and Sustainable Drainage Systems in Urban Development to Achieve Multiple Ecosystem Benefits</u>

5.1: Abstract

Urbanisation is one of the greatest threats to biodiversity and ecological connectivity. Green infrastructure (GI) networks and corridors are promoted as a way to preserve habitat connectivity, even in the context of urbanisation. Yet previous research has a primary focus on the ecological benefits and ecosystem services that may be provided by GI networks, without less attention paid to other benefits these networks can bring. There has also been a lack of research into how the design of urban developments, including the potential of sustainable drainage systems (SuDS), may contribute to such networks. A new methodology was developed using readily available datasets and established approaches to assess different urban designs under four aspects of GI provision – ecosystem services, ecological status, ecological connectivity and proximity to the population – and these also combined into a single score to compare the overall spatial patterning of GI potential across the region under the different design and development approaches. Established metrics with minimal data requirements were used for each aspect to support ease of use and replicability of the developed methodology for future studies. Furthermore, ecosystem status was assessed using both the degree and naturalness and patch size to offer appreciation on extents of fragmentation as well as naturalness. Three key considerations for planners were found: first, the positive role urban spaces can play, even without SuDS or considering the proximity to the population, in contributing to GI potentials; second, the balance required between different

development approaches to manage the trade-offs between each; and third, the range of positive and negative impacts different SuDS infrastructure have on GI potentials under different urban designs.

5.2: Introduction

The current growth of urban populations suggests that by the end of the decade our current urban areas will need to be three times larger than they were in 2000 (Felappi et al. 2020). At the same time, however, an increased call for sustainable practices places a challenge on how we develop in a manner that is both sufficient and sustainable. Research suggests that the biodiversity crisis can largely be attributed to habitation fragmentation as a result of urban development and agricultural intensification (Cannas et al. 2018). Reduced habitat area and connectivity is known to limit biodiversity, reduce ecosystem services, and impact gene flow (Bolliger & Silbernagel 2020), and continued habitat fragmentation is considered among the greatest threats to global biodiversity (Algador et al. 2012). In addition, loss of connectivity inhibits ecosystems' capacity to adapt to climate change (Gurrutxaga, Lozano & del Barrio 2010). The importance of such issues in our contemporary world is highlighted in the inclusion of biodiversity, ecosystem health and the green economy as key themes of the Sustainable Development Goals (SDGs) in the 2030 Agenda (Bolliger & Silbernagel 2020).

The compact city approach in urban planning seeks to minimise urban footprints, promoting high density and multi-functional urban spaces, yet this is often at the cost of urban greenspace elements which can be important in providing ecological connectivity (Bibri, Krogstie & Karrholm 2020). Alternately, green city ideas promote the maintenance of inner-city greenspaces, but often lead to greater urban footprints, frequently resulting in the reduction of large-scale undeveloped areas on the urban periphery, which are an equally important feature for habitat strength (Echenique et al. 2012).

Greenspaces have also been a growing feature in urban planning over the past century, justified by the ecological and social benefits they provide (Ignatieva, Stewart & Meruk 2011), whilst sustainable drainage systems (SuDS) are lauded for the social and ecological benefits they can provide in addition to their drainage functions, usually over and above that of traditional grey drainage systems (Ncube & Arthur 2021). These are both examples of green infrastructure (GI) which look to focus on considering and integrating the protection and enhancement of natural processes into planning endeavours (Hansen & Pauleit 2014).

SuDS, in particular, can be a useful tool in augmenting the provision of these socioenvironmental benefits. Different SuDS infrastructures offer different combinations (and relative strengths) of these co-benefits, and so their inclusion and design can be tailored to provide those sought by the aims of a development or retrofit project (La Rosa & Pappalardo 2021). Some SuDS, for example, involve the introduction of plants or grassed surfaces to a landscape (such as green roofs, bioswales or wetland creation) which leads to the creation of new habitats, whilst others (such as permeable roads and paving) do not. Equally, the more visibly green and aesthetically pleasing infrastructure provide greater mental health benefits than those which make no visual alteration to the landscape (Felappi et al. 2020). Furthermore, installing multiple different types of SuDS in a development can enable a wider array of these co-benefits to be achieved, with one form complementing another – although it is important to consider how, too, these may lead to trade-offs in the extent to which ecosystem services are provided (Hansen & Pauleit 2014).

However, whilst there has been great interest in the concept from researchers and practitioners, the physical implementation has been much more limited (Bolliger & Silbernagel 2020). Many current and previous strategies for biodiversity conservation and ecosystem enhancement have treated spaces for protection as independent, self-contained regions, but there is growing evidence that these are not sufficient to retain ecological processes and value (see Gurrutxaga, Lozano & del Barrio 2010; Bolliger & Silbernagel 2020; Valeri, Zavaterro & Capotorti 2021). Recognition of the importance of habitat connectivity and ecological networks has increased, which act to promote and maintain ecological processes and flows (Baguette et al. 2013). Crucially, these connections need not always be continuous corridors of space but, for some species, can be supported through a semi-fragmented, stepping stone approach (Algador et al. 2012), which increases their potential for retrofitting into existing urban spaces.

In order to create these networks, whether continuous or semi-fragmented, however, regional-scale planning and design is important to have a significant ecological impact (Grădinaru & Hersperger 2019). This has been widely recognised in policies across Europe. Natura 2000, for example, is an existing international GI programme led by the EU, protecting key areas in Europe and encouraging the development of ecological networks to ensure ecological coherence across the continent (Rincón et al. 2022). This is further supported by the EU's more recent Biodiversity Strategy, which recommends maintaining and enhancing ecosystems through establishing GI at a regional-scale (Cannas et al. 2018). Meanwhile, the European Commission (EC) launched their Green Infrastructure Strategy in 2013, aiming to mainstream GI planning at regional, national and international scales (Maes & Jacobs 2015), whilst in England, Natural England has introduced anthropocentric standards for greenspace planning, defining limits such as maximum distances to parks and hectares of nature reserve per population number (Hansen & Pauleit 2014).

5.3: Evaluating the Benefits of Green Infrastructure

The European Commission in their GI strategy define GI as “a strategically planned network of natural and semi-natural areas with other environmental features designed and managed to deliver a wide range of ecosystem services” (Maes & Jacobs 2015).

Due to the spatiality of the concept, many studies looking to assess the strength of, or identify, ecological networks in a region use a cost-distance mapping approach in GIS (see Gurrutxaga, Lozano & del Barrio 2010; Liquete et al. 2015; Cannas et al. 2018). This requires key habitat areas to be identified, however, for which expert opinion is typically used (Rodriguez-Espinosa, Aguilera-Benavente & Gómez-Delgado 2020). Consequently, the resultant maps are often very species-specific and so cannot be used to generalise the situation or condition. To assess or comment on overall biodiversity with such an approach, the process would need to be repeated for a range of species, which can be both data and time intensive.

Restrictions or barriers to movement between these spaces also need to be identified for cost-distance mapping, and for this physical infrastructure is often the only barrier considered (Rodriguez-Espinosa, Aguilera-Benavente & Gómez-Delgado 2020). However, as Algador et al. (2012) identify in their study, which focuses more on climatic variations and their impacts on species movements, anthropocentric factors are not alone in determining ecosystem flows and can lead to different pathway identification – factors which are also currently under flux due to climate change (Rincón et al. 2022). Furthermore, research by Baguette et al. (2013) questions whether these cost-distance pathways accurately reflect ecological migration patterns, as GPS tracking data suggests they are regularly over-simplified.

Many studies to-date are also focused on the Natura 2000 area (or a sub-section of it) as it is a region specially targeted for GI network development in order to protect and enhance biodiversity under the EU’s scheme (such as Algador et al. 2012, Liquete et al. 2015, Rincón et al. 2022). Consequently, data availability and/or regional factors limit the replicability of these methodologies in other locations. Furthermore, this spatial focus leaves questions unanswered on how regionally-variable conditions and the lack of targeted GI corridor projects may affect network success or generation. Notable exceptions to this include Zhang et al. (2019), who focus on Detroit, Michigan and the potential inner-city development could have in network generation, utilising a methodology that could be applied in other urban locations, and Rodriguez-Espinosa, Aguilera-Benavente & Gómez-Delgado (2020) who reviewed previous mapping approaches and developed a general methodology to analyse a range of GI roles (such as recreation potential).

Regardless of focus or spatiality, however, there is widespread agreement that to achieve the regional and connected preservation identified by GI networks, action is of a time-critical nature. Poor or slow interventions can lead to the loss of potentially strong connections (Liquete et al. 2015), particularly as many areas that show good GI potential are those already under threat by urbanisation and development (Gurrutxaga, Lozano & del Barrio 2010). If practices around GI identification and preservation became more actively integrated in planning processes, they would offer a long-term and persistent method in strengthening a range of ecosystems and greenspace benefits (Bolliger & Silbernagel 2020). This integration should occur both regionally and locally as whilst a regional-scale perspective is needed to identify the best locations for such networks (Cannas et al.

2018), without smaller scale considerations, the projects often remain abstract ideas or not realistically implementable (Liquete et al. 2015).

This is not to say, however, that urban development and GI networks are mutually exclusive. In fact, many studies identify several aspects of urban spaces that can assist in developing and supporting these networks. Vacant lots or brownfield sites, for example, can be crucial for inner-city habitat creation, especially wild and indigenous vegetation (Zhang et al. 2019; Ma, Li & Xu 2021). Furthermore, contemporary movements in design of urban spaces offer their own potential for habitat creation and enhancement, such as the green city movement or environmentally-sensitive infrastructure (Ignatieva, Stewart & Meruk 2011; La Rosa & Pappalardo 2021). Little work has been done at present, however, to model or investigate how such movements would be applied to best achieve GI network provision and support, or identify what the resultant urban form may look like.

These ideas of hybridised spaces, forming both part of a GI network and urban environment, are further supported by increased interest in the human elements of GI (see Cannas et al. 2018; Felappi et al. 2020; Ncube & Arthur 2021). The term GI is often seen as synonymous with biodiversity and visually green spaces (Chatzimentor, Apostolopoulou & Mazaris 2020), but it is equally important to consider additional elements. Hansen & Pauleit (2014) highlight in particular the accessibility of natural or near-natural spaces and how poor management can worsen environmental justice in terms of greenspace access – an idea further supported by Felappi et al. (2020), who argue that many benefits of greenspace provision, such as leisure and mental health benefits, include a human element that cannot be achieved without access by humans. Rodriguez-Espinosa, Aguilera-Benavente & Gómez-Delgado (2020) applied this idea to their methodology, introducing a weighting factor that allows different priorities and targets to be represented in GI network mapping.

Considering and including multiple aspects of GI provision, however, the role of trade-offs also becomes important as compromises are required to balance the optimum conditions for different elements (Hansen et al. 2019). In addition, GI network provision will likely overlap and intersect with other schemes and goals in both the urban and rural environments, particularly given its large-scale remit, and thus these trade-offs should be brought to the fore in the local planning domain where expert knowledge can help achieve a balance/prioritisation sensitive to the area's priorities (Cannas et al. 2018). Whilst work remains to be done on more detailed understanding of individual GI elements, Felappi et al. (2020) call for future research to be focused on multi-faceted examinations of GI, offering a more realistic indication of good connectivity through appreciation of these interactions.

Building on the identified strengths and weaknesses from previous approaches, therefore, this research aims to integrate a wider range of GI benefits, applied to a regional scale new development project, to model the impacts of development approach and SuDS interventions on the location and strength of GI networks.

5.4: Methodology

This study utilises a service mapping approach, as seen in previous studies (such as Liquete et al. 2015; Cannas et al. 2018; Rodriguez-Espinosa, Aguilera-Benavente & Gómez-Delgado 2020) to consider multiple facets of GI, and how these may be affected by a regional-scale urban development project. Results from an urban development model (OpenUDM) study for potential urbanisation scenarios were used to simulate developments of varying spatial design and magnitude. The urban layout of these was then represented using an urban tiling approach, which included designs featuring sustainable urban drainage systems (SuDS). These resultant scenarios were then assessed under four conditions representing different elements of GI, and these results combined to provide an overall GI service map for each scenario.

5.4.1: Case Study & Scenarios

The study area is a region of south-east England earmarked for large-scale urban development – the Cambridge to Oxford corridor (see Figure 5.1). The area lies to the northwest of London, stretching across five counties (Bedfordshire, Berkshire, Cambridgeshire, Northamptonshire, Oxfordshire) and encompasses the existing cities of Cambridge, Milton Keynes and Oxford. It is estimated that 1 million new dwellings will be required in the region before 2050 in order to maximise the area's economic potential and counteract the lack of affordable and sufficient housing currently hindering the socioeconomic success of the region (NIC 2019), although planning into the spatial pattern of such development is still ongoing, enabling the potential for sustainability to be embedded into the proposals (Mok et al. 2020).

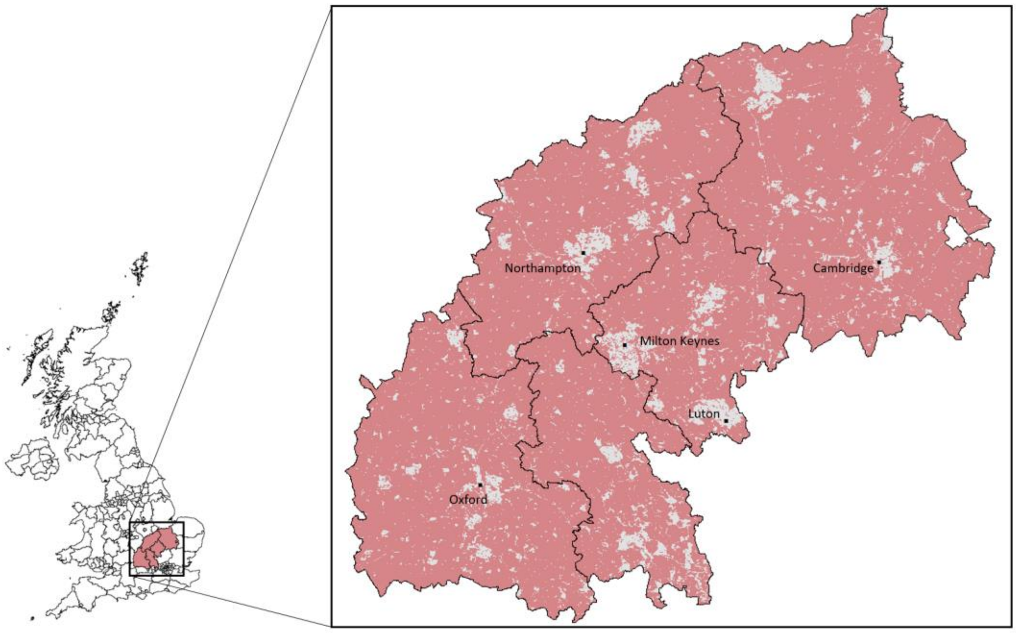

Figure 5.1: An outline map of the case study location showing existing urban development (grey)

5.4.1.1: Urban Development Modelling

Urban development modelling was undertaken using OpenUDM, a spatial optimisation tool combining multi-criteria evaluation and cellular automata approaches to create high resolution scenarios of heterogenous urbanization subject to spatial inputs of attractors and constraints (Ford et al. 2019). Eight future development scenarios were generated for the case study at a 100m grid scale, representing two rates of growth (23,000 and 30,000 dwellings per year), two development patterns (new settlement construction and existing settlement expansion), and two contrasting sets of development constraints (“grey” and “green” – the former placed more emphasis on proximity to roads and less on avoiding natural capital loss, whilst the latter placed more emphasis on proximity to rail stations and avoided development in areas designated nature recovery networks). Further detail on this approach can be found in Chapter 4.

5.4.1.2: Urban Layouts

The identified development locations and densities from the UDM were represented at a lot-scale, based on the urban tiling approach developed by Hargreaves (2015). This allowed the analysis of different urban designs, enabling the role of urban greenspace and sustainable drainage infrastructure to be considered in the service mapping. To represent the heterogeneity of urban spaces, four designs were drawn up for each of four density categories, with footprints and layouts guided by the national Manual for Streets (Department for Transport 2007) and Hargreaves (2015). Figure 5.2 illustrates the sixteen core tiles for urban layout used in this study, and Figure 5.3 illustrates how these changed under the different SuDS designs. The placement and allocation of tiles was not targeted to create or augment specific ecological corridors or networks across the study area, rather tiles were assigned at random to each 1-hectare area dependent on the scale and location of housing development indicated by the OpenUDM output – further information on the assignment process can be found in Chapter 4. This was so done to better illustrate how such connectivity may exist (with or without SuDS) and/or be enhanced under current planning approaches that have not placed ecological connectivity at the heart of their design.

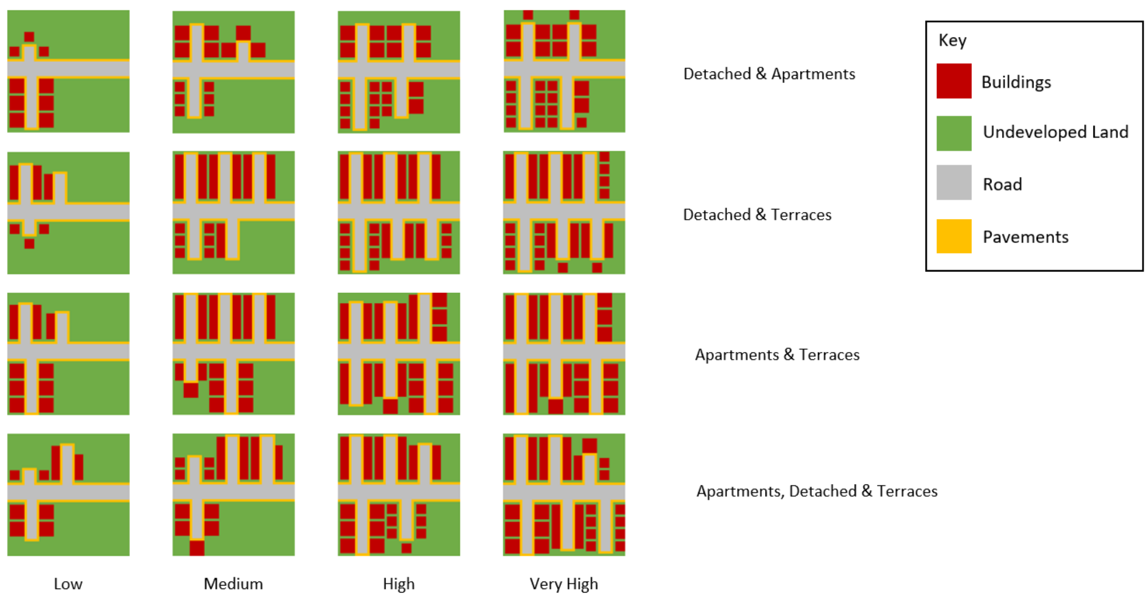

Figure 5.2: The urban tile layouts used in the modelling

Five different SuDS designs were considered in the study:

1. No SuDS

2. Permeable Surfaces (featuring permeable asphalt on minor road surfaces and permeable pavements)

3. Permeable Pavements & Green Roofs

4. Retention Basins & Green Roofs

5. Permeable Surfaces, Retention Basins & Green Roof

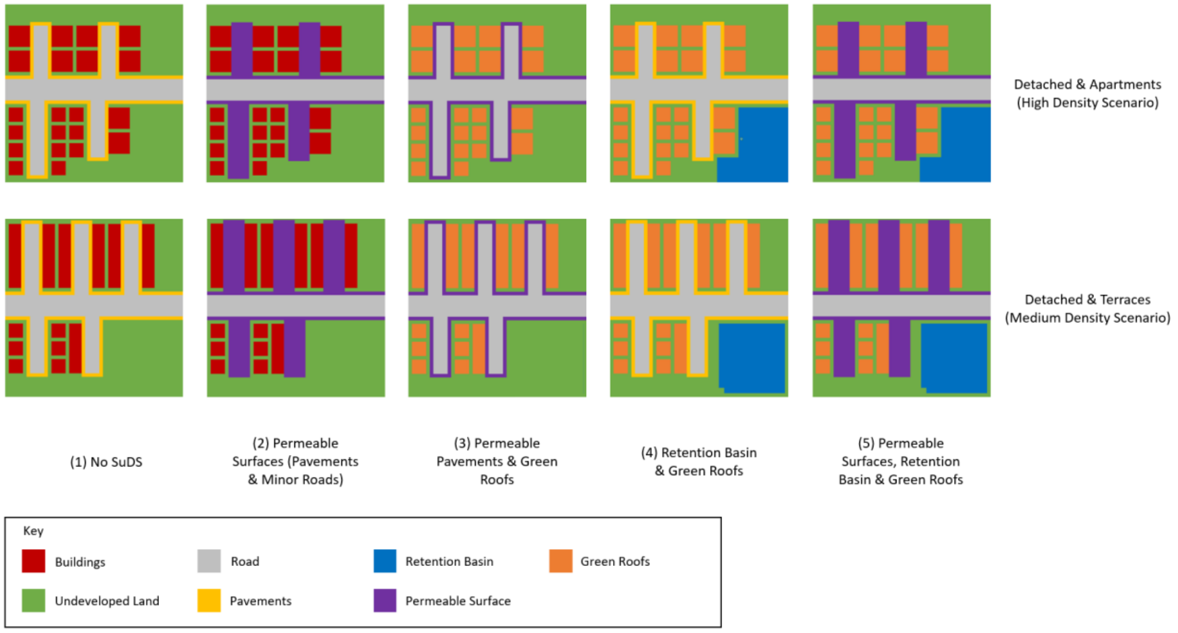

Figure 5.3: Example tiles for the five SuDS scenarios

5.4.2: GI Provision Assessment

Rodriguez-Espinosa, Aguilera-Benavente & Gómez-Delgado (2020) identify four key features to assessing strength of GI based on the definition by the European Commission outlined in section 5.3: ecosystem service provision, ecological status, ecological connectivity, and proximity to the population. These were used as the four main criteria for this study in assessing the strength of GI provision as they allowed consideration of ecological and anthropocentric elements. Each was assessed by a metric, outlined below, and scored accordingly. These scores were then normalised from 0-8 for each criterion, allowing cross-comparison between the factors (Liquete et al. 2015). Through summation, these scores were then combined to create a single "GI score", offering a multi-faceted appreciation of GI provision in that location, as illustrated in Figure 5.4. This combined score allows multiple criteria to be considered and compared across development scenarios simultaneously, and different SuDS designs to be cross-compared for a given development scenario.

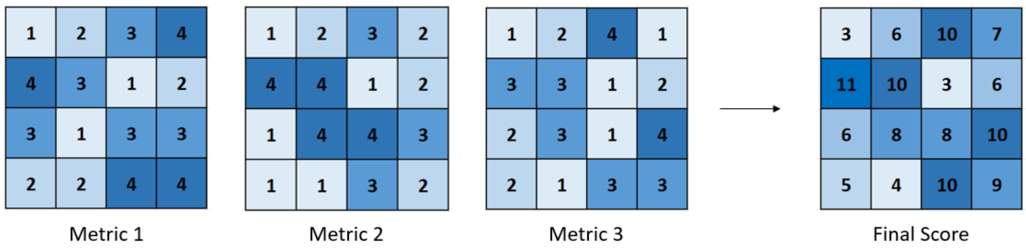

Figure 5.4: A visual representation of the approach used to combine analysed elements of GI into a single score

The benefits of the outlined approach include the use (within specific GI criteria) of established methods from other studies, whilst allowing multiple elements of GI to be examined, combined and compared, and reducing the need for large, ecological base datasets (such as species habitat preferences or movement patterns) (Rodriguez-Espinosa, Aguilera-Benavente & Gómez-Delgado 2020). The spatial outputs generated also offer useful visuals for targeted actions in the planning community. Equal weighting was applied to the metrics as the aim was to identify how provision of GI elements overall were affected by proposed development and each was considered to have its own contributions, however weighting could be applied in future studies to emphasise a particular element of GI.

5.4.2.1: Ecosystem Services

Ecosystem services are the primary basis of many contemporary assessment approaches for GI provision (Chatzimentor, Apostolopoulou & Mazaris 2020). They are the benefits offered by a habitat and can be sub-divided into provisioning (those which provide tangible products or energy), regulating (those which support the healthy operation of ecosystem functions) and cultural (those which provide non-material advantages to humans) (Ma, Li & Xu 2021). Consequently, the standard and range of services provided by an ecosystem vary with both land cover and quality (Liquete et al. 2015).

Smith (2021) developed and outlined a scoring matrix for ecosystem services based on the land cover, which provides individual scores (from 0 to 10) for each of 18 services that can be averaged to provide a generalised ecosystem services score. The method and resultant scores were developed from a 780-paper literature review (see Smith et al. 2017), expert consultation, and comparison with similar scoring approaches. Whilst such an approach is not explicitly linked to a quantified dataset as a proxy, a score-based ecosystem services assessment offers an overview and simple estimate of contributions, which is sufficient for a multi-metric approach (Rodriguez-Espinosa, Aguilera-Benavente & Gómez-Delgado 2020). It is important to note, however, that land cover data provides an indication of the current surface type, but does not always indicate the actual land use or any sub-surface conditions (CEH 2020) –for example, distinction is made for forested land cover between natural woodland and a plantation, but urban land cover cannot be divided between occupied or abandoned.

Information on the land cover in the study area was obtained from a national land cover map from the UK Centre for Ecology & Hydrology (CEH) at a 25m raster scale (Morton et al. 2021). This was used to inform the land cover for all undeveloped pixels (i.e. those not assigned a development tile). For the development tiles (which represented both existing and proposed urban development), the land cover was assigned based on the dominant land cover in the tile. For example, in the low density detached and terraced tile, undeveloped land occupied nearly 70% of the tile, and thus the land cover was assigned that which it would have been were it all undeveloped. Consequently, areas of such spatial design were not reflected in the results as development and thus their impacts as urban environments were overlooked. However, due to the random nature of tile assignment, it is unlikely large coherent areas of development will be obscured by this. This reflects a similar process as to the land cover data obtained from the CEH, which also uses the dominant land cover to determine the assigned value as neither natural nor man-made environments are homogenous spaces.

Average ecosystem services scores were then assigned from Smith's matrix based on this land cover type. Whilst this offers a means of aggregation for ease of comparison, however, it does mask trade-offs where a land type is particularly strong for one service type but relatively weak for another, such as with top-tier agricultural land which, despite high provisioning service scores, is relatively poor performing for cultural services. Nevertheless, such land is highly valuable, and as such future work should look to apply and develop an approach which has a greater appreciation of variability within ecosystem service types. One such approach, using multipliers which are based upon condition and spatial factors as well as ecosystem type, is described in Smith (2021).

5.4.2.2: Ecological Status

Ecological status is defined as the condition of an area relative to a natural or pre-development state (Valeri, Zavaterro & Capotorti 2021). This degree of naturalness is considered a key criteria in understanding the role of spatial planning in nature conservation (Tuluhan Yılmaz, Alphan & Gülçin 2019), and there are many existing scoring metrics in the literature (such as Machado 2004 and Rodriguez-Espinosa, Aguilera-Benavente & Gómez-Delgado 2020). In this study, the scoring system outlined by Rudisser, Tasser & Tappeiner (2012) was used due to its simple metric, which assigns values between 1 and 7 using land cover data as a proxy.

Equally as important to consider as an element of the ecological status is the patch size, which offers commentary on how fragmented and/or clustered the landscape is (Ersoy, Jorgenson & Warren 2019). The size of each land cover patch was therefore calculated and awarded a corresponding value between 0 and 4, dependent on magnitude. This was added to the naturalness value to create the overall value for ecosystem status. Appendices 4 – 6 outline the scores applied for both degree of naturalness and land patch size.

5.4.2.3: Ecological Connectivity

Connectivity is a crucial part of healthy ecosystem functioning, allowing gene dispersal, preserving biodiversity, and increasing ecosystem resilience (Baguette et al. 2013). When these healthy ecosystems form part of a wider network, their interconnectivity is also integral to the functioning of the system as a whole (Rodriguez-Espinosa, Aguilera-Benavente & Gómez-Delgado 2020). It is herein noteworthy that previous studies have indicated the potential for urban greenspaces (including those provided by SuDS infrastructure) to improve ecosystem strengths and coherence (see Algador et al. 2012; Wild, Henneberry & Gill 2017).

To assess the strength and location of any connection, nodes (or base sites) need to be identified, which are the areas looking to be connected. For this metric, the nodes were established by identifying all regions above a set area threshold (5 hectares) where each constituent tile scored ‘1’, ‘2’ or ‘3’ (natural, near-natural or semi-natural) under the degree of naturalness metric (outlined in the previous section). Urban development tiles were also included in these nodes if they met the above criteria. These nodes were then awarded a value of 0 and all other tiles scored from 1-4, dependent on their distance to the nearest of these base sites.

5.4.2.4: Proximity to the Population

As previously identified, many of the benefits we consider arising from successful and healthy ecosystems have a distinct human element, such as recreation potential, and there is increasing interest in the literature in understanding these interactions between landscape and humans (see Hansen et al. 2019; Bibri, Krogstie & Karrholm 2020). In order for these to be fully appreciated, however, there is a need for people to be able to access such spaces, whether physically or visually (Felappi et al. 2020).

This fourth metric used a similar approach to ecological connectivity, identifying base sites and assigning scores dependent on distance from these. In this case, however, the base sites were those tiles with residential buildings and had no set area threshold.

5.5: Results

5.5.1: Ecosystem Services

5.5.1.1: Development Approach

The spatial variation of ecosystem service scores is dependent on the spatial pattern of land cover in the region. Thus, in the development scenarios with a greater magnitude of urban development, there is a greater change in ecosystem service scores as more land is required to be converted to a predominantly urban land cover - the land cover type with the lowest ecosystem services score of those in the study.

However, the magnitude of change in ecosystem service score is dependent both on the location of the development and the type of development. This can be seen in Figure 5.5, which compares the area of land scoring each value for ecosystem service scores between two contrasting development scenarios. A greater area in large magnitude development under grey constraints (30YE) registered low ecosystem services scores, suggesting poor provision, than in the green constrained, smaller magnitude counterpart (23GNS). At times, this was area was twice as large – ecosystem service scores of zero were given to 1577 km^2 of the 30YE scenario, but only 645 km^2 in 23GNS. High ecosystem service scores also covered a much lower area in 30YE, particularly for scores of 8, as a greater spatial coverage of development (from development pattern and magnitude) sees high scoring land covers being urbanised.

Due to the relatively low surface cover of roads and housing in the lower density scenarios, many of these tiles were scored as undeveloped land cover, presenting higher ecosystem service scores than those dominated by the urban infrastructure. It is interesting to note that this was the case for the detached and apartments housing typology at all densities due to the relative minimal urban footprint offered, whereas no other typology scored this way at greater than medium density. Nonetheless, without the inclusion of SuDS, no urban tile improved on the existing ecosystem service score of an area prior to development.

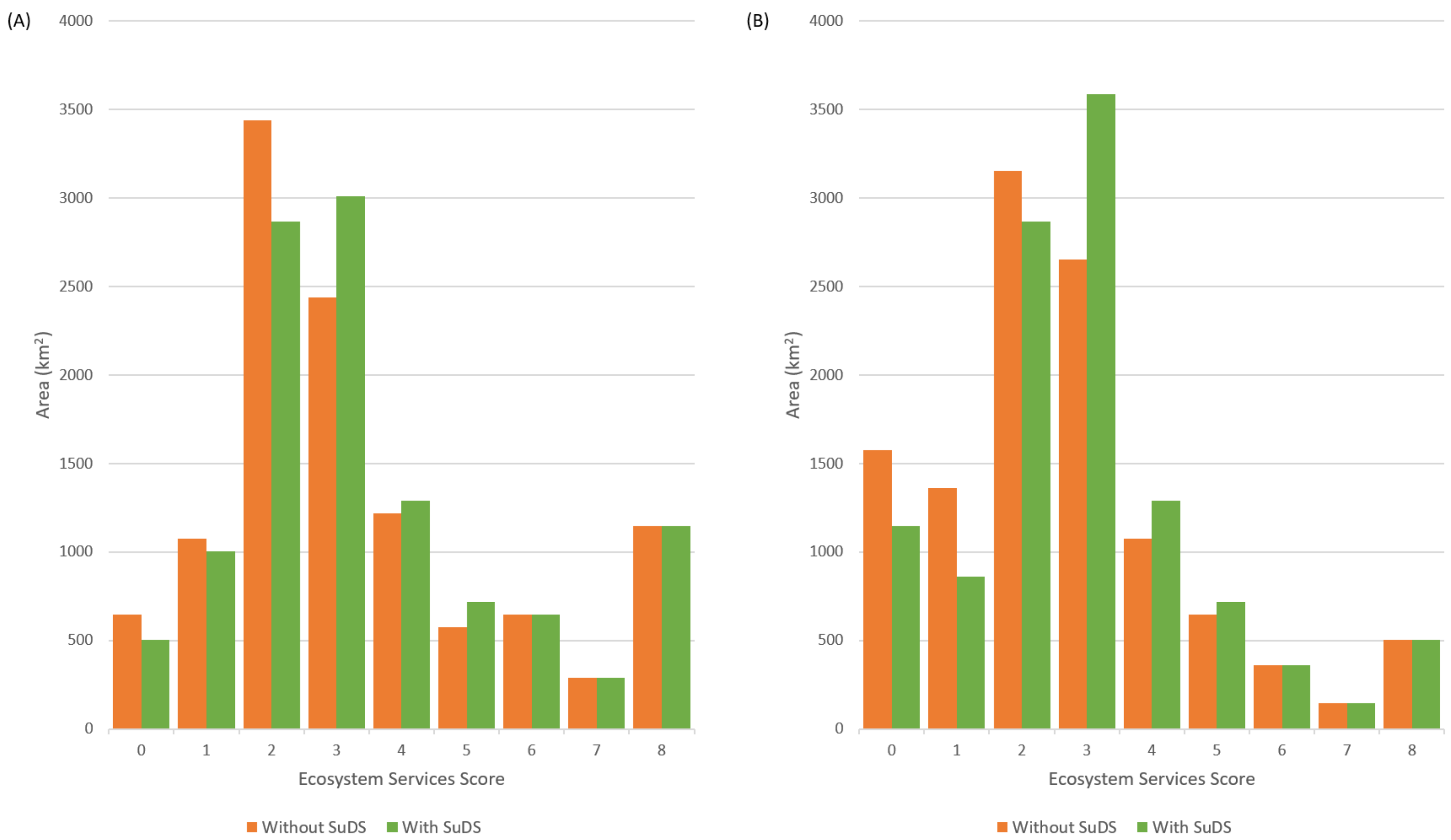

Figure 5.5: Area of ecosystem service scores with and without SuDS for two development scenarios (A - 23GNS, B – 30YE)

The amount of land taken for the grey development scenario over the green of the same magnitude (23,000 new houses per year) is over 50% more, as a greater proportion of development tiles in the green scenarios had higher housing densities, and thus fewer total development tiles were required for the same magnitude. This is evident in the impact on the ecosystem services scores with a greater area showing lower scores, and is particularly noticeable, for example, in the central area of the region around the borders of Berkshire and Buckinghamshire, as this is where much development is proposed in both layout types. Whilst central southern Cambridgeshire sees a similar increase in development scale, however, between development magnitudes, changes in ecosystem services scores are less noticeable due to the relatively lower scores in this region pre-development.

The new settlement approach under a set magnitude and constraint pattern requires a lower footprint of previously undeveloped land than expansion as a greater proportion of development tiles are of high densities to create urban centres, although this difference is relatively small (~10%) and so is less noticeable in ecosystem services scores across the study area.

5.5.1.2: SuDS Implementation

Whilst for all urban typology tiles there is a SuDS scenario which offers the same or an improved ecosystem services score when compared to the same tile without SuDS, this is not consistent across all development or SuDS designs. Furthermore, only rarely was a SuDS design able to raise the ecosystem service score above that had the tile remained undeveloped – notably those where a low scoring undeveloped land type (such as arable or improved grassland) was replaced by an urban tile that was dominated by green roof systems, which offered a higher ecosystem services score. No other SuDS infrastructure scored greater than a natural land use in this study.

As infrastructure-based SuDS (those constructed upon or replacing existing sealed surfaces), the presence of green roofs and permeable surfaces in a SuDS scenario was more likely to increase the overall ecosystem services score for the tile. Conversely, as a freespace SuDS infrastructure (those built upon existing undeveloped land), retention basins (which scored lower than undeveloped land) also reduced the area of undeveloped land contributing to the overall score, and it is in scenarios with this infrastructure present that some tiles saw a reduced ecosystem services score when compared to a non-SuDS scenario. This was not universally the case, however, and those tiles with minimal undeveloped land available (such as the very high density scenarios) saw little impact. That is also not to say that all freespace SuDS will have a similar impact – in areas with poor existing land cover, for example, bioretention systems could have the opposite effect – and thus future work should look to expand the range of SuDS infrastructure and land cover types considered.

The change in ecosystem services scores offered by SuDS are not limited to the changes in permeability they bring about, either. Further co-benefits of SuDS implementation (including habitat creation, leisure opportunities and urban cooling) are all additional services, and these too are captured by the metric. However, challenges in quantifying or comparing the magnitude of some of these in different environments/infrastructure (such as mental health benefits) mean further work is required in the field to improve such measurements and allow a more detailed insight into the relative strengths and weaknesses of different SuDS and their co-benefits.

These impacts of the given SuDS scenario translate directly to the areas of urban development in the study, as this is where the SuDS are located. Whilst an incremental change in ecosystem services score can appear minimal when considering a tile in isolation, when combined with others across an area of development, there is a clearer impact, as shown with permeable surfaces in Figure 5.6. Patterns were less distinct for other SuDS scenarios, such as green roofs, due to their relatively low surface areas at low densities. Due to method design, this meant undeveloped land was the dominant land cover type, and thus low density tiles scored the same as undeveloped tiles (masking their contribution). This highlights the need for future work to develop the approach for applying ecosystem services scores based on land cover.

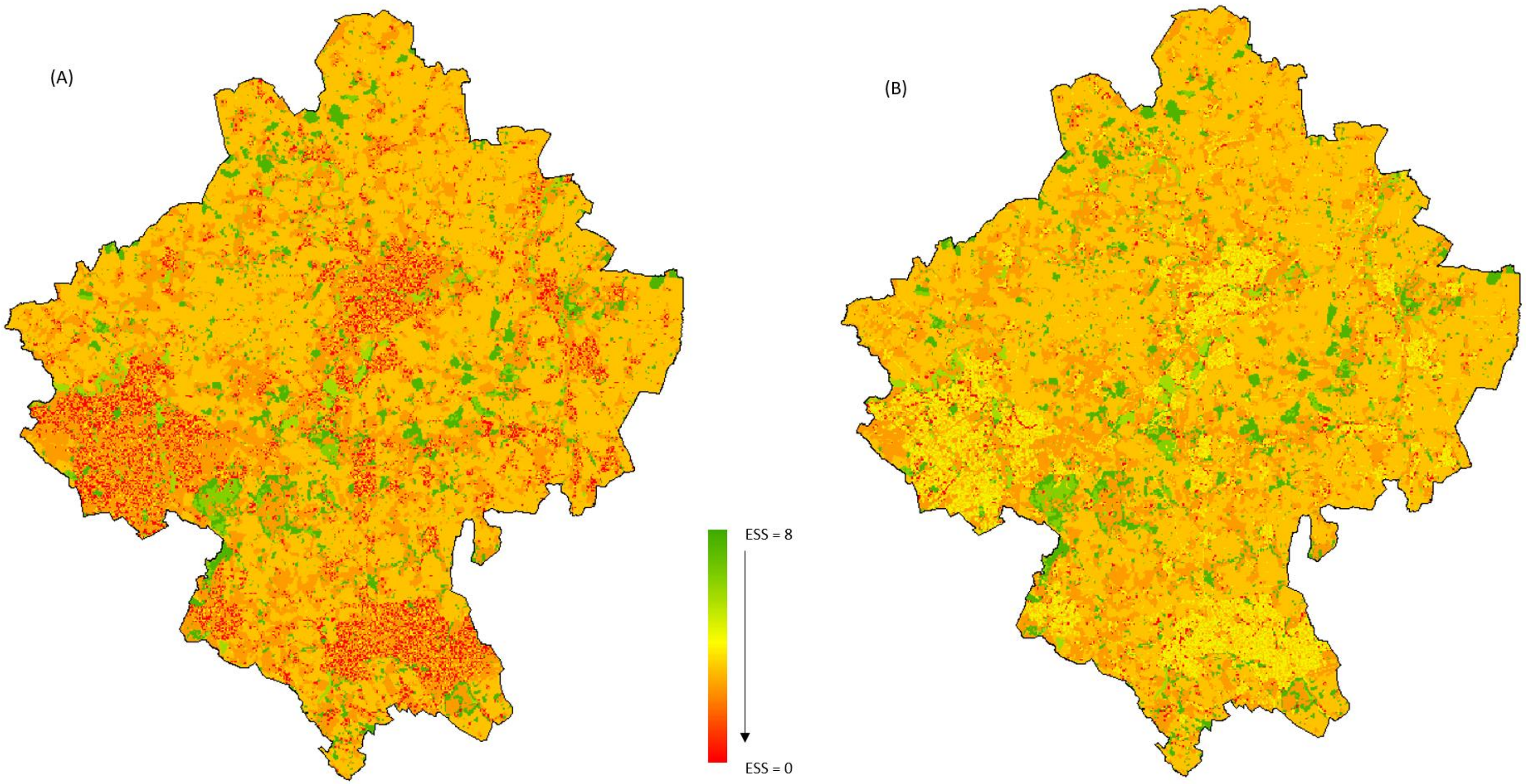

Figure 5.6: Ecosystem services scores for two SuDS scenarios – (A) no SuDS and (B) permeable surfaces – in Bedfordshire for the 23GNS development scenario.

5.5.2: Ecological Status

5.5.2.1: Development Approach

The spatial patterning of scores under this metric are very similar to the first (attributable to the fact that both are based on the same land cover dataset), although contributions from patch size mean that landscape fragmentation contributes to the overall strength or weakness of an area in this metric. Additionally, the variety and extent of ecosystem services a land cover provides is not always directly related to how natural it is perceived to be. Fens and heather, for example, are considered extremely natural (and thus score highly under ecological status) whereas under the averaged ecosystem services, they score similarly to other land classifications. However, they may score highly for specific ecosystem services. In addition, the impact of urban spaces has a greater effect under this metric as the lowest it can score in naturalness is ‘5’, and this only when soil sealing is less than 30%. Thus, the location of urban spaces can be seen much more clearly, as demonstrated in Figure 5.7, with poorly scoring areas clearly correlating with the layout of urban development.

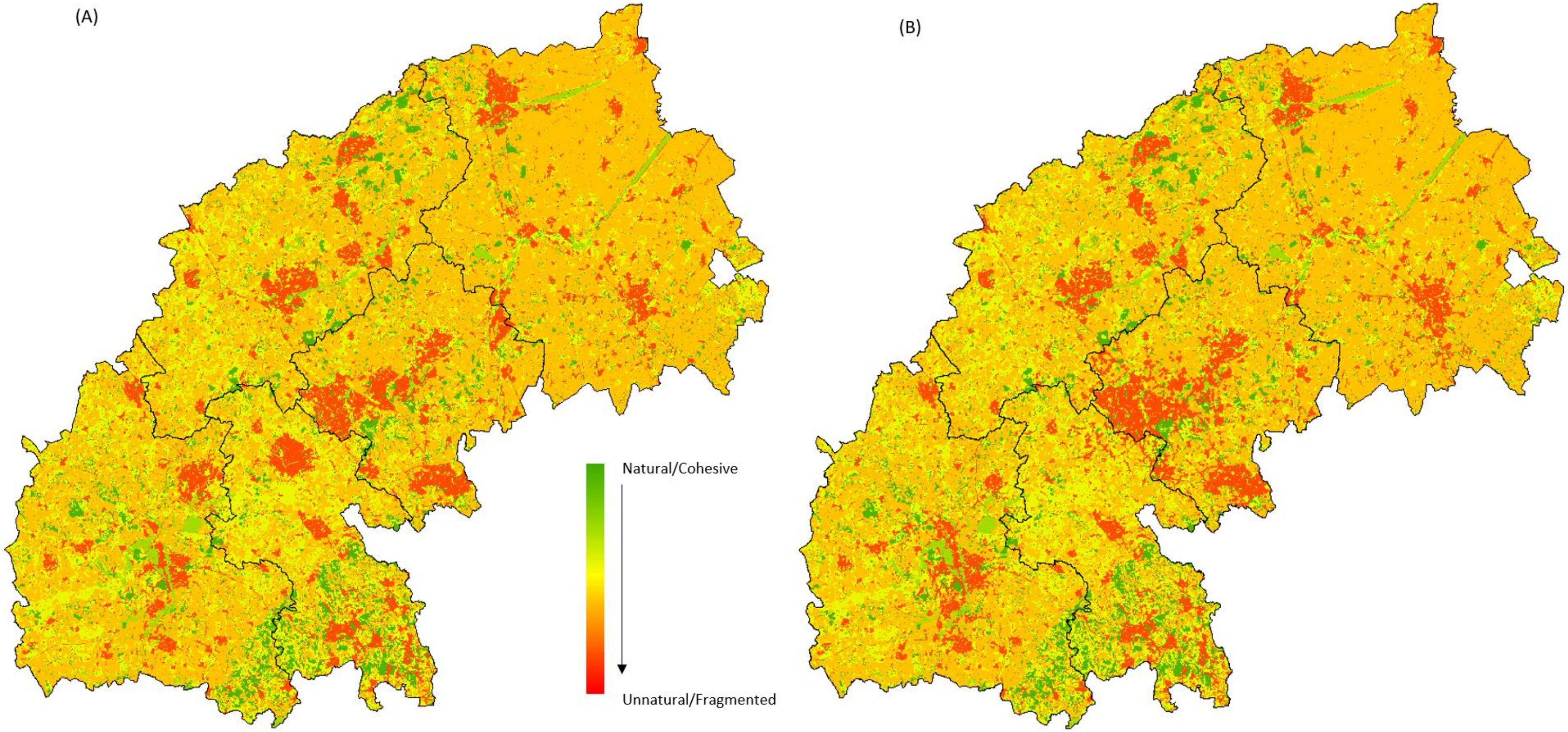

Figure 5.7: Ecological status scores for the two development without SuDS (A – 23GNS, B – 30YE)

The southern area of Buckinghamshire is also of particular note. Under the ecosystem services metric, the area consistently scored relatively high, yet this is much more variable in the status metric, with both top-scoring and bottom-scoring tiles. This represents both a more fragmented landscape and the presence of more land cover types perceived as less natural, and highlights the importance of considering the status of the environment as well as its maximum ecosystem services potential when targeting conservation or environmentally-sensitive development projects.

5.5.2.2: SuDS Implementation

Whilst the inclusion of SuDS and its effect on naturalness scores is limited to a minimum score of ‘5’ (as previously mentioned) due to still remaining an urban location, all tile designs (except two very high density layouts) show an improvement in naturalness score due to increased permeable surface area. This is particularly noticeable in the permeable surfaces scenario as only the trunk road and building areas remain impermeable. Thus, in the overall ecosystem status scores (which also include an appreciation of fragmentation), permeable surfaces see a marked reduction in the area scoring poorly, alongside those featuring green roofs, despite not being necessarily visibly more natural in reality (see Figure 5.8).

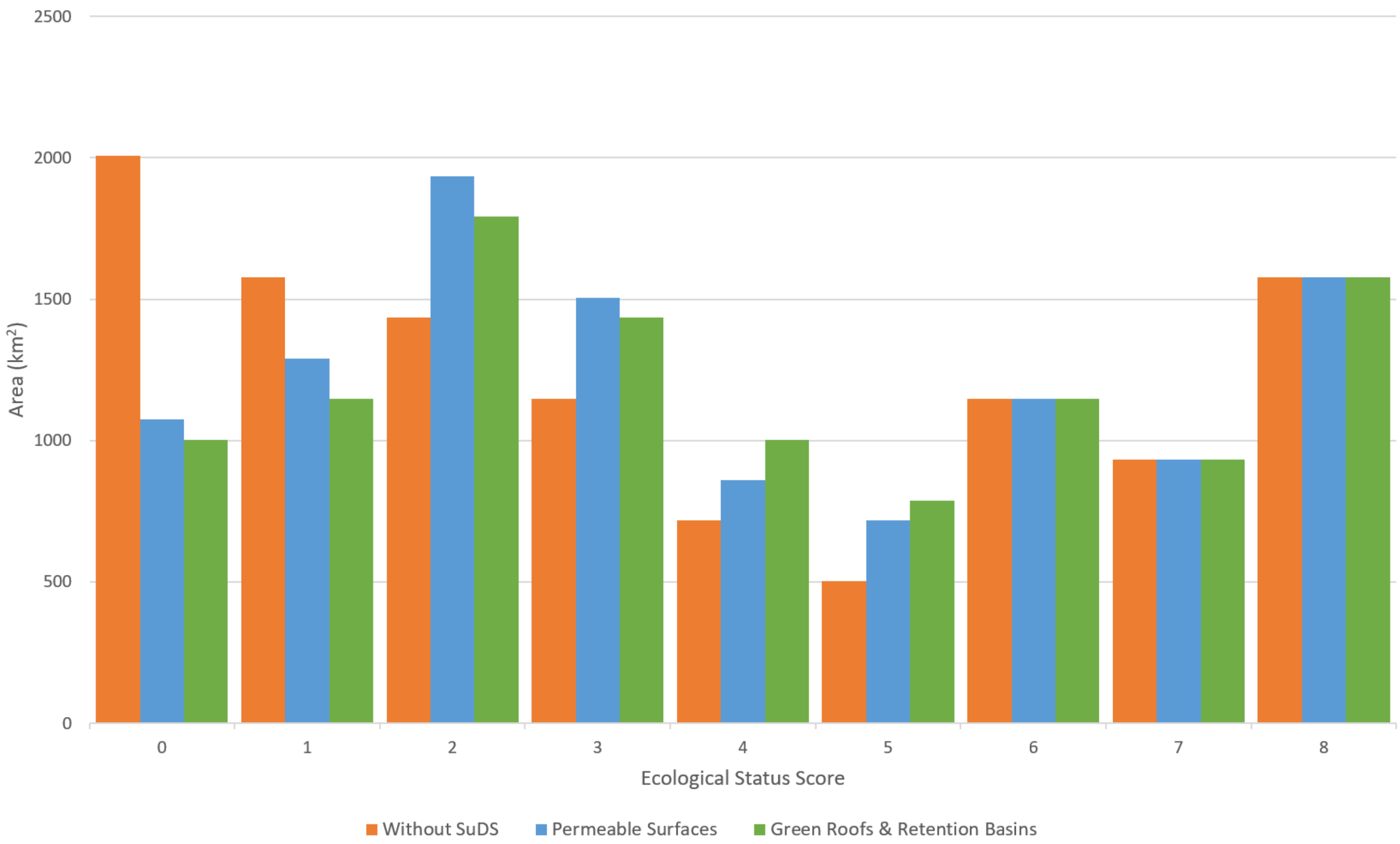

Figure 5.8: Ecological status scores for the 23GE development under three SuDS scenarios

In a similar manner to ecosystem services, scenarios featuring retention basins result in the least impact in improving naturalness scores since they are not constructed on any existing impermeable surface and thus do not reduce the overall impermeable surface area of a scenario. Nonetheless, whilst these infrastructure offer the return or increase of natural hydrological processes, they remain limited in the 'naturalness' due to still being manmade spaces, which is reflected in the small magnitude changes in scores for this metric (which seeks to measure how similar to a natural state these locations are).

5.5.3: Ecological Connectivity

5.5.3.1: Development Approach

Due to the inclusion of some urban tile types in the base areas, these differed between development scenarios, though this difference was fairly minimal as the majority of the base areas were undeveloped parcels in all scenarios. Due to the slight changes in spatial distribution of the development, both green and grey development scenarios saw a slightly different distribution of scores - this difference was more notable, however, between the expansion and new settlements scenario as their contrast in spatial distribution across the study area was greater.

5.5.3.2: SuDS Implementation

Differences between the scenarios were more exaggerated when SuDS were included, too, as some SuDS infrastructures, such as green roofs, provided a greater area of tile that registered as natural (even though these were man-made ‘natural’ spaces). With a greater proportion of urban tiles therefore forming the base tiles for ecological connectivity, distances were reduced and stronger connections across urban areas formed.

However, this is not observed across all SuDS scenarios (see Figure 5.9) as neither permeable surfaces nor retention basins impact those tiles deemed as base sites for connectivity, highlighting the importance of considering individual SuDS types (including those, such as bioswales, not considered in this study). Furthermore, the study provided a generalised overview of GI network potentials, but it is important to note that green roofs, whilst increasing perceived ‘natural’ surface area, would not be accessible for all species (such as land mammals) and thus be an ineffective connector for such genera.

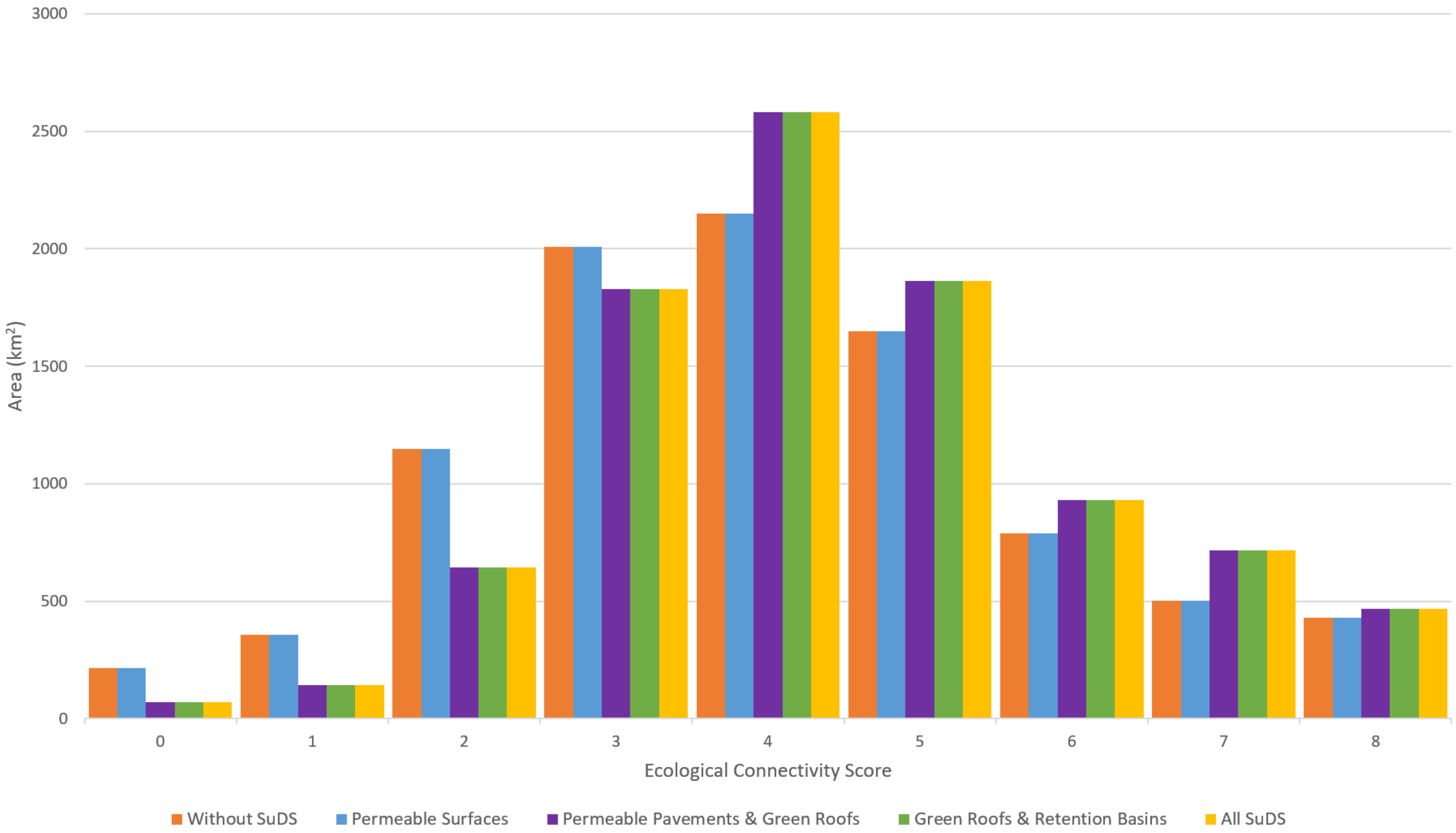

Figure 5.9: Ecological connectivity scores for the 23GNS development scenarios under different SuDS conditions

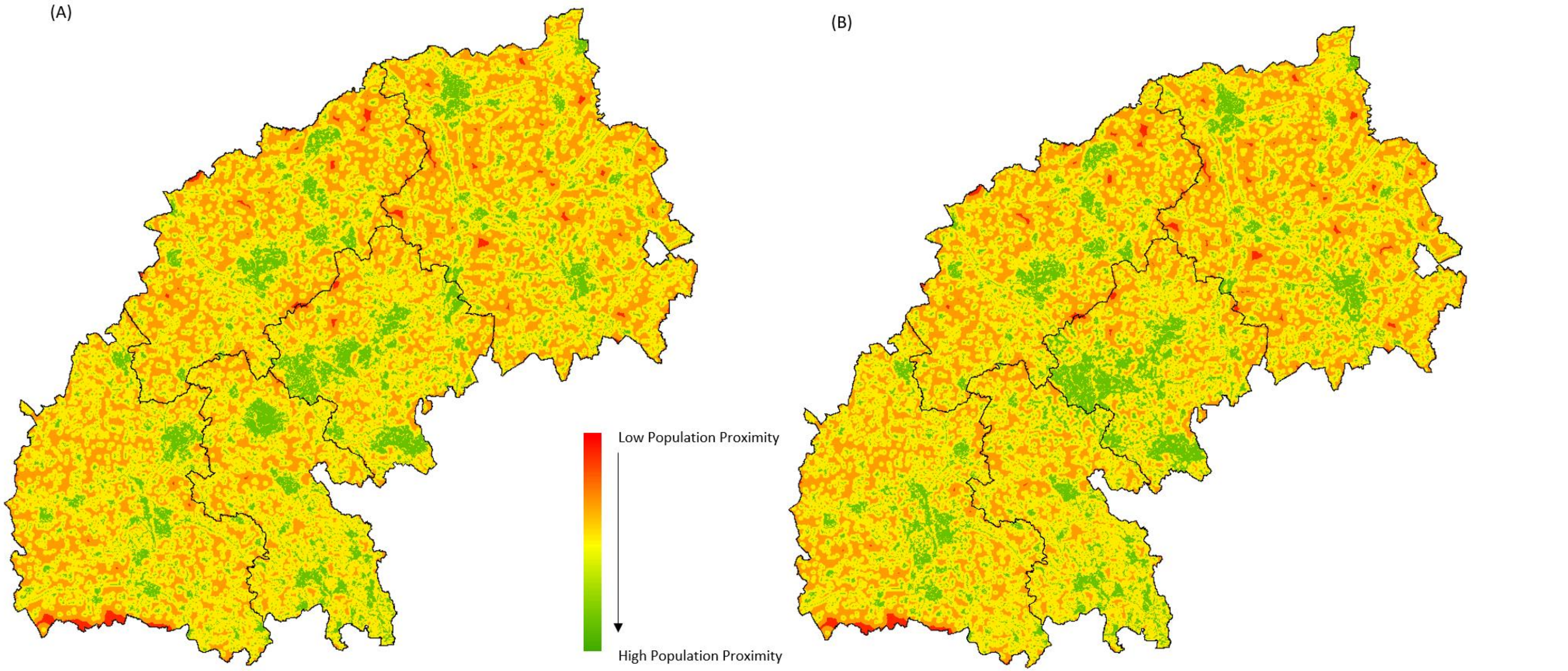

Figure 5.10: Proximity to the population for two development scenarios (A – 23GNS and B - 30YE)

5.5.4: Proximity to the Population

5.5.4.1: Development Approach

A greater overall improvement in scores (i.e. closer to the population) was observed in the expansion scenarios over the new settlements as the urban spaces were more distributed across the study area, whilst the new settlements were more clustered. This was more exaggerated in the greater magnitude developments, too, with the new settlements approach clustering development even more, whilst the expansion scenarios covered more of the study area. The differences in proximity offered by the different scenarios can be seen clearly in northern Buckinghamshire and north-eastern Oxfordshire (see Figure 5.10).

Similarly, the differences observed in both development magnitude and approach were further exaggerated between the green and grey development constraint scenarios. In both development approaches, the green scenarios had more compact urban development whilst the grey were more sprawled (as development clustered around railway stations rather than the more distributed road networks), leading to a wider impact on improving population proximity, as shown in Figure 5.10. In addition, whilst there was proximity to the population across the north of the study area, this remained relatively unaffected by development scenario as no proposed development approach looked to increase urbanisation significantly in this area. It is also worth noting that areas outside the study area were not considered, and so peripheral tiles, such as in Southern Oxfordshire, may score differently due to impacts from urban locales in other neighbouring counties.

5.5.4.2: SuDS Implementation

The presence, or lack thereof, of SuDS had no impact on this metric, and thus there was no difference between the SuDS scenarios for this.

5.5.5: Overall GI Scores

5.5.5.1: Development Approach

With each of the four metrics showed strong spatial patterning, it is no surprise that there are spatial differences in the overall score too. Due to their greater spatial coverage, the expansion scenarios showed a greater spatial spread of higher GI scores, offering greater proximity to the population than the new settlements approaches. However, the increased resultant fragmentation impacted the ecological status in these scenarios, meaning that whilst the spatial distribution of the higher scoring areas was different, the new settlement scenarios generally offered slightly higher overall scores due to reduced fragmentation, supporting better ecological status and connectivity. Since the grey constrained scenarios typically showed greater sprawl, this was more noticeable in these over the green.

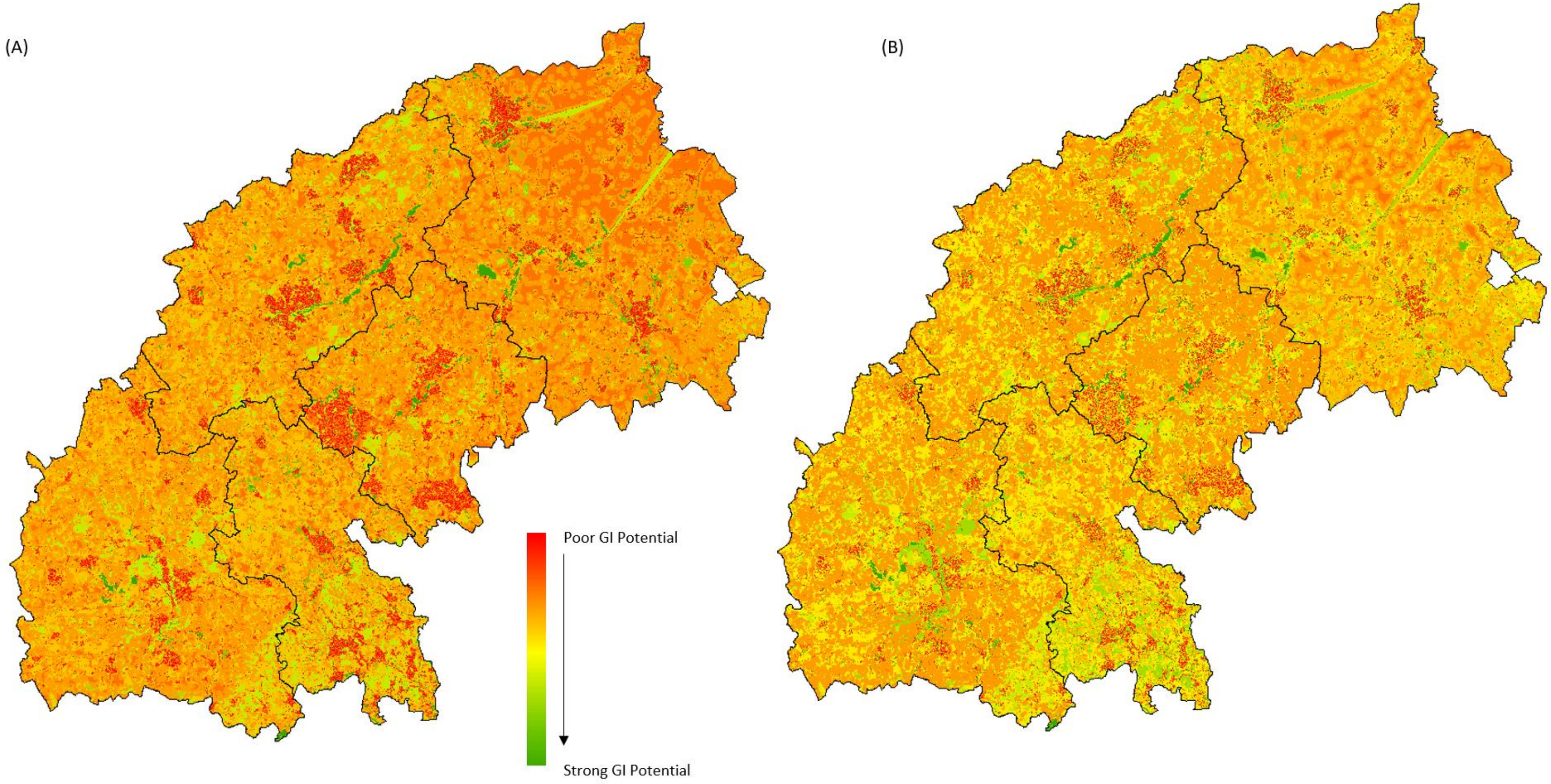

Figure 5.11: The overall score for a development scenario (23GE) without SuDS (A – without and B - with the proximity to population metric)

There was a similar trade-off between the magnitude of development too, with increased development improving the proximity of spaces to the population, but equally fragmenting the landscape and impacting connectivity. However, the contribution of urban spaces was not confined to population proximity, even without SuDS implementation, as Figure 5.11 illustrates. When comparing the overall scores with and without the inclusion of the population proximity metric, there is little change in the spatial pattern of the dominant high scoring areas of GI. This is because some urban tile designs, most particularly those of a low density, scored highly under the other metrics, too.

5.5.5.2: SuDS Implementation

When we include SuDS infrastructure in the proposed developments, these act to improve the scores of urban locations in our GI mapping (see Figure 5.12). However, the extent to which it does so is highly dependent on the SuDS infrastructure involved, as previously discussed, with the infrastructure that are not visibly green or infrastructure-based scoring relatively less well. Nevertheless, since there is not one metric where the inclusion of SuDS consistently worsens the score of urban tiles in terms of GI, the overall scores are consistently higher when compared to the non-SuDS scenario.

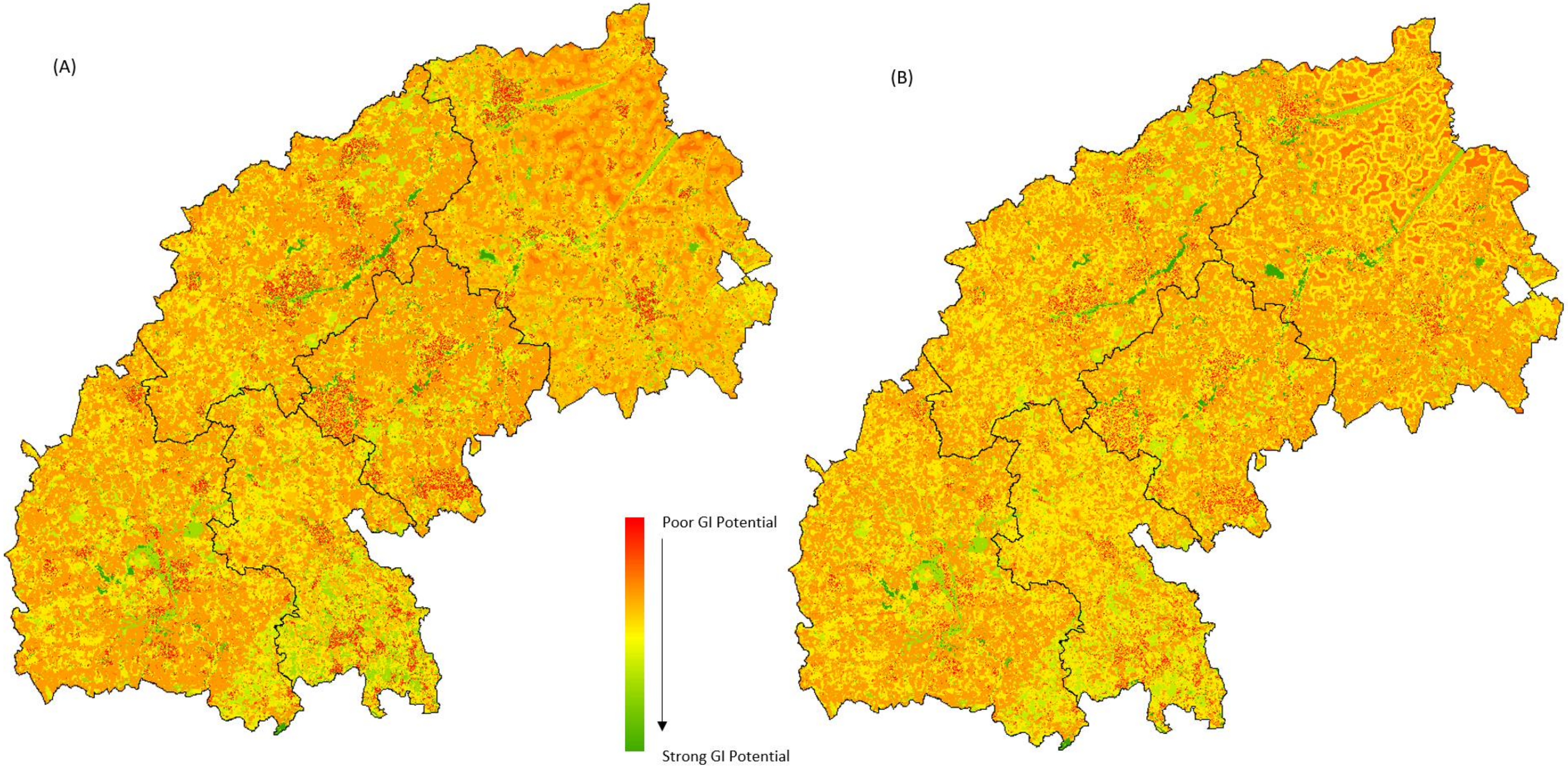

Figure 5.12: The overall score for a development metric (23GE) under two SuDS scenarios (A – without SuDS, B – all SuDS)

Such findings highlight several key considerations when planning development sensitive to the enhancement and preservation of GI networks. First, the contribution urban spaces can have to GI provision, even without the inclusion of SuDS infrastructure. Whilst all urban tile designs scored the minimum value for both ecological services and ecological status without SuDS, the high proportional area of undeveloped space in many of the low density layouts meant that these could be considered as contributors to ecological connectivity. Thus, in urban spaces which were not too dense, connectivity between base areas could be maintained, even where urbanisation interrupted the continuity of these areas.

Second, in both development approaches, there were trade-offs to be balanced in the scale and location of the development. Whilst expansion scenarios allowed good proximity of undeveloped areas to the population to be more widespread, they led to greater fragmentation of the landscape and, if not low density, adversely impacted ecological connectivity too. New settlements, on the other hand, were typically more compact, leading to less landscape fragmentation but equally less spread of population proximity. Furthermore, where these concentrated areas of development were particularly dense, ecological connectivity across them was severed through the lack of urban tiles dominated by undeveloped land.

Third, different SuDS infrastructure (and combinations) offer different contributions to the GI provision, and so need to be carefully considered when planning their use. Whilst, on the whole, these acted to increase a metric's score for the urban tile, this was not always the case, and whilst they improved scores comparative to a non-SuDS urban tile, often these were still lower than would have been scored had there been no development there at all. However, since the need for increased urbanisation has been established, it is equally as important to consider how we can minimise the impact, and with some SuDS scenarios offering ecological connectivity at even high urban densities, this demonstrates the potential that carefully considered SuDS implementations can have in reducing the impact on GI networks, regardless of the other benefits they simultaneously offer in terms of flood mitigation.

5.6: Conclusion

The potential of green infrastructure is increasingly recognised for addressing biodiversity loss and the challenge of providing sustainable, environmentally-sensitive urban developments without considerably compromising natural environmental health (Wild, Henneberry & Gill 2017; Bibri, Krogstie & Karrholm 2020). Despite this popularity, many studies and methods for assessing an area’s suitability consider GI strength largely synonymous with ecosystem services rather than the multi-faceted concept it is, even in the face of growing recognition in biogeography of the importance of ecological connectivity on habitat health.

Rodriguez-Espinosa, Aguilera-Benavente & Gómez-Delgado (2020) outlined a methodology to address this broad spectrum of elements within GI provision, focusing on four key facets: ecosystem services, ecological status, ecological connectivity, and proximity to human populations. This study utilised this subdivision of elements to identify how GI networks may be affected by the size and shape of new development at a regional scale, and what further impact the use of SuDS infrastructure in said developments may have. To do so, eight potential development scenarios were modelled using a UDM for a case study area in south-east England to which different urban layout designs were applied, before being assessed under four metrics (one for each GI element). These scores were then normalised (to allow comparability) and compiled to create an overall score without weighting any elements.

It was found that with greater magnitude development, a greater area saw reduction in ecological services and status scores as more land was required to be developed upon. The extent of this loss for ecosystem services, however, was dependent on the urban tile design as some lower density tiles still offered a dominant undeveloped land use (and these were assigned the same land cover as had they been undeveloped). Similarly, with the grey scenarios utilising more previously-undeveloped land than the green, the score reductions were more widespread in these than their green counterparts. The scale of reductions in score was less noticeable between the two development approaches (new settlement or expansion), but the spatial location of these changes was different due to the different spatial focus of the urbanisation.

Without SuDS, the location of urban development was easily identifiable in the ecological status metric as all designs scored the minimum potential degree of naturalness. The impact of SuDS was limited here, too, as the spaces were still urban locations, even if greater proportions of their area were semi-natural (though artificial). However, some relatively strong areas for ecosystem services scored much lower for ecological status as the landscapes were much more fragmented and/or were not as natural, which highlights the importance of considering a landscape’s ecological status as well as potential contribution to ecosystem services.

The ecological connectivity was, understandably, strongest in areas unaffected by urban development, although even without SuDS some lower density urban layouts were considered as contributors to the ecological network due to their high proportion of undeveloped spaces with an eligible land cover type. When SuDS were introduced, the number of these contributing tile designs increased, allowing urban spaces to score more highly in regards to connectivity. This was not consistent across all SuDS scenarios, however, with retention basins reducing the area of potentially eligible space. Yet it is worth noting that changing the design of such SuDS could alter this ecosystem services provision (such as through natural planting). Furthermore, whilst green roofs are considered by our approach to increase this area, it is important to note that they are not accessible for all taxa, and thus it is also important to consider the types of biodiversity a scheme is aiming to attract or support.

The final metric (proximity to the population) aimed to address the increasing recognition that accessibility is key for many perceived benefits of GI, and was not affected by SuDS implementations. This had higher scores in the expansion and grey scenarios as these both saw increased spatial coverage over their counterparts. This was then further exaggerated by the magnitude of development. However, with urban spaces contributing positively to other GI elements too, this metric largely acted to enhance existing areas of high GI potential rather than create new ones.

When compiled into the single GI score, it is clear to see that each component of the development approach offers its own range of factors which must be considered when planning a development – expansion and grey scenarios show more spread, offering greater proximity to undeveloped green space, but equally therefore lead to a more fragmented landscape, and an increased magnitude of development will increase this difference between approaches even further. Going forward, therefore, we highlight three key considerations for the planning of regional-scale developments that are sensitive to, and/or enhance, GI networks; first, the value and potential contributions of urban spaces to elements of GI even without SuDS infrastructure; second, a balance of the trade-offs presented by each development approach to best achieve the specific aims of the development project; and third, an awareness of the potential impacts different SuDS types and designs will have on the provision of GI, as not all SuDS are created equal.

6. Conclusion

Sustainable drainage systems (SuDS) are often proposed as a potential solution to minimising surface water flood risk, particularly in urban areas – a risk that is expected to increase under current urbanisation pressures and climatic changes. With much work to-date focusing on the optimisation of SuDS within their system design and primarily focusing on the hydrological benefits they could offer, this research looked to understand how their behaviour was affected by built elements in the urban form itself, including co-benefits, and appreciate their potential role at a larger scale. Using rainfall-runoff and spatial modelling tools, a range of drainage infrastructures were assessed in relation to both the hydrological and wider benefits they could offer. It was found that the design of both the infrastructure and the wider development had an impact on their operation and the range and extent of benefits offered.

This chapter looks to provide an overview of the research from the thesis and identify the outcomes of this work. First, the key research findings are summarised. Then, the main contributions to the field are identified, followed by the avenues and opportunities for further work. Finally, the outcomes are looked at in relation to the current policy context, and resultant policy implications are discussed.

6.1: Research Findings

6.1.1: Urban Design & Runoff Dynamics

Urban tiles of 1-hectare were designed, representing different housing typologies, densities, SuDS type and SuDS deployment extent, which were then simulated under a range of design storms using the Stormwater Management Model (SWMM). It was found that impacts on runoff dynamic varied between different SuDS infrastructures. This was due to the different drainage processes and pathways they looked to promote, elements of their individual design, and the area they were able to occupy in different urban designs. All SuDS modelled reduced the time to peak in all rainfall scenarios, but where system storage was full or soil conditions saturated, peak runoff rates were not always reduced. These problems could have been further exacerbated where external environment characteristics, such as the soil type, were different.

Similar tiles were then used to represent multi-typology designs, and these tiles arranged spatially to represent different development designs across the Cambridge to Oxford Arc. SWMM was used again to simulate runoff dynamics for a 1-in-10 year rainfall event. At both lot- and regional-scale, it was found that different housing typologies required different amounts of accompanying urban infrastructure (e.g. streets, pavements) which, combined with different building footprints, led to variations in impermeable surface areas. Runoff dynamics following a simulated rainfall event were strongly influenced by this impermeability, regardless of SuDS intervention, and thus different typologies generated different runoff dynamics. Whilst at lower density scenarios, this meant designs featuring apartments saw greater proportional impermeable surface areas compared to those without, yet as they house a greater population per building unit, this was the opposite in higher density scenarios as fewer buildings were required to achieve “high” and “very high” density conditions.

Regional development design characteristics also influenced the runoff dynamics. Greater peak and total runoff volumes were generated from larger-scale development, due to the increased prevalence of surface sealing. However, these also offered the greatest potential area for infrastructure-based SuDS, and thus greater potential volumes of runoff could be managed by such infrastructure. Higher density scenarios also saw higher runoff volumes, although the relative performance of different housing typologies varied due to their proportional impermeable surface areas. Apartments and terraces, for example, which yielded the highest runoff volumes at low, medium and high densities, generated the lowest runoff volume at a very high density due to the smaller loss of permeable surfaces relative to the other typologies (although it is important to note total dwelling numbers were not consistent across these different typologies). Some of this patterning, however, is masked when considering minimum pipe diameters that would be required to capture peak runoff in these scenarios, with only two of the sixteen housing & density designs showing consistent reductions across all SuDS designs over the non-SuDS scenario.

6.1.2: Urban Form & SuDS Potentials

The housing typology and development density also affected the potential for SuDS inclusion. Greater urban form footprints (such as in a higher density scenario) present greater opportunity for infrastructure-based SuDS (such as green roofs) which are constructed upon other infrastructure, but minimise the opportunities for freespace SuDS (such as bioretention units) which operate on otherwise undeveloped land. Furthermore, specific infrastructure-based SuDS are differently affected by the proportional area of different urban form elements (such as pavements) which are influenced by typology and development design – more compact development, for example, reduces the area of road and pavement required, but also simultaneously reduces the maximum potential area of permeable roads and paving that can be introduced (yet with less hard surface generating less runoff, less permeable roads and paving overall is required).

6.1.3: Development Design & Elements of Green Infrastructure (GI)

A methodology was developed and outlined to assess the impact of different development scenarios on four key elements of green infrastructure (GI) provision – ecosystem services, ecological status, ecological connectivity, and proximity to the population. It was found that larger developments saw a greater loss of ecosystem service provision, but that the magnitude of this loss was determined by urban design. Where SuDS were implemented and/or there was a high proportion of undeveloped land, the extent of this loss was minimised as some services could still be provided. The provision of green roofs on buildings, for example, offered some services that traditional roofs would not, such as habitat provision, even if this level of provision was less than pre-development levels. Other design criteria, such as the layout of the development, had a more spatially-specific impact, with service losses centred around areas of urbanisation. The spatial distribution of impacts, however, is just as important as the impact magnitudes when planning urban development designs.

As with ecosystem services, the inclusion of SuDS helped to minimise the impacts of urban development on the ecological status. However, there was a limit to this impact, even under the most idealised of conditions, since the artificial greenspace or other created environment was still only considered semi-natural. This reinforces the argument that whilst urban spaces can be improved to better support natural processes and minimise impacts, there is no completely recreating a natural landscape, and so where possible the best development for nature is no development at all (Rubiera-Morollon & Garrido-Yserte 2020).

On the other hand, at low densities, it was found that urban spaces had the potential to contribute to ecological connectivity. Some urban designs with a large area of undeveloped land were considered to offer the required characteristics to support ecological connections, and the range of designs and densities this included increased with SuDS implementation. However, of the SuDS modelled, this was not the case for retention basins, and the effectiveness of other infrastructure would vary by species/genus investigated.

The final individual metric for development GI provision – population proximity to green space – was unaffected by the inclusion of SuDS as these infrastructures did not affect population distributions. Development design, however, influenced this metric with the more spatially-spread development scenarios, involving expansion of existing settlements, showing a more widespread distribution of high scoring areas. Yet, due to the performance of urban spaces under the other metrics, this acted to enhance existing strongpoints for GI provision in development rather than simply introduce new ones or cancel out patterns.

When combined into a single score, the trade-offs between different development approaches, designs and SuDS implementation were visible. For example, grey and expansion scenarios saw more widely-spread benefits across the case study area, offering more proximity to green space, but were equally created a more fragmented landscape. These patterns were further amplified by increased magnitudes of development. The different relative performances of the scenarios under the different metrics exemplified the falsehood of “one size fits all” in urban development design, highlighting the need to carefully match the aims of a development with the approach on the ground.

6.2: Research Contributions

Prior to the work of this thesis, there was limited systemic understanding of how elements of built urban form influenced the potentials for, and performances of, different SuDS infrastructures. The methodological approach, which combined urban development modelling, the use of urban tiles for small-scale representation of different drainage infrastructure and urban form layouts, and rainfall-runoff modelling using SWMM (an established model in the field), enabled the simulation of different urban environments, built forms, and SuDS combinations at both local and regional scales. It was shown that both runoff dynamics and SuDS potentials are affected by built form elements, and most crucially their proportional areas of different surface types. Furthermore, it was found that the relationship between development density and the extent of SuDS implementation was not straightforward as in some high-density scenarios a lower proportional surface area of SuDS implementation was required to achieve the same (or greater) reduction in peak runoff compared to their lower density counterparts (although this was due to these still offering a great actual area of SuDS infrastructure). Furthermore, offering consideration of pipe diameters and pipe lengths in different designs, the impact of these runoff reductions on pipe infrastructure requirements could also be observed.

Similarly, wider elements of development design were considered, and in particular how these influenced both the hydrological and non-hydrological performance of different SuDS infrastructures. Whilst this had previously seen more work in the literature than urban built form elements, much analysis had been at a localised scale with comparison between a limited number of situations. In addition, due to their difficulty to quantify, many previous studies into SuDS co-benefits focused on a single or small selection of co-benefits. This research saw investigation of a wider range of development design elements (i.e. magnitude, spatial pattern of development, housing density, housing types, SuDS type, environmental constraints), finding that whilst some elements affected the magnitude of benefit provision, others affected the spatial distribution. It also highlighted the trade-offs present between different elements which, unless adequately balanced, could weaken and undermine otherwise good planning practices.

This research also introduced a new methodology for assessing and cross-comparing co-benefit provision of SuDS. Previously, approaches have required expert knowledge, specialised datasets and/or have focused on a limited number of elements. The outlined approach, however, has no such requirements, using data which is commonly collected in many countries, and some which can also be extracted from existing global datasets. Using a series of metrics, the focus is also not fixed on a particular element (e.g. health benefits) but equally can introduce weighting to facilitate a tailored approach for a specific set of development goals. Furthermore, the method for each measure is already an established approach in the relevant literature, providing a level of robustness to the methodology. As these approaches are further developed in the future, so too can this method appreciate such changes and refinement.

6.3: Recommendations for Future Research

With a focus on the influence of built form elements, the SuDS infrastructure designs considered in the research were informed by best practice guidelines outlined by CIRIA (2015) and the Department for Transport (2007). However, as has been identified by previous research, individual infrastructure design has a strong influence on drainage and co-benefit provision. Future work, therefore, should look to consider designs beyond those of current best practice to identify how these changes may affect hydrological and co-benefit provision in the different urban form scenarios, such as through variation in vegetation types used in bioretention or green roofs.

In addition, due to the scope of the research, there are SuDS infrastructures not investigated by this research, such as artificial wetlands and bioswales, as well as some combinations of multiple SuDS infrastructures. Due to the infrastructure-specific nature of the results from this research, future work should be undertaken to include these other SuDS infrastructures and combinations as they will offer their own extent and range of benefits, and may prove more suitable for a future development. Similarly, as spatial characteristics (such as soil type and rainfall intensity) are influential on SuDS behaviour, additional work should look to investigate these relationships in a range of locations covering different climatic and topological conditions, and across even greater scales to provide a better overall appreciation of SuDS potentials. Furthermore, additional consideration of vegetation types (both within SuDS designs and the wider urban environment) would be useful.

A key finding of this research was the non-linear relationship between development density and the extent of SuDS implementation. This, too, is both specific to urban design and SuDS design, and thus a wider range of multi-form developments should be considered. Future research should look to identify a specific housing density threshold for different development types and SuDS combinations, as this will be a useful benchmark for urban planners, as well as to establish whether such results are affected by different climatic conditions and further design storm events.

Whilst this research considered both urban form elements and larger-scale development designs, a uniform layout was used for each urban design, focusing development around a central linear road. As such, different spatial layouts of urban elements were not considered, such as poly-centric or nucleated neighbourhood designs. Given that these will lead to different proportional impermeable surface layouts, it would be prudent for future work to consider the impact of such spatial variations due to the observed influence of impermeable surface areas in this work. In a similar vein, the modelled urban spaces in this research were solely residential, and thus further research should also look to include the non-domestic building requirements of new developments (such as schools and hospitals) as well as more detailed elements of the domestic units, including features like driveways and private gardens.

SWMM was consistently used throughout this research when modelling runoff dynamics due to its established standing in the field and inclusion of separate modules targeted for SuDS modelling. However, the range of modules available is limited, and many require a significant number of input parameters. Furthermore, due to their relatively new introduction, it is also these that have been subject to mixed feedback recently from researchers, with several studies questioning the sensitivity of various modules (such as Peng & Stovin 2017 and Randall et al. 2020). Refinement of these SuDS modules and an expansion of the range of infrastructure covered would thus be welcomed future developments. As the research was modelling-based throughout the study, too, it would be beneficial for future work to validate the results and conclusions in the field. In so doing, this would also allow limitations presented by the SuDS modules to be identified.

6.4: Policy Implications

The findings of the research highlighted several key planning considerations for the design and implementation of urban developments. First, the impact of the layout of a development. As has been heavily discussed in the literature, there is a balance to be reached between promoting compact city development and urban sprawl, such that greenspace can be provided within urban space without significant negative impact on surrounding environments through large urban footprints. This research has shown the importance of built form elements in relation to surface runoff generation and some of these elements (e.g. streets) will also be required to connect more spatially-spread settlements. This once again highlights the need for a balance as the wider the footprint, the greater the volume of interconnection required, yet equally the greater potential space for freespace SuDS infrastructures. Additionally, in less dense developments, SuDS are more likely to be able to capture a rainfall event (as the surface runoff produced is lower), reducing the need for (or size of) additional piped drainage infrastructure costs. Current planning guidance allows developers to use high costs as a reason for not including SuDS in developments (Vilcan & Potter 2020). Yet, such findings from the study indicate that some designs can generate financial savings elsewhere, and these should be considered before discounting SuDS on an economic basis.

Second, the importance of examining multiple different types of housing typologies. This should be done within the context of the proposed development as both spatial characteristics (such as soil type or slope) and specific development criteria (such as density or additional environmental aims) affect the resultant SuDS implementation and performance (whether in relation to drainage or GI provision). Whilst denser and wider scale urban developments generate increased surface runoff, the increased potential for infrastructure-based SuDS implementation can lead to a greater proportional capture of runoff by SuDS than in lower density developments, reducing piped drainage requirements and, in some cases, increasing co-benefit provision (although this is heavily dependent on undeveloped land cover type).

Third, the requirement to compare multiple forms and types of SuDS within a development as they, too, are influenced by contextual characteristics, and different SuDS infrastructures are as unique as (or even more so) than housing typologies. Wetland creation and green roofs, for example, can both provide new habitats, yet being located at ground-level makes the former more accessible for leisure and aesthetic co-benefits, whilst as an infrastructure-based SuDS, the latter offers greater surface permeability creation in comparison to freespace SuDS and requires less free space in a development. Furthermore, the type of SuDS used in a development could be tailored to the needs of specific priority species or taxa. Admittedly, however, as Vilcan & Potter (2020) identify, a lack of clarity in design guidance and construction principles generates uncertainty and often requires expert knowledge or experience to develop a potential SuDS intervention for a development, although it is widely acknowledged that the SuDS Manual (CIRIA 2015) provides a good overview. This situation should thus be improved to better enable a comprehensive consideration of SuDS solutions by developers and planners without it being a too costly and onerous exercise.

Fourth, the opportunities posed by urban spaces (even without SuDS) in achieving additional ecosystem benefits through GI provision. Building on the points previously mentioned, with a carefully thought-out development elements of GI provision can be achieved that minimise the loss of existing GI through urbanisation practices, and in some locations increase particular elements (although it should be noted that no situation was found in the research that showed consistent improvement in all assessed elements of GI when urbanising a previously undeveloped area). This echoes conclusions from the most recent Adapting to Climate Change Report (CCC 2021) for the UK, which suggests that multiple environmental benefit provision is still under-incentivised.

Considered a method for reducing surface water flood risk whilst providing a range of co-benefits, SuDS offer much potential for helping to minimise the hydrological and environmental impacts of urban development. However, as has been demonstrated in this study, there is no 'one size fits all' approach as the behaviour of SuDS is influenced by both the design of the development and the spatial context into which the development is set. Going forward, therefore, the design of both development and SuDS should be assessed within the proposed development setting to better ascertain how different design principles can be best used to achieve the desired outcomes of the specific project – it is not just about what we build, but how and where we build it.

7. References

Ahern, J. (2013). Urban landscape sustainability and resilience: the promise and challenges of integrating ecology with urban planning and design. *Landscape Ecology*. 28, 1203-1212. doi: 10.1007/s10980-012-9799-z

Ahmadian, E., Sodagar, B., Bingham, C., Elnokaly, A., and Mills, G. (2021). Effect of urban built form and density on building energy performance in temperate climates. *Energy and Buildings*. 236, 110762. doi: 10.1016/j.enbuild.2021.110762

Ahiablame, L., Engel, B.A. and Chaubey, I. (2012). Effectiveness of Low Impact Development Practices: Literature Review and Suggestions for Future Research. *Water, Air & Soil Pollution*. 223, 4253-4273. doi: 10.1007/s11270-012-1189-2

Ahiablame, L. and Shakya, R. (2016). Modelling flood reduction effects of low impact development at a watershed scale. *Journal of Environmental Management*. 171, 81-91. doi: 10.1016/j.envsoft.2012.04.011

Albino, V., Berardi, U. and Dangelico, R.M. (2015). Smart Cities: Definitions, Dimensions, Performance, and Initiatives. *Journal of Urban Technology*. 22, 3-21. doi: 10.1080/10630732.2014.942092

Alexander, M., Priest, S., Micou, A.P., Tapsell, S., Green, C., Parker, D. and Homewood, S. (2016). *Analysing and evaluating flood risk governance in England – Enhancing societal resilience through comprehensive and aligned flood risk governance*. STAR-FLOOD Consortium, Flood Hazard Research Centre, Middlesex University. ISBN 978-94-91933-07-3.

Algador, D., Triviño, M., Cerdeira, J.O., Brás, R., Cabeza, M., and Araújo, M.B. (2012). Linking like with like: optimising connectivity between environmentally-similar habitats. *Landscape Ecology*. 27, 291-301.
doi: 10.1007/s10980-012-9704-9

Alves, A., Gersonius, B., Kapelan, Z., Vojinovic, Z., and Sanchez, A. (2019). Assessing the co-benefits of green-blue-grey infrastructure for sustainable urban flood risk management. *Journal of Environmental Management*. 239, 244-254.
doi: 10.1016/j.jenvman.2019.03.036

Akpınar, A. (2019). Green Exercise: How Are Characteristics of Urban Green Spaces Associated with Adolescents' Physical Activity and Health? *International Journal of Environmental Research and Public Health*. 16(21), 4281.
doi: 10.3390/ijerph16214281

Anim, D.O., Fletcher, T.D., Pasternack, G.B., Vietz, G.J., Duncan, H.P. and Burns, M.J. (2019). Can catchment-scale urban stormwater management measures benefit the stream hydraulic environment?. *Journal of Environmental Management*. 233, 1-11. doi: 10.1016/j.jenvman.2018.12.023

Arjekani, M.O., Sanayei, H.R.Z., Heidarzadeh, H. and Mahabadi, N.A. (2020). Modeling and investigating the effect of the LID methods on collection network of urban runoff using the SWMM model (case study: Shahrekord City). *Modeling Earth Systems & Environment*. doi: 10.1007/s40808-020-00870-2

Arnbjerg-Nielsen, K., Willems, P., Olsson, J., Beecham, S., Pathirana, A., Gregersen, I.B., Madsen, H. and Nguyen, V.T.V. (2013). Impacts of climate change on rainfall extremes and urban drainage systems: a review. *Water Science & Technology*. 68(1), 16-28. doi: 10.2166/wst.2013.251

Aronica, G.T. and Lanza, L.G. (2005). Drainage efficiency in urban areas: a case study. *Hydrological Processes*. 19(5), 1105-1119. doi: 10.1002/hyp.5648

Aronson, M.F.J., Lepczyk, C.A., Evans, K.L., Goddard, M.A., Lerman, S.B., MacIvor, J.S., Nilon, C.H. and Vargo, T. (2017). Biodiversity in the city: key challenges for urban green space management. *Frontiers in Ecology and the Environment*. 15(4), 189-196. doi: 10.1002/fee.1480

Artmann, M., Inostroza, L., and Fan, P. (2019). Urban sprawl, compact urban development and green cities. How much do we know, how much do we agree? *Ecological Indicators*. 96(2), 3-9. doi: 10.1016/j.ecolind.2018.10.059

Artmann, M., Kohler, M., Meinel, G., Gan, J., and Ioja, I.C. (2019). How smart growth and green infrastructure can mutually support each other — A conceptual framework for compact and green cities. *Ecological Indicators*. 96, 10-22. doi: 10.1016/j.ecolind.2017.07.001

Ashley, R., Lundy, L., Ward, S., Shaffer, P., Walker, L., Morgan, C., Saul, A., Wong, T. and Moore, S. 2013. Water-sensitive urban design: Opportunities for the UK. *Municipal Engineer*. 166(2), 65-76. doi: 10.1680/muen.12.00046

Ashley, R., Walker, A.L., D’Arcy, B., Wilson, S., Illman, S., Shaffer, P., Woods-Ballard, B. and Chatfield, P. (2015). UK sustainable drainage systems: past, present and future. *Proceedings of ICE – Civil Engineering*. 168(3), 125-130. doi: 10.1680/cien.15.00011

Ashley, R., Gersonius, B. and Horton, B. (2020). Managing flooding: from a problem to an opportunity. *Philosophical Transactions of the Royal Society A: Mathematical, Physical & Engineering Sciences*. 378. doi: 10.1098/rsta.2019.0214

Bach, P.M., Deletic, A., Urich, C., Sitzenfrei R., Kleidorfer, M., Rauch, W. and McCarthy, D.T. (2013). Modelling Interactions Between Lot-Scale Decentralised Water Infrastructure and Urban Form – a Case Study on Infiltration Systems. *Water Resources Management*. 27, 4845-4863. doi: 10.1007/s11269-013-0442-9

Bach, P.M., Rauch, W., Mikkelsen, P.S., McCarthy, D.T. and Deletic, A. (2014). A critical review of integrated urban water modelling – Urban drainage and beyond. *Environmental Modelling & Software*. 54, 88-107. doi: 10.1016/j.envsoft.2013.12.018

Baek, S.S., Choi, D.H., Jung, J.W., Lee, H.J., Lee, H., Yoon, K.S. and Cho, K.H. (2015). Optimising low impact development (LID) for stormwater runoff treatment in urban area, Korea: Experimental and modeling approach. *Water Research*. 86(1), 122-131. doi: 10.1016/j.watres.2015.08.038

Baek, S.S., Ligaray, M., Pachepsky, Y., Chun, J.A., Yoon, K.S., Park, Y. and Cho, K.H. (2020). Assessment of a green roof practice using the coupled SWMM and HYDRUS models. *Journal of Environmental Management*. 261, 109920. doi: 10.1016/j.jenvman.2019.109920

Baguette, M., Blanchet, S., Legrand, D., Stevens, V.M. and Turlure, C. (2013). Individual dispersal, landscape connectivity and ecological networks. *Biological Reviews*. 88, 310-326. doi: 10.1111/brv.12000

Ball, J.E. and Rankin, K. (2010). The hydrological performance of a permeable pavement. *Urban Water Journal*. 7(2), 79-90. doi: 10.1080/15730620902969773

Barbosa, A.E., Fernandes, J.N. and David, L.M. (2012). Key issues for sustainable urban stormwater management. *Water Research*. 46(20), 6787-6798. doi: 10.1016/j.watres.2012.05.029

Bastien, N., Arthur, S., Wallis, S. and Scholz, M. (2010). The best management of SuDS treatment trains: a holistic approach. *Water Science & Technology*. 61(1), 263-272. doi: 10.2166/wst.2010.806

Benvenuti, S. (2014). Wildflower green roofs for urban landscaping, ecological sustainability and biodiversity. *Landscape & Urban Planning*. 124, 151-161. doi: 10.1016/j.landurbplan.2014.01.004

Bernet, D.B., Prasuhn, V. and Weingartner, R. (2017). Surface water floods in Switzerland: what insurance claim records tell us about the damage in space and time. *Natural Hazards Earth Systems Sciences*. 17, 1659-1682. doi: 10.5194/nhess-17-1659-2017

Bevan J. (2018). *Surface water: The biggest flood risk of all*. Available at: https://www.gov.uk/government/news/surface-water-the-biggest-flood-risk-of-all (date accessed: 21 February 2022)

Beven, K.J. (2012). *Rainfall-Runoff Modelling: The Primer*. Oxford, UK: Wiley-Blackwell.

Bibri, S.E., Krogstie, J., and Karrholm, M. (2020). Compact city planning and development: Emerging practices and strategies for achieving the goals of sustainability. *Developments in the Built Environment*. 4, 100021. doi: 10.1016/j.dibe.2020.100021

Bolliger, J. and Silbernagel, J. (2020). Contribution of Connectivity Assessments to Green Infrastructure (GI). *International Journal of Geo-Information*. 9(4), 212. doi: 10.3390/ijgi9040212

Bos, M.G., Murray-Rust, D.H., Merrey, D.J., Johnson, H.G. and Snellen, W.B. (1993). Methodologies for assessing performance of irrigation and drainage management. *Irrigation & Drainage Systems*. 7(4), 231-261. doi: 10.1007/BF00881553

Boulton, C., Dedekorkut-Howes, A., Holden, M., & Byrne, J. (2020). Under pressure: Factors shaping urban greenspace provision in a mid-sized city. *Cities*. 106, 102816. doi: 10.1016/j.cities.2020.102816

Brachet, A., Schiopu, N. and Clergeau, P. (2019). Biodiversity impact assessment of building's roofs based on Life Cycle Assessment methods. *Building and Environment*. 158, 133-144. doi: 10.1016/j.buildenv.2019.04.014

British Standards Institution. (2018). BS EN 13476-1:2018 Plastics piping systems for non-pressure underground drainage and sewerage - structured-wall piping systems of unplasticized poly (vinyl chloride) (PVC-U), polypropylene (PP) and polyethylene (PE). General requirements and performance characteristics (Incorporating corrigenda October 2019). Available at: https://www.thenbs.com/PublicationIndex/documents/details?DocID=322323 (date accessed: 15 December 2022)

Brody, S., Kim, H. and Gunn, J. (2013). Examining the Impacts of Development Patterns on Flooding on the Gulf of Mexico Coast, *Urban Studies*. 50(4), 789-806. doi: 10.1177/0042098012448551

Bubeck, P., Kreibich, H., Penning-Rowsell, E.C., Botzen, W.J.W., de Moel, H. and Klijn, F. (2015). Explaining differences in flood management approaches in Europe and in the USA – a comparative analysis. *Journal of Flood Risk Management*. doi: 10.1111/jfr3.12151

Campisano, A., Catania, F.V. and Modica, C. (2017). Evaluating the SWMM LID Editor rain barrel option for the estimation of retention potential of rainwater harvesting systems. *Urban Water Journal*. 14(8), 876-881. doi: 10.1080/1573062X.2016.1254259

Cannas, I., Lai, S., Leone, F. and Zoppi, C. (2018). Green Infrastructure and Ecological Corridors: A Regional Study Concerning Sardinia. *Sustainability*. 10(4), 1265. doi: 10.3390/su10041265

Caparros-Midwood, D., Barr, S. and Dawson, R. (2017). Spatial optimisation of future urban development with regards to climate risk and sustainability objectives. *Risk Analysis*. 37(11), 2164-2181. doi: 10.1111/risa.12777

Carter, J.G., Cavan, G., Connelly, A., Guy, S., Handley, J. and Kazmierczak, A. (2015). Climate change and the city: Building capacity for urban adaptation. *Progress in Planning*. 95, 1-66. doi: 10.1016/j.progress.2013.08.001

Centre of Ecology & Hydrology UK (CEH). (2020). *UK CEH Land Cover Maps*. Available at: https://www.ceh.ac.uk/ukceh-land-cover-maps (date accessed: 29 November 2020)

Charlesworth, S.M. (2010). A review of the adaptation and mitigation of global climate change using sustainable drainage in cities. *Journal of Water and Climate Change*. 1(3), 165-180. doi: 10.2166/wcc.2010.035

Chatzimentor, A., Apostolopoulou, E. and Mazaris, A.D. (2020). A review of green infrastructure research in Europe: Challenges and opportunities. *Landscape and Urban Planning*. 198, 103775. doi: 10.1016/j.landurbplan.2020.103775

Chow, M.F., Yusop, Z. and Toriman, M.E. (2012). Modelling runoff quantity and quality in tropical urban catchments using Storm Water Management Model. *International Journal of Environmental Science and Technology*. 9(1), 737-748. doi: 10.1007/s13762-012-0092-0

Cipolla, S., Maglionico, M. and Stojkov, I. (2016). A long-term hydrological modelling of an extensive green roof by means of SWMM. *Ecological Engineering*. 95, 876-887. doi: 10.1016/j.ecoleng.2016.07.009

CIRIA. (2015). *The SuDS Manual (Version 6)*. London, UK: CIRIA.

CivilWeb (2016) *Rainfall and Runoff Calculator Spreadsheet*. Available online at: https://civilweb-spreadsheets.com/drainage-design-spreadsheets/runoff-and-rainfall-intensity-calculator-spreadsheet/ (date downloaded: 3 June 2020)

Climate Change Committee (CCC). (2021). *Progress in Adapting to Climate Change: 2021 Report to Parliament*. Available online at: https://www.theccc.org.uk/publication/2021-progress-report-to-parliament/ (date published: 24 June 2021)

Cotterill, S. and Bracken, L.J. (2020). Assessing the effectiveness of sustainable drainage systems (SuDS): interventions, impacts and challenges. *Water*. 12(11), 3160. doi: 10.3390/w12113160

Cranfield University. (2021). *Soils Data © Cranfield University (NSRI) and for the Controller of HMSO [2021]*

da Schio, N., Phillips, A., Fransen, K., Wolff, M., Haase. D., Ostoić, S.K., Živojinović, I., Vuletić, D., Derks, J., Davies, C. *et al.* (2021). The impact of the COVID-19 pandemic on the use of and attitudes towards urban forests and green spaces: Exploring the instigators of change in Belgium. *Urban Forestry & Urban Greening*. 65, 127305. doi: 10.1016/j.ufug.2021.127305

Dale, M., Wicks, J., Mylne, K., Pappenberger, F., Laeger, S. and Taylor, S. (2014). Probabilistic flood forecasting and decision-making: An innovative risk-based approach. *Natural Hazards*. 70, 159-172. doi: 10.1007/s11069-012-0483-z

Davies, C., Hansen, R., Rall, E., Pauleit, S., Lafortezza, R., DeBellis, Y., Santos, A., Tosics, I. (2015). Green Infrastructure Planning and Implementation (GREEN SURGE). The status of European green space planning and implementation based on an analysis of selected European city-regions, 1-134. doi: 10.13140/RG.2.1.1723.0888

Davis, A.P., Hunt, W.F., Traver, R.G. and Clar, M. (2009). Bioretention Technology: Overview of Current Practice and Future Needs. *Journal of Environmental Engineering*. 135(3), 109. doi: 10.1061/(ASCE)0733-9372(2009)135:3(109)

DEFRA. (2018). *A Green Future: Our 25 Year Plan to Improve the Environment.* Available online at: https://www.gov.uk/government/publications/nature-recovery-network/nature-recovery-network (date accessed: 27 April 2021)

Department for Transport. (2007). *Manual for Streets*. London, UK: Thomas Telford Publishing.

Derkzen, M.L., van Teeffelen, A.J.A., Nagendra, H. and Verburg, P.H. (2017). Shifting roles of urban green space in the context of urban development and global change. *Current Opinion in Environmental Sustainability*. 29, 32-39. doi: 10.1016/j.cosust.2017.10.001

Dietz, M.E. (2007). Low impact development practices: A review of current research and recommendations for future directions. *Water, Air and Soil Pollution*. 186(4), 351-363. doi: 10.1007/s11270-007-9484-z

Disse, M., Johnson, T.G., Leandro, L. and Hartmann, T. (2020). Exploring the relation between flood risk management and flood resilience. *Water Security*. 9, 100059. doi: 10.1016/j.wasec.2020.100059

Dolowitz, D.P., Bell, S. and Keeley, M. (2018). Retrofitting urban drainage infrastructure: green or grey? *Urban Water Journal*. 15, 83-91. doi: 10.1080/1573062X.2017.1396352

Drake, J.A.P., Bradford, A. and Marsalek, J. (2013). Review of environmental performance of permeable pavement systems: state of the knowledge. *Water Quality Research Journal of Canada*. 48(3), 203-222. doi: 10.2166/wqrjc.2013.055

Echenique, M.H., Hargreaves, A.J., Mitchell, G., and Namdeo, A. (2012). Growing cities sustainably – Does urban form really matter? *Journal of the American Planning Association*. 78(2), 121-137. doi: 10.1080/01944363.2012.666731

Eckart, K., McPhee, Z. and Bolisetti, T. (2017). Performance and implementation of low impact development – A review. *Science of the Total Environment*. 607, 413-432. doi: 10.1016/j.scitotenv.2017.06.254

Ellis, J.B. (2013). Sustainable surface water management and green infrastructure in UK urban catchment planning. *Journal of Environmental Planning and Management*. 56(1), 24-41. doi: 10.1080/09640568.2011.648752

Ellis, J.B. and Lundy, L. (2016). Implementing sustainable drainage systems for urban surface water management within the regulatory framework in England and Wales. *Journal of Environmental Management*. 183, 630-636. doi: 10.1016/j.jenvman.2016.09.022

Ellis, J.B. and Viavattene, C. (2013). Sustainable Urban Drainage System Modeling for Managing Urban Surface Water Flood Risk. *CLEAN: Soil, Air, Water*. 42(2), 153-159. doi: 10.1002/clen.201300225

Emanuelsson, M.A.E., McIntyre, N., Hunt, C.F., Mawle, R., Kitson, J. and Voulvoulis, N. (2014). Flood risk assessment for infrastructure networks. *Journal of Flood Risk Management*. 7, 31-41. doi: 10.1111/jfr3.12028

Environment Agency. (2009). *Flooding in England: A national assessment of flood risk*. Available at: https://www.gov.uk/government/publications/flooding-in-england-national-assessment-of-flood-risk (Date accessed: 17th October 2017)

EPA. (2016). Storm Water Management Model Reference Manual Volume I – Hydrology. Washington D.C., USA: EPA.

Ersoy, E., Jorgenson, A. and Warren, P.H. (2019). Identifying multispecies connectivity corridors and the spatial pattern of the landscape. *Urban Forestry & Urban Greening.* 40, 308-322. doi: 10.1016/j.ufug.2018.08.001

Fang, X., Li, J., Gong, Y. and Li, X. (2017). Zero increase in peak discharge for sustainable development. *Frontiers of Environmental Science & Engineering*. 11(4), 2. doi: 10.1007/s11783-017-0935-5

Felappi, J.F., Sommer, J.H., Falkenberg, T., Terlau, W. & Kötter, T. (2020). Green infrastructure through the lens of “One Health”: A systematic review and integrative framework uncovering synergies and trade-offs between mental health and wildlife support in cities. *Science of the Total Environment*. 748, 141589. doi: 10.1016/j.scitotenv.2020.141589

Fenner, R.A. (2017). Spatial evaluation of multiple benefits to encourage multi-functional design of sustainable drainage in blue-green cities. *Water*. 9(12), 953-969. doi: 10.3390/w9120953

Filazzola, A., Shrestha, N. and MacIvor, J.S. (2019). The contribution of constructed green infrastructure to urban biodiversity: A synthesis and meta-analysis. *Journal of Applied Ecology*. 56(9), 2131-2143. doi: 10.1111/1365-2664.13475

Fletcher, T.D., Andrieu, H. and Hamel, P. (2013). Understanding, management and modelling of urban hydrology and its consequences for receiving waters: A state of the art. *Advances in Water Resources*. 51, 261-279. doi: 10.1016/j.advwatres.2012.09.001

Fletcher, T.D., Shuster, W., Hunt, W.F., Ashley, R., Butler, D., Arthur, S., Trowsdale, S., Barraud, S., Semadeni-Davies, A., Bertrand-Krajewski, J.L., *et al.* (2015). SUDS, LID, BMPs, WSUD and more – The evolution and application of terminology surrounding urban drainage. *Urban Water Journal.* 12(7), 525-542. doi: 10.1080/1573062X.2014.916314

Ford, A., Barr, S., Dawson, R.J., Virgo, J., Batty, M., and Hall, J.W. (2019). A multi-scale urban integrated assessment framework for climate change studies. *Computers, Environment and Urban Systems*. 75, 229-243. doi: 10.1016/j.compenvurbsys.2019.02.005

Fu, X., Luan, Q., Wang, H., Liu, J., and Gao, X. (2019). Application Research of SWMM in the Simulation of Large-Scale Urban Rain Flood Process - A Case Study of Yizhuang District, China. In: Dong W., Lian Y. & Zhang Y. (eds.) *Sustainable Development of Water Resources and Hydraulic Engineering in China*. New York, USA: Springer International Publishing. pp. 251-260.

Gill, S.E., Handley, J.F., Ennos, A.R. and Pauleit, S. (2007). Adapting cities for climate change; the role of green infrastructure. *Built Environment*. 33, 115–133. doi: 10.2148/benv.33.1.115

Gómez-Baggethun, E. and Barton, D.N. (2013). Classifying and valuing ecosystem services for urban planning. *Ecological Economics*. 86, 235-245. doi: 10.1016/j.ecolecon.2012.08.019

Grădinaru, S.R. and Hersperger, A.M. (2019). Green infrastructure in strategic spatial plans: Evidence from European urban regions. *Urban Forestry & Urban Greening*. 40, 17-28. doi: 10.1016/j.ufug.2018.04.018

Graham, M., Ogilvie, J., Reid, T. and Mills, T. (2012). Best practices for managing surface water flooding: Applying lessons learned in the UK to Canada. *Proceedings of the Annual Conference of the Canadian Society for Civil Engineering.* 2, 1053-1062.

Granata, F., Gargano, R. and De Marinis, G. (2016). Support Vector Regression for Rainfall-Runoff Modeling in Urban Drainage: A Comparison with the EPA's Storm Water Management Model. *Water*. 8(3), 69. doi: 10.3390/w8030069

Green Roof Guide. (2021). *Best Green Roof Drainage Layers*. Available at: https://www.greenroofguide.com/green-roofs/best-drainage-layers/ (date accessed: 13 July 2021)

Gülbaz, S. and Kazezyılmaz-Alhan, C.M. 2017. An evaluation of hydrologic modeling performance of EPA SWMM for bioretention. *Water Science & Technology*. 76(11), 3035-3043. doi: 10.2166/wst.2017.464

Guo, K., Guan, M. and Yu, D. (2021). Urban surface water flood modelling – a comprehensive review of current models and future challenges. *Hydrology and Earth System Sciences.* 25, 2843-2860. doi: 10.5194/hess-25-2843-2021

Gurrutxaga, M., Lozano, PJ and del Barrio, G. (2010). GIS-based approach for incorporating the connectivity of ecological networks into regional planning. *Journal for Nature Conservation*. 18(4), 318-326. doi: 10.1016/j.jnc.2010.01.005

Haaland, C. and van den Bosch, C.K. (2015). Challenges and strategies for urban green-space planning in cities undergoing densification: A review. *Urban Forestry & Urban Greening*. 14(4), 760-771. doi: 10.1016/j.ufug.2015.07.009

Haase, D., Schwarz, N., Strobach, M., Kroll, F., and Seppelt, R. (2012). Synergies, trade-offs, and losses of ecosystem services in urban regions: An integrated multiscale framework applied to the Leipzig-Halle region, Germany. *Ecology & Society*. 17(3), 22. doi: 10.5751/ES-04853-170322

Halbe, J., Pahl-Wostl, C., Sendzimir, J. and Adamowski, J. (2013). Towards adaptive and integrated management paradigms to meet the challenges of water governance. *Water Science & Technology*. 67(11), 2651-2660. doi: 10.2166/wst.2013.146

Hall, J.W., Meadowcroft, I.C., Sayers, P.B. and Bramley M.E. (2003). Integrated Flood Risk Management in England and Wales. *Natural Hazards Review*. 4(3), 126. doi: 10.1061/(ASCE)1527-6988(2003)4:3(126)

Hamouz, V. and Muthanna, T.M. (2019). Hydrological modelling of green and grey roofs in cold climate with the SWMM model. *Journal of Environmental Management*. 249(1), 109350. doi: 10.1016/j.jenvman.2019.109350

Hamouz, V., Møller-Pedersen, P. and Muthanna, T.M. (2020). Modelling runoff reduction through implementation of green and grey roofs in urban catchments using PCSWMM. *Urban Water Journal*. 17(9), 813-826. doi: 10.1080/1573062X.2020.1828500

Hansen, R., Olafsson, A.S., van der Jagt, A.P.N., Rall, E. & Pauleit, S. (2019). Planning multifunctional green infrastructure for compact cities: What is the state of practice? *Ecological Indicators*. 96(2), 99-110. doi: 10.1016/j.ecolind.2017.09.042

Hansen, R. and Pauleit, S. (2014). From multifunctionality to multiple ecosystem services? A conceptual framework for multifunctionality in green infrastructure planning for urban areas. *AMBIO*. 43, 516-529. doi: 10.1007/s13280-014-0510-2

Haowen, X., Yawen, W., Luping, W., Weilin, L., Wenqi, Z., Hong, Z., Yichen, Y. and Jun, L. (2020) Comparing simulations of green roof hydrological processes by SWMM and HYDRUS-1D. *Water Supply*. 20(1), 130-139. doi: 10.2166/ws.2019.140

Hargreaves, A.J. (2015). Representing the dwelling stock as 3D generic tiles estimated from average residential density. *Computers, Environment and Urban Systems*. 54, 280-300. doi: 10.1016/j.compenvurbsys.2015.08.001

Hargreaves, A.J., Cheng, V., Deshmukh, S., Leach, M., and Steemers, K. (2017). Forecasting how residential urban form affects the regional carbon savings and costs of retrofitting and decentralized energy supply. A*pplied Energy.* 186(3), 549-561. doi: 10.1016/j.apenergy.2016.02.095

Hargreaves, A.J., Farmani, R., Ward, S., and Butler, D. (2019) Modelling the future impacts of urban spatial planning on the viability of alternative water supply. *Water Research*. 162, 200-213. doi: 10.1016/j.watres.2019.06.029

Harris, H., Chow, M.F., Usman, F., Sidek, L.M., Roseli, Z.A. and Norlida, M.D. (2016). Urban stormwater management model and tools for designing stormwater management of green infrastructure practices. *Proceedings from the International Conference on Advances in Renewable Energy and Technologies*. 32, 012022. doi: 10.1088/1755-1315/32/1/012022

Hernes, R.R., Gragne, A.S., Abdalla, E.M.H., Braskerud, B.C., Alfredsen, K. and Muthanna, T.M. (2020). Assessing the effects of four SUDS scenarios on combined sewer overflows in Oslo, Norway: evaluating the low-impact development module of the mike urban model. *Hydrology Research.* 51(6), 1437-1454. doi: 10.2166/nh.2020.070

Hoang, L. and Fenner, R.A. (2016). System interactions of stormwater management using sustainable urban drainage systems and green infrastructure. *Urban Water Journal.* 13(7), 739-758. doi: 10.1080/1573062X.2015.1036083

Hoang, L., Fenner, R.A. and Skenderian, M. (2017). A conceptual approach for evaluating the multiple benefits of urban flood management practices. *Journal for Flood Risk Management.* 11(2), 943-959. doi: 10.1111/jfr3.12267

Hodsons. (2019). *Lamb Drove: Demonstrating SuDS with Permeable Paving*. Available online at: https://www.paving.org.uk/lamb-drove-suds-monitoring/ (date accessed: 23 March 2021)

Huang, Y., Tian, Z., Ke, Q., Liu, J., Irannezhad, M., Fan, D., Hou, M. and Sun, L. (2020). Nature-based solutions for urban pluvial flood risk management. *WIREs Water.* 7(3), e1421. doi: 10.1002/wat2.1421

Hunter, R.F., Cleland, C., Cleary, A., Droomers, M., Wheeler, B.W., Sinnett, D., Nieuwenhuijsen, M.J., and Braubach, M. (2019). Environmental, health, wellbeing, social and equity effects of urban green space interventions: A meta-narrative evidence synthesis. *Environment International*. 130, 104923. doi: 10.1016/j.envint.2019.104923

Ignatieva, M., Stewart, G.H. and Meurk, C. (2011.) Planning and design of ecological networks in urban areas. *Landscape and Ecological Engineering*. 7(1), 17-25. doi: 10.1007/s11355-010-0143-y

Infrastructure Transitions Research Consortium. (2020). *A sustainable Oxford-Cambridge corridor? Spatial analysis of options and futures for the Arc*. Available online at: https://www.itrc.org.uk/wp-content/uploads/2020/01/arc-main-report.pdf (date accessed: 20 April 2021)

INTECH GmbH. (2021) *Pipeline Design and Selection: Optimum Pipeline Diameter*. Available online at: https://intech-gmbh.com/pipelines_calc_and_select/ (date accessed: 24 August 2021)

Jayasooriya, V.M. and Ng, A.W.M. (2014). Tools for Modeling of Stormwater Management and Economics of Green Infrastructure Practices: a Review. *Water, Air & Soil Pollution.* 225, 2055. doi: 10.1007/s11270-014-2055-1

Jia, N., Sitzenfrei, R., Rauch, W., Liang, S. & Liu, Y. (2019). Effects of Urban Forms on Separate Drainage Systems: A Virtual City Perspective. *Water*. 11(4), 758. doi: 10.3390/w11040758

Johnson, D. and Geisendorf, S. (2019). Are Neighbourhood-level SUDS Worth it? An Assessment of the Economic Value of Sustainable Urban Drainage System Scenarios Using Cost-Benefit Analyses. *Ecological Economics.* 158, 194-205. doi: 10.1016/j.ecolecon.2018.12.024

Jose, R., Wade, R., and Jefferies, C. (2015). Smart SUDS: recognising the multiple-benefit potential of sustainable surface water management systems. *Water Science & Technology*. 71(2), 245-251. doi: 10.2166/wst.2014.484

Jovanovic, M. (2007). Role of Detention and Retention Basins in Stormwater Management and Environmental Protection, in Ashley R., Garvin S., Pasche E., Vassilopoulos A. & Zevenbergen C. (eds.) *Advances in Urban Flood Management*. CRC Press: London. pp. 166-194.
doi: 10.1201/9780203945988-13

Karka, P., Manoli, E., Lekkas, D.F. and Assimacopoulos, D. (2007). A case study on integrated urban water modelling using Aquacycle. *Proceedings from the 10th International Conference on Environmental Science & Technology*, Kos: Greece, 5-7th September 2007.

Kavgic, M., Mavrogianni, A., Mumovic, D., Summerfield, A., Stevanovic, Z. & Djurovic-Petrovic, M. (2010). A review of bottom-up building stock models for energy consumption in the residential sector. *Building and Environment.* 45, 1683-1697. doi: 10.1016/j.buildenv.2010.01.021

Kaźmierczak, A. and Cavan, G. (2011). Surface water flooding risk to urban communities: Analysis of vulnerability, hazard and exposure. *Landscape and Urban Planning*. 103(2), 185-197. doi: 10.1016/j.landurbplan.2011.07.008

Keenleyside, C., Badlock, D., Hjerp, B. and Swales, V. (2009). International perspectives on future land use. *Land Use Policy.* 26, 14-29. doi: 10.1016/j.landusepol.2009.08.030

Krebs, G., Kokkonen, T., Valtanen, M., Koivusalo, H. & Setälä, H. (2013). A high resolution application of a stormwater management model (SWMM) using genetic parameter optimization. *Urban Water Journal.* 10(6), 394-410. doi: 10.1080/1573062X.2012.739631

Kreibich, H., Bubeck, P., van Vliet, M. and De Moel, H. (2015). A review of damage-reducing measures to manage fluvial flood risks in climate change. *Mitigation and Adaption Strategies for Global Change*. 20(6), 967-989. doi: 10.1007/s11027-014-9629-5

La Rosa, D. and Pappalardo, V. (2021). Policies and Planning of Urban Green Infrastructure and Sustainable Urban Drainage Systems, in Catalano, C., Andreucci, M.B., Guarino, R., Bretzel, F., Leone, M. and Pasta, S. (eds.) *Urban Services to Ecosystems*, pp.297-316, Springer. doi: 10.1007/978-3-030-75929-2_16

Lafortezza, R., Chen, J., van den Bosch, C.K. and Randrup, T.B. (2018). Nature-based solutions for resilient landscapes and cities. *Environmental Research*. 165, 431-441. doi: 10.1016/j.envres.2017.11.038

Lane, S.N., Landström, C. and Whatmore, S.J. (2011). Imagining flood futures: Risk assessment and management in practice. *Philosophical Transactions of the Royal Society A*. 369: 1784-1806. doi: 10.1098/rsta.2010.0346

Langeveld, J., Nopens, I., Schilperoort, R., Benedetti, L., de Klein, J., Amerlinck, Y. and Weijers, S. (2013). On data requirements for calibration of integrated models for urban water systems. *Water Science and Technology*. 63(3), 728-736. doi: 10.2166/wst.2013.301

Langeveld, J., Cherqui, F., Tscheikner-Gratl, F., Muthanna, T., Juarez, M., Leitao, J., Roghani, B., Kerres, K., do Ceu, A., Werey, C. and Rulleau, B. (2022). Asset management for blue-green infrastructures: a scoping review. *Blue-Green Systems*. 4(2), 272-290. doi: 10.2166/bgs.2022.019

Lashford, C., Rubinato, M., Cai, Y., Hou, J., Abolfathi, S., Coupe, S., Charlesworth, S. and Tait, S. (2019). SUDS and sponge cities: A comparative analysis of the implementation of pluvial flood management in the UK and China. *Sustainability*. 11(1), 213-227. doi: 10.3390/su11010213

Lee, J., Chung, G., Park, H., and Park, I. (2018). Evaluation of the Structure of Urban Stormwater Pipe Network Using Drainage Density. *Water*. 10(10), 1444. doi: 10.3390/w10101444

Lehmann, S. (2016). Sustainable urbanism: towards a framework for quality and optimal density? *Future Cities and Environment*. 2(1), 8-21. doi: 10.1186/s40984-016-0021-3

Leigh, N.G. and Lee, H. (2019). Sustainable and Resilient Urban Water Systems: The Role of Decentralization and Planning. *Sustainability*. 11(3), 918. doi: 10.3390/su11030918

Lennon, M., Scott, M. and O’Neill, E. (2014). Urban Design and Adapting to Flood Risk: The Role of Green Infrastructure. *Journal of Urban Design*. 19(5), 745-758. doi: 10.1080/13574809.2014.944113

Li, W.C. and Yeung, K.K.A. (2014). A comprehensive study of green roof performance from environmental perspective. *International Journal of Sustainable Built Environment*. 3(1), 127-134. doi: 10.1016/j.ijsbe.2014.05.001

Liao, K.H., Deng, S. and Tan, P.Y. (2017). Blue-Green Infrastructure: New Frontier for Sustainable Urban Stormwater Management. In: Tan, P.Y. and Jim, C. (eds.) *Greening Cities - Advances in 21st Century Human Settlements*. Singapore: Springer. pp. 203-226.

Liquete, C., Kleeschulte, S., Dige, G., Maes, J., Grizzetti, B., Olah, B. and Zulian, G. (2015). Mapping green infrastructure based on ecosystem services and ecological networks. A Pan-European case study. *Environmental Science & Policy*. 54, 268–280. doi: 10.1016/j.envsci.2015.07.009

Liu, Y., Cibin, R., Bralts, V.F., Chaubey, I., Bowling, L.C. and Engel, B.A. (2016). Optimal selection and placement of BMPs and LID practices with a rainfall-runoff model. *Environmental Modelling & Software.* 80, 281-296. doi: 10.1016/j.envsoft.2016.03.005

Lumbroso, D.M. and Vinet, F. (2011). A comparison of the causes, effects and aftermaths of the coastal flooding of England in 1953 and France in 2010. *Natural Hazards & Earth System Sciences*. 11(8), 2321 – 2333. doi: 10.5194/nhess-11-2321-2011

Ma, Q., Li, Y. and Xu, L. (2021). Identification of green infrastructure networks based on ecosystem services in a rapidly urbanizing area. *Journal of Cleaner Production*. 300, 126945. doi: 10.1016/j.jclepro.2021.126945

Machado, A. (2004). An index of naturalness. *Journal for Nature Conservation*. 12(2), 95-110. doi: 10.1016/j.jnc.2003.12.002

Maes, J. and Jacobs, S. (2015). Nature-Based Solutions for Europe's Sustainable Development. *Conservation Letters*. 10(1), 121-124. doi: 10.1111/conl.12216

Manesh, S.V., Tadi, M. and Zanni, F. (2012). Integrated sustainable urban design: neighbourhood design proceeded by sustainable urban morphology emergence. *Transactions on Ecology and the Environment.* 155, 631-642. doi: 10.2495/SC120532

Maqbool, R. and Wood, H. (2022). Containing a sustainable urbanized environment through SuDS devices in management trains. *Science of the Total Environment*. 807(2), 150812. doi: 10.1016/j.scitotenv.2021.150812

Martin-Mikle, C.J., de Beurs, K.M., Julian, J.P. and Mayer, P.M. (2015). Identifying priority sites for low impact development (LID) in a mixed-use watershed. *Landscape and Urban Planning*. 140, 29-41. doi: 10.1016/j.landurbplan.2015.04.002

Mayrand, F. and Clergeau, P. (2018). Green Roofs and Green Walls for Biodiversity Conservation: A Contribution to Urban Connectivity? *Sustainability*. 10(4), 985. doi: 10.3390/su10040985

McKinley, E., Pagès, J.F., Wyles, K.J. and Beaumont, N. (2019). Ecosystem services: A bridge or barrier for UK marine stakeholders? *Ecosystem Services*. 37, 100922. doi: 10.1016/j.ecoser.2019.100922

McPhearson, T., Pickett, S.T.A., Grimm, N.B., Niemelä, J., Alberti, M., Elmqvist, T., Weber, C., Haase, D., Breuste, J. & Qureshi, S. (2016). Advancing urban ecology toward a science of cities. *Bioscience*. 66 (3), 198–212. doi: 10.1093/biosci/ biw002

Meierdiercks, K.L., Smith, J.A., Baeck, M.L. and Miller, A.J. (2010). Analyses of urban drainage network structure and its impact on hydrologic response. *Journal of the American Water Resources Association.* 46(5), 932-943. doi: 10.1111/j.1752-1688.2010.00465.x

Melville-Shreeve, P., Cotterill, S., Grant, L., Arahuetes, A., Stovin, V., Farmani, R. and Butler, D. (2018). State of SuDS delivery in the United Kingdom. *Water and Environment Journal*. 32, 9-16. doi: 10.1111/wej.12283

Mentens, J., Raes, D. and Hermy, M. (2006). Green roofs as a tool for solving the rainwater runoff problem in the urbanised 21st century? *Landscape and Urban Planning*. 77(3), 217-226. doi: 10.1016/j.landurbplan.2005.02.010

Miller, J.D. and Hess, T. (2017). Urbanisation impacts on storm runoff along a rural-urban gradient. *Journal of Hydrology*. 552, 474-489. doi: 10.1016/j.jhydrol.2017.06.025

Ministry of Housing. (2015). *Technical housing standards – nationally described space standard,* 27 March 2015. Available at: https://www.gov.uk/government/publications/technical-housing-standards-nationally-described-space-standard

Ministry of Housing, Communities & Local Government. (2010). *The Building Regulations 2010: Drainage and Waste Disposal*. Available online at: https://assets.publishing.service.gov.uk/government/uploads/system/uploads/attachment_data/file/442889/BR_PDF_AD_H_2015.pdf (date accessed: 19 April 2021)

Mok, H., Russell, T., Virgo, R., Smith, A., Hall, J.W. & Ford, A. (2020). *Future of the Oxford Cambridge Arc*. https://nismod.github.io/arc-udm-vis/index.html (date accessed: 03 May 2021)

Morton, R.D., Marston, C.G., O'Neil, A.W. and Rowland, C.S. (2021). *Land Cover Map 2020 (25m rasterised land parcels, GB)*. NERC EDS Environmental Information Data Centre. doi: 10.5285/6c22cf6e-b224-414e-aa85-900325baedbd

Mouratidis, K. (2019). Compact city, urban sprawl, and subjective well-being. *Cities*. 92, 261-272. doi: 10.1016/j.cities.2019.04.013

Mueller, N., Rojas-Rueda, D., Khreis, H., Cirach, M., Andrés, D., Ballester, J., Bartoll, X., Daher, C., Deluca, A., Echave, C., Milà, C., Márquez, S., Palou, J., Pérez, K., Tonne, C., Stevenson, M., Rueda, S. and Nieuwenhuijsen, M. (2020). Changing the urban design of cities for health: The superblock model. *Environment International*. 134, 105132. doi: 10.1016/j.envint.2019.105132

Mullaney, J. & Lucke, T. (2013). Practical Review of Pervious Pavement Designs. *CLEAN: Soil, Air, Water*. 42(2), 111-124. doi: 10.1002/clen.201300118

National Infrastructure Commission (NIC). (2019). *Partnering for Prosperity: A new deal for the Cambridge-Milton Keynes-Oxford Arc.* Available online at: https://nic.org.uk/app/uploads/Partnering-for-Prosperity.pdf (date accessed: 21 April 2021)

Natural England. (2016). *Conservation 21: Natural England's Conservation Strategy for the 21st Century*. Available at: https://assets.publishing.service.gov.uk/government/uploads/system/uploads/attachment_data/file/562046/conservation-21.pdf (date accessed: 09 July 2020)

Ncube, S. and Arthur, S. (2021). Influence of Blue-Green and Grey Infrastructure Combinations on Natural and Human-Derived Capital in Urban Drainage Planning. *Sustainability*. 13, 2571. doi: 10.3390/su13052571

Newman, P. (2014). Density, the Sustainability Multiplier: Some Myths and Truths with Application to Perth, Australia. *Sustainability.* 6(9), 6467-6487. doi: 10.3390/su6096467

Noh, S.J., Lee, S., An, H., Kawaike, K. and Nakagawa, H. (2016). Ensemble urban flood simulation in comparison with laboratory-scale experiments: Impact of interaction models for manhole, sewer pipe, and surface flow. *Advances in Water Resources*. 97, 25-37. doi: 10.1016/j.advwatres.2016.08.015

Oberascher, M., Rauch, W. and Sitzenfrei, R. (2022). Towards a smart water city: A comprehensive review of applications, data requirements, and communication technologies for integrated management. *Sustainable Cities and Society*. 76, 103442. doi: 10.1016/j.scs.2021.103442

Ordnance Survey. (2021). *OS Terrain 5 DTM* © Crown copyright and database rights 2021 Ordnance Survey (100025252)

Osman, M., Yusof, K.W., Takaijudin, H., Goh, H.W., Malek, M.A., Azizan, N.A., Ghani, A.A. and Abdurrasheed, A.S. (2019). A Review of Nitrogen Removal for Urban Stormwater Runoff in Bioretention System. *Sustainability*. 11(19), 5415. doi: 10.3390/su11195415

Ossa-Moreno, J., Smith, K.M. and Mijic, C. (2017). Economic analysis of wider benefits to facilitate SuDS uptake in London, UK. *Sustainable Cities and Society*. 28, 411-419. doi: 10.1016/j.scs.2016.10.002

Oubennaceur, K., Chokmani, K., Nastev, M., Lhissou, R. and El Alem, A. (2019). Flood risk mapping for direct damage to residential buildings in Quebec, Canada. *International Journal of Disaster Risk Reduction*. 33, 44-54. doi: 10.1016/j.ijdrr.2018.09.007

Oxfordshire County Council. (2019). *Introducing the Oxfordshire Plan 2050.* Available online at: https://oxfordshireplan.org/wp-content/uploads/2019/02/Introducing_the_Oxfordshire_Plan_Feb_2019.pdf (date accessed: 27 April 2021)

Oxfordshire Growth Board. (2021). *Oxfordshire's Strategic Vision for Long-Term Sustainable Development*. Available at: https://www.oxfordshiregrowthboard.org/wp-content/uploads/2021/05/Strategic-Vision-full-document.pdf (date accessed: 07 May 2021)

Paithankar, D.N. and Taji, S.G. (2020). Investigating the hydrological performance of green roofs using storm water management model. *Proceedings of the 3rd International Conference on Innovative Technologies for Clean and Sustainable Development*. 32(4), 943-950. doi: 10.1016/j.matpr.2020.05.085

Palla, A. and Gnecco, I. (2015). Hydrologic modeling of Low Impact Development systems at the urban catchment scale. *Journal of Hydrology.* 528, 361-368. doi: 10.1016/j.jhydrol.2015.06.050

Parker, D.J., Priest, S.J. and McCarthy, S.S. (2011). Surface water flood warnings requirements and potential in England and Wales. *Applied Geography*. 31(3), 891-900. doi: 10.1016/j.apgeog.2011.01.002

Pauleit, S., Fryd, O., Backhaus, A. and Jensen, M.B. (2020). Green Infrastructures to Face Climate Change in an Urbanising World, in Loftness V.

(ed.) *Sustainable Built Environments*. Springer, New York, pp. 207-234. doi: 10.1007/978-1-0716-0684-1

Paz, I., Willinger, B., Gires, A., Alves de Souza, B., Monier, L., Cardinal, H., Tisserand, B., Tchiguirinskaia, I. and Schertzer, D. (2019). Small-Scale Rainfall Variability Impacts Analyzed by Fully-Distributed Model Using C-Band and X-Band Radar Data. *Water*. 11(6), 1273. doi: 10.3390/w11061273

Peng, Z. and Stovin, V. (2017). Independent Validation of the SWMM Green Roof Module. *Journal of Hydrologic Engineering*. 22(9). doi: 10.1061/(ASCE)HE.1943-5584.0001558

Pitt, M. (2008). *Learning Lessons from the 2007 Floods,* London, UK: The UK Cabinet Office.

Potter, K. and Vilcan, T. (2020). Managing urban flood resilience through the English planning system: insights from the ‘SuDS-face’. *Philosophical Transactions of the Royal Sociey A.* 378, 20190206. doi: 10.1098/rsta.2019.0206

Price, J.C., Wright, L., Fant, C. and Strzepek, K.M. (2016). Calibrated methodology for assessing climate change adaptation costs for urban drainage systems. *Urban Water Journal.* 13(4), 331-344. doi: 10.1080/1573062X.2014.991740

Priest, S.J., Parker, D.J., Hurford, A.P., Walker, J. and Evans, K. (2011) Assessing options for the development of surface water flood warning in England and Wales. *Journal of Environmental Management*. 92(12), 3038-3048. doi: 10.1016/j.jenvman.2011.06.041

Rahman, M.A., Imteaz, M.A. and Arulrajah, A. (2016). Suitability of reclaimed asphalt pavement and recycled crushed brick as filter media in bioretention applications. *International Journal of Environment and Sustainable Development*. 15(1). doi: 10.1504/IJESD.2016.073333

Randall, M., Støvring, J., Henrichs, M., and Jensen, M.B. (2020). Comparison of SWMM evaporation and discharge to in-field observations from lined permeable pavements. *Urban Water Journal*. 17(6), 491-502. doi: 10.1080/1573062X.2020.1776737

Randall, M., Sun, F., Zhang, Y. and Jensen, M. (2019). Evaluating Sponge City volume capture ratio at the catchment scale using SWMM. *Journal of Environmental Management*. 246, 745-757. doi: 10.1016/j.jenvman.2019.05.134

Rincón, V., Velázquez, J., Pascual, Á., Herráez, F., Gómez, I., Gutiérrez, J., Sánchez, B., Hernando, A., Santamaría, T. and Sánchez-Mata, D. (2022). Connectivity of Natura 2000 potential natural riparian habitats under climate change in the Northwest Iberian Peninsula: implications for their conservation. *Biodiversity and Conservation*. 85. doi: 10.1007/s10531-021-02351-z

Rode, P., Keim, C., Robazza, G., Viejo, P. and Schofield, J. (2014). Cities and Energy: Urban Morphology and Residential Heat-Energy Demand. *Environment & Planning B: Urban Analytics & City Science*. 41(1), 138-162. doi: 10.1068/b39065

Rodriguez, M., Fu, G., Butler, D., Yuan, Z., and Sharma, K. (2021). Exploring the spatial impact of green infrastructure on urban drainage resilience. *Water*. 13(13), 1789. doi: 10.3390/w13131789

Rodríguez-Espinosa, V.M., Aguilera-Benavente, F. and Gómez-Delgado, M. (2020). Green infrastructure design using GIS and spatial analysis: a proposal for the Henares Corridor (Madrid-Guadalajara, Spain). *Landscape Research*. 45(1), 26-43. doi: 10.1080/01426397.2019.1569221

Rosa, D.J., Clausen, J.C. and Dietz, M.E. (2015). Calibration and Verification of SWMM for Low Impact Development. *Journal of the American Water Resources Association.* 51(3), 746-757. doi: 10.1111/jawr.12272

Roy-Poirier, A., Champagne, P. and Filion, Y. (2010). Review of Bioretention System Research and Design: Past, Present, and Future. *Journal of Environmental Engineering*. 136(9), 227.
doi: 10.1061/(ASCE)EE.1943-7870.0000227

Rubiera-Morollón, F. and Garrido-Yserte, R. (2020). Recent Literature about Urban Sprawl: A Renewed Relevance of the Phenomenon from the Perspective of Environmental Sustainability. *Sustainability*. 12(16), 6551.
doi: 10.3390/su12166551

Rudisser, J., Tasser, E. and Tappeiner, U. (2012). Distance to nature—A new biodiversity relevant environmental indicator set at the landscape level. *Ecological Indicators*. 15(1), 208-216. doi: 10.1016/j.ecolind.2011.09.027

Rusche, K., Reimer, M. and Stichmann, R. (2019). Mapping and Assessing Green Infrastructure Connectivity in European City Regions. *Sustainability*. 11(6), 1819-1831. doi: 10.3390/su11061819

Russo, A. and Cirella, G.T. (2018). Modern Compact Cities: How Much Greenery Do We Need? *International Journal of Environmental Research and Public Health*. 15(10), 2180. doi: 10.3390/ijerph15102180

Salomaa, A., Paloniemi, R., Kotiaho, J.S., Kettunen, M., Apostolopoulou, E. and Cent, J. (2016). Can green infrastructure help to conserve biodiversity?. *Environment and Planning C*. 35(2), 265-288. doi: 10.1177/0263774X16649363

Savi, T., Andri, S. and Nardini, A. (2013). Impact of different green roof layering on plant water status and drought survival. *Ecological Engineering*. 57, 188-196. doi: 10.1016/j.ecoleng.2013.04.048

She, L. and You, X.Y. (2019). A Dynamic Flow Forecast Model for Urban Drainage Using the Coupled Artificial Neural Network. *Water Resources Management*. 33, 3143-3153. doi: 10.1007/s11269-019-02294-9

Sitzenfrei, R. and Rauch, W. (2014). Integrated hydraulic modelling of water supply and urban drainage networks for assessment of decentralized options. *Water Science & Technology*. 70(11), 1817-1824. doi: 10.2166/wst.2014.326

Smith, A.C., Harrison, P.A., Pérez Soba, M., Archaux, F., Bilcharska, M., Egoh, B.N., Eros, T., Fabrega Domenech, N., György, A.I., Haines-Young, R., *et al.* (2017). How natural capital delivers ecosystem services: A typology derived from a systematic review. *Ecosystem Services*. 26, 111-126.

doi: 10.1016/j.ecoser.2017.06.006

Smith, A. (2021) Natural capital in Oxfordshire: Short report (version 3.0). Available at: https://www.eci.ox.ac.uk/research/ecosystems/bio-clim-adaptation/downloads/bicester-Natural-capital-mapping-in-Oxfordshire-Short-report-V3.pdf (last accessed: 11 April 2022)

Song, X., Zhang, J., Wang, G., He, R. and Wang, X. (2014). Development and challenges of urban hydrology in a changing environment: II: Urban stormwater modeling and management. *Advances in Water Science*. 25, 752-764. doi: 10.14042/j.cnki.32.1309.2014.04.020

Sörensen, J., Persson, A., Sternudd, C., Aspegren, H., Nilsson, J., Nordström, J., Jönsson, K., Mottaghi, M., Becker, P., Pilesjö, P., *et al.* (2016). Re-Thinking Urban Flood Management—Time for a Regime Shift. *Water*. 8(8), 332. doi: 10.3390/w8080332

Steele, W., Davidson, A. and Reed, A. (2020). Imagining the Dirty Green City. *Australian Geographer*. 51(2), 239-256. doi: 10.1080/00049182.2020.1727127

Stevenson, M. and Gleeson, B. (2018). Complex Urban Systems: Compact Cities, Transport and Health. In: Nieuwenhuijsen, M. and Khreis, H. (eds.) *Integrating Human Health into Urban and Transport Planning*. New York, USA: Springer. pp. 271-285.

Stovin, V., Vesuviano, G. and Kasmin, H. (2012). The hydrological performance of a green roof test bed under UK climatic conditions. *Journal of Hydrology*. 414, 148-161. doi: 10.1016/j.jhydrol.2011.10.022

Tabatabaee, S., Mahdiyar, A., Durdyev, S., Mohandes, S.R. and Ismail, S. (2019). An assessment model of benefits, opportunities, costs, and risks of green roof installation: A multi criteria decision making approach. *Journal of Cleaner Production*. 238, 117956. doi: 10.1016/j.jclepro.2019.117956

Tahvonen, O. (2018). Adapting Bioretention Construction Details to Local Practices in Finland. *Sustainability*. 10(2), 276. doi: 10.3390/su10020276

Thieken, A.H., Kienzier, S., Kreibich, H., Kuhlicke, C., Kunz, M., Mühr, B., Müller, M., Otto, A., Petrow, T., Pisi, S. and Schröter, K. (2016). Review of the flood risk management system in Germany after the major flood in 2013. *Ecology and Society.* 21(2), 51-63. doi: 10.5751/ES-08547-210251

Thomas, K. and Littlewood, S. (2010). From Green Belts to Green Infrastructure? The Evolution of a New Concept in the Emerging Soft Governance of Spatial Strategies. *Planning Practice & Research*. 25(2), 203-222. doi: 10.1080/02697451003740213

Tideway. (2021). The Tunnel. Available at: Available at: https://tideway.london/the-tunnel/ (date accessed: 15 October 2022)

Travis, Q.B. and Mays, L.W. (2008). Optimizing Retention Basin Networks. *Journal of Water Resources Planning and Management*. 134(5), 432. doi: 10.1061/(ASCE)0733-9496(2008)134:5(432)

Tscheikner-Gratl, F., Zeisl, P., Kinzel, C., Leimgruber, J., Ertl, T., Rauch, W. and Kleidorfer, M. (2016). Lost in calibration: why people still do not calibrate their models, and why they still should – a case study from urban drainage modelling. *Water Science & Technology*. 74(10), 2337-2348. doi: 10.2166/wst.2016.395

Tuluhan Yılmaz, K., Alphan, H. and Gülçin, D. (2019). Assessing Degree of Landscape Naturalness in a Mediterranean Coastal Environment Threatened by Human Activities. *Journal of Urban Planning and Development*. 145(2). doi: 10.1061/(ASCE)UP.1943-5444.0000499

US Department of Agriculture. (2014). *Restoring Natural Hydrology*. Available at: https://www.nrcs.usda.gov/wps/portal/nrcs/detailfull/national/technical/alphabetical/water/restoration/?&cid=nrcs143_026903 (date accessed: 11 December 2017)

Valeri, S., Zavattero, L. and Capotorti, G. (2021). Ecological Connectivity in Agricultural Green Infrastructure: Suggested Criteria for Fine Scale Assessment and Planning. *Land*. 10(8), 807. doi: 10.3390/land10080807

Vilcan, T. and Potter, K. (2020). Delivering sustainable drainage systems through the English planning system: A proposed case of institutional void. *Journal of Flood Risk Management*. 13(1), e12591. doi: 10.1111/jfr3.12591

Vineyard, D., Ingwersen, W.W., Hawkins, T.R., Xue, X., Demeke, B. & Shuster, W. (2015). Comparing green and grey infrastructure using life cycle cost and environmental impact: A rain garden case study in Cincinnati, OH. *Journal of the American Water Resources Association.* 51(5), 1342-1360. doi: 10.1111/1752-1688.12320

Water UK. (2019). *Sewerage Sector Guidance Appendix C*. Available online at: https://www.water.org.uk/wp-content/uploads/2019/03/Appendix-C-to-draft-sewerage-Sector-Guidance-Design-and-Construction-Guidance.pdf (date accessed: 24 August 2021)

Webber, J.L., Gibson, M.J., Chen, A.S., Savic, D., Fu, G. & Butler, D. (2017). Rapid assessment of surface-water flood-management options in urban catchments. *Urban Water Journal.* 15(3), 210-217. doi: 10.1080/1573062X.2018.1424212

Weiss, P.T., Kayhanian, M., Gulliver, J.S. and Khazanovich, L. (2017). Permeable pavement in northern North American urban areas: research review and knowledge gaps. *International Journal of Pavement Engineering*. 2, 143-162. doi: 10.1080/10298436.2017.1279482

Whitesides, R.W. (2012). Selecting the Optimal Pipe Size. Available online at: https://www.pdhonline.com/courses/m270/m270content.pdf (date accessed: 20 September 2021)

Wild, T.C., Henneberry, J. and Gill, L. (2017). Comprehending the multiple ‘values’ of green infrastructure – Valuing nature-based solutions for urban water management from multiple perspectives. *Environmental Research*. 158, 179-187. doi: 10.1016/j.envres.2017.05.043

Williams, K., Joynt, J.L.R., Payne, C., Hopkins, D. and Smith, I. (2012). The conditions for, and challenges of, adapting England’s suburbs for climate change. *Building and Environment*. 55, 131-140. doi: 10.1016/j.buildenv.2011.11.015

Williams, K. (2014). Urban Form and Infrastructure: A Morphological Review, *Future Cities: A Working Paper*, Foresight.

Winston, R.J., Al-Rubaei, A., Blecken, G.T., Viklander, M. and Hunt, W.F. (2016). Maintenance measures for preservation and recovery of permeable pavement surface infiltration rate – The effects of street sweeping, vacuum cleaning, high pressure washing, and milling. *Journal of Environmental Management*. 169, 132-144. doi: 10.1016/j.jenvman.2015.12.026

Wolch, J.R., Byrne, J. and Newell, J.P. (2014). Urban green space, public health, and environmental justice: The challenge of making cities ‘just green enough’. *Landscape and Urban Planning*. 125, 234-244. doi: 10.1016/j.landurbplan.2014.01.017

Woods-Ballard, B., Kellagher, R. and Dou, V. (2017). *UK SuDS and Chinese Sponge Cities: solving the problems of urban flood risk management?* Available at: http://eprints.hrwallingford.com/id/eprint/1268 (date accessed: 19 June 2019)

Xu, C., Rahman, M., Haase, D., Wu, Y., Su, M., and Pauleit, S. (2020). Surface runoff in urban areas: The role of residential cover and urban growth form. *Journal of Cleaner Production*. 262, 121421. doi: 10.1016/j.jclepro.2020.121421

Yang, B. and Li, M. (2010). Ecological engineering in a new town development: Drainage design in The Woodlands, Texas. *Ecological Engineering*. 36(12), 1639-1650. doi: 10.1016/j.ecoleng.2010.07.002

Yang, Y. and Chui, T.F.M. (2018). Optimizing surface and contributing areas of bioretention cells for stormwater runoff quality and quantity management. *Journal of Environmental Management*. 206, 1090-1103. doi: 10.1016/j.jenvman.2017.11.064

Yazdi, M.N., Ketabchy, M., Sample, D.J., Scott, D., and Liao, H. (2019). An evaluation of HSPF and SWMM for simulating streamflow regimes in an urban watershed. *Environmental Modelling & Software.* 118, 211-225. doi: 10.1016/j.envsoft.2019.05.008

Yazdanfar, Z. & Sharma, A. (2015). Urban drainage system planning and design–challenges with climate change and urbanization: a review. *Water Science & Technology*. 72(2), 165-179. doi: 10.2166/wst.2015.207

Yin, J., Ye, M., Yin, Z. and Xu, S. (2015). A review of advances in urban flood risk analysis over China. *Stochastic Environmental Research and Risk Analysis.* 29, 1063-1070. doi: 10.1007/s00477-014-0939-7

Zhang, K. and Chui, T.F.M. (2018). A comprehensive review of spatial allocation of LID-BMP-GI practices: Strategies and optimization tools. *Science of the Total Environment.* 621, 915-929. doi: 10.1016/j.scitotenv.2017.11.281

Zhang, Z., Meerow, S., Newell, J.P. and Lindquist, M. (2019). Enhancing landscape connectivity through multifunctional green infrastructure corridor modeling and design. *Urban Forestry & Urban Greening*. 38, 305-317. doi: 10.1016/j.ufug.2018.10.014

Zhang, Z. and Valeo, C. (2022). Verification of PCSWMM's LID processes and their scalability over time and space. *Frontiers in Water*. 4. doi: 10.3389/frwa.2022.1058883

Zuniga-Teran, A.A., Staddon, C., de Vito, L., Gerlak, A.K., Ward, S., Schoeman, Y., Hart, A., and Booth, G. (2020). Challenges of mainstreaming green infrastructure in built environment professions. *Journal of Environmental Planning and Management*. 63(4), 710-732. doi: 10.1080/09640568.2019.1605890

8. Appendices

Appendix 1: Soil parameters used in SWMM for comparison of soil types (Chapter 3) (based on suggested parameter values from EPA, 2016)

	Clay	***Sandy***	***Silty***
Thickness (mm)	150	150	150
Porosity	0.475	0.437	0.501
Effective porosity	0.385	0.417	0.486
Wilting point	0.10	0.10	0.01
Saturated Hydraulic Conductivity (mm/hr)	0.254	120.39	6.604
Suction head (mm)	316.23	49.53	166.87

Appendix 2: Parameters used for SuDS modelling in SWMM

		Bioretention	***Green Roof***	***Permeable Paving***	***Retention Basin***
Drain	Flow coefficient	0.5	-	0.5	0.5
	Flow exponent	0.7	-	0.7	0.5
	Offset (mm)	0.0	-	0.0	0.0
Drainage Mat	Thickness (mm)	-	25	-	-
	Void fraction	-	0.5	-	-
	Roughness	-	0.1	-	-
Pavement	Thickness (mm)	-	-	80	-
	Void ratio	-	-	0.26	-
	Impervious surface fraction	-	-	0	-
	Permeability (mm/hr)	-	-	17693	-
	Clogging factor	-	-	0.05	-
	Regeneration interval (days)	-	-	0.0167	-
	Regeneration fraction	-	-	1	-
Soil	Thickness (mm)	300	80	50	-
	Porosity	0.412	0.464	0.26	-
	Field capacity	0.2	0.2	0.2	-
	Wilting point	0.10	0.10	0.01	-
	Conductivity (mm/hr)	10.90	48.00	14.85	-

	Conductivity slope	27	44	27	-
	Suction head (mm)	110.1	49.8	3.5	-
Storage	Thickness (mm)	150	-	350	2000
	Void ratio	0.99	-	0.99	0.99
	Seepage rate	0	-	0	0
	Clogging factor	0	-	0	0
Surface	Berm height (mm)	0	0	0	0
	Vegetation volume	0.5	0.5	0.0	0.0
	Roughness	0.140	0.100	0.012	0.100
	Slope	*Location specific – sourced from the DTM*			

Appendix 3: Counts of the different tile assignments for each development scenario, including a breakdown by county.

		LOW DENSITY				MEDIUM DENSITY				HIGH DENSITY				VERY HIGH DENSITY			
		DA	DT	AT	ADT	DA	DT	AT	ADT	DA	DT	AT	ADT	DA	DT	AT	ADT
23GE	**Total**	**16547**	**6150**	**12129**	**13792**	**15877**	**18576**	**7793**	**6470**	**2797**	**13693**	**593**	**5763**	**60**	**9267**	**1104**	**925**
	Bedfordshire	3506	1141	3281	3084	3917	3770	1805	2005	768	3204	151	1230	5	1997	329	241
	Buckinghamshire	2511	794	1056	2205	2540	2883	1188	1384	292	2114	61	879	8	1846	115	84
	Cambridgeshire	4284	2311	3042	3941	3870	4554	1392	923	502	3506	108	1037	2	1856	191	240
	Northamptonshire	2773	906	2304	2246	2887	3692	1598	846	439	2211	118	1083	10	1216	183	139
	Oxfordshire	3473	998	2446	2316	2663	3677	1810	1312	796	2658	155	1534	35	2352	286	221
23GNS	**Total**	**16204**	**6241**	**13677**	**14084**	**16583**	**18671**	**8267**	**7551**	**3381**	**14062**	**567**	**5204**	**14**	**8719**	**952**	**866**
	Bedfordshire	3327	1095	3187	2791	3992	3938	2090	2184	1000	3052	146	1201	5	1937	261	269
	Buckinghamshire	2844	852	2437	2288	2862	3223	1361	1467	581	2535	59	866	2	1796	122	162
	Cambridgeshire	3981	2178	3160	3734	3567	4263	1420	1301	479	3379	108	999	0	1708	156	124
	Northamptonshire	2868	1063	2336	2433	3385	3661	1610	1157	584	2384	116	1080	0	1171	179	123
	Oxfordshire	3184	1053	2557	2838	2777	3586	1786	1442	737	2712	138	1058	7	2107	234	188
23YE	**Total**	**16516**	**7616**	**12433**	**15736**	**15134**	**19238**	**7609**	**5361**	**2493**	**12561**	**563**	**5220**	**0**	**7724**	**823**	**588**
	Bedfordshire	4402	2692	3418	5379	3726	5461	1799	1501	541	2705	148	1246	0	1756	194	135
	Buckinghamshire	2792	1162	1931	2638	2375	3273	1139	622	279	1983	59	872	0	1313	75	50
	Cambridgeshire	4323	2586	3309	4399	3660	4805	1444	1280	466	3381	107	1053	0	1794	158	122
	Northamptonshire	2909	1073	2376	2512	3387	3683	1611	1188	585	2421	118	1095	0	1204	178	123
	Oxfordshire	3624	1428	2527	3518	2596	4261	1757	980	639	2505	151	1157	0	1955	229	158
23YNS	**Total**	**17970**	**9530**	**14219**	**19243**	**16071**	**20829**	**7771**	**5611**	**2500**	**13614**	**562**	**5384**	**0**	**8304**	**822**	**592**
	Bedfordshire	3999	2096	3381	4339	3804	4688	1833	1554	541	2915	142	1286	0	1974	192	135
	Buckinghamshire	3353	1751	2512	3479	2662	3857	1200	715	280	2328	59	935	0	1518	76	50
	Cambridgeshire	4290	2932	3409	5383	3585	4581	1406	1288	463	3385	108	1006	0	1728	156	122
	Northamptonshire	2868	1063	2336	2433	3385	3661	1610	1157	584	2384	116	1080	0	1171	179	123
	Oxfordshire	3460	1688	2581	3609	2635	4042	1722	897	632	2602	137	1077	0	1913	219	162

		LOW DENSITY				MEDIUM DENSITY				HIGH DENSITY				VERY HIGH DENSITY			
		DA	DT	AT	ADT	DA	DT	AT	ADT	DA	DT	AT	ADT	DA	DT	AT	ADT
30GE	**Total**	**16564**	**6006**	**13193**	**14643**	**16603**	**19252**	**7943**	**8356**	**3316**	**14135**	**607**	**5292**	**48**	**9452**	**974**	**1084**
	Bedfordshire	3554	951	3047	3111	4045	4308	1889	2219	802	3116	153	1216	8	2048	222	236
	Buckinghamshire	2737	843	2058	2224	2668	3074	1173	1475	343	2328	64	882	6	1863	102	91
	Cambridgeshire	4153	2115	3303	3991	3719	4572	1430	1594	655	3508	109	1014	3	1906	166	287
	Northamptonshire	2865	1063	2335	2425	3380	3639	1614	1216	612	2390	118	1090	4	1221	187	146
	Oxfordshire	3255	1034	2450	2892	2791	3659	1837	1852	904	2793	163	1090	27	2414	297	324
30GNS	**Total**	**18484**	**6241**	**16098**	**17101**	**19206**	**21546**	**8944**	**11292**	**4359**	**16717**	**592**	**5239**	**24**	**10342**	**1106**	**1203**
	Bedfordshire	4091	1095	4064	3768	4938	4833	2434	3216	1509	3953	159	1219	8	2370	348	416
	Buckinghamshire	3669	852	3316	3344	3907	4321	1596	3151	883	3560	58	873	4	2490	177	296
	Cambridgeshire	4068	2178	3232	3889	3645	4378	1431	1394	537	3451	108	998	0	1770	158	138
	Northamptonshire	2868	1063	2336	2433	3385	3661	1610	1157	584	2384	116	1080	0	1171	179	123
	Oxfordshire	3788	1053	3150	3667	3331	4353	1873	2374	846	3369	151	1069	12	2541	244	230
30YE	**Total**	**18628**	**11319**	**13962**	**19639**	**15671**	**22314**	**7832**	**5588**	**2507**	**13459**	**585**	**5443**	**0**	**8068**	**832**	**598**
	Bedfordshire	4522	3918	3587	5649	3635	5746	1809	1477	540	2866	148	1271	0	1762	196	134
	Buckinghamshire	3016	1477	2012	2922	2388	3605	1139	636	279	2122	59	876	0	1354	76	52
	Cambridgeshire	4311	3057	3335	4567	3580	4744	1462	1305	464	3456	108	1058	0	1767	156	122
	Northamptonshire	2939	1148	2430	2570	3377	3742	1627	1178	583	2414	118	1110	0	1206	178	123
	Oxfordshire	3840	1719	2598	3931	2691	4477	1786	992	641	2601	152	1128	0	1979	226	167
30YNS	**Total**	**11238**	**14820**	**10840**	**15334**	**8699**	**11392**	**7823**	**7912**	**7493**	**8997**	**7631**	**7740**	**7728**	**8118**	**7746**	**7489**
	Bedfordshire	2885	3631	2628	3997	2018	2894	1736	1749	1579	2173	1639	1616	1559	1772	1621	1502
	Buckinghamshire	2354	2722	2258	3502	1588	2438	1224	1258	1108	1590	1191	1179	1229	1377	1177	1120
	Cambridgeshire	2320	4488	2231	3345	1966	2273	1867	1899	1825	1951	1901	1953	1902	1895	1927	1934
	Northamptonshire	1468	1477	1571	1518	1481	1512	1526	1508	1489	1520	1500	1515	1521	1509	1503	1524
	Oxfordshire	2211	2502	2152	2972	1646	2275	1470	1498	1492	1763	1400	1477	1517	1565	1518	1409

Appendix 4: Averaged Ecosystem Services and Degree of Naturalness Scores for the Fixed Land Cover Classifications

Land Cover Type	Average Ecosystem Services Score	Degree of Naturalness Score
Deciduous Woodland	7	1
Coniferous Woodland	6	1
Arable	2	5
Improved Grassland	3	4
Neutral Grassland	6	2
Calcareous Grassland	6	2
Acid Grassland	6	2
Fen	5	1
Heather	4	2
Heather Grassland	5	3
Inland Rock	3	1
Freshwater	5	2
Saltmarsh	6	1
Urban/Suburban	(see appendix 5 below)	(see appendix 6 below)

Appendix 5: Average Ecosystem Services Scores for Urban Tiles under Different SuDS Scenarios

Housing	Density	Permeable Surfaces	Green Roofs & Permeable Paving	Green Roofs & Retention Basin	All SuDS
Apartments & Detached	Low	4	4	4	4
	Medium	4	4	4	4
	High	4	4	3	3
	Very High	4	4	3	3
Apartments & Terraced	Low	4	4	4	4
	Medium	4	4	3	3
	High	2	2	2	2
	Very High	1	2	2	1
Detached & Terraced	Low	4	4	4	4
	Medium	4	4	3	3
	High	1	3	2	3
	Very High	1	2	2	2
Apartments, Detached & Terraced	Low	4	4	4	4
	Medium	4	4	3	3
	High	2	2	2	2
	Very High	1	2	1	1

Appendix 6: Degree of Naturalness Scores for Urban Tiles under Different SuDS Scenarios

Housing	Density	Permeable Surfaces	Green Roofs & Permeable Paving	Green Roofs & Retention Basin	All SuDS
Apartments & Detached	Low	5	6	6	6
	Medium	5	6	6	6
	High	6	7	7	7
	Very High	6	7	7	7
Apartments & Terraced	Low	5	6	6	6
	Medium	6	6	7	7
	High	6	6	7	7
	Very High	7	7	7	7
Detached & Terraced	Low	5	6	6	6
	Medium	6	6	7	7
	High	6	7	7	7
	Very High	6	7	7	7
Apartments, Detached & Terraced	Low	5	6	6	6
	Medium	6	6	7	7
	High	6	6	7	7
	Very High	7	7	7	7

Appendix 7: Land Cover Patch Size Scores

Patch Size (ha)	Score
< 10	0
10.1 – 50.0	1
50.1 – 200.0	2
200.1 – 500.0	3
> 500.1	4