Stormwater Disconnection:

Transient Scenario Analysis of Intervention Flexibility

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A thesis submitted to the University of Sheffield in part fulfilment of the requirements for the degree of Doctor of Engineering

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January 2018

Abstract

Urban drainage networks protect people, society, and the environment from the hazards presented by domestic and industrial effluent, and urban stormwater run-off. However, urban drainage networks are financially and carbon intensive, and their failure results in damage to people and the environment. The likelihood and magnitude of failure is anticipated to increase in the future as a result of pressures including climate change and urbanisation. The rate and extent of these pressures manifesting is uncertain.

Sustainable drainage systems (SuDS) are structural measures that can be retrofitted to replace or augment an urban drainage network, reducing the likelihood of failure now and in the future.

Adaptation of infrastructure to encroaching future pressures requires infrastructure constructed in the present to be flexible. An existing method for assessing flexibility is combined with transient scenario analysis to enable the flexibility of conventional solutions, and source-control and regional-control retrofit SuDS interventions to be compared in two real-world case-study catchments. A new multi-criteria assessment framework is proposed for the comparison of these interventions.

A method for distributing retrofit SuDS within an urban drainage catchment is developed from first principles. It is a hydraulic modelling method based on identifying potentially disparate locations within an urban drainage catchment that possess similar times of concentration to a point of interest within the network. The concept of the efficiency of stormwater disconnection is introduced. The developed method is shown to be more effective at identifying efficient disconnection locations than existing methods in two real-world case study catchments.

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1. Introduction

1.1 Motivation for Research

Combined urban drainage networks capture stormwater run-off and domestic and trade wastewater, and facilitate their flow to wastewater treatment plants, protecting residents and the environment from the deleterious effects these flows can cause. It is estimated that over 11 billion litres of stormwater run-off and domestic wastewater is collected by urban drainage networks in the UK in an average day (DEFRA, 2012). Combined networks account for approximately 70% of the total 624,000 kilometres of sewerage infrastructure in the United Kingdom (Butler and Davies, 2009).

Large rainfall events can cause urban drainage networks to fail; resulting in flooding and pollution of the aquatic environment. Pressures such as climate change and urbanisation are predicted to lead to greater and more frequent instances of flooding and polluting in the future. The lack of ability to predict the future with precision is the biggest challenge in developing long term plans for stormwater management infrastructure (Manocha and Babovic, 2017). Adaptive management, under which the strategy is modified as one learns more about how the future is unfolding, is an appealing approach to dealing with uncertainty (Colombo and Byer 2012). Adaptive management approaches require flexibility (Colombo and Byer, 2012), which is the property of infrastructure, after implementation, to keep options open to cope with new requirements as a response to unknown future developments (Spiller *et al.,* 2015).

The conventional solution to failure, which is to increase the capacity of the network, is financially expensive and carbon intensive, and has been described as being unsustainable (Ashley and Hopkinson, 2002). Conventional solutions are inflexible as they are long-lived, large-scale, expensive infrastructure and difficult to modify (Pahl-Wostl, 2007).

Stormwater disconnection describes the act of severing the hydraulic connection between existing impermeable surfaces, such as roads and roofs, and the urban drainage network. Stormwater run-off generated by disconnected surfaces can subsequently be managed through installation of retrofit sustainable drainage systems (SuDS). Stormwater disconnection can help to remediate urban drainage systems that are deemed to be failing, and protect performance levels in the face of future pressures. Some retrofit SuDS can provide amenity, societal and bio-diversity benefits to local residents. Badger *et al.* (2014) identified that guidance literature for retrofit SuDS use in the UK is inherently weighted towards source-control SuDS due to adherence with the SuDS Management Train. Some studies have demonstrated that SuDS are more flexible than conventional drainage infrastructure in new developments (Eckart *et al.*, 2012). However there has been little academic research on the design and evaluation of flexibility in water and wastewater engineering (Spiller *et al*., 2015).

The research presented in this thesis has been supported by Scottish Water and Anglian Water as part of the EPSRC-funded STREAM Industrial Doctorate Centre for the Water Sector.

1.2 Aim and Objectives of the Thesis

The overall aim of this thesis is to compare the Flexibility of conventional solutions, and source-control and regional-control retrofit SuDS, when designed for the improvement of urban drainage network performance in the UK.

The objectives to achieve this aim are:

1. The development of a transient scenario framework for the assessment of flexibility that can be used in urban drainage studies;
2. The application of the transient scenario framework to real world case-study urban drainage catchments to examine the relative flexibility of source-control SuDS, regional-control SuDS, and conventional solutions;
3. The creation of a method to identify efficient stormwater disconnection locations, when the intervention is for the improvement of urban drainage network performance metrics.

1.3 Thesis Outline

The remainder of the thesis is organised into the following chapters:

Chapter 2: Literature Review

This chapter provides background information on urban drainage networks and sustainable drainage systems. A review of adaptive management methods is presented, and the concept of flexibility is discussed. There is a review of scenario forecasting methods, methods to distribute retrofit SuDS and pertinent assessment criteria for retrofit SuDS.

Chapter 3: Prioritising Stormwater Disconnection Locations for Urban Drainage Network Performance

A new method for the distribution of stormwater disconnection within an urban drainage network is developed from first principles and applied to two real-world case study catchments. The results obtained by this method are compared against those provided by existing distribution methods. Discussion of uncertainty and calibration issues is presented.

Chapter 4: Developing Transient Scenarios for the Assessment of Flexibility

Existing work on scenario analysis in urban drainage systems is modified to allow transient scenario analysis, which is required for flexibility testing. The manifestation of future pressures in each scenario is presented, and perspective theory is used to identify preferred stormwater management measures in each scenario

Chapter 5: Application of Transient Scenarios to Case-Study Catchments

The transient scenarios are applied to two real-world case-study urban drainage catchments in order to test the relative flexibility of conventional solutions, source-control SuDS and regional-control SuDS.

Chapter 6: Results and Discussion

The costs and benefits incurred during the transient scenarios are quantified and used to identify which intervention is most flexible through the use of minimax regret.

Chapter 7: Conclusions

This Chapter summarises the conclusions reached in the thesis and makes recommendations for future work.

1.4 Published Work

Aspects of this thesis have been presented in the following conference paper:

Badger, M., Stovin, V., Shucksmith, J., McCreath, K., Winter, D., and Brookes, A. (2014). Unanswered Questions When Optimising Retrofit SuDS *13th International Conference on Urban Drainage*, Sarawak, Malaysia.

2 Literature Review

2.1 Introduction

This chapter presents the problematic aspects of combined urban drainage networks, and identifies future pressures that are likely to exacerbate these problems in the future. The concepts of sustainable drainage and retrofit SuDS are introduced, and a design dichotomy that could be restricting the use of retrofit SuDS in the UK is identified. Methods to undertake adaptive management are presented, and the concept of flexibility is discussed. A method to assess the relative flexibility of different urban drainage infrastructure is identified. This method requires scenario planning, and the development and assessment of retrofit SuDS options. Approaches to these requirements are subsequently reviewed.

2.2 Sewer-cide and SuDS

Urban drainage networks protect urban areas from the risk of flooding. However, combined systems are of finite hydraulic capacity, and therefore provide inherently limited flood risk protection. Large rainfall events can generate stormwater run-off that exceeds the conveyance capacity of the network, resulting in hydraulic overload. Stormwater run-off which unintentionally escapes from the sewer system, or is unable to enter a hydraulically overloaded sewer system, is called exceedance flow. Exceedance flow can lead to flooding. Sewer flooding is a costly phenomenon; in 2007, two-thirds of the 57,000 flooded properties in the UK were inundated through the mechanism of sewer flooding, at a cost of £270 million (POST, 2007). As sewer flooding causes combined flow to be discharged into properties, this event is distasteful and unhygienic, with distress to customers supplementing associated financial burdens, environmental damage, and reputational damage to wastewater service providers. Combined sewer overflows (CSO) are features within urban drainage networks that allow combined flow to spill into a watercourse such that the risk of upstream uncontrolled sewer flooding is reduced, and that downstream sewerage infrastructure is a cost-effective size. The operation of CSO is a major contributing factor to the pollution of watercourses, for instance, the presence of faecal indicators (Stapleton *et al.*, 2008). Often the receiving watercourse is littered with sanitary detritus, which is an eyesore.

The primary function of urban drainage networks is the provision of hydraulic capacity; the ability to effectually drain some urban area under some prescribed storm condition. Where an urban drainage network can have no further urban area connected to it without exhibiting additional flooding or CSO metrics, then it may be said to be at hydraulic capacity. This can restrict economic growth, which is considered undesirable. The term “performance” may be used to describe the extent to which flooding, CSO and hydraulic capacity metrics are observed.

Pumping flow through urban drainage networks and treating flow at wastewater treatment works incurs considerable expense. The total power demand for wastewater pumping and treatment represents 1.5% of the total UK energy consumption, totalling 7,900 GWh per year, and costing £44.4 million for pumping and £149 million for treatment (UKWIR, 2010). Wastewater service provision is accountable for the emission of over 5 million tonnes of carbon dioxide equivalents (CO_2e) per year (DEFRA, 2008) in the UK. Additionally, some treatment process units require chemical dosing, incurring further financial expense.

Urban drainage networks have long service lives, due to a low rate of renewal. Reynolds (2000) reports that the current life expectancy for sewer pipes in the UK is 570 years. The length of this service life means the conditions in which they operate are likely to undergo continuous and unpredictable changes (Milly *et al.*, 2008).

2.2.1 Future Threats to Urban Drainage Networks

The performance of existing water infrastructure is likely be affected by climate change, urbanisation, and asset deterioration (Marlow *et al.*, 2013; Butler *et al.*, 2016). Changes in perception through time will contextualise the acceptability of performance changes. This section describes these phenomena.

The rate and magnitude of climate change that will occur is uncertain because it is correlated with the emission of greenhouse gases into the atmosphere, and these emissions are a product of complex and dynamic systems (Nakicenovic *et al.*, 2000). However it is possible to forecast likely general trends, albeit these vary around the globe (IPCC, 2014b). In the UK, the main effect of climate change on urban drainage infrastructure is the likely increase in winter precipitation (DEFRA, 2009). Indeed, an upward trend in rainfall extremes has been observed in the UK since the 1960s, and in particular the intensity of winter storms has increased (Osborn *et al.*, 2000), implying that flooding events may occur with increased frequency and severity as a result of climate change (Ekström *et al.*, 2005).

Recognising the importance of representing climate change effects on rainfall within urban drainage models to assess future performance, Butler & McEntee (2007) identified three methods in which this may be undertaken; precipitation intensity uplift, precipitation peakedness uplift, and climate model simulations.

Global or regional climate model (G/RCM) simulations can be used to assess changes in extreme rainfall resultant from climate change (Anandhi *et al*., 2011). In the context of the UK, the UK Climate Projections 2009 (UKCP09) project provides projections of rainfall conditions under high, medium and low climate change scenarios (Murphy *et al.*, 2010). The climate change scenarios are based on emissions scenarios found within the Intergovernmental Panel on Climate Change (IPCC) Special Report on Emissions Scenarios (SRES) (Nakicenovic *et al.*, 2000). No information is provided on the likelihood of each scenario manifesting (Gersonius *et al.*, 2013). Data derived from climate models has been used to assess the effect of climate change on urban drainage systems (e.g. Prudhomme *et al*., 2003; Semadeni-Davies *et al*., 2008; OFWAT 2011b), and these studies have corroborated that climate change is likely to degrade the performance of urban drainage systems.

Arnbjerg-Nielsen (2008) identified that climatic models simulate a coarse resolution of the world, which is often inappropriate for use in the context of water management (Figure 2-1), however techniques such as interpolation, statistical downscaling and high-resolution dynamic modelling can be used to generate more data of greater spatial and temporal resolution, which is more useful (Ekström *et al.*, 2005).

The second method identified by Butler & McEntee (2007) involves increasing the intensity of present-day rainfall events by some factor. A greater factor effectively represents a high rate of greenhouse gas emissions, and therefore represents a greater extent of climate change manifesting. This method is described as being attractively simply, and it can be used for both time-series rainfall and design storms (Butler and McEntee, 2007). The factor which is used to adjust present-day rainfall can be derived from global and regional climate models (Willems *et al*., 2012).

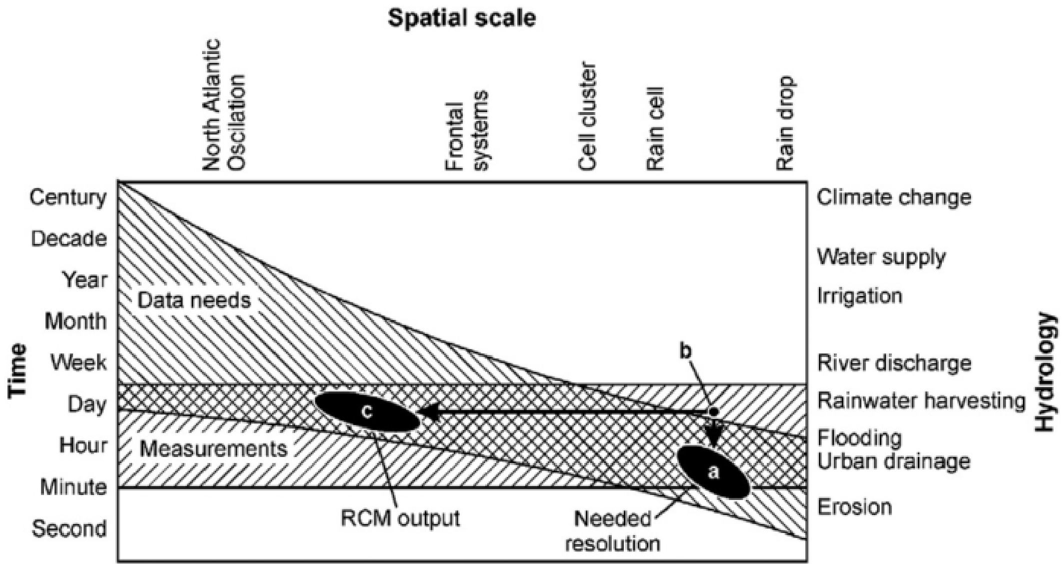


Figure 2-1: Scale mismatch between RCM outputs and required resolution for urban drainage studies (Arnbjerg-Nielsen, 2008).

Arnbjerg-Nielsen (2008) used global climate model data to derive climate change factors between 1.1 and 1.5 for Denmark for return periods between 2 and 100 years, and durations between 10 minutes and 24 hours over the next 100 years. Urich *et al.* (2013) used this work to justify increasing rainfall intensity by +10%, +30% and +50% compared to the present day. The increase was assumed to manifest linearly. Dong *et al.* (2017) used Arnbjerg-Nielsen's work to justify increasing rainfall intensity by +5%, +10%, +15% and +20% in their representation of climate change.

DEFRA (2006) generated climate change factors for application to UK rainfall events based on climate model simulations. These factors used epochs to represent how climate change may occur through time, as follows: +5% (from the present) for the period 1990-2025, +10% for the period 2025-2055, +20% for the period 2055-2085, and +30% for the period 2085-2115.

As part of the Adaptable Urban Drainage – Addressing Change in Intensity, Occurrence and Uncertainty of Stormwater (AUDACIOUS) project, rainfall intensity was increased by up to 40% to represent climate change (Ashley *et al.*, 2008).

A third option for the representation of climate change is to increase the "peakedness" of rainfall events (Butler and McEntee, 2007). Peakedness is the ratio of maximum to mean intensity of a rainfall event (Butler and Davies, 2009). There are difficulties in using an increase in peakedness, however, including applying this procedure to time-series rainfall data, which have more than one peak, and altering peakedness for different return periods (Butler and McEntee, 2007).

Urbanisation occurs in two forms; urban creep and urban expansion, however both phenomena generally stop rain from infiltrating to groundwater, and therefore increase the volume and rate of stormwater flowing into urban drainage systems. Urban creep describes the transformation of presently permeable areas within a catchment to impermeability, typically caused by the expansion of the area of individual residences as homeowners construct extensions or pave over gardens to create patios or car parking facilities (Allitt and Tewkesbury, 2009). Sampling over 533,000 houses in the UK via high-resolution aerial surveys (UKWIR, 2009b) demonstrated that urban creep has occurred at an average rate of 0.4 to 1.1 m^2/house/year, as a result of a series of diverse and inter-related factors. Allitt *et al.* (2009) propose several methods to calculate future rates of urban creep, including regression trees, and relating urban creep to type and density of residences. In one case-study, urban creep was predicted to increase flood volumes by 20% over a 20 year horizon (UKWIR, 2009b). Urban expansion describes the construction or redevelopment of buildings and infrastructure that previously had no hydraulic connection to the urban drainage network (Coombes *et al.*, 2002).

Dong *et al.* (2017) increased total impervious area by +5%, +10%, +15%, and 20%, and noted that urbanisation scenarios are related to local socio-economic development trends.

Casal-Campos *et al.* (2015) accounted for urban expansion by relating new impermeable area to forecast population growth, assuming an occupancy rate of 2.4 inhabitants per property, an urban density of 90 houses/ha and an impermeable area rate of 77% (34% roofs and 43% roads); typical values in UK terraced residential developments (Ward *et al.*, 2012). The rates of population growth were informed by UK-specific estimates by the Environment Agency and the Office of National Statistics.

OFTWAT (2011b) used values between 5% and 10% and suggested a 5% increase of impermeable area for 2040 horizon where data is unavailable

The deterioration of urban drainage networks as a function of age can result in structural problems and increased siltation (CIRIA, 2006). Ackers, Butler and May (1996) noted that sedimentation could reduce the pipe-full capacity of sewer by 10-20%. Casal-Campos (2016) used a reduction in pipe-full capacity to represent sedimentation to indicate deterioration of infrastructure in the future.

Any urban drainage network will exhibit an objective performance level under any set of conditions. The classification of the acceptability of this performance level is subjective, and is typically derived from legislation; for example, urban drainage systems in the UK are subject to the classifications stipulated under the pertinent enactments of the Urban Waste Water Treatment Directive (European Commission, 1991) and the Water Framework Directive (European Commission, 2000).

One insight into how the expectation of performance of urban drainage systems has changed through time is to look at legislation. There has been a general trend in recent decades for legislation governing urban drainage networks to become more stringent through time (Marsalek and Chocat, 2002). It is possible that this will continue. This pressure will exacerbate degradation in urban drainage network performance resulting from urbanisation, infrastructure deterioration and climate change.

It is important to note that pressures are likely to manifest in combination. The combined effect of the climate change, urban creep and urbanisation pressures is estimated to lead to an average increase in sewer flood volumes of 51% by 2040 for the 1 in 10 year return period (OFWAT, 2011b).

Uncertainty about the conditions under which urban drainage infrastructure will operate in the future has led to comment that the "current model" of providing urban drainage services is inappropriate or unsustainable (Butler *et al.,* 2003). In the current model, conventional solutions are typically applied, which increase the capacity of the network. This may be achieved through enlargement of its component parts, such as pipe upsizing, or by introducing dedicated flow storage tanks. Other conventional solutions include the creation of CSO to release excess flow to local watercourses, or improved screening of CSO effluent to collect larger, more unsightly debris that would otherwise pollute the watercourse. Intensifying the screening process, either through smaller aperture sizes or more rigorous maintenance regimes, can reduce the damage to the environment caused by CSO.

An alternative approach is to use retrofit Sustainable Drainage Systems (SuDS).

2.2.2 Sustainable Drainage Systems

Sustainable drainage is a term that envelopes a broad range of structural and non-structural measures that may be used to manage the risks presented by urban stormwater run-off. The guiding objective of sustainable drainage is that stormwater run-off is managed in a way that provides water quality, water quantity and enviro-societal benefits. Sustainable drainage is often compared to conventional urban drainage networks, which, it is suggested, emphasise the control of water quantity at the expense of the other two attributes (Chow *et al.*, 2013).

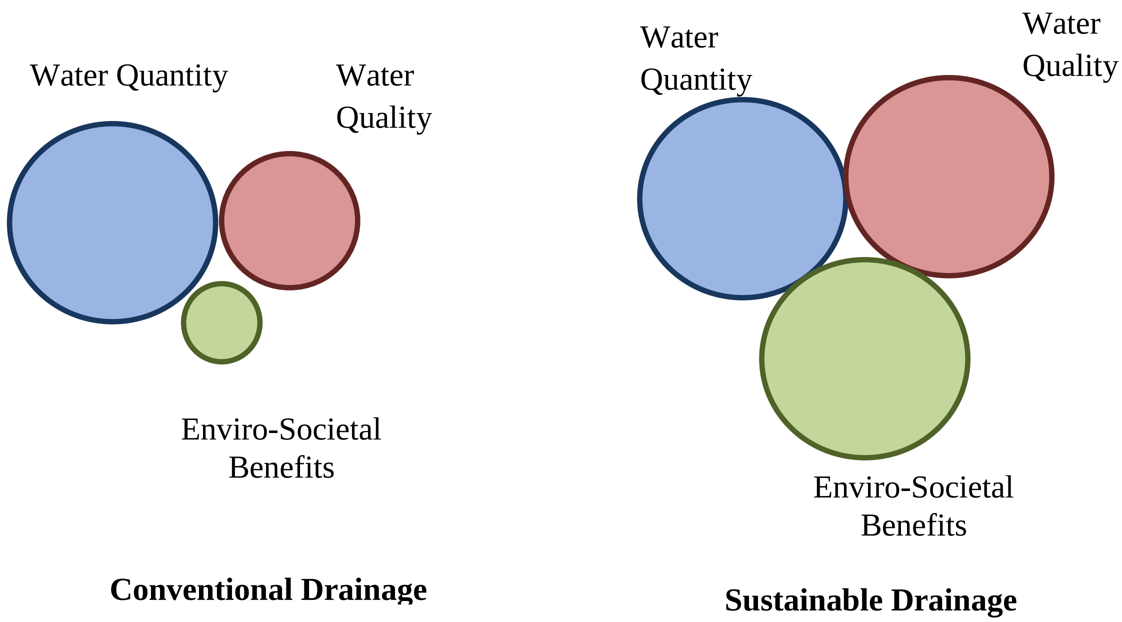


Figure 2-2: The emphasis of traditional and sustainable drainage systems (Chow *et al*., 2013).

Sustainable Drainage Systems (SuDS) are the structural component of sustainable drainage. SuDS use natural hydrological processes, such as infiltration and evapotranspiration, to manage the risks presented by urban stormwater run-off and contribute wherever possible to provide environmental enhancement (Woods-Ballard *et al.*, 2015). SuDS are part of the best management practice techniques used in the USA and are seen as contributing to water-sensitive urban design in Australia (Fletcher *et al.*, 2015; Scholz, 2015).

SuDS are typically located on the surface of urban landscape, another distinguishing facet compared to tradition drainage (DEFRA, 2008). Examples of the enviro-societal benefits that SuDS can provide include recreation value, air quality improvements, urban heat island mitigation, CO_2 reductions, noise reduction and the provision of biodiversity and habitat benefits (Center for Neighborhood Technology, 2010). As a result, SuDS can be used in place-making or urban regeneration schemes.

2.2.3 Stormwater Disconnection and Retrofit SuDS

Stormwater disconnection is the act of severing the flow connection between surfaces in the urban landscape and the urban drainage network (Ashley *et al.*, 2010). Stormwater run-off generated by the surfaces is therefore unable to flow into the urban drainage network. Disconnected stormwater is still required to be managed, and this management can be achieved through the construction of retrofit SuDS. Stormwater disconnection and retrofit SuDS are used to replace or augment an existing drainage system (Stovin, Swan and Moore, 2007).

Augustenborg, an inner city district of Malmo, Sweden underwent an extensive programme of retrofit disconnection between 1998 and 2002 in order to resolve both flooding and CSO problems; 90% of stormwater run-off generated by impervious surfaces in now fed into an open stormwater system (Kazmierczak and Carter, 2010), leading to significant improvements in the performance of the combined urban drainage system, particularly with regard to flooding and CSO metrics, even under extreme rainfall events (Villarreal *et al*. 2004). When describing this project, (Villarreal, Semadeni-Davies and Bengtsson, 2004) called retrofit surface water disconnection “unusual”, indicating that this is one of the earliest examples of stormwater disconnection in the literature (Villarreal, Semadeni-Davies and Bengtsson, 2004). The use of retrofit SuDS in Augustenborg helped to improve local biodiversity; green roofs have attracted birds and insects, and above-ground management of stormwater has encouraged local plants and wildlife (Kazmierczak and Carter, 2010).

In North America, cities such as Philadelphia, Seattle and Portland, Oregon have pursued large-scale institutional stormwater disconnection programmes. These programmes aim to reduce the impact of CSO operations on natural watercourses, as obliged by Clean Water Act, and to reduce the cost of operation and maintenance of piped infrastructure in the future

(USEPA, 2010). The Philadelphia Water Department are investing $1.6 billion over 20 years to convert a third of the city's impervious area to green stormwater infrastructure. This area is in excess of 4,000 acres, 55% of which is privately owned (Philadelphia Water Department, 2017). Portland invested $8 million in residential downspout disconnection, which diverts an annual 5.5 billion litres from 38,000 properties away from the sewer system. The investment required to obtain similar results from conventional solutions has been valued at $250 million, excluding operational savings (Foster, Lowe and Winkelman, 2011). Seattle Street Edge Alternatives (SEA) programme has seen extensive investment in the disconnection of roads, and the use of retrofit SuDS to "bulb" out into the road to provide traffic-calming properties. An exemplar SEA street reduced runoff to offsite by 98% for a typical 1 in 2 year storm event (Seattle Government, 2009).

2.2.4 The Source- vs. Regional- Control SuDS Dichotomy

Retrofit SuDS appear to represent a viable and cost-effective alternative to conventional piped drainage, however English and Welsh wastewater service providers have been slow to implement them (Stovin and Swan, 2007). OFWAT (2011a) identified a range of contextualised reasons why the use of retrofit SuDS has been more quickly and widely adapted in other countries compared to England and Wales; for example water shortage in Australia has prompted the retrofit of rainwater tanks to roofs. One prime explanation is that the responsibility for stormwater typically rests with a municipality or other public body, which appears to give the necessary public ownership to lead to the use of more innovative measures (OFWAT, 2011a). This can be contrasted with the private companies that are responsible for urban stormwater run-off in England.

Badger *et al*. (2014) identified that design guidance in the UK continually advocates for the design of retrofit SuDS in accordance with the SuDS Management Train, which is a conceptual aid to maximise the benefits of SuDS installations (Woods-Ballard *et al.*, 2007). The SuDS Management Train aims to ensure that stormwater run-off is directed through a number of SuDS in series such that alternative and complementary treatment regimes are applied to the run-off, and that run-off is attenuated and released to the environment at a rate that causes no deterioration in the environment through, for example, river scour. The Management Train categorises SuDS into three distinct hierarchical levels (Figure 2-3).

The first is "source-control"; through good-housekeeping measures, stormwater run-off should be returned to the environment as close to the source as possible. It is only in cases where the quantity or quality of the surface water is such that it cannot be dealt with at source that it should be directed into a "site-control" SuDS, the second stage in the Management Train. This logic is extended to assert that only if the quality or quantity of the stormwater run-off cannot be managed at a site-level should the third stage, the "regional-control" system, be employed.

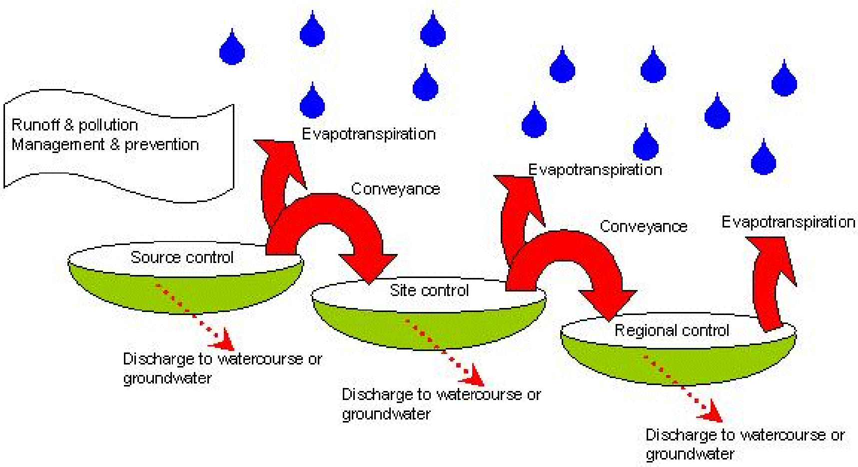


Figure 2-3: The SuDS Management Train (Woods-Ballard *et al*., 2007).

Although the Management Train was developed to maximise the benefits of SuDS in new developments (Woods-Ballard *et al*., 2007), guidance for retrofit SuDS (e.g. Digman, Ashley, Balmforth, Balmforth, *et al.*, 2012) has continued to advocate the Management Train design hierarchy. This design hierarchy is inherently weighted towards source-control SuDS and does not allow for the examination of the absolute or relative benefits of, for example, a regional-control SuDS alternative (Badger *et al.*, 2014). The constraints that an existing site can apply to retrofit SuDS are numerous, and this can oblige the creation of innovative solutions, that may lie outside the hierarchical approach (Singh *et al.*, 2005). A regional control SuDS, such as a pond or basin, may provide greater flood protection and amenity and biodiversity benefits than a source-control permeable pavement, although a higher capital cost may be incurred; the appreciation and negotiation of such trade-offs is an important aspect of retrofit SuDS design (Badger *et al.*, 2014). Examples of alternative, non-hierarchical designs of retrofit SuDS solutions are shown in Figure 2-4. Note that the source control SuDS options may require site- and regional-control SuDS but these are not shown.

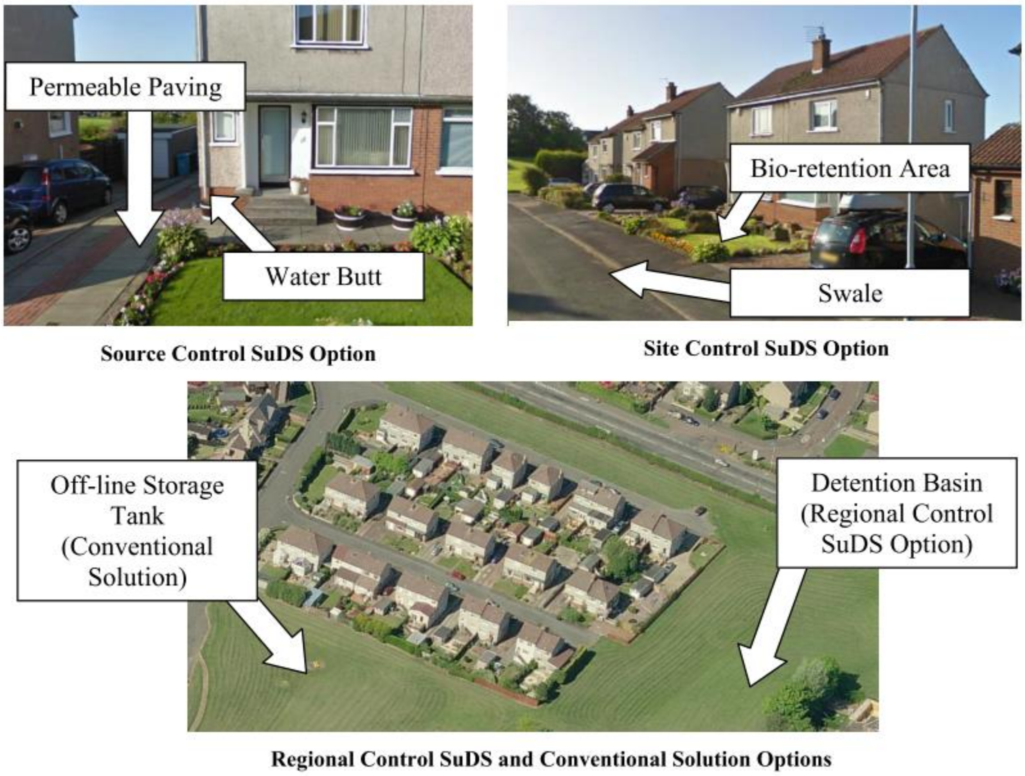


Figure 2-4: Examples on non-hierarchical retrofit SuDS solutions (after Badger *et al*., 2014).

The dedication of retrofit SuDS guidance to the SuDS management train does not align with the behaviour of important stakeholders to retrofit SuDS propagation in the UK; some wastewater service providers in the UK favour regional-control SuDS (e.g. Scottish Water 2015). It is therefore possible to identify a dichotomy in the design of retrofit SuDS in the UK; academic and technical guidance literature emphasises the importance of source-control based SuDS solutions, but the organisations responsible for managing stormwater favour regional-control SuDS.

Singh *et al.*, (2005) demonstrated that the characteristics of urban drainage networks may influence retrofit SuDS intervention selection by noting that end-of-pipe regional-control solutions managing flows from a disconnection storm sewer are likely to constitute a more simple method of removing stormwater flows from the network, resulting in reduced project cost and risk of failure.

Bastien *et al.*, (2009) compared different SuDS configurations, ranging from a single end-of-pipe regional-control SuDS to a multi-stage management train, for the use in an urban regeneration project in Glasgow, Scotland, and determined that the use of the management train was able to reduce the whole-life cost, and improve the water treatment capabilities, of the installation.

Moore *et al.,* (2012) presented a GIS-based methodology for the selection of stormwater disconnection opportunities to improve the performance of a CSO. The logic used to preferentially order the type of retrofit SuDS to be used was informed by the SuDS management train hierarchy.

2.3 Managing Uncertainty through Adaptive Management

As awareness of the encroachment of future pressures has increased, there have been calls to ensure urban drainage networks are resilient to future uncertainty (Moddemeyer, 2015). The concept of resilience emerged in ecological studies to describe the capacity of an ecosystem to survive, adapt and grow in the face of unforeseen changes (Holling, 1973). The concept of resilience has since been applied to other disciplines (Juan-García *et al.*, 2017), including engineering where resilience has come to focus on ensuring continuity and efficiency of system function during and after failure (Mugume *et al.*, 2015). One way of ensuring resilience is to successfully adapt to new conditions (Tran *et al.*, 2017).

Adaptation is common term in climate change literature, where it describes the act of natural or human systems changing in response to climatic stimuli (IPCC, 2007). Adaptation is one of two general responses to climate change, along with mitigation, which describes attempts to reduce greenhouse gas emission in order to eliminate or reduce the rate of climate change (Colombo and Byer, 2012). It is possible to broaden these definitions to apply to concepts other than climate change that threaten urban drainage network performance. Typically, the focus for urban drainage studies is on adaptation to future pressures as opposed to mitigation (e.g. Ashley *et al*., 2008), because not all pressures can be mitigated (Butler *et al.*, 2016).

Wastewater service providers in the UK spend billions of pounds (OFWAT, 2010) in capital investment programmes to remediate urban drainage networks. Remediation is the modification of an urban drainage network to achieve an objective performance level. If it is assumed that urban drainage networks provided the desired performance level at the time of their construction, then it follows that current capital investment is an example of adaptation.

There are broadly two approaches to adapting infrastructure (Pahl-Worstl, 2007), the "prediction and control" approach, and the "adaptive management" approach.

A "prediction-and-control" or "predict-then-adapt" approach is characterised by the assumption that future conditions and expected performance are predictable and system behaviour is deterministic (Medellin-Azuara *et al.*, 2007; Pahl-Wostl, 2007). This is the dominant historic approach to planning water management infrastructure and broadly requires the following four steps (Gersonius *et al.*, 2013):

1. Identification of the source of uncertainty, e.g. changing climate
2. Estimation of the consequent pressure, e.g. increased runoff
3. Assessment of the impact on the system, e.g. flooding increase
4. Responses developed to respond in order to maintain expected performance

Infrastructure developed under prediction and control approaches are characterised by being large and centralised, with prescriptive design procedures (Pahl-Wostl, 2007). Such approaches are well suited to the design of non-adaptable infrastructure where a single irreversible decision is made prior to project commencement (Colombo & Byer, 2012), such as reservoir construction. However, Dessai & Hulme (2009) conclude that these methods are significantly flawed, and can lead to uneconomic investment decisions (Gersonius *et al.*, 2013), particularly regarding infrastructure that could be altered through time as greater insight into the emerging hazards becomes available.

An alternative to prediction and control approaches is termed adaptive management. The core conceit of adaptive management is that the ability to predict future conditions is inherently limited (Pahl-Wostl, 2007), and therefore an preferable strategy is one that can be modified as one learns more about how the future is unfolding. Colombo and Byer (2012) describe adaptive management as an "appealing" approach to dealing with uncertainty. However, it has also been viewed as a way to defer the problem to a later date (Lee, 1999).

Early applications of adaptive plans can be found in the field of environmental management (Holling, 1978; Lee, 1993), and involve the ability to change plans based on new experience and insights (Pahl-Wostl *et al*., 2007).

An early method to improve the adaptability of plans is Assumption-Based Planning (ABP) (Dewar, 2002). Developed in the late 1980s, ABP requires that every assumption underlying a proposed or functioning plan is considered false, and the overall impact that this would cause on the plan is assessed. This enables the so-called “load-bearing” and “vulnerable” assumptions to be identified. Load-bearing assumptions are those that cause the greatest perturbation on the objectives of the original plan. Vulnerable assumptions are those that are most likely to be overturned by future events. Assumptions that are both load-bearing and vulnerable are the most likely cause of a plan being derailed. The benefit of using ABP is that it leads to the identification of “signposts”, an event or threshold that indicates an assumption is being broken, which indicates that some adaptation of the original plan is required. Adaptations are subsequently classified as either “shaping actions” or “hedging actions”. A shaping action is any step taken to protect an assumption. A hedging action is any step taken to prepare for an assumption failing.

The use of such contingency plans and triggers prompted Walker *et al*. (2013) to describe ABP as a “first step towards adaptive planning” and identify four general adaptive planning approaches that have been developed since the introduction of ABP.

2.3.1 Robust Decision Making

Robust Decision Making (RDM) is a planned, anticipatory, long term and widespread approach to decision making under uncertainty. The objective of RDM frameworks is to identify “robust” solutions, i.e. solutions that reduce vulnerability over the largest possible range of conditions (Walker *et al*. 2013). RDM frameworks are characterised by their use of scenarios to identify a single, static preferred plan for implementation (Hall *et al.*, 2012). The benefits of RDM include avoidance of commitment to a plan that would fail to meet its objectives, and it provides clear understanding of the conditions that would cause the plan to fail. RDM approaches typically include the following five steps (Hall *et al.*, 2012; Keefe, 2012):

1. Scoping: identify sources of uncertainty, policy options, key relationships and performance metrics
2. Simulation: identify a candidate policy and assess it against possible future scenarios
3. Scenario discovery: identify the range of sources of uncertainty that cause the policy to fail
4. Adaptation: identify hedging actions to address the vulnerabilities identified in 3.

5. Display: Plot outcomes of policies and probabilities of vulnerable scenarios, and chose the most robust plan for implementation.

Urich & Rauch (2014) used an RDM approach for the assessment of stormwater management strategies for Innsbuck, Austria. This study compared a do-nothing baseline with three stormwater disconnection-based adaptation strategies to future pressures, including climate change. These three strategies involved disconnecting 1%, 3% and 5% of the impermeable area and managing run-off via infiltration trenches. Ulrich and Rauch conclude that the 3% disconnection rate is preferable because the 1% rate is likely to lead to insufficient capacity for climate change, and the 5% rate is likely to present an over-adaptation to climate change.

Casal-Campos *et al.* (2015) compared the robustness of six stormwater management strategies; the exclusive use of the following interventions: permeable pavement, bio-retention planters, raingardens, surface water sewers, improved sewers and storage, and on-site treatment of wastewater. This study demonstrated that decentralised, green infrastructure strategies are more robust than centralised, grey infrastructure alternatives.

2.3.2 Adaptive Policymaking

Developing upon RDM by allowing for robust plans to be "dynamic", or changing through time, Adaptive Policymaking (APM) makes adaptation explicit from the outset. Changes that are inevitably required as external conditions change are managed as part of a larger process which aims to ensure the system meets its original goals. APM is also called Dynamic Adaptive Planning (DAP) in the literature.

APM is achieved in two phases. In phase 1, known as the design phase, the dynamic adaptive plan is developed. This involves analysing the current status of the system and the objectives for the future, and therefore the design of an appropriate monitoring program.

In phase 2, known as the implementation phase, the plan and monitoring program are implemented and adaptive actions are taken as required.

Within APM, there are two types of adaptive action; anticipatory and concurrent actions, and reactive actions.

Examples of anticipatory and concurrent actions include the following:

1. Mitigating actions: these reduce the likely adverse effects of a plan
2. Hedging actions: these reduce the uncertain adverse effects of a plan
3. Seizing actions: these seize likely available opportunities
4. Shaping actions: these reduce failure or enhance success

Examples of reactive actions include the following:

1. Defensive actions: these clarify the basic plan, preserve its benefits or react to external triggers.
2. Corrective actions: adjustments to the basic plan
3. Capitalising actions: these take advantage of opportunities to improve the performance of the basic plan
4. Reassessment: initiated when the plan has lost validity

An example of a capitalising action is provided by Gersonius *et al*. (2012). This study advocates that adaptation actions to mitigate increased precipitation intensity caused by climate change are undertaken in harmony with ongoing urban modification, regeneration or renewal activities. Such early integration of adaptation options is termed "mainstreaming adaptation".

Within the study by Ulrich and Rauch (2014) described in Section 2.3.3, there is the examination of an Adaptive Policymaking strategy, where the rate of impermeable area disconnected from the urban drainage network is determined at five-yearly intervals into the future. This approach is proven to be considerably more robust than the selection and continuation of a single disconnection rate.

Gersonius *et al*. (2013) built on Real Options (RO) analysis to demonstrate the value of adaptive policymaking approaches. RO analysis developed in the finance industry as a mechanism to decide when to implement options over an assessment period (Myers, 1984). RO analysis has been extended to the (re)design of infrastructure systems, leading to the development of Real In Options (RIO) analysis (deNeufville, 2003). RIO analysis focuses on assessing adaptations that can be undertaken as uncertainty about future pressures is reduced (Colombo and Byer, 2012; Gersonius *et al.*, 2013). RIO optimisation aims to minimise the costs of such adaptations (Gersonius *et al.*, 2015). Wang & Neufville (2004) presented a generic set of procedures to apply RIO analysis. These generic procedures were modified by

Gersonius *et al*. (2013) for the context of urban drainage, and were reported as a set of four steps:

1. Specify the uncertainty parameters
2. Identify possible adaptation measures
3. Formulate assessment methods
4. Conduct assessment

Applying this procedure to the adaptation of an urban drainage network in West Garforth, England as climate change manifests, Gersonius *et al*. (2013) demonstrated that the costs of adapting to climate change could be reduced by more than 20% by using a managed adaptive strategy rather than a predict-then-adapt approach.

Typically, adaptive policymaking approaches are "cause-based", in that they initially consider a pressure, such as rainfall increase caused by climate change, and then formulate responses in order to maintain an expected level of performance (Gersonius *et al.*, 2015).

2.3.3 Adaptation Tipping Points and Adaptation Pathways

Similarly to ABP methods, Adaptation Tipping Points (ATP) methods focus on the conditions that will cause a plan to fail. The condition under which a plan no longer meets its objectives is an adaptation tipping point (Kwakkel, Haasnoot and Walker, 2015). ATP approaches are "effect-based", in that initially an acceptable performance level is defined, and then the likelihood of this being achieved or maintained as future pressures manifest is assessed (Gersonius *et al.*, 2015).

ATP methods appreciate that when or even if the failure conditions will manifest is unknown, and so the requirement for adaptations is assessed through monitoring programmes. This is considered preferable to reliance on estimates of future conditions, such as climate projections, because such estimates may not be applicable for the scale of the system in question, and because the estimates are not immediately useful for influencing adaptation policy (Kwadijk *et al.*, 2010)

Undertaking an ATP approach requires the following five steps (Kwadijk *et al.*, 2010; Gersonius *et al.*, 2015):

1. Determine the system and conditions of interest, and identify the plan to achieve objectives
2. Quantify the acceptable performance level
3. Change the conditions affecting the system to identify the adaptation tipping points, assessed as the point the performance levels are compromised
4. Estimate the likely time at which this adaptation tipping point may occur
5. (Optional) Repeat stages 3 and 4 for potential adaptation strategies

Gersonius *et al*. (2012) applied the ATP method to a combined urban drainage network in Dordrecht, the Netherlands, and identified that an unacceptable level of sewer flooding corresponds with an increase in precipitation intensity of 25%, which is forecast to manifest in the worst case situation in 2055.

When an ATP is reached, an adaptation is required to ensure the system continues to function as required. Adaptation Pathways (AP) methods extend ATP methods by incorporating decision making about adaptation options at various ATPs. AP methods typically lend themselves to a presentation similar to a "route-map", whereby a journey into the future is made by transferring between carriages (adaptation options) at stations (adaptation tipping points), see Figure 2-5.

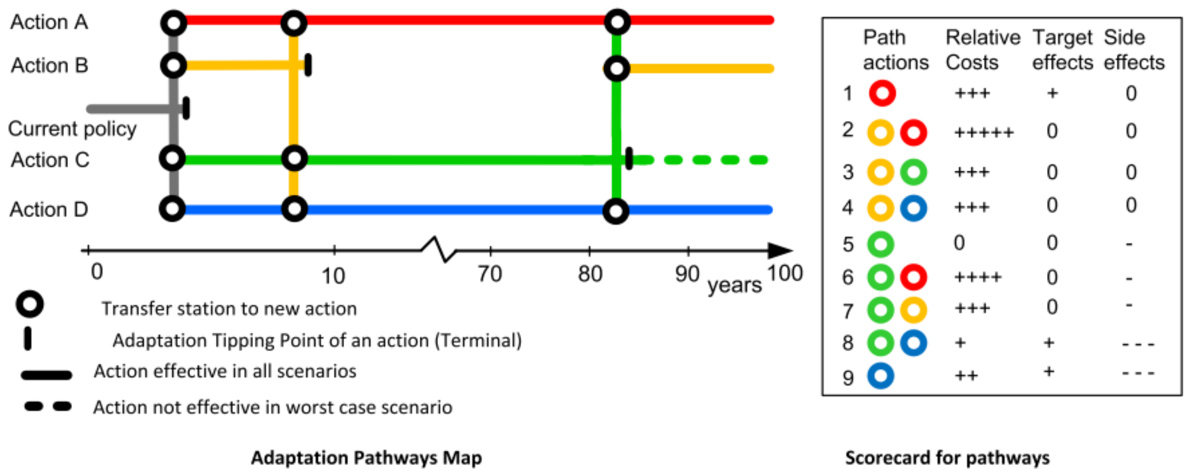


Figure 2-5: An example of an Adaptation Pathways (AP) route map (Kwakkel *et al*., 2015).

Manocha & Babovic (2017) undertook an assessment of stormwater infrastructure adaptation options in light of climate change scenarios in Singapore using adaptation tipping points and adaptation pathways. The adaptation options were increases to the minor system and source-control SuDS; there was no assessment of the benefits of regional-control SuDS.

2.3.4 Dynamic Adaptive Policy Pathways

Dynamic Adaptive Policy Pathways (DAPP) methods combine ABP, APM and AP, including identifying ways a plan may fail and designing actions to mitigate such failures, preparing future actions, and monitoring to understand when such actions should be implemented with adaptation pathways maps to visualise sequences of possible actions through time. In DAPP methods, a plan is conceptualised as a series of actions taken over time (Kwakkel, Haasnoot and Walker, 2015). As in other approaches, DAPP uses the identification of objectives, constraints, and uncertainties, bundled into scenarios. This enables assessment of problems or opportunities, and if and when reactive policy actions are required (Walker *et al.*, 2013). These actions are used as the building blocks for the creation of adaptation pathways. The DAPP approach has been applied to the lower Rhine Delta in the Netherlands ((Haasnoot *et al.*, 2013). The process for undertaking a DAPP approach requires the following ten steps (Haasnoot *et al.*, 2013), as part of an iterative process (Figure 2-6).

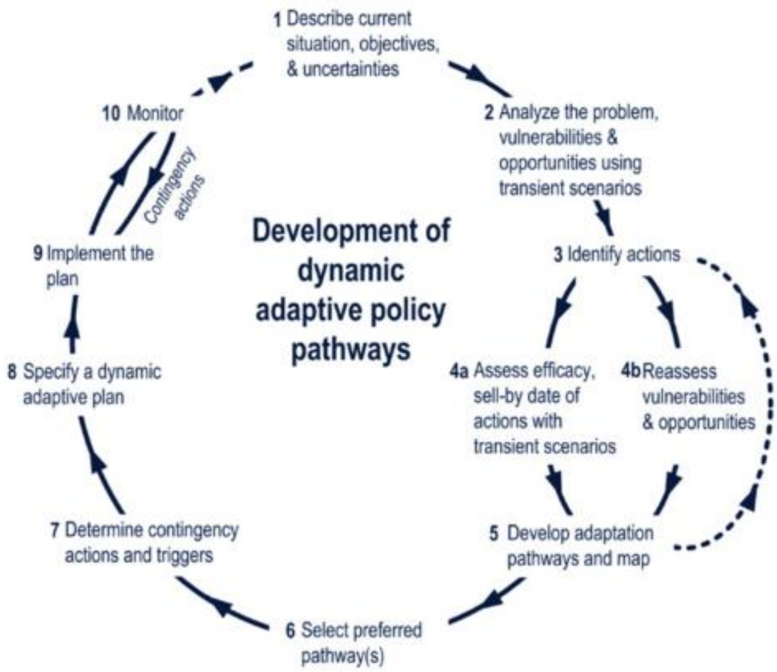


Figure 2-6: The Dynamic Adaptive Policy Pathways (DAPP) approach (Haasnoot *et al.*, 2013).

2.3 Flexibility

While there is no one agreed procedure for the development of an adaptive strategy to climate change (Manocha and Babovic, 2017), one target when undertaking an adaptive management approach is to increase adaptive capacity, where adaptive capacity is defined as the potential or capability of a system to change in order to perform better to current and future stresses (Pahl-Wostl, 2007). Generically, limited adaptive capacity is correlated with increased risk to climate change (IPCC, 2014a). The ability of adaptive management to adjust a system to future uncertainties as they unfold is derived from the inherent flexibility of the system (Gersonius *et al.*, 2013). In other terms, flexibility is the fundamental premise of adaptive management (Colombo and Byer, 2012). An assessment of flexibility should involve the comparison of the relative flexibility of different actions (Difrancesco and Tullos, 2015). The comparison of conventional solutions, source-control SuDS and regional-control SuDS to assess the relative flexibility inherent in each type of intervention would therefore be beneficial to enabling managed adaptive strategies.

Flexibility is thought to contribute to robustness (Difrancesco & Tullos, 2015), as well as being the fundamental premise of adaptive management (Colombo and Byer, 2012). Flexibility has been identified as an indicator of resilience (Yazdani, Appiah Otoo and Jeffrey, 2011), and a key element in planning water infrastructure (ASCE, 2012) as well as a key criteria for sustainability (Chocat *et al.*, 2004). However, there is a lack of clarity about how to prioritise improvements to flood risk management systems in order to achieve flexibility (Difrancesco and Tullos, 2015). Spiller *et al.* (2015) identified that there is little academic research on the design and evaluation of flexibility in water and wastewater engineering, although the use of SuDS has been shown to provide more flexible stormwater management infrastructure than conventional piped systems for new developments (Eckart *et al.*, 2012). Adaptation Pathways methods encourage decision makers to make adaptations while maintaining flexibility (Jeuken and Reeder, 2011), and this is achieved by adapting to changing external conditions while ensuring that options are left open to deal with plausible future scenarios (Walker *et al.*, 2013).

One type of flexibility related to the practice of adaptive management is institutional flexibility. Institutional flexibility can be recognised in organisations that have a formal mandate for adaptive management, leadership to support actions that push against historic institutional bounds, and a culture of participation and learning (Peat *et al.*, 2017). Institutional inflexibility has historically hindered attempts to use adaptive management approaches in the water sector (Peat *et al.*, 2017). Furthermore, non-structural actions are easier to reverse than structural actions, and thus represent greater flexibility (Kundzewicz, 2002). This research project focusses on the flexibility of structural actions, conventional solutions, source-control SuDS, and regional-control SuDS, taken to manage stormwater, and therefore assessing institutional flexibility and the flexibility of non-structural actions are topics outside the scope of this thesis.

There are multiple ways to define flexibility, including "the ability of infrastructure to under-go adaptation without incurring excessive cost" (UK Government, 2011), the ability, but not the obligation, to change a system (Eckart *et al.*, 2012), and the ability to reprioritise policy actions within a predefined time frame (van der Voorn *et al.*, 2017)

Conducting a review of literature from product design, civil engineering, aerospace and car manufacturing sectors in order to define flexibility for water and wastewater engineering, Spiller *et al.* (2015) settled on flexibility as the property of infrastructure, after implementation, to keep options open to cope with new requirements as a response to unknown future developments. Spiller *et al.* (2015) identified four types of flexibility:

1. Robust design: overdesign for probable future requirements, such as increasing the size of pipes where urban growth is forecast in the short-term. Robust design options are likely to incur additional costs in construction and maintenance.
2. Modular design: suitable for highly uncertain and dynamic conditions that require a fast response, such as containerised treatment facilities.
3. Phased design: where options to expand or improve the system are kept open into the future. Because phased designs are only provide flexibility for expansion, not capacity reduction, phased designs are only appropriate when demand or growth is highly likely to occur.
4. Design for remanufacturing: changing of infrastructure harmonised with the asset renewal process, for instance removing steel water distribution pipes and replacing with PVC.

Spiller *et al.* (2015) suggest that flexibility types 3 and 4 are appropriate for the water sector because the primary concerns are slowly-manifesting variables, such as climate change and urban development, rather than, for instance, market dynamics. Of particular interest, given the low rate of renewal of urban water infrastructure, is phased design flexibility.

Difrancesco & Tullos (2015) identified five characteristics of flexible water management systems:

1. Slack; the degree of excess capacity
2. Redundancy; the diversity of options available to meet the system's objectives
3. Connectivity; the ability of system components to interact
4. Adjustability; the ability to add, modify, and remove system components
5. Co-operation; the ability to share and use information

The characteristic of flexibility of interest when comparing conventional solutions, source-control SuDS and regional-control SuDS is the adjustability.

Flexibility has been of interest to water and wastewater engineering practitioners for some years. Despite this, there are few established methods to support the evaluation and assessment of Flexibility within urban drainage contexts.

Flexibility was identified as a criterion to assess the relative sustainability of urban drainage networks by Foxon (2000), and was subsequently used within the Sustainable Water industry Asset Resource Decisions (SWARD) framework (Ashley and Hopkinson, 2002).

SUDSLOC used the criteria "ease of retrofitting" (high, medium or low) and "design freeboard" (% or volume), to assess "System Adaptability" (Ellis *et al.*, 2011). These criteria are analogous to the characteristics of flexibility of "adjustability" and "slack" respectively.

The SWITCH project, which aimed to provoke a switch towards sustainability in urban water management practices, presented two approaches for the assessment of flexibility. Firstly, the Comparing the Flexibility of Alternative Solutions (COFAS) approach compares the performance of alternative solutions against a range of predefined metrics in different future scenarios (Peters *et al.*, 2010). A study using the COFAS approach in the case-study town of Kupferzell in Baden-Wurttemerg, Germany concluded that managing urban stormwater run-off from newly developed areas with decentralised infiltration SuDS provided greater flexibility than using combined or separate sewer systems (Sieker *et al.*, 2008). However, no adaptations were applied to these solutions in the future scenarios, so this study did not determine infrastructure flexibility so much as robustness.

Secondly, SWITCH presented a framework for “more detailed” of flexibility (Eckart *et al.*, 2012). This assessment of flexibility uses a wide range of future scenarios, the performance of the system, and the costs of adapting the system to the future scenarios (Eckart *et al*., 2012), and is undertaken in the following five step process:

1. Relevant future drivers for urban drainage systems are identified, and the range of future development is described, and the drivers are packaged into scenarios;
2. Alternative solutions are generated;
3. For all alternative solutions and future scenarios, the system performance is assessed via predefined performance metrics. Where system performance falls below an identified trigger level, flexibility options are implemented. Whole-life cost data is generated;
4. The performance and life-cycle costs of the different alternatives is ascertained;
5. The alternatives are compared.

In order to undertake the flexibility comparison of conventional, source-control SuDS and regional-control SuDS, it is proposed to use the “more-detailed” SWITCH framework as it represents a generic framework for the assessment of the flexibility of infrastructure options that has been applied within the context of urban drainage studies, for stormwater management infrastructure in new developments (Eckart *et al.*, 2012). Strictly, this assessment will be of the adjustability of the phased design of urban drainage networks.

Flexibility is a relative characteristic; it can only by assessed through comparison between different alternative options (Eckart, Tsegaye and Vairavamoorthy, 2013). Upton (1994) and Koste & Malhotra (1999) identified that three characteristics of flexibility; range, mobility, and uniformity. Hocke (2004) described how to assess these characteristics.

Range is defined as either the number of alternative actions that remain open (“range-number”), or the ability of the infrastructure to be altered in different ways (“range-heterogeneity”). Range may be assessed by the range of future states that can be managed by a particular option. The COFAS method for the assessment of flexibility called this metric the “capability of change”. Mobility is the ease with which change can occur. Mobility is typically assessed by the costs or duration required to undertake the change (MWH, 2014), where lower costs or durations are associated with higher flexibility. Uniformity is the ability to maintain system performance under different future states.

The framework for detailed measurement of flexibility incorporates the consideration of these three characteristics of flexibility (Eckart, Tsegaye and Vairavamoorthy, 2013); capability of change is assessed through the use of scenarios, mobility is assessed through placing costs on the process of applying flexibility options, and the performance of the system under future states is assessed using regret.

Regret is the difference between the performance of a strategy, and the performance of the best performing strategy in the same future scenario (Lempert, 2003). Low regret is an indicator of high flexibility (MWH, 2014). Peters *et al.*, (2011) recommend the use of minimax-regret principle for measuring the flexibility in urban drainage studies. Regret was used by Gersonius *et al.*, (2013) to demonstrate a SuDS-based adaptive management approach can lead to cost savings compared to conventional drainage.

Future drivers and their estimated ranges were presented in Section 2.2. The remainder of this literature review will examine scenario forecasting (Section 2.5), methods to generate alternative solutions (Section 2.6), appropriate performance and cost metrics (Section 2.7).

2.4 Scenario Planning

Scenarios narratives are one way to manage the inherent uncertainty associated with forecasting the future. Scenario planning is based on creating plausible, credible and internally consistent, but not expected, visions of potential futures, against which uncertain pressures can be mapped. Within the literature, there are broadly two ways to create scenarios; driver scenario frameworks and descriptive axes scenario frameworks.

Driver scenarios frameworks use a Driver-Pressure-State-Impact-Response (DPSIR) framework, early examples of which refer to a PSIR framework (Figure 2-7). DPSIR describe the interactions between external phenomena, the resultant impact on water systems, and the society affected (Haasnoot *et al*., 2012). For the example of climate change increasing winter precipitation in the UK:

- the driver is the release of greenhouse gasses changing the climate;
- the pressure is the consequential increase in rainfall as a result of climate change;
- the state is an increase in flooding or pollution from combined urban drainage networks;

- the impact is the disruption to people's lives and/or environmental damage;
- an example of a response is the construction of a stormwater storage tank.

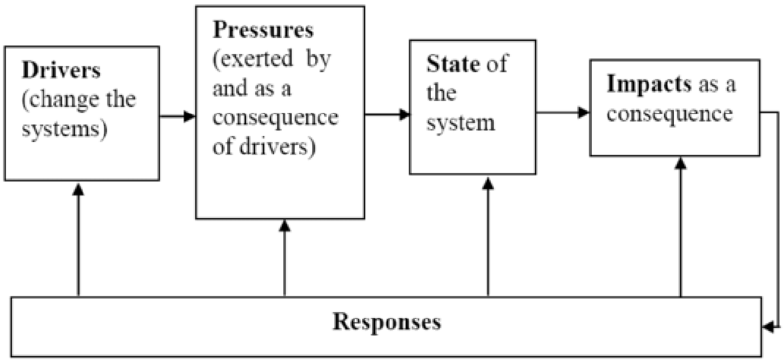


Figure 2-7: A Driver-Pressure-State-Impact-Response (DPSIR) scenario framework (Kwadijk *et al*., 2010).

Previous work to examine urban drainage flexibility has used a DPSIR framework to identify future states. Eckart *et al*. (2012) combined all future pressures into four scenarios; no change to current conditions, +20% and +70% changes to current conditions, and -40% change to current conditions over an 80 year period. Gersonius *et al*. (2013) examined flexibility under a single future pressure; a change in rainfall due to climate change.

Driver scenario frameworks are typically used to assess responses to on a single pressure, such as climate change. Descriptive axes scenario frameworks are useful for reducing the multiple permutations of future pressure manifestations into a small number of representative scenarios (Evans, Ashley and Hall, 2004).

The Foresight Futures 2020 scenarios present possible long-term social, economic and technological pathways that the UK may experience by mapping an axis describing the competing values of community and consumerism against an axis describing the competing governance structures of autonomy and interdependence (Figure 2-8) (OST, 2002). Each scenario was assigned descriptive summaries for traits such as social values, economic development, and income. Despite the Foresight Futures 2020 socio-techno-economic scenarios relating specifically to the UK, Evans *et al*. (2004) associated each scenario with outputs from global emissions scenarios from UKCIP0 based on generating a narrative link between the similar scenario futures.

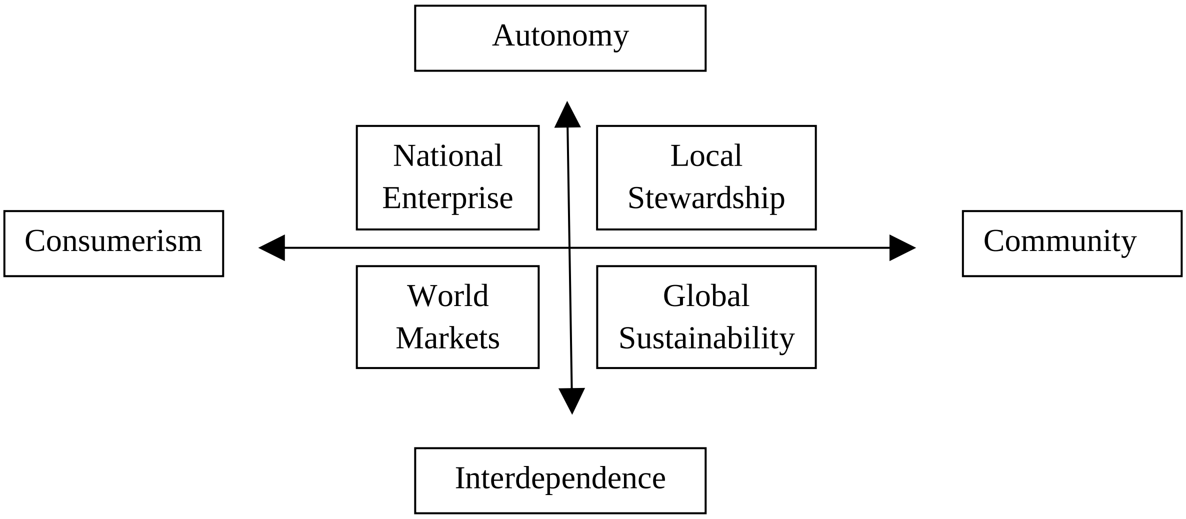


Figure 2-8: The Foresight Futures 2020 scenarios (OST, 2002)

Ashley and Tait (2012) built on this work by directly plotting an axis describing high and low climate change futures against an axis describing the "socio-economic capacity" of the future society, where socio-economic capacity refers to "adaptation potential", the capability to adapt urban drainage infrastructure to pressures as they arise. This work was intended to provide a generic mechanism to generate scenarios water infrastructure, and is malleable for use in different contexts. Ashley and Tait (2012) use three epochs (2025-2030s, 2050s, and 2080s-2100s), aligning with the UKCIP climate forecasts, to provide greater assurance that decisions are likely to be robust.

The scenarios constructed by Evans *et al*. (2004) and developed by Ashley and Tait (2012) provide narratives describing macro trends in, for example, GDP growth and governance structures. Therefore, it would be required to translate these macro trends into quantified representations of specific future pressures affecting urban drainage systems.

Casal-Campos *et al*. (2015) created four scenarios by mapping an axis describing the competing values of economic growth and environmental awareness against an axis describing the competing governance structures of consumerism and conservationism creating four scenarios; titled Markets, Innovation, Austerity and Lifestyles (Figure 2-9). Casal-Campos (2016) used this socio-economic scenario framework to provide a quantified estimation of some urban drainage-specific future pressures

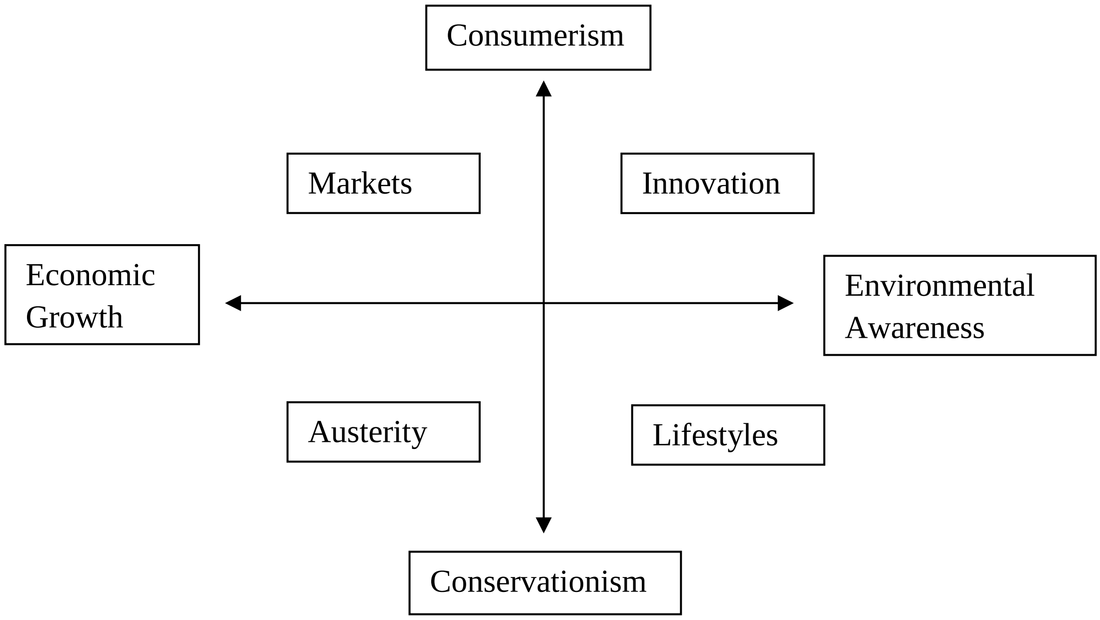


Figure 2-9: The axes of Casal-Campos' (2016) work.

The Markets scenario society is materialist, consumerist and highly motivated by personal financial gain. There is little emphasis on resource-efficiency, and environmental, amenity and biodiversity factors are considered to a very limited extent. Governmental and policy action focuses on economic growth and short-term issues. Cost is not a barrier to the society in this narrative, and there is the expectation that performance standards are maintained by profit-driven institutions. The regulation of utilities is lenient, and the aspiration is to keep prices low to maintain high demand of goods and services and economic growth. The objective of reducing flooding is therefore of high importance due to the economic implications of flooding.

The Innovation scenario is characterised by an emphasis on environmentalism and sustainability, however, people are not willing to compromising their quality of life to achieve these goals. The responsibility for sustainability lies with institutions which are empowered by strong policy making and legislation. There is a high-technology, high-wage economy. The society is underpinned by a desire to improve equality and prosperity for all, and this translates into environmental, financial and societal concerns. Safety from flooding is a priority in this world since life is expected to continue undisrupted. Any other environmental, social and economic objectives are equally valued to achieve more sustainable outcomes.

The Austerity scenario is characterised by economic decline. Public services suffer from under-investment, and cannot be relied upon to fulfil their service obligations, resulting citizens becoming more involved with decision making, an emphasis on self-reliance, and the decentralisation of previously centralised services. There is little technological innovation. Economic concerns are a noted priority within this society; there is little capital available and it must be spent thriftily. This comes at the expense of environmental issues, which have become secondary under this world. Although economic objectives are paramount in this world, social views are also relevant due to the decentralization of power structures.

The Lifestyles scenario is characterised by an absolute prioritisation the quality of the environment. It is strongly believed that inconsiderate individual lifestyles are the biggest hurdle to sustainability, and overcoming this is the responsibility of both government and the individual. Economic gain is a secondary issue, and people have learned to live with the risk of flooding as a small price to pay for the greater good. Thus, this state of the world prioritises environmental and social objectives, which are felt to have been abandoned for too long.

As this study assessed the robustness of intervention strategies that could be employed from the present day to 2050 (i.e. this study was not developed to support a managed adaptation approach), Casal-Campos did not associate preferred interventions to each scenario. Casal-Campos also focused on the objective performance of urban drainage systems and therefore did not consider how the perception of acceptable performance may change in the future.

Scenarios that take account of changing conditions between the present and some time in the future are called transient scenarios. Kwakkel *et al*. (2015) identified two types of transient scenarios; external transient scenarios, that describe only the rate and magnitude of pressures manifesting through time, and; complete transient scenarios, which provide “story lines” that include natural and socio-economic events (e.g. floods and economic crises), trends (e.g. climate change and changing public perceptions), and interactions between society and water infrastructure (flood impacts and interventions) (Haasnoot *et al.*, 2011). Transient scenarios are the only way to ensure the interplay between the unfolding scenario narrative and adaptations through time is explored (Kwakkel, Haasnoot and Walker, 2015). A complete transient scenario approach was achieved by Haasnoot *et al*. (2012) by using the DPSIR framework in an iterative fashion, with cycles of the framework occurring at every timestep. This aligns with Pahl-Wostl’s (2007) suggestion that adaptive management processes involve the ability to change plans based on new experience and insights. The uncertainty relating to future pressures is because they are the result of multiple, complex and dynamic socio-economic factors (Nakicenovic *et al.*, 2000; UKWIR, 2009b). Furthermore, the pressures interact; for instance, Tscheikner-Gratl *et al*., (2014) demonstrated that adapting centralised piped infrastructure to climate change as part of ongoing sewer rehabilitation, by increasing pipe sizes, can help offset the effects of climate change on system performance. Therefore, relating their manifestation in the future to narrative scenarios describing possible socio-economic futures is a valid approach.

Within complete transient scenario analysis, therefore, there is a requirement to assess the impact of pressures, and generate responses, in line with the values and perceptions of the society in question (Haasnoot *et al*., 2011). Forecasting is the process of quantifying phenomena within a descriptive narrative scenario framework (Bunn and Salo, 1993), in order to generate specific differences between scenarios.

Undertaking forecasting presupposes that the process of change into the future can be understood. Martelli (2014) identified three theories that describe how change occurs; the life-cycle theory, the teleological theory, and the evolutionary theory. The life-cycle theory suggests that, like in organisms, change is imminent, and moves the entity towards a future state that is related to the current state, and external factors can influence how the entity changes (Van De Ven & Poole, 1995). The teleological theory explains change as a conscious effort undertaken to obtain an envisioned, predefined end state, typically requiring adaptation along the way (Martelli, 2014; Van De Ven & Poole, 1995). Evolutionary theories suggest change is a response to external forces that selects based on structural elements within groups and organisations (Martelli, 2014; Van De Ven & Poole, 1995). While it can be seen that the adaptive management approach to handling uncertainty is a teleological method, Martelli suggested that life-cycle theories are the most suitable for forecasting scenario traits

Van De Ven & Poole (1995) identified perspective theory, group dynamic theory, and organisation system theory as types of life-cycle change theory. Perspective theory concentrates on the behaviour of people in carrying out change. Group dynamic theory assigns importance to the role of collective norms and group pressures. Organisation system theory views change as programmed or determined by structural transformations.

Perspective theory is derived from Cultural Theory (Douglas 1970; Thompson *et al*., 1990). The premise of cultural theory is that the acts of building, modifying and rejecting are intrinsically related to the individual preferences and world view of those making the choices (Thompson and Wildavsky, 1986). Cultural theory identifies two sets of constraints on human actions; "grid", and "group" (Douglas, 1982). Grid relates to the extent to which people are constrained by convention and regulation. Group measures the extent to which an individual tends to form collection or collaborative relationships. Within a "high group" society, choices are weighted towards solidarity, cooperation, reciprocity, and mutuality. This leads to four dominant cultural biases, or "perspectives"; the filters through which people value and interpret the world, and which acts to influence their actions (Van Asselt, 2000). These perspectives are Egalitarianism, Hierarchy, Individualism, and Fatalism (Figure 2-10).

Perspective theory has been applied by Dutch academics (e.g. Offermans *et al.*, 2011; Haasnoot *et al.*, 2012; Haasnoot *et al.*, 2013) to water management, who have described water management preferences for three perspectives; the Hierarchist, the Egalitarian, and the Individualist. The descriptions of the perspectives provided below combines general preferences from cultural theory studies and the water management-specific preferences generated by the stated studies.

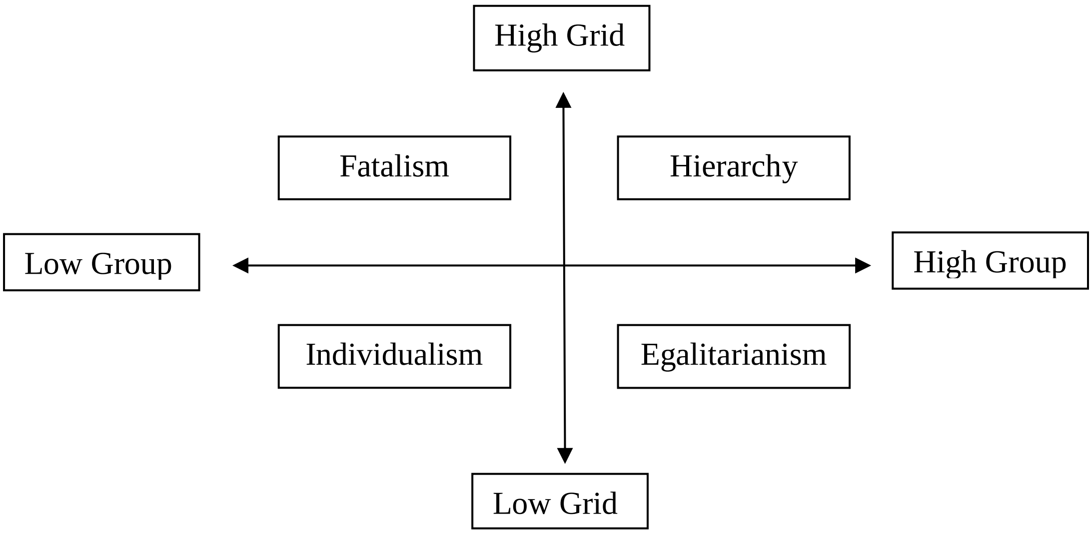


Figure 2-10: Location of perspectives on the grid vs group constraints (Van Asselt, 2000).

Egalitarians hold that nature is very fragile, and that small disturbances may have catastrophic results (Van Asselt, 2000). Man-made changes are likely to be detrimental to the environment, and therefore activities that are likely to harm the environment should be abandoned (Van Asselt 2000). Water management in the Egalitarian perspective is based on providing space for nature and water, with an emphasis on natural and ecological recovery, based on community decision making (Offermans, Haasnoot and Valkering, 2011).

Individualists are agents seeking to fulfil their ever increasing materialistic needs (Van Asselt, 2000), and can be characterised as risk seeking. The individualist considers human quality of life as the priority, with nature providing resources that can be exploited (Van Asselt 2000). The responsibility for water management rests with private companies, who are tasked with controlling water so it does not affect the maintenance of high economic growth (Offermans, Haasnoot and Valkering, 2011). There is high innovation but little attention to the environment and social solidarity (Offermans, Haasnoot and Valkering, 2011).

Hierarchists are risk-accepting, and believe that nature is robust within certain limits and is able to cope with small disturbances, and therefore stress that the relationship between humanity and nature is mutually dependant and must be balanced (Van Asselt 2000). Water management is characterised by an emphasis on safety and flood prevention, but it leaves space for economic and natural development (Offermans, Haasnoot and Valkering, 2011). Centralised governmental agencies are responsible for managing water (Offermans, Haasnoot and Valkering, 2011).

These studies examine water management from only three perspectives; there was no consideration of the Fatalistic perspective. Offermans, Haasnoot and Valkering (2011) state that the fatalist believes everything is determined by destiny, and that therefore policy and future strategies do not exist. Offermans, Haasnoot and Valkering were studying the development of strategies and policies, and as such concluded that the Fatalist perspective could be excluded from analysis.

Fatalistic tendencies are likely to manifest in a world with no escapes and few rewards, because passivity and resignation are more rational than entrepreneurial optimism (Ellis and Coyle, 1994). The only rational strategy for fatalists is to minimise the expenditure of resources (Chai and Wildavsky, 1994).

The use of perspective theory allowed the identification of preferred water management philosophies, and indeed measures. Although these studies focussed on sustainable adaptations to fluvial flood risk, and therefore the specific measures identified are not transferable to this thesis, this work demonstrates that the use of perspective theory to identify preferred measures is possible and valid.

Forecasting of future pressures to the Markets, Innovation, Austerity and Lifestyles scenarios was undertaken by Casal-Campos (2016) in a 4–step process:

1. The narrative depictions of scenarios are distilled to four scenario factors; regulations, maintenance, public attitudes, and technology.
2. The likely extent of future pressures of interest are related to these four scenario factors. For example, the magnitude of urban creep within a scenario is a function of the emphasis placed on regulations (limiting the uncontrolled paving-over of impermeable surfaces) and public attitudes (towards urban water management) in each scenario;
3. The qualitative magnitude of each future pressure is assessed under each scenario, ranked using High, Medium, and Low indicators, based on the depiction of the scenario against the scenario factors. For example, the level of urban creep in the "Markets" scenario is "High" because in this scenario there is little regulation of urban creep and low public attitudes for urban water management.
4. The qualitative magnitude of each future pressure is developed into a value based on a review of estimates within the literature.

However, this process led to some inconsistencies in the quantification of future pressures in each scenario. For example, Casal-Campos suggested that urban creep is likely to manifest to a greater extent in the Austerity scenario compared to the Innovation scenario, contradicting evidence presented by Allitt *et al*., (2009) that increased urban creep is associated with more affluent demographics, and seemingly at odds with the narrative descriptions of each scenario. This may be because the selection of scenario factors within Casal-Camos' work is based on the management of urban drainage systems, and are of limited validity when forecasting external factors.

Casal-Campos did not examine how climate change could differ between scenarios, and examined the objective performance of urban drainage systems, and thus did not consider the subjective concept of acceptable performance, which may change through time.

Futhermore, Casal-Campos examined the robustness of alternative management strategies that could be undertaken in the present day and continued in future years; this study did not examine a managed adaptation approach to future pressures. Therefore, it was not necessary for Casal-Campos to forecast which particular stormwater interventions would be preferable in each scenario. This is required in this thesis for two reasons:

1. To inform how stormwater run-off generated by urban area expansion will be managed in each scenario; i.e. stormwater management in new developments;

2. To inform which interventions will be used to provide solutions to ensure the urban drainage system performance is acceptable; i.e. retrofit SuDS or conventional solutions in response to failure.

As shown, a complete transient scenario framework, which includes both changing environmental factors in different scenarios and feedback between the external pressures, society and water infrastructure is required for the assessment of flexibility (Kwakkel *et al*., 2015; Haasnoot *et al*., 2011). Therefore the following modifications to Casal-Campos' scenarios are required to present a complete transient scenario framework that can be applied in this thesis:

1. The pressures of climate change and acceptable performance of urban drainage systems need to be described in each scenario, and all pressures need to be forecasted in each scenario;

2. The societal preference for different stormwater management infrastructure in different scenarios needs to be depicted. These preferences can be identified using perspective theory, and because we are not developing strategies or policies, rather identifying preferred interventions, it is appropriate to use four perspectives.

2.5 Alternative Solutions

SuDS were initially used to manage stormwater runoff from new developments on greenfield sites. An undeveloped site is a blank canvas. Application of structural SuDS in a retrofit context axiomatically involves the insertion of physical stormwater management measures into an existing urban landscape; the canvas is no longer blank, and must be altered. This Section examines some methods which have been developed to aid the task of distributing retrofit SuDS within an existing, developed urban area. The distribution of retrofit SuDS describes the physical, spatial location of retrofit SuDS within an urban drainage catchment. Distributing retrofit SuDS describes the act of deciding upon the distribution based on some logical rules.

Kuller *et al*., (2017) identified four types of decision support tools that can be used to inform the distribution of retrofit SuDS; planning simulation, technology selection, technology evaluation and spatial suitability evaluation tools.

Planning simulation tools are a relatively new approach to the distribution problem, and combine simulations of both the urban form and hydrology to direct the distribution of retrofit SuDS (Kuller *et al*., 2017).

The Urban Water Optioneering Tool (UWOT) (Makropoulos *et al.*, 2008) models the urban water cycle and provides a means of exploring the design and placement of retrofit SuDS in urban environments to improve sustainability (Bach, McCarthy and Deletic, 2015). UWOT focusses on relatively small sections of the urban landscape, up to the "development" scale, representing a group of households. As such, it is more suited more the design of new developments that for the identification of the optimal locations to distribute retrofit SuDS within a catchment.

The siting module of SUSTAIN (Lee *et al.*, 2012) was developed by the USEPA as a tool for evaluating, selecting, and placing retrofit SuDS based on cost and effectiveness to support the achievement of improvements to water quality in natural water bodies. Within SUSTAIN, the user must define the design of the retrofit SuDS intervention which is then located using the Optimisation Module based on feasibility criteria such as slope, soil and ground water characteristics. This limits its use for identifying optimal location for retrofit SuDS,

UWOT and SUSTAIN were identified as being unsuitable for use in exploring adaptive strategies as they are incapable of modelling dynamic changes in urban drainage infrastructure (Bach, McCarthy and Deletic, 2015).

UrbanBEATS (Bach, 2014) also uses feasibility criteria and urban form analysis, and augments this with the input of planning regulations, to support planning and policy-making for stormwater infrastructure by exploring possible futures. UrbanBEATS uses a two-stage process to automate the design of stormwater infrastructure; initially there is an assessment of all possible locations and scales at which SuDS can be implemented, followed by the generation of random combinations within the simulation region (Bach, McCarthy and Deletic, 2013). Each random combination is ranked based on a multi-criteria framework, and the top-scoring combinations are then used to simulate the resultant system performance (Bach, McCarthy and Deletic, 2013). This automation also does not allow for the innovative design of solutions outside those considered within the UrbanBEATS database. Innovative design is a key element of retrofit SuDS design (Digman *et al.*, 2012).

The Adaptation Support Tool (Voskamp and Van de Ven, 2015) allows its users to place systems on a map and evaluates the costs and benefits of the defined interventions.

Technology selection tools use multi-criteria assessment techniques to rank retrofit SuDS based on their suitability to certain locations or contexts once the location has been chosen (Kuller *et al*., 2017). Scholz (2006) identified 17 separate criteria to define suitability (Figure 2-11).

Criteria		
Contamination	Car park run-off	Soil infiltration
Possible SuDS area	Roads run-off	Impermeable surface
Catchment size	Road Type	Slope
Land values	Drainage type	Ownership
Run off quantity	Groundwater	Ecological impact
Roof run-off	Groundwater	Site classification

Figure 2-11: Constraints to retrofit SuDS use (Scholz, 2006).

Kuller *et al*. (2017) identified SUDSLOC (Viavattene *et al.*, 2008) as being a "rigorous" technology selection tool. SUDSLOC combines hydraulic and hydrological modelling, site selection criteria and data on the general suitability of different types of SuDS for those selection criteria from the DayWater comparison tool (Ellis *et al.*, 2006). Recently, there have been demonstrations of SUDSLOC's capability to distribute retrofit SuDS to reduce flooding (Ellis and Viavattene, 2014). However, this distribution is structured on the SuDS Management Train, and also the design and placement of the SuDS intervention occurs prior to assessment of the impact of the retrofit SuDS on the performance of the urban drainage system (Ellis and Viavattene, 2014). This leaves the possibility that there are locations at which retrofit SuDS installation is more challenging, but would provide a greater benefit to the urban drainage system.

Technology evaluation tools provide quantification of the multiple benefits provided by retrofit SuDS and thereby justification of investment made in retrofit SuDS (Kuller *et al.*,

2017) through a multi-criteria assessment framework. These tools can also be used to compare the benefits and dis-benefits of two or more competing retrofit SuDS designs, and have been constructed by industrial (e.g. Urrutiaguer *et al*., 2008), academic (e.g. Chow *et al*., 2013), and technical guidance (Digman *et al.*, 2015) practitioners. A review of multi-criteria assessment frameworks is presented in Section 2.6. However, their purpose is to assess the relative merits of retrofit SuDS designs, rather than inform their distribution.

Spatial suitability evaluation tools are spatially-explicit, meaning they assess the suitability of a location rather than the suitability of a particular SuDS. Spatial evaluation tools constitute the earliest attempt to impose some logic on the distribution of retrofit SuDS in an urban area to efficiently direct practitioner engineers to consider locations at which retrofit SuDS may be applicable.

Swan and Stovin (2002) proposed that stormwater disconnection should preferentially be undertaken on institutional roofs due to the relative ease with which retrofit SuDS schemes can be promoted, implemented, managed and monitored compared to, for example, highways. Recent work by Backhaus & Fryd (2012) has corroborated the early work by Swan and Stovin. Backhaus & Fryd undertook a multi-disciplinary investigation into the design process for retrofit SuDS in Copenhagen, Denmark, and identified that a similar coarse hierarchy for directing retrofit SuDS installations by ease of installation could be useful. This Danish study identified a wider range of urban land uses, the results show some symmetry with Swan and Stovin; notably, that institutional roofs should be targeted for stormwater disconnection ahead of residential roofs (Figure 2-12). The USEPA has developed a number of guidance documents to encourage improved management of stormwater in urban areas for environmental improvement. The manual on Urban Stormwater Retrofit Practices (Schueler *et al*., 2014) uses land-use types to describe typical areas which may be amenable to the introduction of retrofit SuDS. The practitioning engineer is enabled to match the generic examples provided by the manual to similar urban landforms within their study area. However, the engineer cannot be confident that the disconnection of stormwater in the locations suggest by the methods is likely to provide significant benefit to the performance of the combined sewer system. An array of industrial roofs may be easily and cheaply disconnected from the public sewer system, but if a flooding problem is located on a separate sewer branch, undertaking this disconnection may be non-beneficial. In this way, land-use driven methods effectively present a scatter-gun distribution tactic.

In addition to the four approaches types of decision support tool identified by Kuller *et al*. (2017) to distribute retrofit SuDS, there are two general approaches to distributing retrofit SuDS that do not require specific decision support tools; opportunistic and indiscriminate approaches.

	Urban surface type	
	Swan & Stovin, 2002	**Backhaus & Fryd, 2012**
Decreasing order of preference ↓	Institutional roofs Car parks Residential roofs Highways	Public institutions Transformational areas (i.e. brownfield sites) Industrial areas Areas bordering green areas/water "Open" apartment blocks Perimeter block development Row houses Single family homes

Figure 2-12: Demonstrating the similarities in Swan and Stovin's (2002) and Backhaus & Fryd's (2012) technology support distribution hierarchies

Opportunistic approach methods advocate installing retrofit SuDS "one by one as opportunities arise" (Ashley *et al.*, 2011) by aligning the retrofit of SuDS into an urban area with other works, external to the management of stormwater run-off (Digman *et al.*, 2012). For example, the renovation of a road surface provides an opportunity to direct the stormwater run-off from the road to a retrofit SuDS. Each individual retrofit will deliver minimal immediate improvement to the management of urban stormwater run-off, but the aggregated effects of many installations, it is argued, will produce substantial benefits in the long term. This approach of "nibbling" or "mainstreaming adaptation" was examined by Gersonius *et al.* (2012), who found that aligning opportunistic SuDS retrofit with refurbishment and redevelopment presents a cost-effective way to protect urban drainage systems from the pressures posed by climate change. Opportunistic approaches have been shown to be practicable in the UK for installing retrofit SuDS; examples from Digman *et al.* (2012) in Blackpool, Sheffield and Bristol have introduced retrofit SuDS by aligning with external urban regeneration or refurbishment works. The use of opportunistic approaches to retrofit SuDS distribution is not realistic for the engineer, however. This is due to the tension between the long timescale for urban renovation to occur and the short timescale in which urban drainage problems are expected to be resolved. While opportunistic retrofit SuDS cannot be relied upon to produce meaningful combined sewer system performance improvements in the short term, an outlier may be if a large scale regeneration or redevelopment was to be undertaken, and opportunistic retrofit at this location serendipitously provided benefits to performance metrics.

While opportunistic approaches suggest that retrofit SuDS should be undertaken whenever possible, indiscriminate methods advocate the installation of retrofit SuDS wherever possible. Indiscriminate methods advocate large-scale repetition of either a given type of retrofit SuDS opportunity or intervention. The indiscriminate approach is characterised by the involvement of governmental or municipal bodies actively pursuing, mandating or supporting the use of retrofit SuDS across an urban area. An example of the indiscriminate approach in practice is Portland's downspout disconnection programme (Foster *et al*. 2011).

Distributing retrofit SuDS in a catchment is an open-ended problem. Noting the reciprocating interaction inherent in the distribution of retrofit SuDS; the location at which the SuDS is installed affects the functioning of the SuDS, and the presence of SuDS affects the function and quality of the surroundings, Kuller *et al*. (2017), reviewing decision support tools for sustainable drainage, developed two key questions regarding the distribution of retrofit SuDS; "what do the SuDS need for optimal functioning?", and "where is the need for SuDS the highest?" This review of methods to distribute retrofit SuDS has identified that all methods that currently exist to inform the distribution of retrofit SuDS are centred on the former question, corresponding to the classical perception of "suitability" (Kuller *et al.*, 2017). Some methods do identify the resultant impact of a retrofit SuDS design on urban drainage performance, and thus try to answer the latter question; however, as discussed above, these cannot be used with confidence. For example, SUDSLOC and UrbanBEATS examine the resultant impact on the urban drainage system after the design of the retrofit SuDS option which means potential superior locations are not assessed. No method exists to inform the distribution of retrofit SuDS with the central objective of improving a pre-defined urban drainage performance metric, and this can be seen in industrial attempts to use retrofit SuDS in the UK (e.g. Hyder Consulting, 2004) which used Swan and Stovin's land-use hierarchical approach to stormwater disconnection distribution.

There is therefore a requirement for the development of an approach to distribute retrofit SuDS in order to provide a pre-defined improvement to the performance of an urban drainage system. The usefulness and therefore uptake of decision support tools can be improved by ensuring the tools are simple and heuristic (te Brömmelstroet and Bertolini, 2008).

2.6 Assessing System Performance

A typical assessment of a conventional stormwater intervention within the UK water industry is a cost-benefit assessment whereby the total financial cost of the intervention is estimated and the contribution the intervention may make towards achieving some stated performance objective is understood, usually through representation of the intervention within hydraulic modelling software. By undertaking this process for a number of intervention options for a known problem, it is possible to understand which intervention option provides the greatest performance improvement for the least financial outlay. This data is used to justify the selection and subsequent construction of a preferred intervention. The use of SuDS to improve urban drainage system performance does not fit easily into this assessment process. SuDS have a wider range of associated costs and benefits than conventional solutions. For example, some SuDS can provide benefits to biodiversity.

Multi-criteria assessment frameworks (MCAF) provide one approach to negotiate this problem. MCAF incorporate more than two criteria within a decision making process, allowing a greater range of aspects of a stormwater intervention to be accommodated within the assessment process, including data with heterogeneous units. For this reason, MCAF have been widely-used in the environmental context (Salminen *et al.,* 1998).

By definition, the purpose of the construction of stormwater interventions to improve urban drainage network performance is to achieve performance improvements. Typically there will be a specific design objective that forms the primary motivating factor for the construction of the intervention. This may be, for example, ensuring that no flooding occurs at a specific location under specified rainfall conditions. However, stormwater interventions may be able to provide positive contributions to other aspects of urban drainage network performance as secondary benefits. It is important that such secondary benefits are captured when constructing a suitable MCAF for this study. A report commissioned by Scottish Water (Atkins, 2004) identified that use of retrofit SuDS interventions could provide the benefits to a wastewater service provider described in Table 2-1. However, not all SuDS would provide all the benefits described.

Table 2-1: The benefits of retrofit SuDS accruing to wastewater service providers (Atkins, 2004).

Aspect	Description	Benefit
Reduction in sewer flooding risk	Internal and/or external flooding as a result of the hydraulic overload of urban drainage infrastructure by stormwater run-off	Improved customer relations. Improved performance statistics
Water quality improvements	Reducing the impact on the environment caused by the operation of overflows triggered by stormwater run-off	Compliance with environmental legislation
Increased headroom	Reduction of peak flow rates through existing piped infrastructure	Reduction of capital costs associated with construction of new piped infrastructure and allowing economic growth
Financial and energy savings	Reduced volumes of flow passing through pumping stations and/or wastewater treatment works	Decrease in electrical energy use. Reduction in the carbon footprint of the company.

Furthermore, certain SuDS provide a range of intangible benefits, such as improved biodiversity and area for recreational activity. These benefits do not accrue directly to the wastewater service provider, but instead accrue to a variety of stakeholders including the public in close proximity to the infrastructure, society more widely, and the environment. These intangible benefits are often classified under the headings “amenity”, “biodiversity” and “societal” benefits, which when taken together form an assessment of the “social and urban community benefits” (Ellis *et al.,* 2006) provided by a stormwater intervention. The assessment of the social and urban community benefits provided by a stormwater intervention is particularly important when assessing a retrofit SuDS intervention

A point of particular note is the inadequate level of carbon consideration within existing decision support systems. The assessment of carbon is of increasing importance within the UK water industry; as such it is vitally important that a carbon impact assessment is included within the decision making process for compliance with this objective to be secured. The assessment of carbon is of increasing importance within the UK water industry. Wastewater service providers have been set the objective to achieve an 80% reduction (from the 1990 baseline) in carbon emissions by 2050 under the CRC Energy Efficiency Scheme (UK Government, 2010).

Some SuDS may reduce the volume of stormwater flowing through pumping stations and treatment, reducing energy demand and therefore reducing the carbon impact of the urban drainage network as a whole. Additionally, the construction of SuDS and conventional solutions incurs a carbon cost in terms of embodied carbon. In order to provide the important, required holistic assessment of the carbon impact of a SuDS option, it is necessary to ensure the assessment of embodied carbon is made alongside the full assessment of offset carbon.

Based on these remarks, Badger *et al.*, (2014) presented a range of criteria pertinent to the assessment of retrofit SuDS installed for urban drainage performance improvement (Figure 2-13). The criteria can be grouped into four categories; Financial, Technical, Social and Urban Community Benefits, and Carbon criteria. Table 2-2 demonstrates that no existing SuDS decision support system meets assesses this range of criteria.

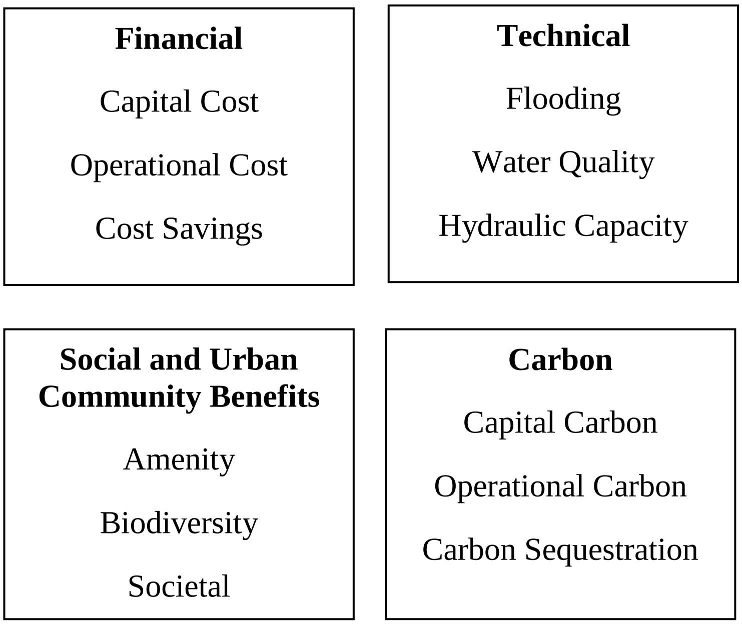


Figure 2-12: Pertinent retrofit SuDS assessment criteria (Badger et al., 2014)

Table 2-2: Pertinent retrofit SuDS assessment criteria in existing assessment frameworks.

Criteria	Indicator	Scholz, (2006)	DayWater (2005)	SNIFFER (2006)	Urrutiaguer (2008)	Wade *et al.* (2009)	Bastien *et al.* (2010)	Digman *et al.* (2012)	Chow *et al.* (2013)	Digman *et al.* (2015)
Financial	Capital Cost	N	Y	Y	Y	Y	Y	Y	Y	N
	Operational Cost	N	Y	Y	Y	Y	Y	Y	Y	N
	Cost Savings	N	N	N	N	Y	N	Y	P	Y
Technical	Flooding	N	Y	N	N	Y	Y	N	Y	Y
	Water Quality	P	P	Y	Y	Y	Y	Y	Y	Y
	Hydraulic Capacity	N	Y	N	N	N	N	N	N	N
Social and Urban Community Benefits	Amenity	N	Y	Y	N	N	N	P	Y	Y
	Biodiversity	P	N	N	N	N	N	P	Y	Y
	Societal	N	P	Y	P	N	N	N	P	Y
Carbon	Capital Carbon	N	N	N	N	N	N	P	N	N
	Operational Carbon	N	N	N	N	Y	N	P	N	N
	Carbon Sequestration	N	N	N	N	N	N	P	P	Y

Key: N-Not Considered; P-Partially Considered; Y-Satisfactorily Considered

2.7 Uncertainty and Sensitivity

The use of computational models of urban drainage systems is universal within the UK water industry when assessing the performance of real-world drainage systems, however uncertainty is intrinsic in such models, and uncertainty can have a significant effect of the decisions made based on such models (Sriwastava *et al.*, 2016). Deletic *et al.* (2012) presented a framework for the global assessment of modelling uncertainties for urban drainage models, and identified three sources of uncertainty: model input uncertainty, calibration uncertainty, and model structure uncertainty.

Model input uncertainty relates to uncertainty in "input data" and "model parameters". Input data uncertainty relates to uncertainty in any measured or estimated input data, such as the effective impermeable area, which is estimated and often used to calibrate the model. Model parameter uncertainty relates to "the sensitivity of a model to its parameters". Calibration uncertainty relates to such uncertainties as those within measured data used during model calibration and whether appropriate calibration variables and data sets are used during calibration. Model structure uncertainty relates to simplifications and/or inadequacies in the description of real-world processes.

Sensitivity analysis is the study of how important various input parameters are with respect to model outputs, and the subsequent identification of the sources of uncertainty which have the greatest impact on model response (Song *et al.*, 2015). Saltelli *et al.* (2006) identified three classifications of sensitivity analysis: local sensitivity analysis, global sensitivity analysis, and screening methods.

Local sensitivity analysis, alternatively called "one factor at a time" (OAT) sensitivity analysis, approaches modify a single source of uncertainty while the rest are held constant. The resultant impact on model output is assessed. The variation of the modified uncertainty source is typically undertaken for a range of published potential values. Local sensitivity analysis approaches are computationally inexpensive. However, because hydrodynamic models are typically non-linear, local sensitivity analysis can only deliver a rough estimation of sensitivity. Global sensitivity analyses assess how model outputs are influenced by multiple sources of uncertainty changing over their entire range (e.g. Dotto *et al.*, 2012), and as such enable a fuller understanding of sensitivity to be obtained than with local sensitivity analysis, at the cost of increased computational expense. While local and global sensitivity analyses typically attempt to highlight which input has the highest impact on the model result, screening methods attempt to identify model inputs that may be fixed at a given value without significantly influencing the model output. This helps to provide an overview of important model inputs and non-linearity. Such attempts at model simplification may be used as a precursor to global sensitivity analysis methods to reduce the computational expense incurred.

There is deep uncertainty associated with the prediction of the manifestation of urban drainage model inputs that may change over decadal time scales, such as climate change and changes in the urban landscape (Dessai *et al.*, 2009). Phenomena that have been predicted to change in the future and put pressure on the performance on urban drainage networks are described in Section 2.2.1. Section 2.2.1 also identifies a number of industrial and academic studies that have used sensitivity analysis methods to test the response of urban drainage models to a range of manifestations of uncertain parameters. Typically, increased rainfall intensity and changes to the urban landscape provoke the greatest deterioration in network performance (e.g. Mark *et al.*, 2008; Kleidorfer *et al.*, 2014; Urich and Rauch, 2014).

2.8 Chapter Summary

Urban drainage networks are financially- and carbon-expensive, and their failure causes damage to society and the environment. The likelihood and magnitude of network failure is predicted to increase in the future due to climate change, urbanisation and deterioration of infrastructure, although the rate and extent of these phenomena that will occur is uncertain. Compounding this, there is a trend of tightening legislation; expectations on urban drainage networks are increasing.

The lack of ability to predict the future with precision is the biggest challenge in developing long term plans for stormwater management infrastructure (Manocha and Babovic, 2017). Adaptive management approaches, under which the strategy is modified as one learns more about how the future is unfolding, is an appealing approach to dealing with uncertainty (Colombo and Byer 2012). Adaptive management approaches require flexibility (Colombo and Byer, 2012), which is the property of infrastructure, after implementation, to keep options open to cope with new requirements as a response to unknown future developments (Spiller *et al.*, 2015).

The conventional solution to failure, which is to increase the capacity of the network, is financially expensive and carbon intensive, and has been described as being unsustainable (Ashley and Hopkinson, 2002). Conventional solutions are inflexible as they are long-lived, large-scale, expensive infrastructure and difficult to modify (Pahl-Wostl, 2007).

Stormwater disconnection describes the act of severing the hydraulic connection between existing impermeable surfaces, such as roads and roofs, and the urban drainage network. Stormwater run-off generated by disconnected surfaces can subsequently be managed through installation of retrofit sustainable drainage systems (SuDS). Stormwater disconnection can help to remediate urban drainage systems that are deemed to be failing, and protect performance levels in the face of future pressures. Some retrofit SuDS can provide amenity, societal and bio-diversity benefits to local residents. Badger *et al.* (2014) identified that guidance literature for retrofit SuDS use in the UK is inherently weighted towards source-control SuDS due to adherence with the SuDS Management Train. Some studies have demonstrated that SuDS are more flexible than conventional drainage infrastructure in new developments (Eckart *et al.*, 2012), however there has been little academic research on the design and evaluation of flexibility in water and wastewater engineering (Spiller *et al.*, 2015).

Two alternative schools of retrofit SuDS design in the UK have been identified. The first advocates for the hierarchical use of SuDS (source-, then site-, then regional- control), which is a design philosophy inherited from the context of managing stormwater run-off from new developments. This approach is generally supported by academic studies and best practice guidance documentation. In contrast, retrofit of SuDS can require designs outside of this hierarchy, and wastewater service providers generally prefer regional-control SuDS placed on publicly accessible land as these are more easily and reliably maintained than a plethora of source-control SuDS on private land. A direct comparison of these schools of retrofit SuDS design in a real-world location to understand the trade-offs between each design school would be a useful, novel and interesting contribution to knowledge. As flexibility is such a fundamental aspect of adaptive management approaches, it would be useful to identify which of these design approaches provides the more flexible infrastructure. When assessing the benefits and costs of adaptation actions, environmental, and economic costs and benefits need to be accounted for (Allan, Xia and Pahl-Wostl, 2013). It was identified that no existing SuDS assessment framework allows the assessment of all the pertinent criteria involved in the retrofit SuDS for urban drainage performance improvement context.

Current methods to distribute retrofit SuDS use the classical perception of suitability, which may be inappropriate when attempting to resolve an identified urban drainage failure (Kuller *et al.*, 2017). Therefore, developing an approach to targeting stormwater disconnection which uses objective performance metrics to assess the relative merits of alternative stormwater disconnection distributions and extents will be a novel and useful contribution to this important aspect of retrofit SuDS design.

3 Prioritising Stormwater Disconnection Locations for Urban Drainage Network Performance Improvement

3.1 Introduction

Chapter 2 noted that Sustainable Drainage Systems (SuDS) techniques may be used to control the risks associated with urban stormwater run-off, a service traditionally provided by urban drainage systems. Stormwater disconnection describes the process of severing the flow path between an urban area and the urban drainage network. The use of SuDS to manage disconnected stormwater is termed retrofit SuDS. The act of stormwater disconnection, and the subsequent use of retrofit SuDS, can improve the performance of urban drainage systems.

The range of structural SuDS that are recognised in guidance texts, for example the SuDS Manual (Woods-Ballard *et al.,* 2007), is large, such that structural SuDS may conceptually be feasibly retrofit to many types of impermeable surface found in the urban landscape (e.g. Backhaus and Fryd, 2012). The universal applicability of the SuDS concept in urban areas could be considered a positive, but such universality creates a problem for practitioning engineers; there is uncertainty about where retrofit SuDS should be located within the urban area. As identified within Section 2.6, current decision support tools for SuDS focus on identifying which SuDS are most suitable for a given biophysical environment, rather than where the need for SuDS is the highest (Kuller *et al.*, 2017). From the perspective of urban drainage engineers, the location that is of greatest priority to undertake stormwater disconnection, and subsequently use retrofit SuDS, is that location which provides the greatest benefit to the performance metric(s) of interest. There is a clear requirement for decision support mechanisms to assist urban drainage practitioners with the distribution of retrofit SuDS in an urban area.

The objectives of this Chapter are:

1. To develop a new method for distributing stormwater disconnection within an urban drainage catchment that has the improvement of the performance of the urban drainage network as its principle objective;

2. To test the new method in (a) some generic representations of urban drainage catchments, and (b) a more complex representation of a real-world catchment, in order to demonstrate the efficacy of the new method compared to existing methods;

Section 3.2 describes the principles of a new method to distribute stormwater disconnection, which is based on fundamental hydraulic principles. It is hypothesised that this approach can lead to more efficient stormwater disconnection distributions. Section 3.3 details the construction and use of a spreadsheet-based model to allow for rapid catchment representation, modification and assessment in order to test this hypothesis. Sections 3.4 and 3.5 describe the application of the new method in two real-world case study catchments. Section 3.6 discusses calibration and uncertainty issues.

3.2 Stormwater Disconnection: Areal Co-Contribution Distribution Method

A new method to inform the distribution of stormwater disconnection within a catchment has been conceived. It is based on ensuring that the peak flow from a catchment at a specified point of interest within the urban drainage network is reduced. The basis of achieving this is the concept of areal co-contribution.

3.2.1 Superposition of Flows from Subcatchments

Consider a linear branch of sewer (Figure 3-1). A single pipe receives flow from a catchment comprising three identical subcatchments. There is a flow monitoring point (A) downstream of the catchment. A symmetrical peaked rainfall event falls uniformly across the catchment, generating stormwater runoff which flows through point A.

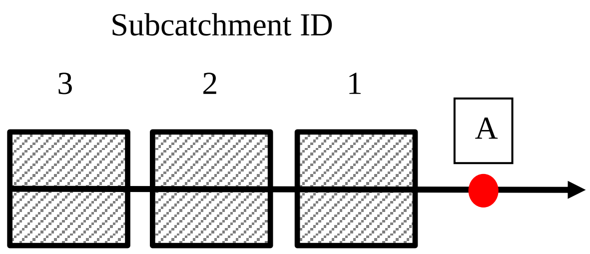


Figure 3-1: Schematic of linear catchment.

The arrival of stormwater runoff at point A is dictated by the time of concentration (t_c) of the subcatchment from which the stormwater was generated. Time of concentration comprises two components; time of entry (t_e) and time of flow (t_f), and is described by Equation 3-1.

$$t_c = t_e + t_f \qquad \text{(Equation 3-1)}$$

Let us assume that each subcatchment is identical, and as such possesses equal times of entry. The arrival of stormwater runoff at A is therefore dependent on the time of flow from each subcatchment. For simplicity, each subcatchment may be ascribed consistently incremental values for time of flow. Figure 3-2 shows the disaggregated arrival of stormwater flow at point A. The actual flow through point A may be calculated through application of the theory of superposition.

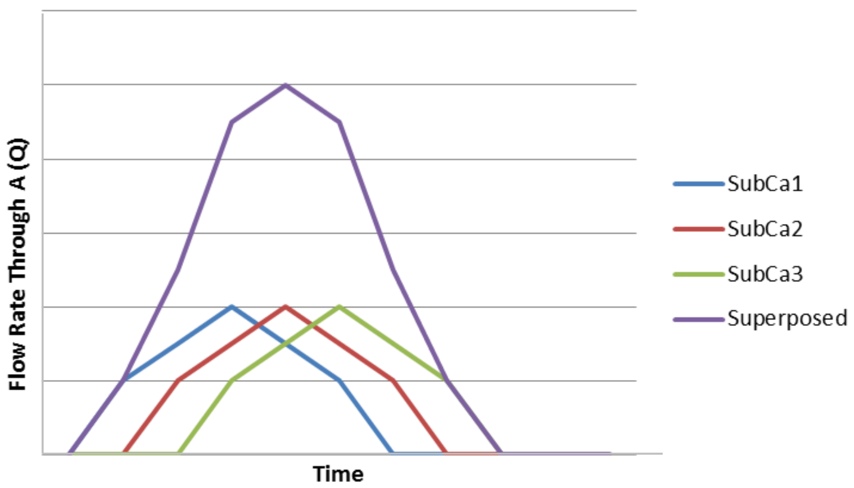


Figure 3-2: Depiction of stormwater flows through point A in Figure 3-1.

3.2.2 Peak Flow and Sewer Failure

A rainfall event falling on a catchment generates a flow profile, determined by the superposition of the flow profiles of its subcatchments. Any section of an urban drainage network will have an associated conveyance capacity. When the flow rate within the system exceeds the conveyance capacity, surcharge may be observed, leading to flooding events. CSO are set to allow a design flow rate to be retained within the urban drainage network, and excess flow is spilled. Sewer flooding and CSO spills are caused by the exceedance of the conveyance capacity of the sewer system by the flow profile from a catchment. The duration of the flow rate above the conveyance capacity of the system determines the volume of flood water and the volume and duration of a CSO spill. The reduction of the peak flow rate is therefore an important objective of stormwater disconnection for urban drainage performance improvement.

3.2.3 Areal Co-Contribution

To achieve the objective of reducing the peak flow rate, it is useful to understand if the generation of peak flows can be ascribed to particular locations in the catchment. This would enable engineers to "target" stormwater disconnection in appropriate parts of the catchment.

The characteristics of a catchment affect the flow hydrograph generated by that catchment. Real-world catchments, and therefore the associated flow hydrographs, are heterogeneous. Some examples of heterogeneity are slopes within catchments, the proportion of impermeable to permeable area and the type of urban area found in the catchment, which affects the run-off coefficient of the catchment.

One additional and important characteristic of urban drainage systems is areal co-contribution. Any urban drainage system can be sub-divided through the use of isochrones. Isochrones are imaginary lines that split a catchment into sections based on the time taken for stormwater run-off to flow to a designated point in the system.

An example of an urban drainage catchment split using isochrones is provided in Figure 3-3, showing a catchment of 11 subcatchments. Each subcatchment has a time of entry = 5 minutes. The time of flow from one subcatchment to the downstream subcatchment is 10 minutes. This allows the calculation of the time of concentration ($t_c = t_e + t_f$) for each subcatchment to Point A. In Figure 3-3, subcatchments with the same time of concentration to A have been split using isochrones.

Areal co-contribution is the concept of disparate areas located within a catchment possessing similar times of concentration to some pre-identified point of interest in the catchment; run-off from these areas will arrive at the point of interest simultaneously. Summing the amount of area within each isochrone segment gives the areal co-contribution at that time of concentration.

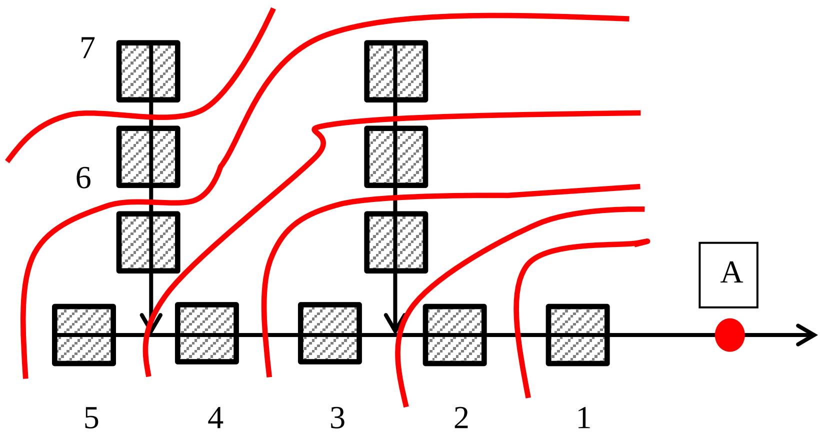


Figure 3-3: A simple catchment split via isochrones.

3.2.4 The Disaggregated Unit Hydrograph

Originally used in the field of natural hydrology, the unit hydrograph describes the hydrograph inherent to the specific catchment under investigation (Shaw, 1998). A unit depth of rainfall is applied uniformly to the catchment at a constant rate for some duration. The resultant hydrograph is the unit hydrograph for the rainfall conditions. The unit hydrograph can subsequently be adjusted to describe the response of the associated catchment to any rainfall event.

The unit hydrograph can theoretically be spatially disaggregated to understand how run-off from different isochrones contributes to it. Figure 3-4 shows the complete and disaggregated unit hydrograph for the catchment presented in Figure 3-3. It can be seen from Figure 3-4 that isochrone 5 produces the greatest peak flow response from any co-contributing area within the catchment, that flow from isochrones 3, 4, and 5 are contemporaneous with the overall peak flow response from the catchment, and that the greatest contribution to the peak flow is associated with isochrones 5. This suggests that stormwater disconnection may be usefully targeted within isochrones with the greatest areal co-contribution.

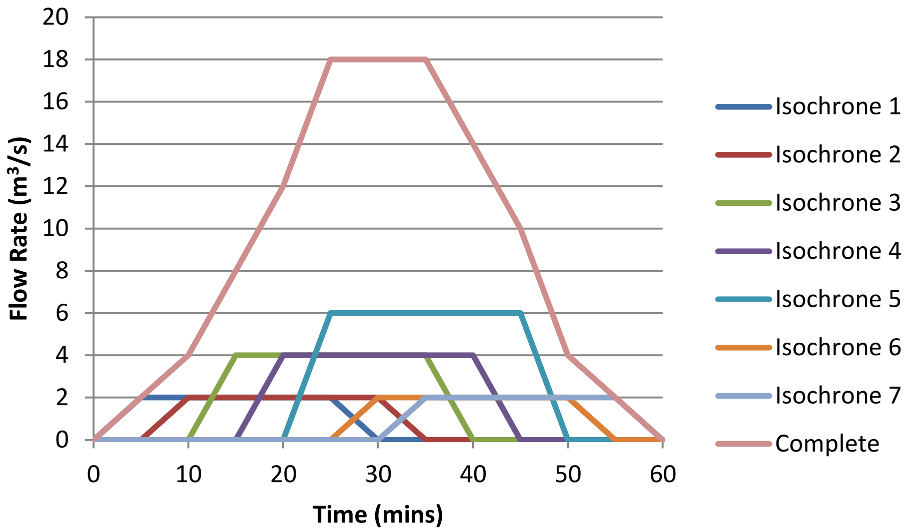


Figure 3-4: The resultant and disaggregated unit hydrograph for the catchment in Figure 3-3.

3.3 Application of the Areal Co-Contribution Method in Generic Catchments

It is hypothesised that by targeting stormwater disconnection in areas of greater areal co-contribution, stormwater disconnection interventions may be made more efficient compared to interventions informed by existing distribution guidance methods. The construction of a disaggregated unit hydrograph for the catchment under consideration is a key tool in targeting stormwater disconnection as it allows understanding of which subcatchments have similar times of concentration. In this context, the term "efficiency" is used to describe the reduction in peak flow rate per unit area disconnected from the catchment.

3.3.1 Model Construction

A spreadsheet-based model was developed to represent the flows from generic urban drainage catchments under user-specified rainfall events. This model comprised a simple run-off module which converts a rainfall event into a run-off hydrograph based on the characteristics of the urban drainage catchment being investigated. The conceptual schematic of this model is provided in Figure 3-5.

Two options were considered for the simplified run-off prediction; the Rational Method and the Time-Area Method. The Rational Method is a simple technique that allows estimation of the peak flow rate from a catchment under storm conditions through application of Equation 3-2.

$$Q = 2.78CiA \qquad \text{(Equation 3-2)}$$

Q peak flow rate (l/s)
C runoff coefficient (-)
i rainfall intensity (mm/h)
A catchment area (ha)

While the Rational Method is useful for some design purposes, it has been developed to create the Modified Rational Method. Within the Modified Rational Method (Equation 3-3), the runoff coefficient, C, is considered to be comprised of two components:

$$C = C_v C_R \qquad \text{(Equation 3-3)}$$

C_V volumetric runoff coefficient (-)
C_R dimensionless routing coefficient (-)

The volumetric runoff coefficient represents the proportion of rainfall the catchment receives which manifests as runoff. The dimensionless routing coefficient allows rainfall and catchment characteristics to influence the magnitude of peak flow rate.

A limitation of both the Rational Method and the Modified Rational Method is that area is treated as a constant, which means only peak flow rate from the catchment is calculated.

The Time-Area Method provides both the peak flow rate and a flow hydrograph from a catchment under storm conditions. The Time-Area Method splits a catchment through use of isochrones delineating sections of the catchment with similar times of concentration, with the closest sections of the catchment to the area under consideration contributing first. The flow from the catchment at incremental time-steps may be calculated to provide insight to how the flow from the catchment varies through time.

The use of the Time-Area Method is appropriate for this study for the following reasons:

a. Spatial variability in the catchment is accommodated;
b. It produces a peak flow rate and a flow hydrograph;
c. The speed of model construction, amendment and calculation.

The use of the Time-Area Method requires a rainfall hyetograph to be applied, rather than a set intensity as in the Rational Method and Modified Rational Method. The starting point for the generation of a rainfall hyetograph is setting the duration of storm event equal to the time of concentration of the catchment. This ensures that the peak flow rate from the catchment is at its maximum. A duration less than the catchment's time of concentration produces a reduced peak flow rate as not all parts of the catchment are contributing to flow. A duration greater than the catchment's time of concentration lowers the peak flow rate because duration and intensity of a storm are inversely proportional.

Having established the duration of the design storm, it is typical to estimate the average intensity of the storm through interrogation of an intensity-duration-frequency (IDF) relationship. IDF relationships are location specific, and so may not be appropriate for use on synthetic, generic catchments.

Rather, Holland's (1967) relationship between intensity, duration and frequency may be used. Holland presented a formula for an IDF relationship valid up to rainfall durations of 25 hours (Equation 3-4).

$$N = D(\frac{I}{25.4})^{-3.14} \qquad \text{(Equation 3-4)}$$

N number of times in 10 years during which rainfall occurs
I rainfall depth (mm)
D duration (h)

The use of Holland's formula is preferable to the use of an IDF curve for the following reasons:

a. It is applicable to the UK rather than a designated location within the UK;
b. It is simple to calculate numerically. IDF curves must be interrogated visually.

Using Equation 3-4 the average rainfall intensity for the design storm may be calculated. This must be translated to a time-varying hyetograph.

The Flood Studies Report (NERC, 1975) produced a set of standard, symmetrical rainfall profiles. The profile shape was found to not vary significantly with storm duration, return period or geographic region within the UK. Kellagher (1981) recommends the use of the 50 percentile summer profile for design of drainage systems. Summer storms were found to be more peaked than winter storms. A rainfall hyetograph can be generated by distributing the mean intensity over the storm duration for the Flood Studies Report 50 percentile summer storm profile.

The parameters used to define catchments within this model are the catchment lay-out, and the area of each subcatchment; both of which are defined by the user to represent a catchment of interest. The lay-out of the catchment was represented in the model through simple adjustment of the spreadsheet. The user selects the return period for the storm event, which is used in conjunction with the design of the catchment to create a rainfall an appropriate rainfall hyetograph. Time of entry was set to 1 minute and the time of flow between subcatchments was set to be 5 minutes.

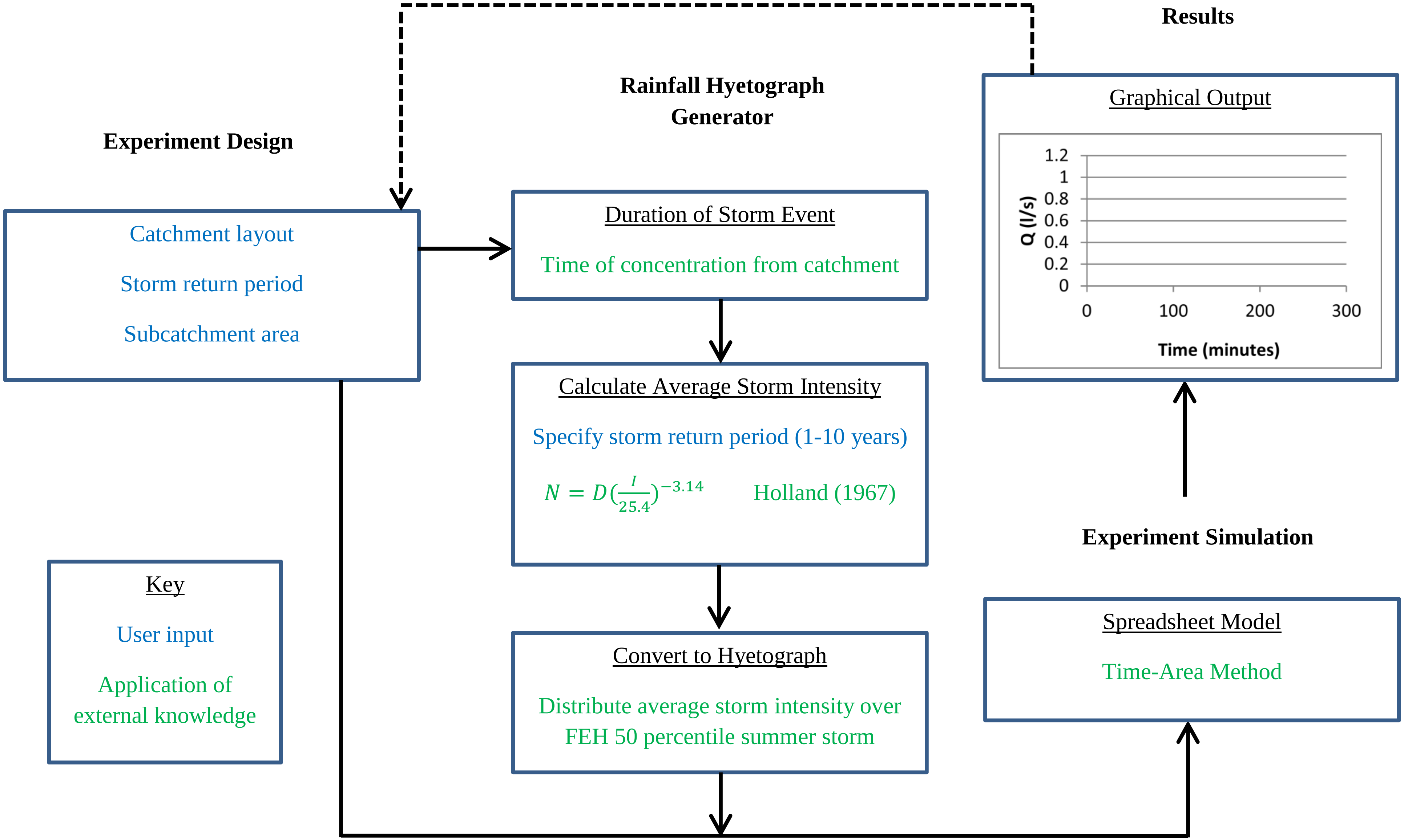


Figure 3-5: Conceptual schematic of a model for the representation of generic urban drainage catchments.

3.3.2 Examples and Results

Three representations of arbitrary generic urban drainage catchments were created within the model. The disaggregated unit hydrograph for each catchment was created based on the characteristics of the catchment, provided in Figures 3-6 to 3-8.

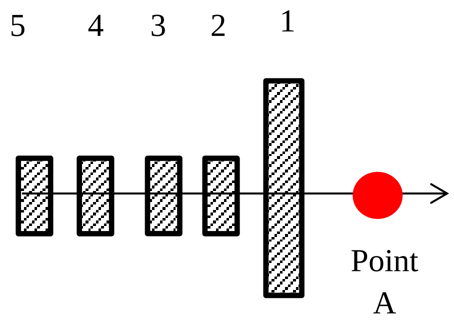
To test the usefulness of a maximum areal co-contribution disconnection approach, the three catchments were subject to three stormwater disconnection distribution strategies, and the peak flow rate resultant from the baseline and modified catchments was noted. Each disconnection strategy was limited to a disconnection budget of 20% of the area of the baseline catchment. Urich and Rauch (2014) used rates of 1, 3, and 5% annual disconnection of impermeable area from the combined urban drainage system over a 20 year period, so the figure of 20% may be considered an achievable amount of disconnection, representing stormwater disconnection over a period between four and 20 years at these rates. A disconnection budget was set because the effect of the spatial distribution, rather than amount, of stormwater distribution is of interest.

The disconnection approaches applied to each catchment representation were:

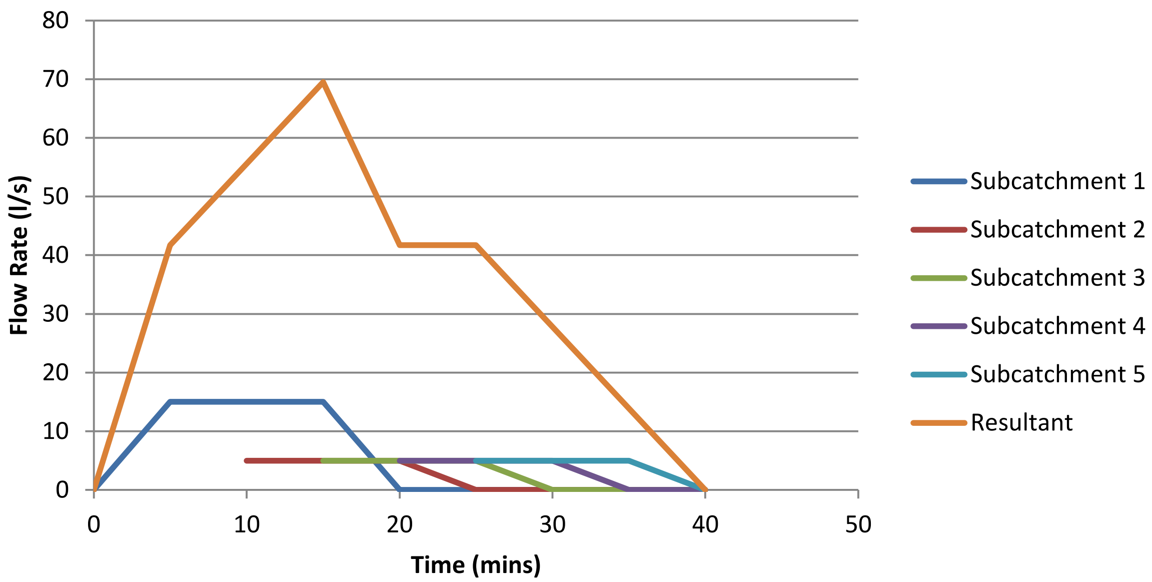
1. Indiscriminate: A uniform disconnection was applied across the catchment. An arbitrary disconnection of 20% of each subcatchment was conducted, representing the indiscriminate approach to stormwater disconnection;
2. Random: A random number generator was used to direct the disconnection of area from the catchment. Conducting stormwater disconnection with no regard to the resultant flow profile under investigation is analogous to targeting stormwater disconnection based on the classical perception of "suitability" (Kuller *et al.*, 2017) identified in the literature;
3. Unit hydrograph-Informed Disconnection: Disconnection of area from the catchment in locations indicated by the disaggregated unit hydrograph of the catchment.

The results presented for Catchments A, B and C demonstrate that maximum areal co-contribution is a primary cause of peak flow generation, and that targeting stormwater disconnection in larger areas of co-contribution is an effective method of reducing the peak flow rate resulting from the catchment. This is because the stormwater disconnection is being used to interrupt the generation of peak flows.

<u>Catchment A</u>



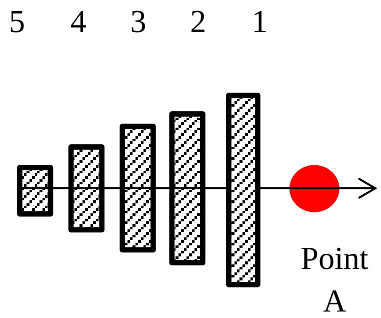
Ref	Area (ha)	Time of entry (mins)	Time of flow (mins)
1	3	1	5
2	1	1	10
3	1	1	15
4	1	1	20
5	1	1	25



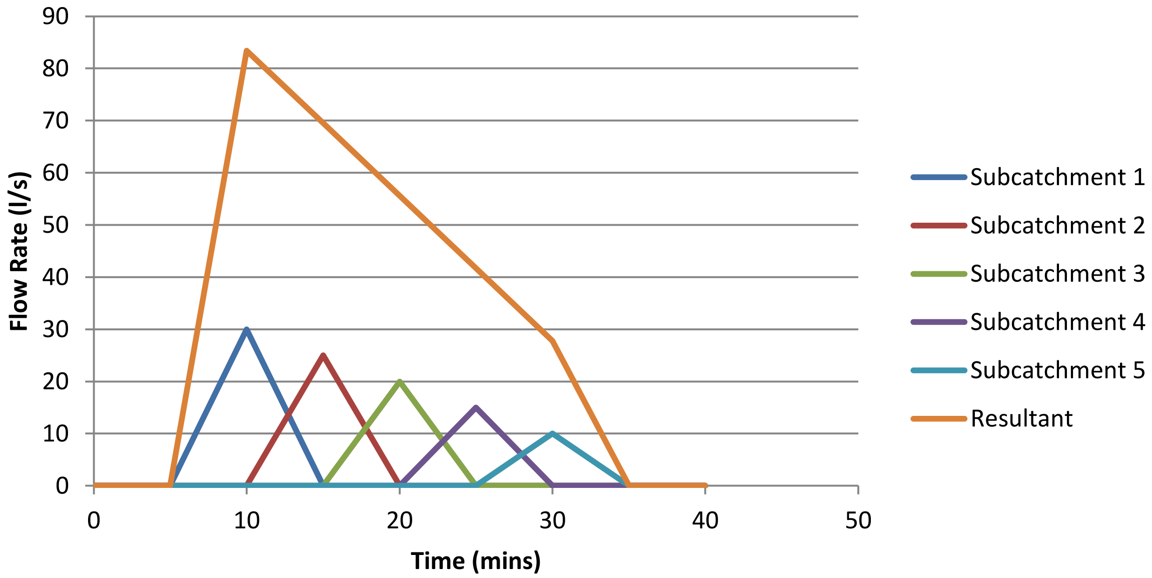
Disconnection Scenario	Peak Flow Rate (l/s)	Peak Flow Reduction (l/s)	Proportion Baseline (%)
Baseline	69.5	-	-
Indiscriminate	55.6	13.9	20
Random	66.2	3.3	4.7
Unit Hydrograph Informed	50.0	19.5	28

Figure 3-6: Generic catchment A.

Catchment B

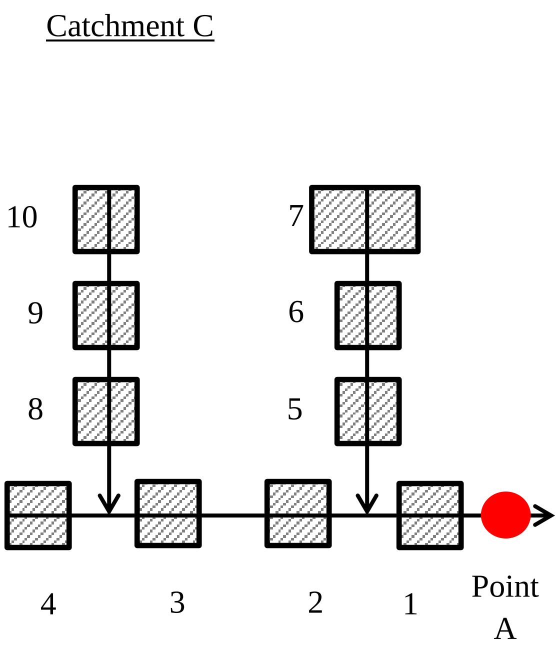


Ref	Area (ha)	Time of entry (mins)	Time of flow (mins)
1	3	1	5
2	2.5	1	10
3	2	1	15
4	1.5	1	20
5	1	1	25

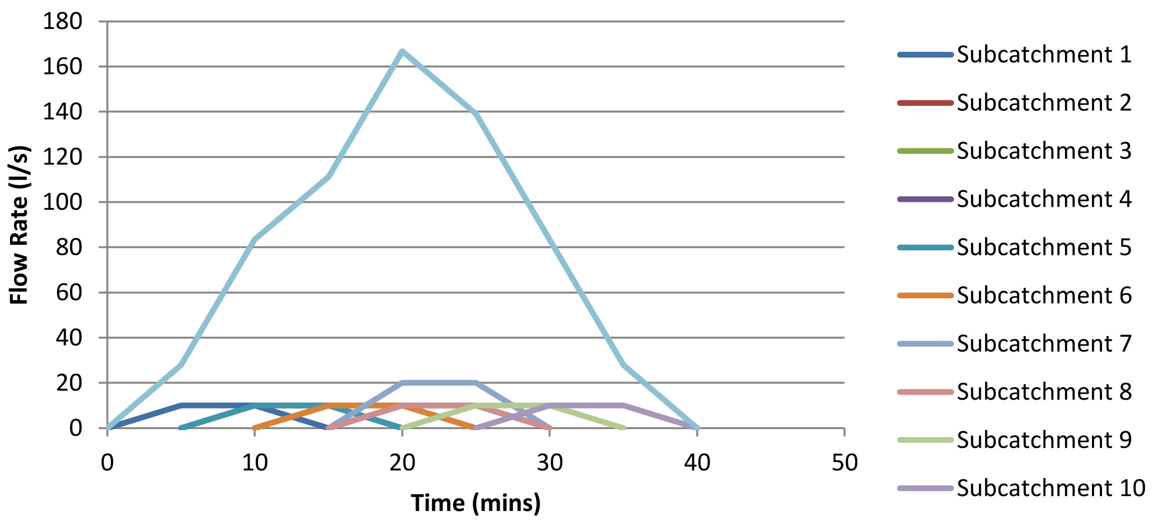


Disconnection Scenario	Peak Flow Rate (l/s)	Peak Flow Reduction (l/s)	Proportion Baseline (%)
Baseline	83.4	-	-
Indiscriminate	66.7	16.7	20
Random	78.8	4.6	5.5
Unit Hydrograph Informed	55.6	27.8	33.3

Figure 3-7: Generic catchment B.



Ref	Area (ha)	Time of entry (mins)	Time of flow (mins)
1	1	1	5
2	1	1	10
3	1	1	15
4	1	1	20
5	1	1	25
6	1	1	30
7	2	1	35
8	1	1	20
9	1	1	25
10	1	1	30



Disconnection Scenario	Peak Flow Rate (l/s)	Peak Flow Reduction (l/s)	Proportion Baseline (%)
Baseline	166.8	-	-
Indiscriminate	93.4	73.4	20
Random	144.3	22.5	13.5
Unit Hydrograph Informed	71.9	94.9	56.9

Figure 3-8: Generic catchment C.

Further examples of generic catchments are provided in Appendix 1. Stormwater disconnection targeted to interrupt peak flow generation is proven in all cases to be beneficial compared to existing methods of stormwater disconnection distribution. Specifically, the areal co-contribution method allows a distribution of stormwater disconnection within a catchment to be identified that is more efficient than that provided by approaches advocated currently in academic and best-practice literature. The term efficiency relates to the performance improvement obtained per area removed from the representation of the catchment. Removal of area represents the implementation of stormwater disconnection.

In all cases, the use of a random disconnection distribution, representing suitability-based approaches to stormwater disconnection, proved to be the least beneficial approach. Disaggregated unit hydrographs can be used to identify locations within catchments that are priority locations for stormwater disconnection. This logically implies that other locations within the catchment are lower-priority locations for stormwater disconnection; that is, disconnection in these locations is of lower efficiency for the improvement of the metric under consideration. Random disconnection may target stormwater disconnection within the priority, efficient locations. However, it is more likely (due to the larger area of non-priority locations compared to priority locations) to target stormwater disconnection in inefficient areas. In these generic catchments, the random disconnection approach targeted stormwater disconnection in low-priority locations for stormwater disconnection, producing small benefits to the performance metric.

Whereas the random approach may or may not target the stormwater disconnection in a high-priority location, the indiscriminate approach will certainly target stormwater disconnection in both high- and low-priority locations. The indiscriminate approach can provide some substantial reductions in peak flow rates. By guaranteeing that some disconnection will occur in high-priority locations, the indiscriminate approach may be considered a superior disconnection approach to the random approach, albeit that on some occasions the random approach will target disconnection in high-priority locations.

The disaggregated unit hydrograph approach guarantees that disconnection will be targeted within high-priority locations. Implementation of stormwater disconnection targeted by the unit hydrograph-informed approach can provide the greatest reduction in peak flow rates compared to any other approach to distributing stormwater disconnection for the same amount of disconnected area.

These trial catchments demonstrate that the use of a disaggregated unit hydrograph to inform stormwater disconnection provides a useful tool to achieving efficient disconnection distributions. It is appropriate to test the viability of this approach in a real-world catchment.

3.4 Application of the Areal Co-Contribution Method in The Urquhart Catchment

The model for Urquhart, a village in the Scottish Highlands, is an upstream component of a model representing the larger urban areas of Elgin and Lossiemouth. As Urquhart is an upstream component of this larger model, it may be evaluated in isolation. The geo-plan of the Urquhart model is provided in Figure 3-9. There are 9 subcatchments, labelled numerically, which drain towards a pumped section of sewer (disappearing from the Figure to the South) which transports the flow to treatment. A CSO spills to the east of the catchment. This study will examine how alternative disconnection distributions improve the performance of the CSO. The total area of the Urquhart catchment is 9.598 ha, comprising 12 links. This is comparable to the 12 ha catchment drained by five surface water pipes used by Viavattene and Ellis (2013) to test the SUDSLOC tool. The model representing Urquhart has been provided by Scottish Water has been built and verified using the WaPUG Code of Practice (WaPUG, 2002).

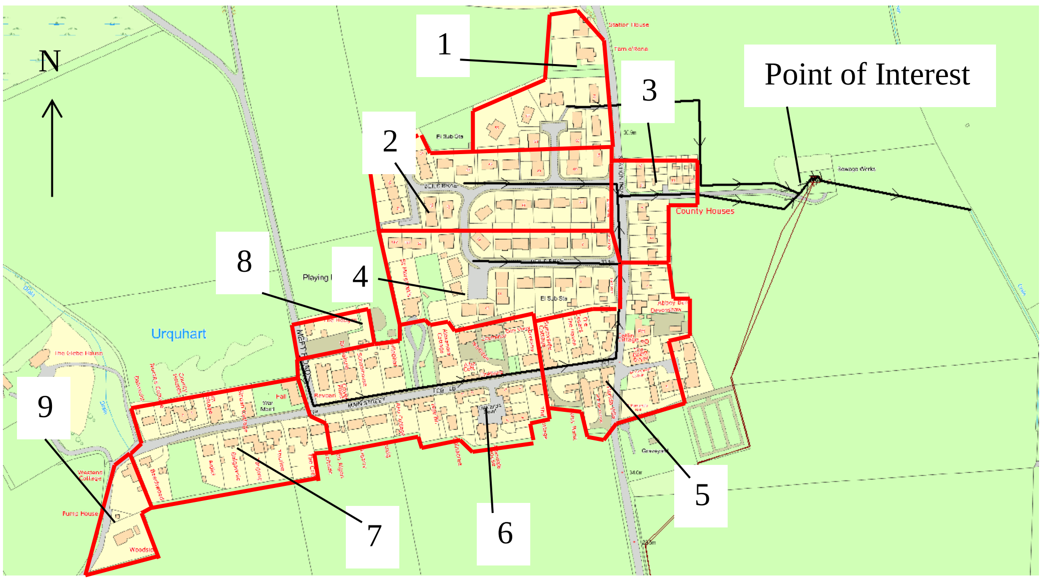


Figure 3-9: Map of Urquhart, showing subcatchments and point of interest.

3.4.1 Creating a Disaggregated Unit Hydrograph in InfoWorks CS

InfoWorks CS is a hydraulic modelling software tool that may be used to represent urban drainage systems, such that their performance under different conditions may be assessed and interventions to improve under-performance may be designed. InfoWorks CS is the standard software for hydraulic modelling of urban drainage systems in the UK water industry. The resultant unit hydrograph is easily obtained through InfoWorks CS simulation of the catchment model against a defined rainfall event. However, InfoWorks CS does not provide a mechanism for the automatic disaggregation of the resultant hydrograph. It was therefore necessary to develop a method to use the disaggregated unit hydrograph approach of stormwater disconnection in InfoWorks CS to enable the use of this method by UK urban drainage practitioners.

InfoWorks CS allows a user-generated "pollutograph" to be applied at any node within the model to represent a point-source pollutant entering the sewer system. This is achieved through construction of a profile of pollutant concentration (mg/l) against time being applied to a node in conjunction with a user-generated inflow (m^3/s) at the same node. The software allows for 19 types of pollutants to be applied to the system. However, in addition to the point-source pollutograph entering the system, InfoWorks CS assesses how diffuse pollution built up on the surface of each subcatchment enters and transports through the sewer system upon mobilisation by a rainfall event. There is overlap in the types of pollutant represented from these two different sources; for example the representation of BOD from both sources. It was not possible to identify a way to activate the representation of a pollutant from a point-source application at a node without activating the representation of diffuse pollution from the subcatchments.

However, included in the 19 types of pollutant are eight user-defined pollutants; these are "spare" to allow new or uncommon pollutants to be represented in the model as required. These eight pollutants are not represented in the diffuse pollution from each subcatchment, and as such each of the eight pollutants may be applied to the model with confidence that any trace of the pollutant identified within the network can be associated with a user-defined inflow of that pollutant. These eight spare pollutants may therefore by used as tracers.

A tracer is applied at a subcatchment, and is observed at the point of interest at some point later in time. An elegant use of the tracer mechanism in InfoWorks CS is to apply a tracer at a subcatchment and observe the rate at which it manifests at some point of interest; for example at a CSO or flooding location.

Figure 3-10 shows how a tracer applied at any subcatchment may manifest downstream at the point of interest. The profile of the tracer observed at the point of interest can be adjusted with the subcatchment at which it was applied to describe the unit hydrograph of that subcatchment. The area of each subcatchment may be easily observed in an InfoWorks CS catchment model through interrogation of the Subcatchment dataset. By repeating this exercise for each subcatchment in the catchment, the disaggregated unit hydrograph for the catchment may be produced.

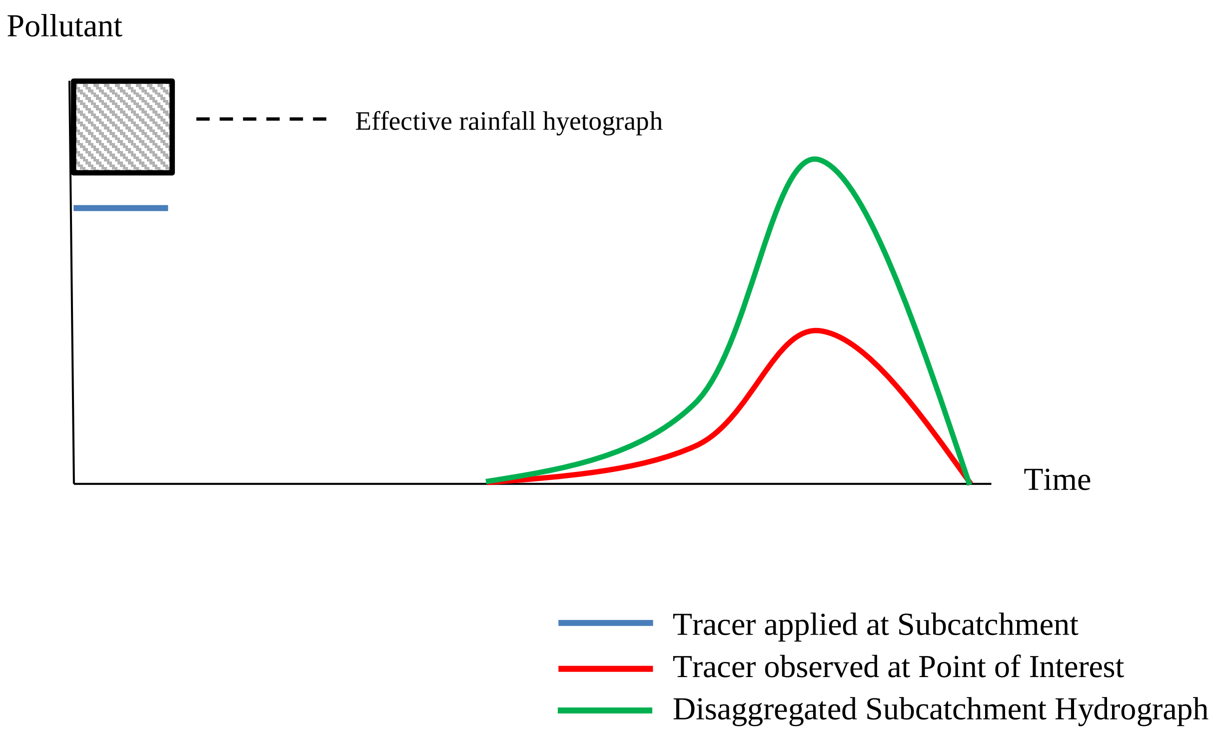


Figure 3-10: Generating a subcatchment hydrograph from a tracer and unit rainfall hyetograph.

3.4.2 Areal Co-Contribution Disconnection in Urquhart

A 10 minute rainfall event of constant intensity of 5 mm/hr was applied to the Urquhart model. Tracers were applied at each subcatchment at a constant rate of 100 mg/l, via application of an inflow of 1 l/min for 10 minutes. The tracer profiles observed downstream were provided by InfoWorks CS in units of kg/s.

The tracer profiles were converted from units of kg/s to m^3/s by the following steps:

1. The total observed downstream mass load of each tracer was inconsistent. It was known that 0.06 kg of tracer was applied at each subcatchment, however the observed total downstream mass load varied between 0.016 and 0.06 kg. It was subsequently identified that this “loss” of tracer load is because there was a flow of tracer within the model that was below the software’s detection level. This may be avoided by increasing the absolute volume of tracer load applied to a node, which decreases the proportion lost. To offset this phenomenon, the observed tracer profiles were adjusted such that the profile shapes were maintained, but the total tracer load was constant at 0.06 kg for each tracer. This detection level was triggered where the peak flow contribution was made, and as such the results remain valid.
2. Each ordinate of every tracer profile was multiplied by the area of the subcatchment at which it was applied. This gave units of m^2.kg/s.
3. Each ordinate was subsequently multiplied by the rainfall depth at each 1 minute timestep. For the 5mm/hr rainfall event this was 0.000083 m, resulting in units of m^3.kg/s
4. Each ordinate was divided by the load of tracer applied at each 1 minute timestep (0.006 kg). This provided the required units of m^3/s.

To verify that this method is suitable, the resultant unit hydrograph built from tracer observations was compared to the hydrograph provided by InfoWorks CS for the same location. Figure 3-11 presents the resultant and disaggregated unit hydrograph for the Urquhart catchment, and the hydrograph provided by InfoWorks CS.

The use of tracers to disaggregate the hydrograph from the Urquhart catchment can be considered successful due to the following points:

1. The relative contributions from each subcatchment can be clearly observed. It is apparent that subcatchments 5, 6, and 7 are the primary contributing subcatchments to the peak of the unit hydrograph. In particular, subcatchment 6 is both contemporaneous to the peak of the unit hydrograph and provides the greatest and most peaked flow response.
2. When the resultant unit hydrograph from the use of tracers is compared to the resultant hydrograph provided by InfoWorks CS, there is good correlation in the time (93.3%) and magnitude (90.2%) of the peak flow rate. However, it is clear that the

InfoWorks CS hydrograph begins earlier and ends later than the tracer hydrograph, and shows that a greater volume of flow is predicted to pass the observation point overall. The InfoWorks CS hydrograph ending later may be explained by the time of entry of each subcatchment delaying run-off manifesting in the drainage system, while the tracers are applied directly into the nodes.

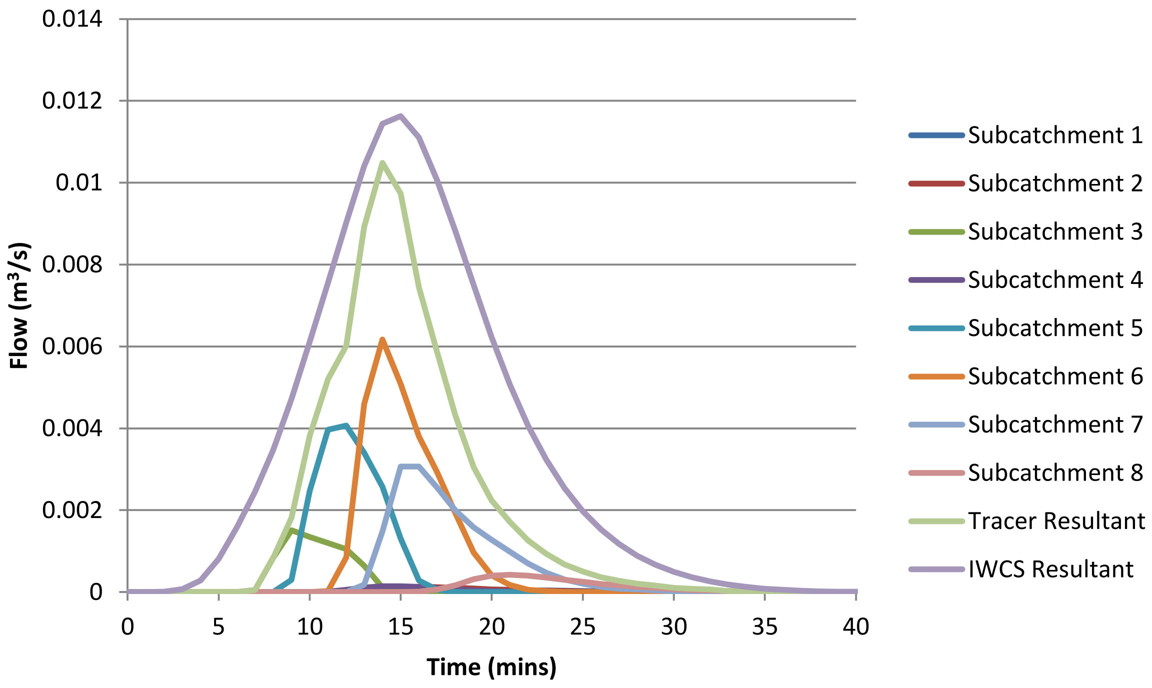


Figure 3-11: Tracer profiles at the downstream point of interest, adjusted to m^3/s.

3.4.3 Comparing Disconnection Approaches

The total impermeable area in the village of Urquhart is 1.9317 ha. Assuming an arbitrary disconnection budget of 20%, which may be seen as reasonable (e.g. Urich and Rauch, 2014), 0.39 ha is available for removal from the model. The removal of this amount of impermeable area from the Urquhart model was undertaken, with this amount of removal being similarly to the disconnection strategies used in Section 3.3.2:

1. Unit Hydrograph Informed; clearly, subcatchments 5 and 6 provide the greatest contribution to the unit hydrograph peak, and is therefore the priority locations for stormwater disconnection. Subcatchment 5 comprises 0.289 ha of road surface, 0.245 haof roof surface, and 0.794 ha of impermeable area Subcatchment 6 comprises 0.351

ha of road surface, 0.338 of roof surface, and 1.240 ha permeable area. The budget of 0.39 ha was split equally between the impermeable areas of subcatchments 5 and 6.

2. Indiscriminate; the impermeable area of each subcatchment in the Urquhart model was reduced by 20%.
3. SuDS Suitability; As identified in Section 2.6, and corroborated by (Kuller *et al.*, 2017), most SuDS distribution approaches are based on a perception of suitability. Whereas in Section 3.3.2, a random number generator was used to distribute stormwater disconnection within a generic catchment, using a real-world catchment, provides the opportunity to trial the application of a disconnection approach identified in the literature. In this instance, the preferential hierarchy created by Swan and Stovin (2002), was used (Figure 3-12). This hierarchy suggests that it is preferable to disconnect institutional roofs, car parks, residential roofs, then highways due to the relative ease with which retrofit SuDS schemes can be promoted, implemented, managed and monitored. In Urquhart, there are no institutional roofs or car parks, so a disconnection of 0.39 from residential roofs across the catchment was undertaken.

Decreasing order of preference	**Urban surface type**
	Institutional roofs
	Car parks
	Residential roofs
	Highways

Figure 3-12: Prioritising stormwater disconnection (after Swan and Stovin, 2002).

The baseline and modified representations of the Urquhart catchment were subject to a typical year rainfall event groups comprising roughly 150 individual rainfall events that are statistically representative of observed rainfall patterns observed in the UK 1961-1991. Typical year rainfall event groups are not time-series rainfall profiles for a year. Table 3-1 presents the results of this modelling exercise of alternative disconnection approaches.

Table 3-1: Disconnection approaches and the impact on CSO performance metrics.

Scenario	**Spill Count (no.)**	**Duration of Spills (min)**
Baseline	7	73

Indiscriminate	5	48.3
Suitability	4	42.1
Unit Hydrograph	4	38.2

As expected, the unit hydrograph informed approach provided the most effective disconnection distribution. The suitability approach provided marginally superior results than the indiscriminate approach because of the increased density of residential roofs in the high-priority locations of the catchment. This suggests that the use of catchment unit hydrographs to inform disconnection distributions can be effective in identifying efficient areas of the catchment in which to undertake stormwater disconnection in both simple and complex representations of catchments. To verify this finding, the method was applied in a second real-world catchment; Winterton.

3.5 Application in Winterton

Winterton is a town in North Lincolnshire, England. Similarly to the Urquhart catchment, the Winterton catchment is part of an urban drainage network that serves three distinct urban areas. Foul flow from the villages of Roxby and Appleby is pumped north and joins flow from Winterton to the East of Winterton, where additional pumping directs the total flow from the three conurbations to treatment on the southern bank of the Humber estuary. Stormwater run-off generated within Roxby and Appleby is lost to soakaways and local water courses. Within Winterton, there is a clear split in the drainage system; the historic east of the town is primarily served by a combined network, while the post-war residential development to the west of the town is served by separate storm- and foul-sewers. In the north of Winterton, a hydraulically-distinct area drains via a mix of combined and separate sewer systems to the east, which connects to the rising main directing flows from the rest of the catchment to treatment. There is a CSO along the connecting length of sewer, which spills to a watercourse. Figure 3-13 shows this hydraulically-distinct area.

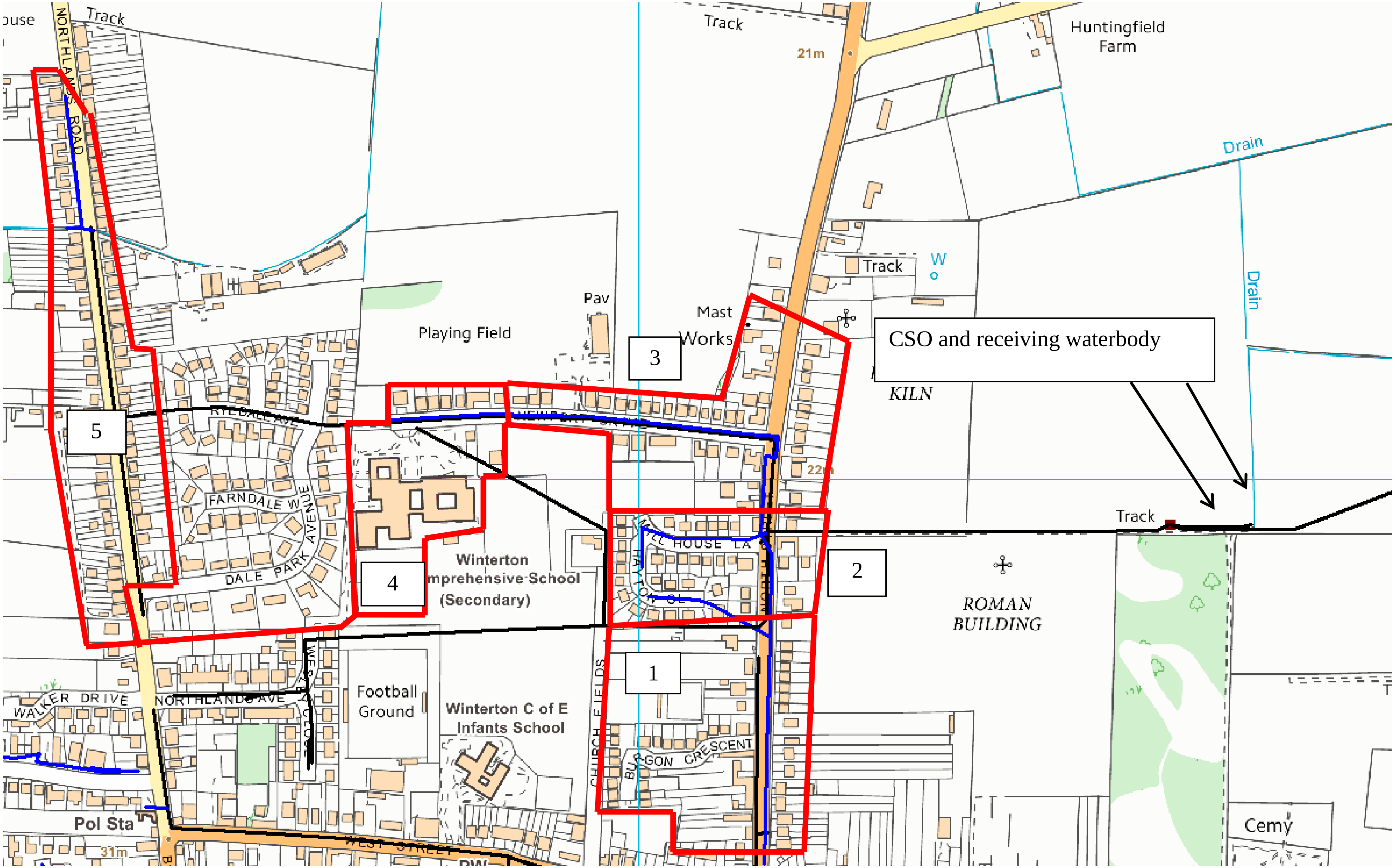


Figure 3-13: The hydraulically distinct Winterton catchment, showing subcatchments and CSO location.

The InfoWorks CS model representing this hydraulically distinct section of Winterton describes a catchment of total area 25.4 ha and a represented population of 742. This catchment is large compared to Urquhart, and as such the InfoWorks CS model is split into more subcatchments; 46 subcatchments in total, 35 that drain to combined sewer and 11 that drain to separate systems. Because of the large number of subcatchments in the sizeable hydraulically-distinct Winterton catchment, application of the areal co-contribution tracer method was considered impractical; 45 subcatchments with 8 pollutant slots would have increased modelling time, especially under a typical year rainfall event group.

The identification of efficient areas within the Winterton catchment was undertaken by grouping distinct groups of InfoWorks CS subcatchments together. Grouping was undertaken by examining the structure of the urban drainage network and the land-use. These aggregated areas were disconnected from the Winterton catchment, and the result of their disconnection on the performance of the CSO was observed. Figure 3-14 shows the aggregated areas. A thirty year return period, thirty minute duration storm event was used; this is not the storm profile that a CSO would be assessed against but it was used to understand the relative effect on stormwater disconnection achieved by each area.

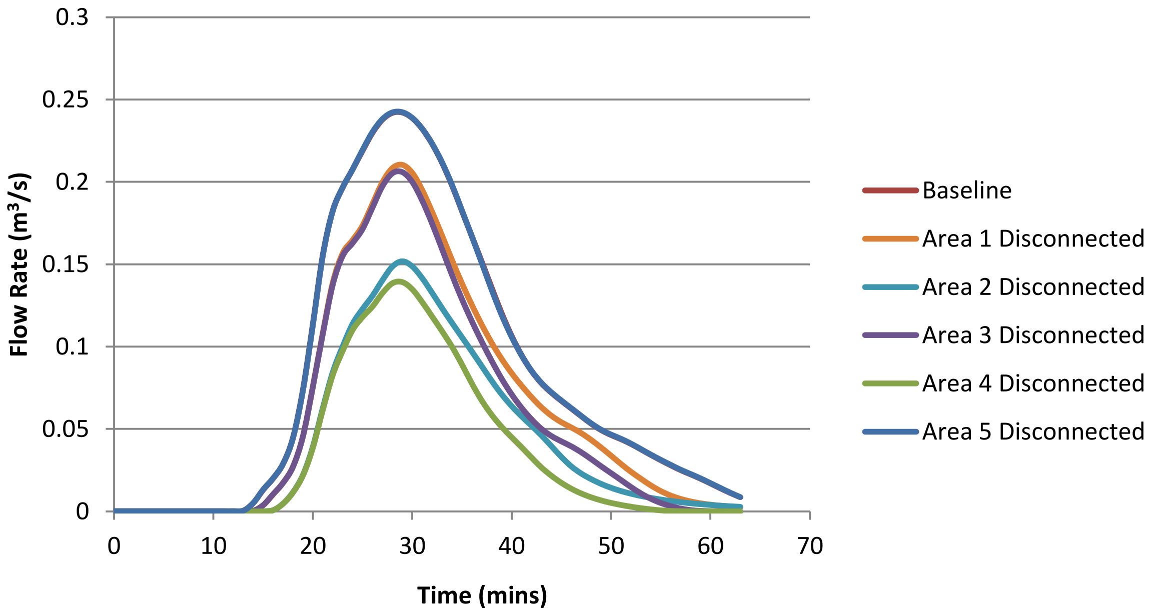


Figure 3-14: The response of the Winterton CSO to alternative disconnection distributions under an M30:30 storm.

To verify that the areas suggested by Figure 3-14 as priority areas for disconnection – areas 2 and 4 – were efficient as well as effective, the peak flow rate reduction was normalised against the area removed from the model (Table 3-2). It is clear that subcatchment 4 presents the highest priority, most efficient location for stormwater disconnection.

Table 3-2: Normalised disconnected efficiency in Winterton.

Area	Peak Flow Rate Reduction (m^3/s)	Disconnected Area (ha)	Reduction per Disconnected Area ($m^3/s/ha$) ($x10^3$)
1	0.032	3.996	8.01
2	0.090	3.864	23.29
3	0.036	1.038	34.68
4	0.103	2.675	38.50
5	0	2.538	-

To confirm that area 4 remains the highest priority area for stormwater disconnection when considering traditional CSO metrics, the areas were sequentially disconnected from the Winterton model subject to a storm profile from the typical year rainfall event group. This storm profile (Figure 3-15) was observed to produce the greatest CSO metrics. The baseline and area disconnection scenarios are presented in Table 3-3.

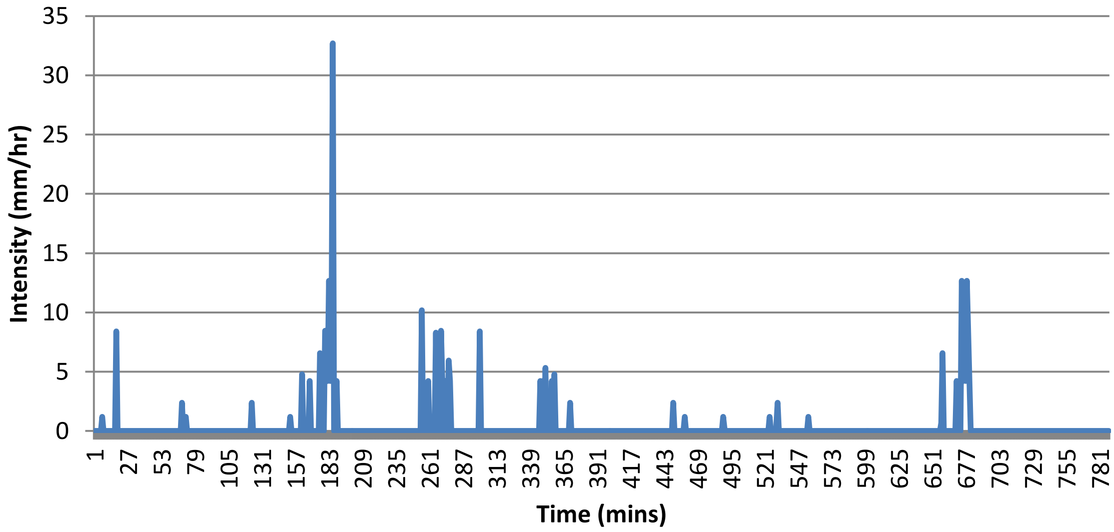


Figure 3-15: Time-series of impactful storm profile.

Table 3-3: Total volume spilled from Winterton CSO under Disconnection Scenarios under impactful storm profile

Disconnection Scenario	**Total Volume (m^3)**
Baseline	82.4
1	46.5
2	66.7
3	46.0
4	14.0
5	82.4

This process has identified that area 4 is the preferential location in which to undertake stormwater disconnection.

3.6 Uncertainty and Sensitivity

The work presented in this Chapter demonstrates that some locations within an urban drainage catchment may provide more prefereable locations at which to undertake stormwater disconnection than others. However, as with any decision based on modelling, the results are likely to be sensitive to sources of uncertainty; differences between the physical reality of the catchment in question and the computational representation.

For confidence in the model outputs, Deletic *et al.* (2012) recommended that the model development and calibration process should be strongly related to the model application. Calibration of the Urquhart model has been conducted in accordance with guidance from WaPUG (2002), which is appropriate for the use of this model to predict stormwater flows within the urban drainage network.

Based on a global sensitivity analysis, Sriwastava *et al.* (2016) identified that the three most important model input parameters within an InfoWorks CS model for prediction of CSO spills are runoff coefficient, weir crest and conduit roughness. However, difference in run-off characteristics and conduit roughness between the representation in the model and will affect the validity of the modelling results and potentially lead to stormwater disconnection being undertaken in an inefficient area.

The areal co-contribution method is based on the concept of the unit hydrograph. Typically, a unit hydrograph for a catchment is obtained under a single rainfall event, with the assumption that the behaviour of the catchment under alternative rainfall events can be linearly related to the behaviour observed under that single rainfall event. Unit hydrographs were developed for river basin catchments. In the context of urban drainage networks, alternative rainfall events can be expected to change the response of catchments in non-linear ways. Certain types of rainfall event may trigger the manifestation of such phenomena as backlogging effects within the drainage network, pipe flow velocities increasing as the pipe becomes full, and flow lost from the network at CSO or through surcharge flooding. These phenomena will change the flow regime within the urban drainage network. The assumption that there is an inherent linear transferability from any given unit hydrograph to describe the behaviour of a catchment under an alternative rainfall event may be flawed in the urban drainage context.

In Section 3.4 the Urquhart catchment was subject to a ten minute duration, 5 mm/hr intensity rainfall event. Under these conditions, subcatchments 5, 7, and particularly 6 were identified as being priority locations for stormwater disconnection. The Urquhart catchment is here subjected to a one hour duration, 30 mm/hr intensity event; a rainfall event of greater duration and intensity. The tracer mechanism was again used to generate some inferred unit hydrographs, presented in Figure 3-13. It is clear that the subcatchments identified in Chapter 3 as priority locations for stormwater disconnection, namely subcatchments 5, 6 and 7, retain this status. This is observed by noting that the peak flow rates of these disaggregated hydrographs for these subcatchments are contemporaneous with the peak flow rate of the resultant hydrograph.

Table 3-4 presents the peak flow rate from these three priority subcatchments as a proportion of the peak resultant flow rate under the two rainfall events described; 10 minute, 5 mm/hr and 1 hour, 30 mm/hr. It is clear that in both cases the peak flow rate from Subcatchment 6 provides the greatest contribution to the resultant peak flow. As such, it may be noted that Subcatchment 6 remains the overall priority location for stormwater disconnection. Furthermore, the relative priority of each subcatchment (5 to 7) is maintained.

The results of this investigation suggest that increasing the size of rainfall event, in order to hydraulically overload the urban drainage network, does not alter the results provided by the areal-contribution method; the same locations are indicated as being priority locations for stormwater disconnection.

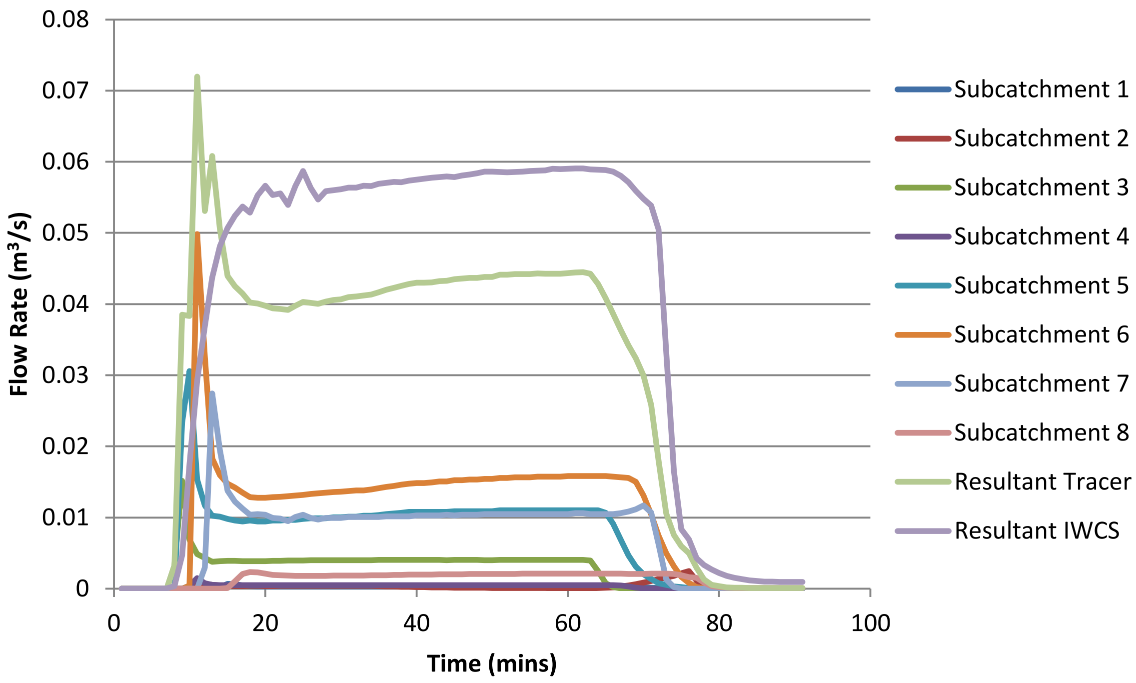


Figure 3-16: Resultant, disaggregated and IWCS hydrographs for the Urquhart catchment for a one hour, 30 mm/hr design storm.

Table 3-4: Peak flow rates per Urquhart subcatchment under medium and large rainfall events.

Subcatchment Number	% Tracer Resultant Peak Flow Rate	
	10 minute, 5 mm/hr	**One hour, 30 mm/hr**
5	34.9	42.5
6	58.8	69.2
7	29.2	38.1

3.7 Chapter Summary

This chapter has presented the development of the areal co-contribution method to stormwater disconnection distribution. The areal co-contribution method can provide certainty that stormwater disconnection is undertaken in priority locations within a catchment. The priority of a location is determined by the effect that a unit area of disconnected impermeable surface has on the peak flow rate at some determined point in the network. This metric is used as a proxy for cost and/or difficulty of undertaking stormwater disconnection. Application of the areal co-contribution method to identify stormwater disconnection locations was proven in three generic catchments to generate disconnection distributions that are more efficient than existing distribution methods, where efficiency is the reduction of peak flow rate per unit area disconnected.

The application of the areal co-contribution method to an InfoWorks CS representation of a real-world case-study catchment; Urquhart in Moray, Scotland. InfoWorks CS is a common hydraulic modelling software in the UK water industry. An innovative method was developed to identify times of concentration from subcatchments within InfoWorks CS models; using the native pollutant mechanism as a tracer. Using this tracer allowed individual subcatchment times of concentration to be identified, and therefore the areal co-contribution method was able to identify priority disconnection locations. Again, the distribution of stormwater disconnection suggested by the areal co-contribution method was observed to result in more efficient distributions than suggested by existing methods. A simplified approach was used to identify efficient locations for stormwater disconnection in Winterton, North Lincolnshire.

4 Developing Transient Scenarios for the Assessment of Flexibility

4.1 Introduction

Adaptive management of infrastructure allows modification of infrastructure as one learns more about how the future is unfolding. The fundamental premise of adaptive management is flexibility (Colombo & Byer, 2012). It would be a useful contribution to knowledge to compare the flexibility of retrofit SuDS interventions designed according to the two approaches in the UK; the academic preference for source-control, and the industrial preference for regional-control. SWITCH presented a framework for the detailed measurement of flexibility. This framework requires transient scenarios to be developed to enable its application to completely assess flexibility, as transient scenarios are the only way to ensure the interplay between scenario narratives and adaptations is explored (Kwakkel *et al.*, 2015). This chapter presents the modification of Casal-Campos' (2016) scenarios to transient scenarios by defining how pressures will vary in scenarios and using using perspective theory to establish each society's likely preferred stormwater infrastructure. Representations of stormwater infrastructure are presented, and values are assigned to the costs and benefits pertinent to retrofit SuDS assessment.

4.2 Relative Forecasting of Pressures in Scenarios

Casal-Campos (2016) used High, High-Medium, Medium, Low-Medium, and Low indicators to describe the relative magnitude of pressures in each scenario. These indicators were then used to identify appropriate values from the literature that could be used to represent the pressure. This range of indicators is considered sufficient because the exactness of the values is not crucial; a consistent relationship between inter-scenario narratives to define contrasting, plausible futures is the objective (Casal-Campos, 2016), and the method provides a logical approach to link the forecast of scenario aspects to the scenario descriptions. This approach to forecasting is similar to the use of "driver impact scores", which numerically indicate the relative likely manifestation of future pressures between scenarios, used in the preparation of the Foresight reports (Evans *et al.*, 2004), and can therefore be described as valid.

Casal-Campos (2016) did not examine how climate change could differ between scenarios. Climate change is related to greenhouse gas emissions, which are the result of dynamic socio-economic interactions (Nakicenovic *et al.*, 2000), and the scenarios described by Casal-Campos depict possible future socio-economic states. It is therefore valid that climate change could occur differently in the different Casal-Campos scenarios, and that the depictions of the scenarios can be used to inform the forecast of the different manifestation scenarios. The Lifestyles scenario explicitly describes a society that places great emphasis on environmental concerns, and has actively pursued economic realignment to reduce the society's environmental impact. The Lifestyles society can therefore be described as having a Low climate change indicator. The Innovation scenario also places a relatively high emphasis on environmental concerns, and this is supported by a competent and powerful governance structure. However, the society is not willing to compromise the standard of living to achieve environmental outcomes, and so there is likely to be Low-Medium climate change. On the contrary, the Markets society is highly consumptive, with little regard for environmental or resource-efficiency, and the governance is centred on short-term economic growth. As such, this society is affected by High climate change. The Austerity scenario is similar to the Markets society in that it is geared towards economic concerns at the expense of environmental concerns. However, the lower economic output in the Austerity scenario compared to Markets means that there is likely to be a lower level of climate change, and can therefore be attributed a High-Medium indicator.

Casal-Campos (2016) commented that the level of urban creep occurring in a scenario may be related to two scenario factors; the level of regulation limiting the uncontrolled resurfacing of impermeable areas and public attitudes towards urban water management. As noted in Section 4.1, urban creep can be correlated with affluence (Allitt *et al*., 2009). The society within the Markets scenario is highly motivated by personal financial gain and is highly materialistic, with little regulation. Urban creep is therefore High in the Markets scenario. Actions within the Austerity scenario are also driven by economic demands; however the motivation is the avoidance of poverty rather than the acquisition of wealth. As there is such low affluence in the Austerity scenario, and there is likely to be a Low level of urban creep. Within the Lifestyles scenario, there is great emphasis on environmental concerns and therefore some key motivators for urban creep, such as expanding families or increased car parking facilities, are likely to be reduced compared to the present day. Furthermore, there is some regulation of activities which cause environmental degradation by both the governance and the individual, resulting in a Low urban creep indicator. The Innovation scenario has an understanding of environmental concerns, but these are not of greater importance than the society's quality of life, and the responsibility for sustainability lies with institutions rather than the individual. There is therefore likely to be a Medium level of urban creep.

The method proposed to evaluate urban area expansion is to relate population growth and urban area expansion. Relating population growth to urban area expansion has been used by previous studies examining possible future impacts on urban drainage networks e.g. Kleidorfer *et al*. (2009). Discussion of how these population trends are likely to affect the level of urbanisation under each scenario, and the forecast of infrastructure used to manage stormwater run-off from new developments, which will affect the impact of urbanisation on the urban drainage system, will be made in Section 4.4. In Casal-Campos (2016), indicators of population growth were assigned to each scenario based on the socio-economic depictions of the future. The definition of population growth used by Casal-Campos is analogous to that used in this thesis, and there has been no change in the scenario narratives. It is therefore valid to incorporate the use of population growth indicators from Casal-Campos to this thesis. Specifically, the population growth is likely to be High in Markets, High-Medium in Innovation, Low in Lifestyles and Low-Medium in Austerity. This depiction of population growth is in agreement with the current understanding that population and economic growth occurs in tandem (Berry, 2014).

Maintenance is important to ensure the performance of urban drainage systems. The maintenance of assets is distributed over time, and such incremental, piecemeal rehabilitation has the effect of perpetuating the same type of centralised infrastructure into the future (Marlow *et al.*, 2013). It is therefore unlikely that in any scenario there will be a complete transition away from centralised infrastructure. However, the capacity and willingness to undertake maintenance of urban drainage systems will be related to socio-economic changes in the future, and therefore it is likely that alternative socio-economic futures will result in different levels of infrastructure deterioration. For example, in the Innovation scenario, which is characterised by a high level of technological innovation and strong institutions, there is likely to be a high level of maintenance, and therefore a Low level of infrastructure deterioration. In contrast, the Austerity scenario is characterised by low technological development and the provision of public services is unreliable, and therefore there is likely to be low levels of maintenance, and a consequentially High level of infrastructure deterioration. In the Lifestyles scenario, there is a low level of technological capability to undertake rehabilitation activities, but there is a strong emphasis on ensuring that urban drainage networks impacts on the environment are minimised. Therefore there is a medium level of maintenance and a Medium level of infrastructure degradation. The Markets scenario, utilities companies have the capability to undertake maintenance, but the motivation of these for-profit institutions is to keep costs low. Therefore there is likely to be a medium level of maintenance and a Medium level of infrastructure degradation.

Casal-Campos (2016) did not examine how the subjective acceptable performance of urban drainage networks could change in the future. The expectation that a society places on urban drainage networks is related to the socio-economic outlook of that society; it can therefore be defined according to the socio-economic narratives that depict societies in Casal-Campos' scenarios. The Markets scenario requires urban drainage networks to eliminate the risk of flooding in order to maintain high economic growth, so this pressure is High. The Innovation scenario expects that urban drainage networks are managed well to reduce disruption to society and to ensure the environment is not degraded; there is a Medium-High extent of this pressure. In the Austerity scenario, the society is geared towards economic concerns, but acknowledges that with little resource available, realistic expectations of urban drainage performance must be maintained. In the Lifestyles scenario, economic gain is a secondary issue, and people have learned to live with the risk of flooding as a small price to pay for the greater good; therefore the expectations on the urban drainage system are reduced compared to today.

The extent of manifestation of these pressures in each scenario is presented in Table 4-1.

Table 4-1: Extent of manifestation of pressures in each scenario

Future Pressure	Markets	Innovation	Austerity	Lifestyles
Climate change	H	L-M	H-M	L
Urban creep	H	M	L	L
Population growth	H	H-M	L	L-M
Asset deterioration	M	L	H	M
Acceptable performance	H	H-M	L-M	L

(H: High, H-M: High-Medium, M: Medium, L-M: Low-Medium, L: Low)

4.3 Quantification and Representation of Future Pressures

Arnbjerg-Nielsen (2008) pioneered the use of proportional increases in rainfall intensity to represent the effects of climate change. Dong *et al*. (2017) used this work to justify increasing rainfall intensity by between 5 and 20%. DEFRA (2006) suggested using a 20% increase in rainfall intensity to represent rainfall in the 2050s. Based on these sources, climate change will be represented through an increase in rainfall intensity of range 5-20%. The Markets scenario will experience a High level of climate change; this will be represented by a 20% increase to rainfall precipitation. The Austerity and Innovation scenarios will experience High-Medium and Low-Medium levels of climate change; these will be represented by 15% and 10% increases in rainfall intensity. The Lifestyles scenario will experience a Low level of climate change; this will be represented by a 5% increase in rainfall intensity.

OFWAT (2011b) used a range of 5-15% increases to impermeable area to represent differing extents of urban creep, and it is proposed to use this range to inform the extent of urban creep within each scenario. The Markets scenario will experience a High manifestation of urban creep; this will be a 15% increase of impermeable area. The Innovation scenario will experience a Medium manifestation of urban creep; this will be a 10% increase of impermeable area. The Austerity and Lifestyles scenarios will experience a Low manifestation of urban creep; this will be represented by a 5% increase of impermeable area. Urban creep can be represented by the transfer of permeable to impermeable area within a hydraulic model.

As discussed in Section 4.2, the population growth methodology used within Casal-Campos (2016) will be replicated here to inform urban area expansion. The Markets scenario will experience 10% increase, the Innovation scenario will experience 8% increase, the Austerity scenario will experience 4% increase, and the Lifestyles scenario will experience 2% increase. This range of increases was informed by Environment Agency (2010). Further discussion of the representation of this pressure is provided in Section 4.4 as it is related to each society's preferential stormwater infrastructure.

Casal-Campos *et al*. (2015) represented the deterioration of urban drainage infrastructure as a reduction in conveyance capacity due to sediment, after Ackers *et al*. (2016). Based on these works, a range of 0-13% is appropriate. The Austerity scenario is expected to experience a High level of asset deterioration. Based on prior work by Casal-Camos and Ackers *et al*., this is represented by 13% reduction in pipe area. Markets and Lifestyle scenarios are expected to experience a Medium level of asset deterioration, which will be represented by a 5% reduction in pipe area. Due to technological innovation, the Innovation society is likely to reduce sewer sedimentation, which will be represented by no sedimentation within the network. This pressure will be represented the reduction of pipe area.

Expected performance is related to the socio-economic depiction of the society in each scenario. No prior examples of this pressure have been identified in the literature; it is proposed to use multipliers of the expected level of service between 1.2 and 0.9 to represent how expectations may vary. A multiplier greater than 1 indicates that the expected performance of the network has increased; a multiplier less than 1 indicates that the expected performance of the network has decreased. For example, for a CSO with a performance standard in the present of 50 m^3/year spilt to a water body, an expected performance multiplier of 1.2 would translate this to (50/1.2 =) 41.7 m^3/year. The manifestation of the expected performance pressure is High in the Markets scenario; this will be allocated a multiplier of 1.2. The Innovation and Austerity scenarios are likely to experience High-Medium and Low-Medium manifestations of this pressure respectively; therefore multipliers of 1.1 and 1 will be used. The Lifestyles scenario will experience a Low manifestation of this pressure; therefore a multiplier of 0.9 will be used.

These pressures are being represented as manifesting linearly through time. This is acceptable because the detailed measurement of flexibility testing framework developed by SWITCH uses one future epoch. These details are summarised in Table 4-2.

Table 4-2: Quantified extent of manifestation of pressures in each scenario.

Future Pressure	Representation	Markets	Innovation	Austerity	Lifestyles
Climate change	Precipitation intensity increase (%)	20	10	15	5
Urban creep	Permeable to impermeable (% existing impermeable area)	15	10	5	5
Population growth	*Discussed in Section 4.4*	10	8	4	2
Asset deterioration	Pipe area reduction (%)	5	0	15	5
Acceptable performance	Expected performance multiplier	1.2	1.1	1	0.9

4.4 Forecast of Appropriate Interventions

It is necessary to forecast which types of stormwater management infrastructure would be preferred in each scenario to in order to appropriately represent the management of stormwater run-off from new developments and the augmentation of the existing urban drainage system. This interaction between society and infrastructure is a key part of complete transient scenario analysis. One mechanism to identify the preference of water infrastructure within a society is to use perspective theory (e.g. Offermans *et al*., 2011), in which decision making is described as being influenced by four dominant cultural biases, or "perspectives"; the egalitarian, hierarchist, individualist and fatalist perspectives. Despite the perspectives describing a broad range of attitudes, it is typical for a dominant perspective to emerge, reflecting the beliefs of the majority of people in a given group or society, and changes in dominant perspectives can occur gradually over time or in the aftermath of surprises that show that reality is different from the perspective holder's expectation about reality (Offermans *et al*., 2011). Scenarios depict a range of plausible future societies, and the perspective theory is used to rationalise decision making in different groups, therefore linking scenarios and perspective theory is valid.

Offermans *et al*., (2011) and Valkering *et al*., (2008) inferred infrastructure preference under three perspectives for fluvial flood risk management in the Netherlands qualitatively. Casal-

Campos (2016) presented a forecasting approach, used to quantify future pressures within scenarios, which linked both scenario narratives and magnitude of future pressures, independently, to a series of “key scenario factors”. Comparing the relationship between the narratives and the scenario factors to the relationship between the future pressures and the scenario factors enabled the logical forecast of future pressures within the narratives. It is proposed to use a similar method to forecast preferences for stormwater management measures in each perspective, and therefore each scenario. This requires a series of “infrastructure factors” to be identified.

The method of identifying the types of infrastructure that would be favoured in each scenario is therefore a two-stage process:

1. Describe the likely dominant perspective in each scenario;
2. Use the traits of the dominant perspectives to identify appropriate infrastructure.

Table 4-3 shows how there is a clear match between the perspective traits described both in cultural theory literature and studies using perspective theory in the context of water management, and the scenarios presented by Casal-Campos. From this matching of traits, it is appropriate to use the Egalitarian perspective to understand the views of the Lifestyles society, the Individualist perspective to understand the views of the Markets society, the Hierarchist perspective to understand the views of the Innovation perspective, and the Fatalist perspective to understand the views of the Austerity scenario.

Criteria pertinent to the assessment of retrofit SuDS were identified in Section 2.6, and are here used as infrastructure factors, with the following modifications; as this is an assessment of SuDS components, rather than their resultant effect on an urban drainage network, the ability of each SuDS to manage stormwater quantity and quality has been assessed, and the assessment of carbon has been excluded as this is primarily included in the pertinent criteria framework so that reduction in treatment and pumping carbon is assessed.

Table 4-3: Comparison of perspectives and scenarios.

Perspective	Scenario
Egalitarian: advocate for the abandonment of environmentally harmful activities, emphasis on natural and ecological recovery, community based decision making	**Lifestyles:** absolute prioritisation of the quality of the environment, high prioritisation of environmental and social objectives, emphasis on de-centralised decision making
Individualist: highly materialistic and motivated by financial acquisition, low regard for environmental concerns	**Markets:** materialist and consumerist society highly motivated by personal financial gain, little emphasis on resource efficiency, nor environmental, amenity or biodiversity
Hierarchist: favours a balance between quality of life and environmental and societal concerns, centralised governmental agencies are responsible for managing water	**Innovation:** although people are not willing to compromise their quality of life, the institutions attempt to achieve sustainability empowered by strong policy and legislation
Fatalist: minimisation of the expenditure of resources due to lack of rewards, little entrepreneurial vigour	**Austerity:** under-investment in infrastructure, little capital available, little technological innovation

The relative performance of each infrastructure, except conventional solutions, for each factor criteria is taken from The SuDS Manual (Woods-Ballard *et al.*, 2007; Woods Ballard *et al.*, 2015), which uses a qualitative scale from None-Low-Medium-High or Poor-Medium-Good-High. This is justified because the exactness of the values is not crucial; a consistent relationship between inter-scenario narratives to define contrasting, plausible futures is the objective (Casal-Campos, 2016). The centralised or decentralised factor is included because perspective theory incorporates a preference for community or institutional action. These factors are assigned based on the typical location of each infrastructure within the SuDS Management Train.

For the Financial Cost criteria, the infrastructure options are ranked Low-Medium-High, where Low indicates that the infrastructure requires little financial expense to construct (capital) or operate. The financial cost of maintenance is inferred from the "maintenance burden" indicator in The SuDS Manual.

The criteria Amenity, Bio-diversity and Contribution to Water Quality are used to indicate the environmental concerns of the scenario, and the contribution of the infrastructure options to providing environmental benefits. The Contribution to Water Quality criteria indicates the extent to which the infrastructure provides treatment to stormwater run-off. Conventional solutions are judged to have a low contribution to water quality because they create the risk of CSO spills. These criteria are ranked Poor-Medium-Good-High, where High indicates a high degree of amenity, bio-diversity or water quality benefit is provided. Bio-diversity is taken from the "ecological potential" value in The SuDS Manual.

The Land Sterilisation criterion is used to represent the surface area required by each infrastructure option that subsequently cannot be used for other purposes, ranked None-Low-Medium-High where None indicates that no land sterilisation is suffered through the use of the infrastructure. Soakaways and permeable pavements are judged to have a low land sterilisation because their surface area can still be used. Detention and infiltration basins can be used outside of storm events for recreation purposes. This benefit is considered within the Amenity criteria, and so to avoid double-counting, basins are judged sterilise the land which they occupy; this represents that basins prevent development on the land they occupy.

The Contribution to Water Quantity criteria represent the extent to which the infrastructure reduces the rate (Peak Flow Reduction) or volume (Volumetric Control) of stormwater run-off flowing to the combined urban drainage system, assessed Poor-Medium-Good-High where High indicates the infrastructure reduces the rate or volume of stormwater significantly. Table 4-4 presents the results of the review of stormwater management measures for each criterion.

Table 4-4: A qualitative assessment of infrastructure factors (after Woods-Ballard *et al*., 2007).

	Centralised/ Decentralised	**Financial Cost**		**Amenity**	**Bio-diversity**	**Contribution to Water Quality**	**Land Sterilisation**	**Contribution to Water Quantity**	
		Capital	**Operational**					**Peak Flow Reduction**	**Volumetric Control**
Green Roofs	De	Low-High	Medium	Good	Good	Good	None	Medium	Medium
Soakaways	De	Low	Low	Poor	Poor	Good	Low	Good	Good
Rainwater Harvesting	De	High	Medium	Poor	Poor	Poor	None	High	High
Filter Strips	-	Low	Low	Medium	Medium	Medium	High	Poor	Poor
Infiltration Trench	De	Low	Medium	Low	Low	High	Low	Medium	High
Filtration Trench	De	Low-Medium	Medium	Low	Low	High	Low	Medium	Low
Swales	-	Low	Medium	Medium	Medium	Good	High	Medium	Medium
Tree Planting	-	High	Medium	Good	Good	Good	Medium	Medium	Medium
Bioretention	De	Low	Medium	Good	Medium	Good	High	Medium	Medium-High
Permeable Pavement	De	Medium	Low	Poor	Poor	Good	Low	Good	Good
Geocellular Storage	Cent	Low	Low	Poor	Poor	Poor	Low	Poor	Poor-Good
Infiltration Basins	Cent	Low	Low	Good	Good	Good	High	Average	Good
Detention Basins	Cent	Low	Low	Good	Medium	Medium	Medium	Good	Poor
Ponds	Cent	Med-High	Medium	Good	Good	Good	High	Good	Poor
Conventional Solns	Cent	High	High	Poor	Poor	Low	Low	Good	Poor

The egalitarian perspective places high importance on environmental benefits, so the favoured infrastructure will contribute to amenity, bio-diversity and water quality, and there is a greater emphasis on this than the contribution of the infrastructure to water management. Due to the importance of this aspect of infrastructure, the egalitarian would accept the sterilisation of land to deliver sustainable water management. There is an emphasis on community action, which is likely to result in the selection of decentralised infrastructure. This means that the egalitarian is likely to prefer bioretention systems.

Regarding new development, the Lifestyles society does not expect to occupy newly built residences, preferring to renovate where possible. The urban area expansion is therefore likely to be related to industry or leisure. The high environmental concerns exhibited by this society meant new developments are likely to include areas for residents to enjoy nature (25%). Due to environmental concerns and a preference for decentralised infrastructure, stormwater run-off in this society is managed on-site; there is no connection to the urban drainage network.

The individualist perspective places the responsibility for water management with private companies, who are tasked with controlling water so it does not affect the maintenance of high economic growth (Offermans, Haasnoot and Valkering, 2011), so high capital and operational costs are acceptable. There is high innovation but little attention to the environment and social solidarity (Offermans, Haasnoot and Valkering, 2011). There is low emphasis on environmental factors, so the preferred infrastructure is not required to provide amenity, biodiversity and water quality benefits. The individualist will require that no land is sterilised by the infrastructure. It is therefore likely that the individualist's favoured intervention will be centralised infrastructure.

Regarding new development, in Markets it is assumed that new urban area would be associated with a low ratio (5%) of permeable area; there is little emphasis on gardens or public green space, and new developments are likely to be industrial. The Markets scenario is likely to manage stormwater run-off by directing it to the existing urban drainage network.

Hierarchists are risk-accepting, and believe that nature is robust within certain limits and is able to cope with small disturbances, and therefore stress that the relationship between humanity and nature is mutually dependant and must be balanced (Van Asselt 2000). Water management is characterised by an emphasis on safety and flood prevention, but it leaves space for economic and natural development (Offermans, Haasnoot and Valkering, 2011).

Centralised governmental agencies are responsible for managing water (Offermans, Haasnoot and Valkering, 2011). The hierarcist perspective has no preference between centralised and decentralised infrastructure. Cost is not a concern. There is some motivation for environmental sustainability, so the preferred infrastructure will require to contribute towards amenity, bio-diversity and water quality. The hierarchist accepts some sterilisation of land to enable water management sustainably. It is proposed that the preferred infrastructure within the Innovation scenario is therefore Swales and Tree Planting. These are site-control level SuDS that are likely to be constructed on public land, e.g. swales managing run-off from roads, reflecting the management of water by institutions.

Regarding new development, there is a high wage economy in Innovation, and society will expect detached housing with large gardens. This corresponds to a high ratio of permeable to impermeable development (50%). Stormwater is likely to be routed through an on-site swale, providing some volume loss, and good peak flow attenuation. An overflow to the urban drainage network is provided.

The fatalist perspective is characterised by a minimisation of resource expenditure (Chai and Wildavsky 1994), and there is little optimism about the future or the environment. The preferred infrastructure will therefore be very inexpensive, and will not necessarily contribute towards the environment.

Regarding new development, new developments are likely to be inexpensive residences because of the lack of economic growth. There is high emphasis on minimising costs, so green space associated with new residences is likely to be low (10%). It could be envisaged that the new development would comprise flats. Stormwater run-off is likely to be connected to the existing urban drainage system or managed using soakaways or basins.

Table 4-5 presents the preferred criteria value in each perspective; the qualitative values associated with each perspective represent the minimum expectation that perspective is likely to have.

The four scenarios have been interpreted to draw out a mix of different SuDS types. One drawback of this mapping process is that only one or two interventions have been mapped to each scenario; this simplification excludes some types of SuDS, such as green roofs, from the analysis.

Table 4-5: Qualitative perspective preferences and infrastructure characteristics for the infrastructure factors.

	Centralised/ Decentralised	Financial Cost		Amenity	Bio-diversity	Land Take	Contribution to Water Quality	Contribution to Water Quantity	
		Capital	Operational					Peak Flow Reduction	Volumetric Control
Individualist Preference	Cent	High	High	Poor	Poor	Low	Low	Good	Poor
Conventional Solutions	Cent	High	High	Poor	Poor	Low	Low	Good	Poor
Fatalist Preference	-	Low	Low	Low	Low	Medium	Medium	Good	Medium
Soakaways	De	Low	Low	Poor	Poor	Low	Good	Good	Good
Detention Basins	Cent	Low	Low	Good	Medium	Medium	Medium	Good	Poor
Hierarchist Preference	-	High	High	Medium	Medium	Medium	Good	Medium	Medium
Swales	-	Low	Medium	Medium	Medium	High	Good	Medium	Medium
Tree Planting	-	High	Medium	Good	Good	Medium	Good	Medium	Medium
Egalitarian Preference	De	Low	Medium	Good	Good	Good	Good	Medium	Medium
Bioretention	De	Low	Medium	Good	Medium	High	Good	Medium	Med-High

4.5 Representation of Adaptations

Section 4.4 mapped stormwater interventions to each of the four scenario narratives based on a qualitative assessment of the traits of the interventions and the traits that would be favoured by each of the four societies. These societies are likely to use the appropriate interventions to ensure that the performance of the urban drainage network does not degrade despite the manifestations of some identified future pressures.

This Section notes that the representation of stormwater interventions can be more completely undertaken external to InfoWorks CS software (4.5.1), and describes how reservoir routing models can be used to achieve this external modelling (4.5.2).

For each stormwater intervention, a description and conceptual schematic of the modelling process is provided. The interventions are Storage Tanks (4.5.3), Soakaways (4.5.4), Swales (4.5.5), Basins (4.5.6), Trees (4.5.7) and Bioretention Areas (4.5.8).

4.5.1 The Representation of Interventions

The representation of interventions is conducted by initially removing the surfaces (either road or roof) from the InfoWorks CS model. The area (m^2) of disconnected surface is multiplied by the rainfall depth per timestep (mm) per timestep, to give a runoff hydrograph from the disconnected surface that serves as the inflow hydrograph to a model using reservoir routing to represent the appropriate intervention.

Reservoir routing is a mechanism for accounting for alterations to flow patterns resulting from control structures such as weirs. It involves a modification of the continuity equation, which is given by:

$$I = O + \frac{dS}{dt}$$

I	Inflow	(m^3/s)
O	Outflow	(m^3/s)
S	Storage	(m^3)

For a time interval, t, the continuity equation may be expressed as:

$$\frac{I_1 + I_2}{2}.t = \frac{O_1 + O_2}{2}.t + (S_2 - S_1)$$

This may be arranged to give:

$$\frac{I_1 + I_2}{2}.t - \frac{O_1}{2}.t + S_1 = \frac{O_2}{2}.t + S_2$$

Outflow and storage parameters are typically a function of H, the depth of water with respect to a feature of the system. This allows an iterative solution to solve this equation. An outflow hydrograph can, in this way, be generated based on any inflow hydrograph and the design parameters of the system being modelled.

Where flow control structures are used to induce storage within the interventions, outflow from the intervention ($Q_{outflow}$) is dictated by the orifice equation:

$$Q_{outflow} = A_o C_d \sqrt{2gH}$$

A_o	Area of orifice	(m2)
C_d	Orifice coefficient	(-)
g	Gravity	(9.81 m/s^2)
H	Depth of water	(m)

Where interventions provide infiltration to groundwater, this Outflow component was represented by:

$$Q_{outflow} = A_w.f$$

A_w	Wetted area	(m^2)
f	soil infiltration rate	(m/h)

Although some intervention are likely to provide further losses through vegetative process, these were assumed to be negligible, particularly in the context of peak flow reduction. Furthermore, infiltration through media within systems was assumed to be immediate.

Total Outflow for a timestep is given by the sum of orifice outflow and infiltration outflow.

This process has been used to develop spreadsheet-based models of the stormwater interventions used in this study. By adjusting the design parameters of the interventions within the spreadsheet-based reservoir routing models based on the idiosyncratic features of the site at which they may be installed, and then routing outflow hydrographs into the InfoWorks CS model, an accurate representation of the interventions' impact on the urban drainage network may be generated.

The following sub-sections (4.53 to 4.58) present a description of these interventions, the modelling methodologies used to represent each intervention, and parameters used to ensure interventions are located in technically-feasible and appropriate locations. Design and feasibility parameters taken from Woods-Ballard *et al.,* (2015) unless otherwise referenced.

4.5.2 Storage Tanks

Storage tanks increase the capacity of the network by retaining stormwater flow. The stormwater is returned into the urban drainage network at a prescribed rate, and ideally the majority of the flow is released following the end of storm, such that there is no superposition of flows within the network. Storage tanks therefore account for no loss of volume from the network.

Outflow from the storage tank in the reservoir routing model is dictated by an orifice. This is a conservative approach as flow into the network from the storage tank will be observed synchronous with other stormwater-induced flows in the network.

Design Parameters:

- Storage tanks are represented as being cuboid.
- They are sub-surface, regional control structures, requiring no land-take.
- To be located underneath public land in the vicinity of trunk sewer lines.

A conceptual schematic of the processes modelled to represent a Storage Tank is shown in Figure 4-3.

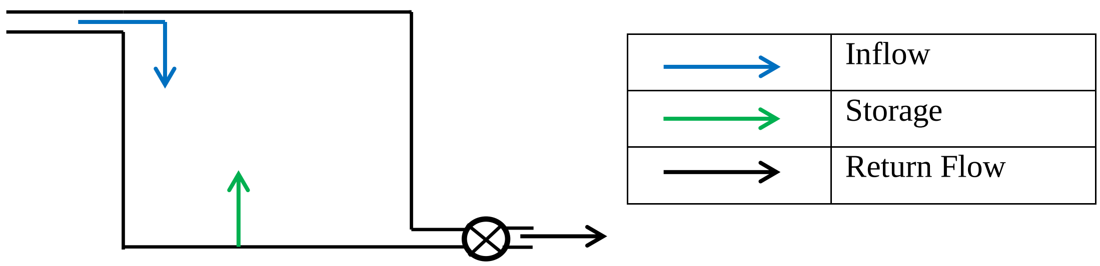


Figure 4-1: A conceptual schematic of a storage tank.

4.5.3 Soakaways

Stormwater directed to soakaways is stored and gradually returned to groundwater through infiltration. To facilitate this, soakaways are constructed with a high void ratio and porous sides. Stormwater that is infiltrated to groundwater can be considered lost from the urban drainage network. When the volume of stormwater entering the soakaway exceeds the capacity of the device to store and infiltrate, there is an overflow to prevent upstream flooding. This overflow can be directed to the urban drainage network or to a local water body, depending on the characteristics of the site.

In accordance with (BRE, 2003) the base of the soakaway is assumed to clog in the long term and therefore no infiltration loss from the base is included. Infiltration from the sides of the soakaway is calculated as a function of the hydraulic conductivity of the surrounding soil, and wetted area on the porous sides of the device, a function of the depth of water in the device.

Design Parameters:

1. Soakaways are assumed to be cylindrical in design.
2. They are a sub-surface, source-control feature, requiring no land-take.
3. They may be located on private or public land.

A conceptual schematic of the processes modelled to represent a Soakaway is shown in Figure 4-4.

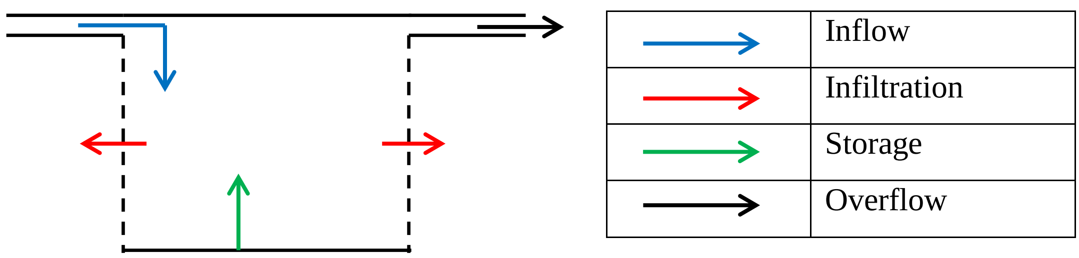


Figure 4-2: A conceptual schematic of a soakaway.

4.5.4 Swales

Swales are landscaped open channels that are typically used to convey stormwater run-off across a site and/or between SuDS. Swales are usually lined with vegetation and as such provide some treatment to the stormwater as it flows through the system. Being vegetated, they provide some environmental benefits.

Provided that associated conditions allow it, swales can enable infiltration of stormwater run-off to groundwater.

In this study, where the swale outflow is directed to the urban drainage network, Swales will be designed to promote attenuation of stormwater run-off.

Design Parameters:

1. Maximum base width: 2 m
2. Maximum linear slope: 6%
3. Maximum side slope: 33%
4. Maximum swale depth: 0.6 m

A conceptual schematic of the processes modelled to represent a Swale is shown in Figure 4-5.

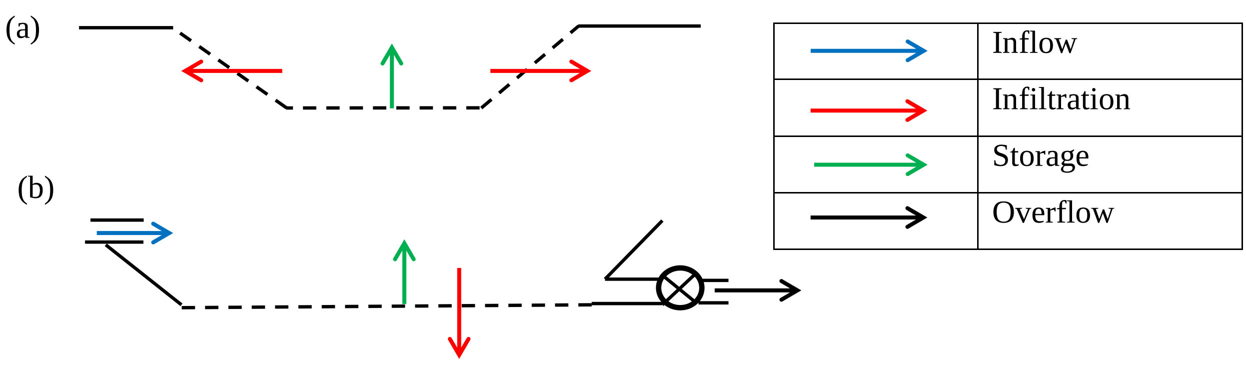


Figure 4-3: A conceptual schematic of a swale in (a) cross-section and (b) long-section.

4.5.5 Infiltration/Detention Basins

Normally dry except during and immediately following rainfall, basins are landscaped depressions that can collect significant volumes of stormwater run-off. During dry spells basins may serve as a recreational facility. Detention basins solely provide; infiltration basins provide storage and enable infiltration to groundwater. Infiltration and detention basins restrict pass forward flow by an orifice. Pass forward flow can be returned to the urban drainage network or to a local water body depending on the site characteristics.

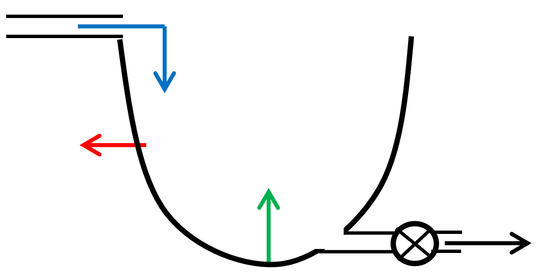
Some losses may be expected to be observed in reality from both detention and infiltration basins resulting from interception by vegetation and evapo-transpiration while stormwater is in residence in the basin. For simplicity and to provide conservative results, these losses are assumed to be negligible.

Basins may be either vegetated or hard-landscaped; the basins in this study are assumed to be vegetated. Detention and infiltration basins are a regional-control feature that should be located on public land.

Design Parameters:

1. Maximum depth: 2 m
2. Effectively flat base
3. Maximum length:width ratio: 5:1
4. Maximum side slope: 33%

A conceptual schematic of the processes modelled to represent Basins are shown in Figure 4-6.



	Inflow
	Infiltration (Infiltration Basins only)
	Storage
	Pass Forward Flow

Figure 4-4: A conceptual schematic of a basin.

4.5.6 Tree Planting

The mechanisms by which trees reduce the volume of stormwater flow to piped urban drainage systems include transpiration (whereby water is absorbed by tree roots and evaporated by leaves), interception (absorbtion of rain that fall directly onto the tree), and increased soil infiltration capacity and rate in the vicinity of the tree.

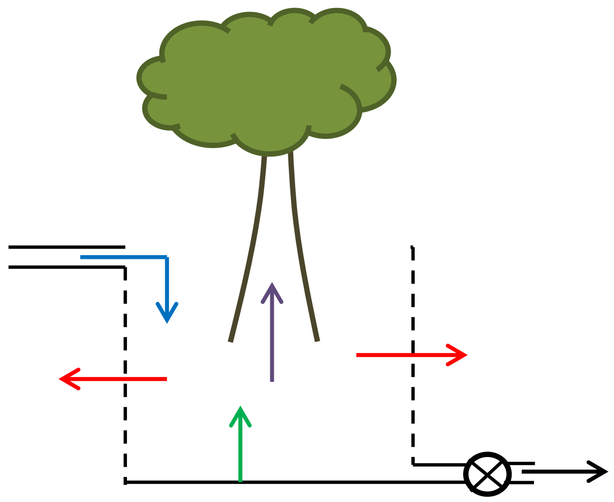
Tree planting interventions for stormwater management can be connected by a continuous underground trench (Trees & Design Action Group, 2014). A continuous trench arrangement increases soil volume for tree root expansion and surface water detention.

Stormwater run-off that is not infiltrated or lost to vegetation-based processes is passed forward via an underdrain to either the urban drainage network or a local water body. Tree planting is assumed to be undertaken on public land, trees connected in series by an underground, linear trench.

Design Parameters:

1. Maximum depth: 2 m
2. To be located over 5 metres from structures, overhead electricity cable and existing urban drainage network pipes (DEFRA, 2011)

A conceptual schematic of the processes modelled to represent tree planting is shown in Figure 4-7.



	Inflow
	Infiltration (Infiltration Basins only
	Storage
	Losses to Vegetation
	Pass Forward Flow

Figure 4-5: A conceptual schematic of tree planting.

4.5.7 Bioretention Areas

Bioretention areas are constructions of engineered soils and vegetation through which stormwater run-off drains. In doing so, the run-off is treated by filtration pollutant removal processes in the vegetated and soil layers. Due to their highly vegetated nature, bioretention areas are considered attractive and environmentally-advantageous stormwater interventions (Robert Bray Associates, 2012). Bioretention areas are very flexible in design, as the configuration of the system can be changed to fit local site characteristics.

Inflow to a bioretention area is typically situated at the top of the system, such that the stormwater flows through the engineered soil levels under the action of gravity. Where possible, infiltration to groundwater can be encouraged. Bioretention areas are typically underdrained, passing forward any flows not lost from the system due to infiltration, and this pass forward may be directed to the urban drainage system or a local water body.

Bioretention areas are a source-control system that can be placed on either public or private land.

Design Parameters:

1. Maximum depth: 1.6 m (typical)
2. Maximum area drained to a bioretention area: 0.8 ha
3. Typical surface area: 2-4% drained area
4. Maximum depth: 0.65 (draining individual properties)
5. Bioretention provides losses of stormwater run-off volume through vegetative processes. A loss rate of 5 mm is used to represent these processes.

A conceptual schematic of the processes modelled to represent Bioretention is shown in Figure 4-8.

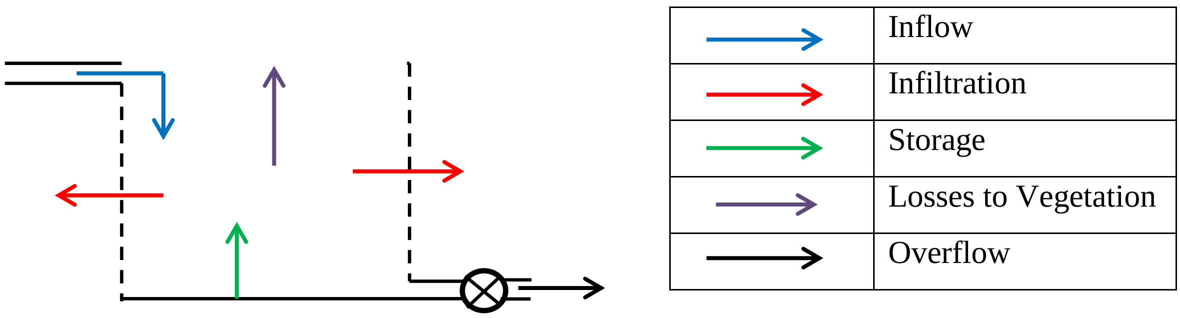


Figure 4-6: A conceptual schematic of bioretention areas.

4.6 Specifying Assessment Methods

Note that throughout this section, cost data has been updated to 2015, the latest year for which data is available, equivalents using the consumer price index (Bank of England, 2017).

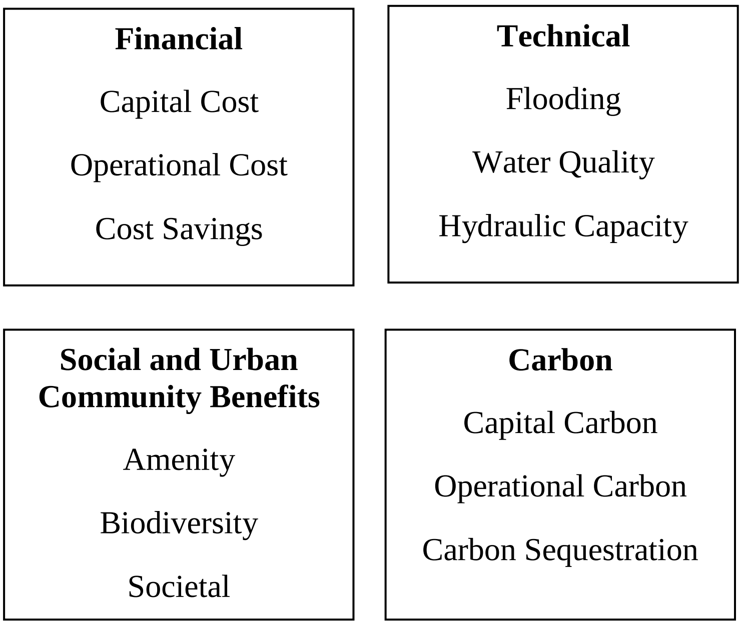


Figure 4-7: Criteria pertinent to the assessment of retrofit SuDS (Badger et al., 2014).

4.6.1 Financial Criteria

The financial category comprises three criteria; Capital Cost, Operational Cost, and Cost Savings. Capital Cost is defined as the financial expense incurred to construct the intervention. Operational Cost is defined as the financial expense incurred annually to operate the intervention, including maintenance costs. Cost Savings are defined as the financial expense that is not incurred as a result of the intervention being operational.

There are a number of factors which limit the availability of cost information for SuDS. The technology is relatively new; private firms that have experience with SuDS guard their cost data from competitors, and public agencies have not had time or resources to compile and release information on their limited experience (Houdeshel *et al.*, 2011). Furthermore, design standards vary across the globe which can mean that reported costs are often incomparable (Houdeshel *et al.*, 2011). Also, many examples of SuDS infrastructure are showcase installations and may not be representative of practical design and construction costs

(Houdeshel *et al.*, 2011). Because SuDS provide multiple benefits, a number of parties may be undertaking maintenance activities on the same SuDS based on the benefits that accrue to each party, and maintenance may be undertaken on a reactive rather than proactive basis (HR Wallingford, 2004b).

HR Wallingford (2004b) presented capital and annual maintenance costs for SuDS in the UK based on an "extensive consultation exercise". Capital cost data was provided for wetlands, retention ponds, infiltration basins, permeable pavements, infiltration trenches, soakaways, filter drains and swales. These costs were accompanied by the region of the UK in which the SuDS is located and the treatment volume of the SuDS. Annual maintenance cost data was provided for retention ponds, detention basins, infiltration basins, filter drains and swales. These costs were accompanied by the region of the UK in which the SuDS is located, the surface area of the system, and the proportion of the capital cost that the annual maintenance represents, where available.

The absence of robust, comparable cost data from case studies for SuDS has prompted the development of an approach that identifies the construction and maintenance activities required for different SuDS types, and associate each activity with a cost based on unit rates, typically taken from civil engineering cost databases.

HR Wallingford, (2004a) provided generic maintenance schedules for different types of SuDS, and presented the costs of annual maintenance of SuDS installed at two motorway service areas; one near Oxford, UK, and the second at Hopwood in Worcestershire, UK. This report published a number of tenders for the contract to undertake maintenance that these two site, and noted the large range of cost estimates submitted. HR Wallingford also noted that the cost of long-term maintenance is related to the initial design characteristics of the infrastructure.

Stovin and Swan (2007) generated indicative capital cost estimates for retrofit soakaways, infiltration trenches, basins, ponds, permeable pavements, swales, and filter drains. These SuDS were designed to manage stormwater run-off from a range of impermeable areas, and the cost of construction was estimated using civil engineering cost databases. Stovin and Swan provide High and Low estimates of capital costs, corresponding to cost differences resulting from site characteristics or construction materials.

This approach to capital cost estimation has been used within spreadsheet tools which allow the user to input characteristics of their SuDS design which are required to develop a cost estimate. The design of retrofit SuDS is frequently required to innovative in order to take account of the pre-existing biophysical landscape (Digman *et al*., 2012), and spreadsheet cost estimation tools can take account of idiosyncratic designs. The SUDS for Roads cost estimation tool developed for the Society of Chief Officers of Transportation in Scotland (SCOTS) (Pittner and Allerton, 2010) provides similar capability for swales, filter drains, permeable paving, ponds, wetlands, basins, bio-retention areas, and filter strips.

Royal Haskoning (2012) used the consultation undertaken by HR Wallingford, and Stovin and Swan's work, to produce databases for capital and maintenance unit costs. Despite the small sample size of costs that inform these works (for example, there are only two examples of soakaway construction costs in the UK in HR Wallingford's survey) they have been used extensively within the literature (e.g. Chow *et al.*, 2013). An alternative approach to estimate the cost of SuDS is to use unit rates for activities associated with the construction and maintenance of SuDS. These sources have been used to generate high and low cost estimates for construction and operational costs (Table 4-6), which will be used to undertake sensitivity testing (Digman *et al.*, 2015). In both cases, the cost estimates for tree planting were generated using guidance from the Trees & Design Action Group (2014) that tree planting could be assessed as a swale system with trees, costing £250 per tree every 10 metres and requiring £6.88 to maintain per year.

The capital cost of conventional storage tanks and sewer laying is taken from (Environment Agency, 2015).

For example, HR Wallingford estimate the cost of swales to be £14.93-22.39/m^2 swale area (adjusted to 2016 values). Stovin and Swan estimate the cost of swales to be £22.92-25.47/m^3 swale volume (adjusted to 2016 values). The SuDS for Roads tool was used to generate high and low estimates for the construction of swales, where the range of costs was achieved using high and low values for items that influence cost, e.g. type of inlet structure. A swale was considered to have a width of 1 metre to enable comparison between these sources. The range of costs generated is presented in Figure 4-8.

Table 4-6: High and low capital cost estimate data.

Intervention	Capital Cost Estimate (£)		
	Low	**High**	**Unit**
Storage Tanks	452	1,239	/m^3 volume
Sewer Laying	123	383	/m length
Soakaways	42.95	140.9	/m^3 volume
Swales	14.93	74.66	/m^2 area
Infiltration Basins	14.93	72.77	/m^3 volume
Detention Basins	16.55	79.35	/m^3 volume
Tree Planting	39.93	99.66	/m^2 area
Bioretention Areas	41.95	65.51	/m^3 volume

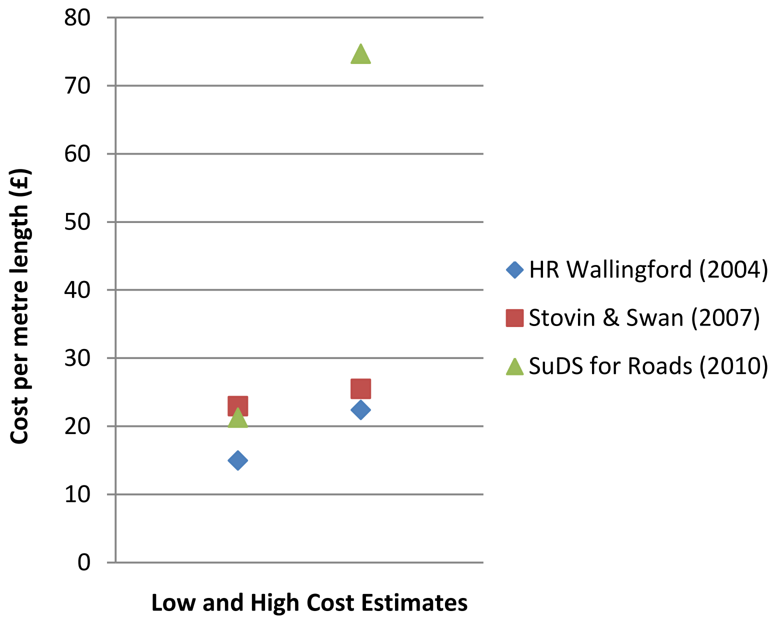


Figure 4-8: High and low cost estimates for the construction cost of swales.

High and low estimates of maintenance costs were generated from Royal Haskoning's database of costs and the SuDS for Roads unit rate cost tool. In both sources, maintenance costs are generated through the attribution of a unit cost to a maintenance activity. However, Royal Haskoning noted that the frequency of different maintenance activities can vary (from

Low to High); this variable was additionally used to generate low and high cost estimates (Table 4-7).

The majority of maintenance costs are incurred from inspection, reporting and information management services. SuDS for Roads calculates the costs of such activities based on monthly visits, Royal Haskoning allows the user to define the frequency of visits between every three years (Low) and ten times per year (High). The cost associated with such visits is very high (£76 per visit in SuDS for Roads) compared to the costs of most maintenance activities that are undertaken (e.g. grass cutting for basins costs £1.48 per $100m^2$), due to labour rates which are used to calculate the costs of site visits. Royal Haskoning estimate the costs of a site visit to be £49.36, while SuDS for Roads puts it at £76 (both prices adjusted for inflation).

The maintenance demands of SuDS within Royal Haskoning and SuDS for Roads are both derived from The SuDS Manual, however the interpretation of the maintenance demands can lead to varying cost estimates; for example Royal Haskoning provides a cost of silt disposal from swales on-site, which is much lower than SuDS for Roads' interpretation of silt disposal which is undertaken off-site, incurring greater expense. This may be due to the SuDS for Roads tool's focus on SuDS for roads; stormwater run-off from roads is likely to be contaminated by hydrocarbons that require specialist disposal. Uncertainty about the required frequency of inspection and maintenance of SuDS, and the different regimes required based on the SuDS application, can result in large differences in the operational cost estimates (Table 4-7).

SuDS for Roads estimates the cost of bioretention systems maintenance including the cost removing and replacing mulching on an annual basis. However, The SuDS Manual suggests that the use of mulching is optional, and should generally be avoided. In the high cost estimate of the annual cost of bioretention systems, the cost associated with mulching is included; however in the low cost estimate it is excluded. These operational cost estimates refer to new maintenance demands as a result of new SuDS infrastructure. Where an urban drainage system exists already, the maintenance of storage tanks is assumed to be negligible.

Table 4-7: High and low annual operational cost estimates.

Intervention	Annual Operational Cost Estimate (£)	
	Low	High
Soakaways	17.14 + 0.11/m^3	762 + 0.11/m^3
Swales	17.14 + 0.7/m^3 + 0.44/m^2	762 + 7/m^3 + 0.18/m^2 + 4/m
Infiltration Basins	110.14 + 0.34/m^2 + 0.89/m^3	762 + 2.41/m^2 + 7.59/m^3
Detention Basins	110.14 + 0.41/m^2 + 0.89/m^3	762 + 2.41/m^2 + 7.59/m^3
Tree Planting	17.83 + 0.7/m^3 + 0.44/m^2	763 + 7/m^3 + 0.18/m^2 + 4/m
Bioretention Areas	17.14 + 2.4/m^2	762 + 2.4/m^2 + 83/m^3

Estimation of the cost savings achieved by installation of a stormwater intervention is based on avoiding two expenses; the costs associated with pumping and treatment of stormwater in the urban drainage network. Hydraulic simulation of the urban drainage network against a typical year rainfall profile using InfoWorks CS allows the user to obtain the total volume of flow being pumped through the network and arriving at the treatment works in a typical year. Comparing these data sets in the baseline and post-intervention construction scenarios provides total avoided pumped and treated volumes. Conversion of pumped and treated volumes into financial units would allow a complete whole life cost of each intervention to be made. The cost of pumping wastewater through a combined urban drainage network is assumed to be £0.09/ML/pumping station (Smith *et al.*, 2011). The cost of treating wastewater is assumed to be £1.89/ML (Georges *et al.*, 2009; DECC 2013) at a medium activated sludge treatment works.

The calculation of the net financial expense incurred is undertaken by converting the total costs incurred and avoided between 2016 and 2050 to a present value, through application of the following equation. A discount rate of 3.5% is used (HM Treasury, 2003).

$$PV = \sum_{t=2016}^{t=2050} \frac{C_t - CA_t}{(1+r)^t}$$

C_t Costs incurred in year t
CA_t Costs avoided in year t
r Discount rate

4.6.2 Technical Criteria

The Technical category comprises Flooding, Water Quality, and Hydraulic Capacity criteria. These criteria assess the contribution of the intervention towards improving these attributes of urban drainage performance. Most instances of stormwater intervention installation to be assessed using this multi-criteria assessment framework will be undertaken with at least one of these performance criteria as the primary design objective. In such cases, all interventions will be designed to achieve the design objective, and will therefore provide similar contributions towards the objective. The contribution of the intervention towards achieving the primary design objective will therefore not be assessed. However, assessment methods for each criterion need to be specified.

The primary paradigm for SuDS assessment involves an analysis of each SuDS type's inherent performance for water quality and quantity management. In the context of retrofit SuDS undertaken by a wastewater service provider, it is more apt to consider the resultant impact on the hydraulic performance of the existing combined sewer system. The assessment of resultant urban drainage system performance rather than the performance of stormwater by SuDS components has been used in previous studies on retrofit SuDS (e.g. Stovin *et al.*, 2013). Table 4-8 presents assessment methods for each of the three Technical criteria.

Some studies have attempted to monetise the value of avoiding CSO and flooding events; the cost saving for each unsatisfactory CSO is estimated at £51,000 per CSO (Gordon-Walker, Harle and Naismith, 2007), and the cost saving for avoiding flooding is £39,000 per flooding (Royal Haskoning, 2012).

Table 4-8: Assessment methods for the Technical Criteria category.

<table>
<tr><th colspan="2">Criteria</th><th>Assessment Method</th></tr>
<tr><td colspan="2">Flooding</td><td>Volume lost from the urban drainage network (m^3)</td></tr>
<tr><td rowspan="2">Water Quality</td><td>Outflow directed to Urban Drainage Network</td><td>Annual volume spilled from the CSO (m^3)
Annual total spill duration (min)
Annual spill count (no.)</td></tr>
<tr><td>Outflow directed to Local Water Body</td><td>Compliance with Table 4-9</td></tr>
<tr><td colspan="2">Hydraulic Capacity</td><td>Duration pipe is at capacity (min)</td></tr>
</table>

Table 4-9 may be used to understand the impact of stormwater run-off on the aquatic environment where the run-off is disconnected from the urban drainage network. This matrix describes an appropriate number of different SuDS treatment processes that stormwater run-off should pass through as a function of the source of the run-off and the sensitivity of the receiving water body.

Table 4-9: Appropriate number of SuDS treatment train components relative to drained site and receiving water characteristics (Woods-Ballard *et al*., 2007).

Run-off catchment characteristics	**Receiving Water Sensitivity**		
	Low	**Medium**	**High**
Roofs only	1	1	1
Residential roads, parking areas, commercial zones	2	2	3
Refuse collection/industrial areas/highways	3	3	4

4.6.3 Social and Urban Community Benefits

Some stormwater interventions provide Social and Urban Community Benefits to the local environment and population. The assessment of these benefits is simultaneously acknowledged as being important and under-developed aspect of the assessment of SuDS, and it is difficult to provide unambiguous conclusions regarding intangible benefits because of a lack of evidence (Demuzere *et al.*, 2014). The majority of studies examine costs in much greater depth than benefits (MWH, 2014). Table 4-10 presents values for social and urban community benefits taken from (Digman *et al.*, 2015). Soakaways are not associated with any Social and Urban Community benefits and are therefore excluded from this assessment.

Table 4-10: Assessment methods for Social and Urban Community benefits.

Component	Assessment Method				Source
	Swales	**Basins**	**Trees**	**Bioretention**	
Amenity	£1.91/resident/ month	£11.14/household/m onth	£1.91/resident/ month	£1.91/resident/ month	Mell *et al.*, (2013); Bastien *et al.*, (2011)
Biodiversity		£208-£4,475/ha/year		£23.87/ha/year	eftec (2010)
Air Quality	-	-	£78.04/tree/year for small urban area	-	McPherson *et al.*, (2002); DEFRA (2013)
Health	Emotional health: £305/person/year for view over green space* £254/person/year for use of green space**				UK NEA (2011)
Groundwater recharge	£0.45/m^3 infiltrated to groundwater				Environment Agency (2013)

* where there was no view of green space previously
** where green space was used sub-monthly previously

4.6.4 Carbon

The Carbon category comprises three criteria; Capital Carbon, Operational Carbon, and Carbon Sequestration. Capital Carbon is defined as the carbon expense incurred to construct the intervention. Operational Carbon is defined as the carbon expense incurred annually to operate the intervention, including maintenance costs. Carbon Sequestration is defined as the removal of carbon from the atmosphere undertaken by vegetation. The reduction in carbon expense associated with reduced volumes passing through pumping and treatment is acknowledged, but the benefits of this reduction are quantified within the Financial category as the Cost Savings criterion, and are not quantified here to avoid double-counting.

Royal Haskoning do not provide estimates of carbon emissions related to the construction or operation of stormwater interventions, however the SuDS for Roads tool does enable emissions associated with the construction and maintenance of SuDS to be calculated.

In order for High and Low estimates of the carbon emission to be estimated, the SuDS for Roads tool has been used to simulate site characteristics that could affect price, for example whether an inlet structure is required, which is an approach used by Stovin and Swan (2007).

Operational carbon estimates are generally related to the removal and disposal of silt. SuDS for Roads suggest this should be undertaken every five years, but Royal Haskoning suggest it could occur as infrequently as every 50 years. This accounts for the factor of ten difference between High and Low estimates for soakaways, infiltration basins, and detention basins. Similarly to operational costs, bioretention areas are specified within SuDS for Roads to have mulch replaced; this is the sole activity associated with a carbon cost for bioretention operation, so a low operational carbon cost is zero, assuming no mulching as suggested by the SuDS Manual.

The capital carbon estimates for soakaways assumes they are constructed of precast concrete (0.059kg$CO2_e$/kg), and storage tanks are constructed of general construction concrete (0.035kg$CO2_e$/kg) (Hammond and Jones, 2008).

Carbon sequestration rates are taken from values reported in Moore and Hunt (2013); where a range of values is reported, this is used to present High and Low estimates. It is possible to monetize these benefits using the factor £5.91/tCO_2e (DECC, 2013). Table 4-11 presents the assessment methods for each Carbon criterion for each intervention.

Table 4-11: Values for the assessment of Carbon criteria.

Intervention	Capital Carbon Estimates ($kgCO2_e$/unit)		Annual Operational Carbon Estimates ($kgCO2_e$/unit)		Annual Carbon Sequestration ($kgCO2_e$/unit)	
	High	**Low**	**High**	**Low**	**High**	**Low**
Storage Tanks	$385/m^3$	-	-		-	
Soakaways	$576/m^3$	-	$0.09/m^3$	$0.009m^3$	-	
Swales	$23.4/m^3$	$6.1/m^3$	$0.06/m^3$	$0.006/m^3$	$0.09/m^2$	$0.015/m^2$
Infiltration Basins	$23.4/m^3$	$14.3/m^3$	$0.09/m^3$	$0.009/m^3$	$0.11/m^2$	$0.087/m^2$
Detention Basins	$23.4/m^3 + 4.75/m^2$	$14.3/m^3 +4.75/m^2$	$0.09/m^3$	$0.009/m^3$	$0.11/m^2$	$0.087m^2$
Tree Planting	$29.2/m^2$	$18.5/m^2$	$0.09/m^3$	$0.009/m^3$	$0.589 + 0.426t + 3.6/t^2$ t = number of trees	
Bioretention Areas	$15.7/m^3$	8/m3	$0.13/m^3$	-	$0.63/m^2$	

4.7 Chapter Summary

The application of the SWITCH framework for the assessment of flexibility requires transient scenario analysis. In this chapter, transient scenarios have been derived from Casal-Campos' (2016) scenario framework, which has previously been used in the context of urban drainage studies.

Transient scenario generation was achieved by forecasting the qualitative extent of future pressures within the four scenario narratives by 2050. Realistic quantified rates of the future pressures identified within the literature were then associated with each scenario. Modelling methods for each pressure were identified.

A further aspect of transient scenario analysis is that there is a link between the external (e.g. environmental) pressures, the impact on society, and the actions a society takes. Each scenario depicts a contrasting society. It was therefore necessary to forecast the types of infrastructure that each society would be likely to prefer. This was achieved using perspective theory. The hydraulic and hydrological processes that need to be modelled to appropriately represent each intervention were identified.

To provide an objective assessment of the costs and benefits incurred, high and low cost estimates were identified from the literature for the pertinent criteria for retrofit SuDS assessment.

5 Application of the Transient Scenarios to Case Study Catchments

5.1 Introduction

The SWITCH framework for the assessment of flexibility requires the application of transient scenarios to case-study catchments. The flexibility of three types of interventions is being tested; conventional solutions, retrofit source-control SuDS and retrofit regional-control SuDS. A conceptual overview of this process is presented in Figure 5-1. This chapter presents the application of Stages 1, 2 and 3 to two real-world case study catchments; Urquhart in the Scottish Highlands and Winterton in North Lincolnshire.

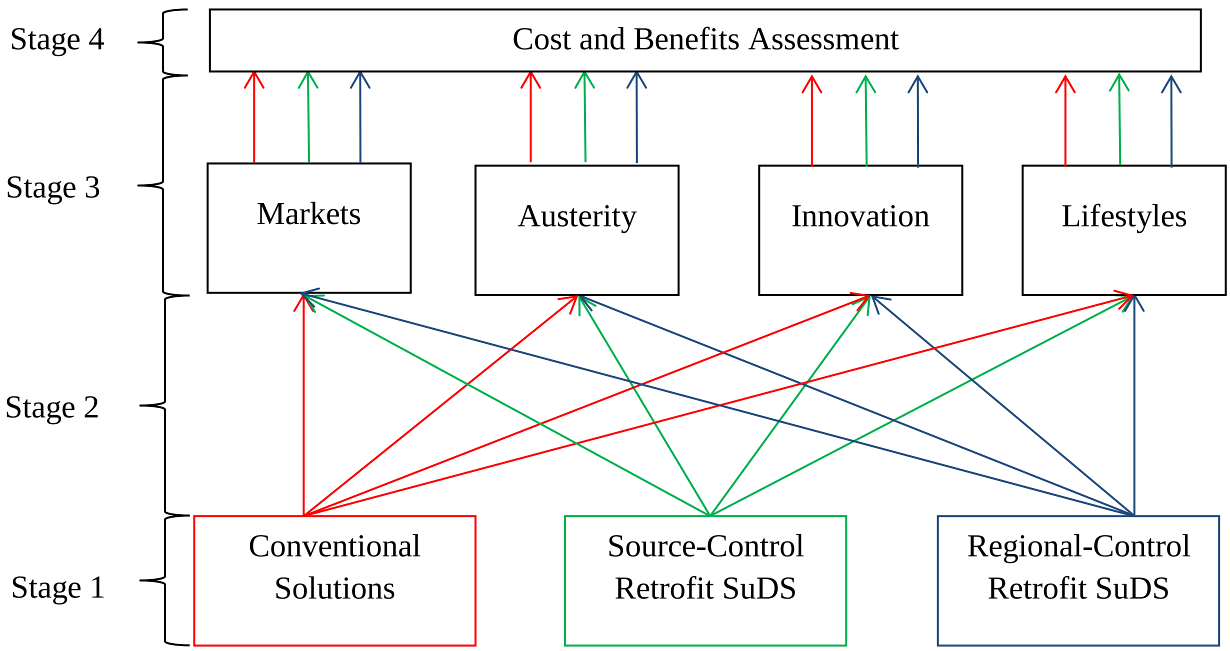


Figure 5-1: Overview of flexibility testing protocol.

5.2 The Urquhart Catchment

As described in section 3.4, Urquhart is a historic village in Moray, Scotland. The urban drainage network that serves Urquhart is the most upstream of three catchments that serve Urquhart, Lhanbryde and Elgin. Wastewater and stormwater run-off generated in Urquhart is pumped to Lossiemouth, onwards to Elgin, and finally North to treatment near Lossiemouth on the North Sea coast. Figure 5-2 provides an overview of the drainage catchment serving Urquhart, Lhanbryde and Elgin.

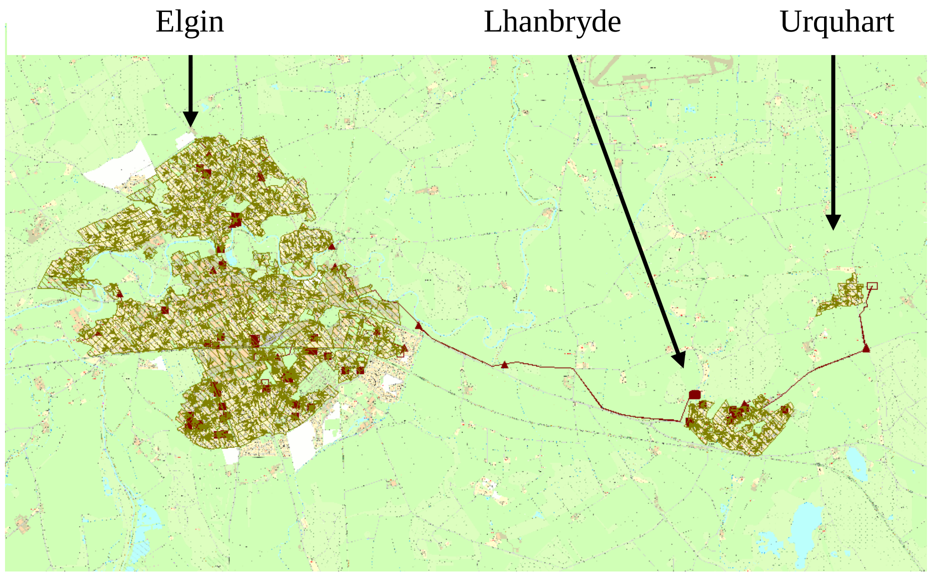


Figure 5-2: Urquhart within the Elgin urban drainage datchment.

The InfoWorks CS model representing Urquhart describes a catchment of 9.6 ha with a population of 270. Urquhart is served by a total length of 645.6 metres of combined sewer. A CSO spills to a local water body to the east of the village. All retained flows are pumped forwards as described previously.

Based on an estimation of the extent to which geology in the area allows infiltration, Moray has been described as being potentially suitable for infiltration SuDS (BGS, 2013). This estimation is based on such factors as depth to groundwater and soil permeability. Physical construction of infiltration infrastructure would require field survey data to verify local ground characteristics; however the data provided by BGS is used as representative of this area.

Equation 5-1 is commonly used to define the infiltration rate based on soil parameters (Horton, 1940).

$$f_t = f_c + (f_o - f_c)e^{-k_e t} \qquad \text{(Equation 5-1)}$$

f_t	infiltration rate at time t	(mm/h)
f_c	final steady state infiltration rate	(mm/h)
f_o	initial infiltration rate	(mm/h)
k_2	decay constant	(h^{-1})

To generate a design infiltration rate, the geology of Moray is assumed to be accurately represented by fine textured soils, a soil type that allows infiltration at a moderate rate. This is representative of the description provided by BGS (2013). The soil parameters associated with fine textures soils, and the f_t value given by application of Equation 5-1, are presented in Table 5-1 (Butler and Davies, 2009). The infiltration rate calculated, 22 mm/h, is used in the design of interventions.

Table 5-1: Soil parameters and resultant design infiltration rate for Moray (Butler and Davies, 2009).

Surface Type	**f_o (mm/h)**	**f_c (mm/h)**	**k_2 (h^{-1})**	**f_t (mm/h)**
Fine textured soils	125	6	2	22

Receiving water quality in the Urquhart area is assumed to be of medium sensitivity, as described by Table 4-9. However, due to the lack of accessible local water bodies, all interventions were hydraulically connected to the existing urban drainage network.

Under a one year return period, thirty minute duration storm event (M1:30) (total depth 5.9 mm symmetrically distributed around a peak intensity of 28.3 mm/h), the flow rate immediately downstream of the catchment is 0.056 m^3/s (Figure 5-3). This rainfall event was uplifted by 20% as advised for accounting for climate change (Environment Agency, 2016), and application of this storm (total depth 7.1 mm symmetrically distributed around a peak intensity of 33.9 mm/h) increased the peak flow rate to 0.058 m^3/s, and the duration of this peak flow rate was extended to ten minutes. The baseline sedimentation in the pipes is assumed to be 3% (Casal-Campos, 2016).

It is assumed that the design objective for the Stage 1 remediation phase is the reduction of the peak flow rate from Urquhart to 0.04m^3/s, in response to a M1:30 +20% storm event. This objective is representative of a flooding problem, as the objective is the reduction of a peak flow rate against a specified design storm to a determined performance objective. Although this does not represent a current problem within the Urquhart catchment, it provides an appropriate objective metric against which comparable interventions can be designed.

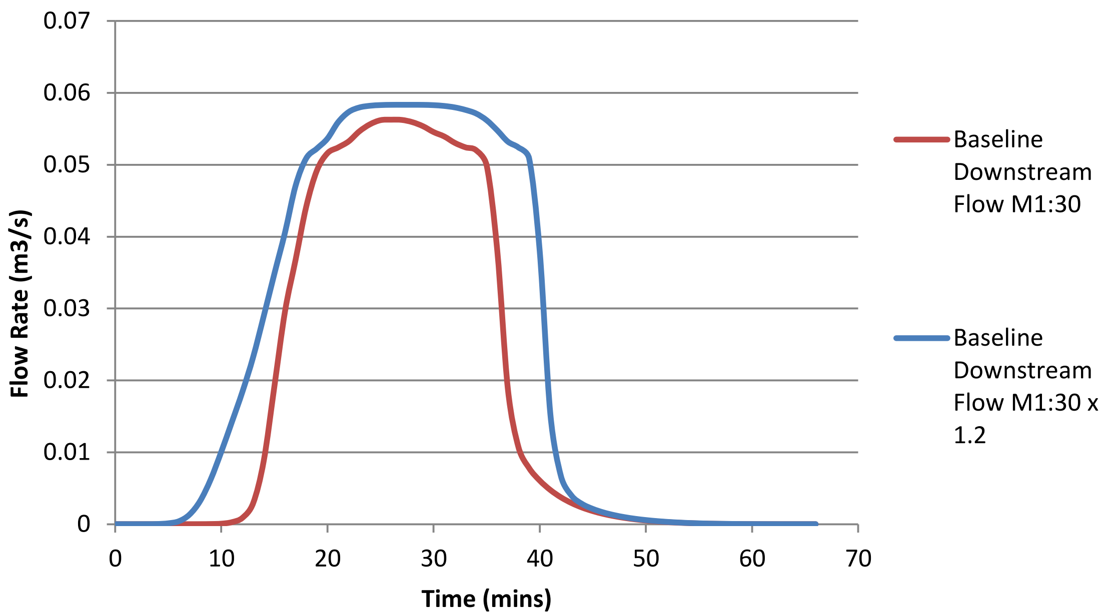


Figure 5-3: Baseline flow from Urquhart under a five-year return period, 60-minute duration storm event.

5.3 Stage 1: 2016 Remediation

It is known from Chapter 3 that subcatchments 5, 6, and 7 are priority locations for stormwater disconnection within Urquhart, and that subcatchment 6 represented the hightest priority location for stormwater disconnection. This section describes the logical processes used to identify the location and size of conventional solutions, and source-control and regional-control retrofit SuDS interventions that reduce the peak flow rate downstream of Urquhart to 0.04. An overview of the process is presented in Figure 5-3.

Through iteration, it was identified that disconnection of 0.35 ha of impermeable surface within subcatchment 6 reduced the flow rate downstream of Urquhart to 0.04 m^3/s in response to the M1:30 (+20%) design storm.

Three interventions were subsequently designed based on this removal of area from the model; a conventional solution comprising a sub-surface off-line concrete storage tank, a source-control retrofit SuDS intervention comprising residential downpipe disconnection to bioretention systems, and a regional-control retrofit SuDS intervention, comprising an end-of-pipe infiltration basin.

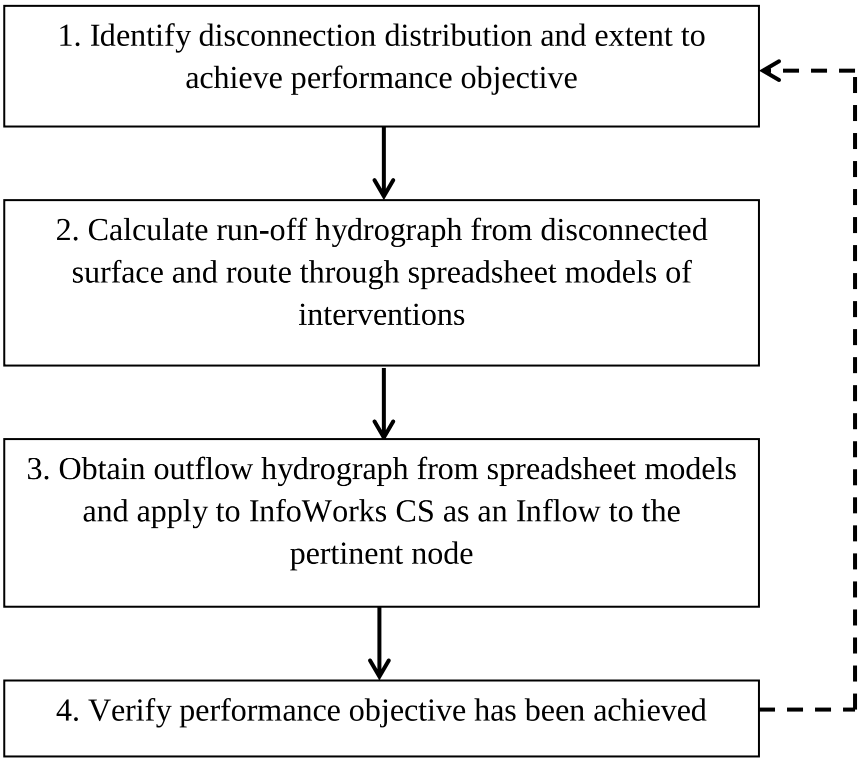


Figure 5-4: Process to identify and size interventions.

The objective of the conventional solution design is to represent a typical conventional solution that a wastewater service provider may construct. It was assumed that increasing pipe sizes downstream of Urquhart was impractical due to the long distance between Urquhart and the treatment plan, and no receiving water bodies were identified upstream of the problematic location, which means that the construction of additional CSO are not viable. Therefore an off-line concrete storage tank has been used to attenuate flows within the urban drainage network. Based on the process shown in Figure 5-3, a storage tank of 225 m^3 was required.

The objective of the source-control retrofit SuDS intervention is to represent current best practise guidance on SuDS; that stormwater should be managed close to the point where it falls on the urban landscape, and that environmental and other intangible benefits should be achieved (Woods-Ballard *et al.*, 2007). The SuDS used to represent this perspective are bioretention systems. Bioretention systems have been used in the UK to achieve stormwater disconnection from residences (Robert Bray Associates, 2012), so residential roofs were targeted in this design. This is appropriate because residential roofs are the majority roof type within Urquhart. The source-control intervention is the disconnection of 0.35 ha of residential roof surface in subcatchment 6 to bioretention systems. A total of 13 bioretention systems of roughly 15 m^3 capacity each were designed (total storage 195 m^3).

A regional-control retrofit SuDS intervention was designed to represent the type of SuDS-based intervention that may presently be favoured by wastewater service providers in the UK; either detention or infiltration basins would be acceptable to a wastewater service provider (Scottish Water, 2015). This intervention was guided by the principles that the wastewater service providers would prefer the intervention(s) not to rely on drainage infrastructure located within private land for reasons of potential maintenance neglection by the land owner, and issues regarding right of access. This resulted in the preferred disconnection of road surfaces within Urquhart. 0.35 ha of road surface was disconnected from subcatchment 6. It was assumed that currently stormwater run-off from road surfaces within Urquhart flows directly to the existing urban drainage network. This requires the separation of storm and foul flows, and therefore the installation of a storm sewer to collect stormwater run-off and direct it to the regional-control SuDS. The longest distance between disconnected road surface and the infiltration basin is 300m following the path of the road. An infiltration basin of 180 m^3 was designed. The location of the infiltration basin is shown in Figure 5-5. This location was selected based of the availability of land, and the natural gradient of the land which falls from south to north towards the sea.

5.4 Stage 2: Application of Future Pressures

Background information presented in Chapter 2 identified additional future pressures; urban creep, urban area expansion, climate change, the degradation of existing infrastructure and uncertain future expected performance. Anticipated manifestation extents for these pressures were presented in Chapter 4, and modelling techniques to represent each pressure in InfoWorks CS were described. The extent of these pressures has been mapped to four diverse scenarios, which describe possible future world states. This Section describes the application of these scenario narratives to the three intervention types.

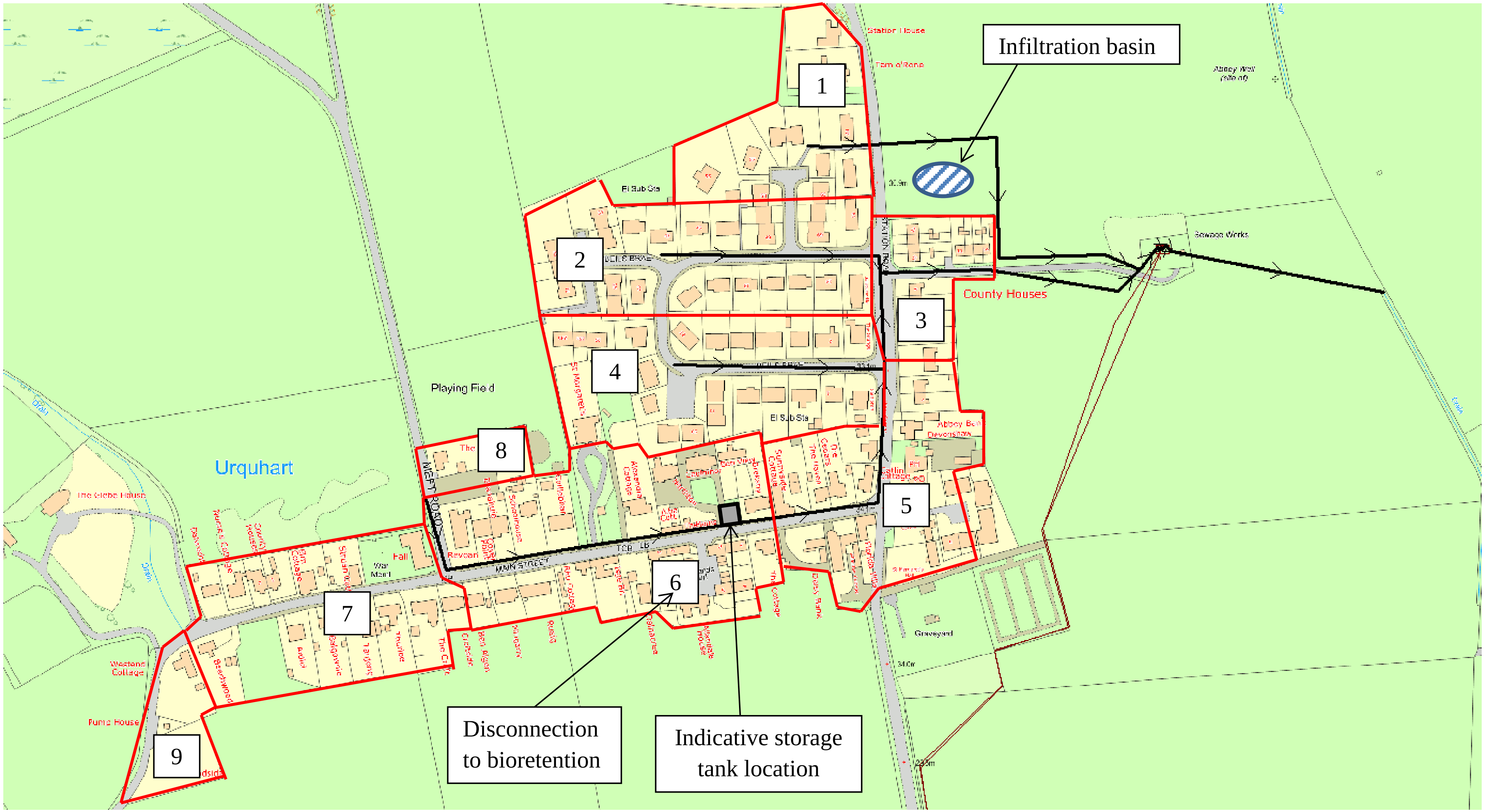


Figure 5-5: Urquhart catchment: subcatchments and interventions (2016 epoch).

5.4.1 Urban Area Expansion Representation

The baseline catchment for Urquhart was developed in three separate ways in Section 5.2.2. Each of these three catchments was “rolled forwards” through four scenario narratives to a time horizon of the year 2050, creating 12 catchments. Of the five future pressures applied to the Urquhart catchment, one, the urban area expansion pressure, is dependent on the nature of the catchment. The logical processes used to apply the urban expansion pressure are described here.

The baseline total impermeable area represented within the Urquhart InfoWorks CS model is 2.03 ha. This value was used as the basis of the urban area expansion calculation for the four scenarios.

Application of the values for urban area expansion under each scenario narrative allowed the forecast urban area expansion area to be calculated. New impermeable area would be likely to be associated with new permeable area, also hydraulically connected to the urban drainage network, and that the extent of the associated permeable area would vary under each scenario narrative. Descriptions of the logic used to map associated permeable ratios to each scenario are given below. Quantification of Urban Area Expansion values is provided in Table 5-2

In Markets, there is forecast to be a population growth of 10%, resulting in an impermeable expansion of 0.203 ha. Based on the transient scenario depiction for Markets, new urban area would be associated with a low ratio of permeable area; there is little emphasis on gardens or public green space, and new developments are likely to be industrial. The increase in catchment area was located in the Playing Fields, north of Catchment 9, as it is deemed that public amenities are of low value in this society. Stormwater run-off from the development would be connected to the existing urban drainage network.

In Austerity, land prices are likely to be low. New developments are likely to be residential, because of the lack of economic growth. There is high emphasis on minimising costs that green space associated with new residences is likely to be low. The increase in catchment area was located in the Playing Fields, north of Catchment 9, as it is deemed that while public amenities are of moderate value, public institutions would be obliged to sell such facilities in order to pay for some subjectively higher priority services. Stormwater run-off from the development is connected to the existing urban drainage network because infiltration is ineffectual and there is no accessible local water body.

Table 5-2: Calculation of urban area expansion.

	Markets	Austerity	Innovation	Lifestyles
Forecast Impermeable Growth (ha)	0.203	0.081	0.173	0.041
Associated Permeable Land Ratio (%)	5	10	50	25
Forecast Permeable Growth (ha)	0.010	0.008	0.087	0.01
Total Urban Area Expansion Area (ha)	0.213	0.088	0.26	0.051

In Innovation, there is a high wage economy, and society's expectation for residences is detached housing with large gardens. This corresponds to a high ratio of permeable to impermeable development. The increase in catchment area was located to the South of subcatchment 5, on the main road through Urquhart. Stormwater is routed through an on-site swale, providing some volume loss, and good peak flow attenuation. An overflow to the urban drainage network is provided.

In Lifestyles, the society does not expect to occupy newly built residences, preferring to renovate where possible. The urban area expansion is therefore likely to be related to industry or leisure. The high environmental concerns exhibited by this society meant new developments are likely to include areas for residents to enjoy nature (25%). Stormwater run-off in this society is managed on-site; there is no connection to the urban drainage network. This development was located in the gap in housing in the South of subcatchment 7.

Connections to the urban drainage network are represented as Inflow hydrographs to the closest node in the InfoWorks CS model.

5.4.2 Performance Deterioration in 2050

The future pressures were applied to the three post-2016 intervention catchments in four scenario-based bundles over twelve independent simulations. Figure 5-4 shows an illustrative representation of the results obtained for the flow profiles immediately downstream of the

Urquhart catchment. Specifically, Figure 5-4 presents the results obtained for the catchment in which some roads were disconnected from the urban drainage network and routed through a regional-control SuDS, namely an infiltration basin. This represents the type of retrofit SuDS intervention a wastewater service provider may construct presently.

The profile “M1:30 (+20% Uplift)” displays the flow rate observed downstream of a non-remediated present-day Urquhart catchment, under a one year return period, thirty minute duration storm event increased by 20% to represent the extent to which climate change may alter rainfall characteristics. This describes the condition against which the regional-control SuDS intervention, as well as the two other intervention options, was designed. It can be seen from Figure 5-6 that in all cases, the peak flow rate observed under a 20% uplifted storm event was not reached. The profile “Baseline” displays the flow rate observed downstream of the same catchment under non-uplifted rainfall conditions.

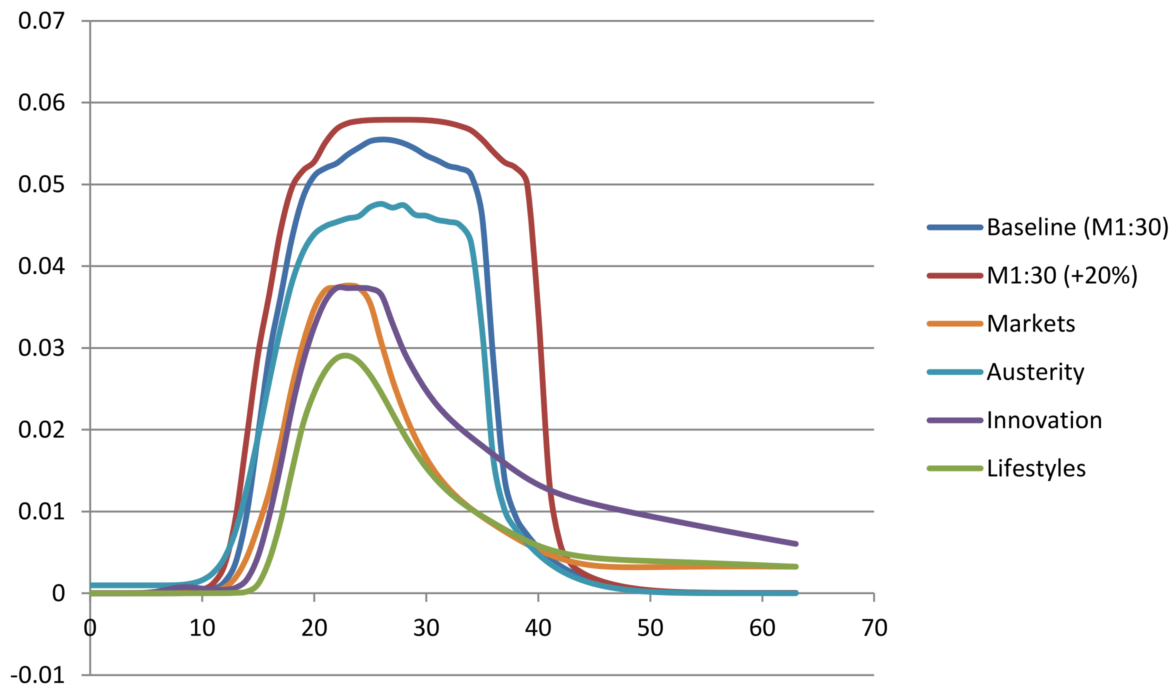


Figure 5-6: The performance of the Urquhart urban drainage network in 2050 under four scenario narratives; infiltration basin constructed in 2016.

In the Markets simulation, some additional impermeable area was included and connected to the urban drainage network. These additional flows, however, have seemingly been offset by the improved capacity of the network resulting from a high degree of maintenance. The Markets narrative describes an increase in the expected performance of the urban drainage network, calculated by a factor of 1.2. The performance objective of 0.04 m^3/s therefore becomes (0.04/1.2 =) 0.033 m^3/s. As such, additional improvement to the network is forecast to be required in 2050. Under the Markets narrative, this improvement is likely to be achieved through the use of conventional solutions.

Under Austerity, although the climate change uplift factor was not as large as the 20% uplift factor used in the design, the Austerity narrative depicts significant degradation in pipe condition, and flows from the increased catchment area are routed directly into the urban drainage network. The Austerity narrative describes an identical level of expected performance to the current level. The performance objective of 0.04 m^3/s therefore remains. As such, additional improvement is forecast to be required in 2050. Under the Scenario B/Austerity narrative, this improvement is likely to be achieved through further use of detention basins.

The Innovations narrative forecasts 10% uplift in rainfall intensity resulting from climate change, whereas the intervention was designed to a 20% uplift. The presence of extensive additional catchment area due to Urban Area Expansion and Urban Creep has evidently been mitigated through the use of on-site, new-development SuDS and an optimal network maintenance regime. The Innovation narrative describes an increase in the expected performance of the urban drainage network, calculated by a factor of 1.1. The performance objective of 0.04 m3/s therefore becomes (0.04/1.1 =) 0.036 m3/s. As such, some minor additional improvement to the network is forecast to be required in 2050. Under the Innovation narrative, this improvement is likely to be achieved through the formal, institutional disconnection of public impermeable areas.

The Lifestyles narrative forecasts 5% uplift in rainfall intensity as a result of climate change, whereas the intervention was designed to 20% uplift. Additionally, the small increases in urban area in Urquhart forecast in Lifestyles were assumed to be drained to source-control SuDS that had no connection to the urban drainage network. Some reduction in urban drainage network maintenance ensured that the improvement observed under Lifestyles conditions was not greater. The Lifestyles narrative describes a decrease in the expected performance of the urban drainage network, calculated by a factor of 0.9. The performance objective of 0.05 m^3/s therefore becomes (0.04/0.9 =) 0.044 m^3/s. As such, additional improvement to the network is not forecast to be required in 2050.

A summary of the modelled and scenario-expectation flow rates for each starting intervention in each scenario depiction of 2050 is presented in Table 5-3, Section 5.5.

5.5 Stage 3: 2050 Remediation

The application of some anticipated future pressures to a model of the Urquhart urban drainage system was undertaken. It was observed that despite remediation in 2016, undertaken in three alternative ways, the urban drainage network is anticipated to require further remediation based on scenario-bundled pressures to 2050 (Table 5-3) in most pathways.

The remediation of the urban drainage network in 2050 conditions was undertaken. The logical processes used to inform the development of intervention can be assumed to be undertaken using the same iterative method presented in Section 5.3.

Table 5-3 summarises the interventions designed for each of the three starting interventions under four alternative narrative scenarios describing the manifestation of pressures on urban drainage networks to the year 2050. The scenario narratives inform the likely types of stormwater intervention used in each narrative (Section 4.3). Stormwater interventions have been designed that remediate the 2016 interventions to the expected performance under each scenario.

It may be noted that stormwater disconnection was undertaken preferentially in order of efficient subcatchments. It was assumed that the hierarchy of efficient subcatchments remained similar to that understood in the present.

Table 5-3: Summary of the performance of the Urquhart urban drainage network in 12 representations of the 2050 catchment, and interventions designed to remediate the 2050 catchment where the network is judged to fail again.

2016 Intervention	**2050 Narrative Scenario**	**2050 Modelled Performance (m^3/s)**	**2050 Performance Objective (m^3/s)**	**Achieved?**	**2050 Intervention**
Conventional (Storage Tank)	Markets	0.037	0.033	✘	Expansion of storage tank by 60m^3, roads and roof disconnection in Subcatchment 7
	Austerity	0.048	0.040	✘	New 110 m^3 infiltration basin, roads disconnection in Subcatchment 5
	Innovation	0.037	0.036	✘	10 linear metre tree planting, roads disconnection in subcatchment 5
	Lifestyles	0.029	0.044	✓	n/a
Source-Control Retrofit SuDS (Roofs to Bioretention)	Markets	0.037	0.033	✘	New 50 m^3 storage tank, roads and roof disconnection in subcatchment 6
	Austerity	0.048	0.040	✘	New 85 m^3 infiltration basin, roads disconnection in subcatchment 6
	Innovation	0.037	0.036	✘	7 linear metre tree planting, roads disconnection in subcatchment 6
	Lifestyles	0.029	0.044	✓	n/a
Regional Control Retrofit SuDS (Roads to Infiltration Basins)	Markets	0.037	0.033	✘	New 75 m^3 storage tank, disconnection of roofs in subcatchment 6 and roads in subcatchment 7
	Austerity	0.048	0.040	✘	Expansion of infiltration basin by 110 m^3, roads disconnection in subcatchment 5
	Innovation	0.037	0.036	✘	10 linear metre tree planning, roads disconnection in subcatchment 5
	Lifestyles	0.029	0.044	✓	n/a

5.6 The Application of the Protocol to the Winterton Catchment

This hydraulically-distinct section of the Winterton catchment has been selected for use in this thesis for the following reasons:

1. CSO metrics may be used as the performance objective of the interventions. This provides a contrast with the Urquhart catchment, the design object of which was representative of a flooding problem;
2. North Lincolnshire is reported to allow greater infiltration than Moray;
3. The presence of significant non-residential roof surface contrasts with Urquhart.

To understand the performance of the CSO under baseline conditions, a typical year rainfall event group was applied to the Winterton model in InfoWorks CS. Typical year rainfall event groups comprise roughly 150 individual rainfall events that are statistically representative of observed rainfall patterns observed in the UK 1961-1991. Typical year rainfall event groups are not time-series rainfall profiles for a year. The typical year rainfall event group was uplifted by 20%, and applied to the Winterton model. The CSO spilling to the East of the Winterton catchment exhibited the performance metrics presented in Table 5-4 under these conditions.

Table 5-4: Performance of the Winterton CSO for a typical year rainfall event group (+20%).

Location	**CSO Metric***		
	Total Spill Count* (no.)	**Total Spill Volume (m3)**	**Total Spill Duration (m)**
CSO Winterton	5	361	411

Spills with sub-25m^3 volumes are discounted from this summary in accordance as the margin of error of InfoWorks CS is assumed to be ± 25m^3 (Irish Water, 2015)

Of the rainfall events applied to the model, meaningfully large CSO spill metrics (judged by total spill volume) were observed from four events. For the purposes of time- and resource-efficiency, these four storm events were used through the subsequent design of interventions in Winterton.

Following the same method for the derivation of soil infiltration rate presented in Section 5-2, the infiltration rate in North Lincolnshire is given by 37.4 mm/h for medium textured soils

(BGS, 2013, Butler & Davies, 2009; Horton, 1940). Receiving water quality is assumed to be low in Winterton, as OS Mastermap data suggests that local watercourses are dedicated rural drainage channels.

Area 4 was identified as the most efficient are in which to undertake stormwater disconnection in Winterton for the improvement of the CSO (Figure 3-13). Area 4 is presented in Figure 5-6. The methodology used in the design of the option in the Winterton catchment is similar to that used the Urquhart catchment (Figure 5-3), and so it not described here. The Stage 1 design objective is the reduction of the CSO performance metrics, shown in Table 5-4 for a 20% uplifted typical year, to 2 spills per year, maximum 200 m^3 total volume spilled per year. An interesting characteristic of area 4, shown in Figure 5-7, is that it comprises both institutional and residential roofs. Through modelling, it was observed that 50% disconnection of impermeable area within Area 4 achieved the performance objective under the impactful storm event.

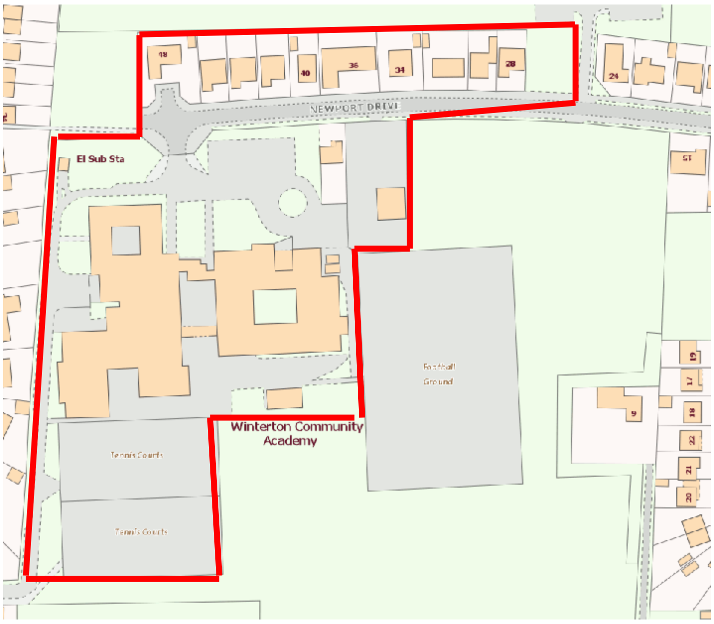


Figure 5-7: Area 4 in Winterton.

The Stage 1 interventions were therefore designed as follows:

The conventional solution is an 80 m^3 storage tank; institutional, residential roofs, and roads amounting to 50% of the impermeable area are stored within this tank.

The regional-control retrofit SuDS solution is a 65m^3 infiltration basin. The school roof is assumed to connect to the basin, and it is located on the school grounds.

The source-control retrofit SuDS solution is the disconnection of the ten residential roofs in the north through the construction of ten bioretention systems of 7 m^3 storage capacity each.

Table 5-5 describes the Stage 3 adaptations required following application of the scenario-based pressures to the urban drainage network. Because all disconnection was undertaken in the same area, there are no efficiency losses in 2050; hence the solutions are of comparable sizes.

5.7 Chapter Summary

This chapter presented the application of the transient scenarios to two real-world case-study catchments; Urquhart in Scotland and Winterton in North Lincolnshire. Stage 1 adaptations, used to remediate the performance of the urban drainage network in the present, were designed to represent typical examples of conventional solutions, and retrofit source-control and regional-control SuDS. Scenario-specific future pressures were applied to the catchments, and a process for generating urban area expansion within transient scenarios was described. If required, Stage 3 interventions were designed; these are infrastructure constructed in the post-2050 catchment to remediate the network to an acceptable standard. The choice of infrastructure in each scenario was related to the transient scenarios through the use of perspective theory presented in chapter 4.

Table 5-5: Summary of the performance of the Winterton urban drainage network in 12 representations of the 2050 catchment, and interventions designed to remediate the 2050 catchment where the network is judged to fail again.

2016 Intervention	**2050 Narrative Scenario**	**2050 Modelled Performance**		**2050 Performance Objective**			**2050 Intervention**
		m^3	**No.**	**(m^3/s)**	**No.**	**Achieved?**	
Conventional (Storage Tank)	Markets	324	2	167	2	✗	Expansion of storage tank by 70m^3
	Austerity	355	2	200	2	✗	New 70m^3 infiltration basin
	Innovation	244	2	182	2	✗	New tree planting 40 linear metres
	Lifestyles	182	2	222	2	✓	n/a
Source-Control Retrofit SuDS (Roofs to Bioretention)	Markets	324	2	167	2	✗	New 70 m^3 storage tank (institutional roof disconnection)
	Austerity	355	2	200	2	✗	New 70m^3 infiltration basin (institutional roof disconnection)
	Innovation	244	2	182	2	✗	New tree planting 40 linear metres (institutional roof and road disconnection)
	Lifestyles	182	2	222	2	✓	n/a
Regional Control Retrofit SuDS (School Roof to Infiltration Basins)	Markets	324	2	167	2	✗	New 70 m^3 storage tank (residential roofs and roads to disconnection)
	Austerity	355	2	200	2	✗	New 70m^3 infiltration basin (residential roofs and roads to disconnection)
	Innovation	244	2	182	2	✗	New tree planting 40 linear metres (residential roofs)
	Lifestyles	182	2	222	2	✓	n/a

6 Results and Discussion

6.1 Introduction

Following application of the transient scenarios to the Urquhart and Winterton catchments in Chapter 5, this chapter monetises the costs and benefits associated with three types of intervention, conventional solutions, regional-control SuDS, and source-control SuDS, for the period 2016-2050. The minimax-regret principle is applied to identify which intervention is the most flexible.

6.2 Urquhart Results

This section presents the cost and benefit data associated with the application of the Flexibility testing protocol in the Urquhart catchment. The calculation of costs assumes that the total expense of the construction of the Stage 1 intervention is met in the present year, 2016, the assumed year of construction, and annual operational costs are met between the years 2017 to 2050 inclusive. The total expense of the construction of the Stage 3 intervention is assumed to be met in the year 2051.

6.2.1 Stage 1 Intervention

This Section presents the calculation of the costs associated with Stage 1 interventions. The process used to generate the cost data for the regional-control retrofit SuDS intervention, a 180 m^3 infiltration basin, are presented, and the cost data for the other two interventions is presented in tabular form.

From Table 416, the construction costs of an infiltration basin range from £14.93 to £72.77 per m^3 volume. The infiltration basin has a volume of 180 m^3. The estimated capital cost range is therefore (14.93 x 180) = £2,911 to (72.77 x 180) = £14,191.

The design of the infiltration basin was premised on the disconnection of the public road surface, which is assumed to drain directly to the urban drainage network. Therefore, 300 m of sewer separation is required, costing between (123 x 300) = £36,900 and (383 x 300) = £114,900

From Table 4-11, the carbon cost of constructing an infiltration basin is estimated to range from 14.3kgCO_2e/m^3 to 23.4kgCO_2e/m^3. The estimated capital carbon cost range is therefore (14.3 x 180) = 2,574kgCO_2e to (23.4 x 180) = 4,212kgCO_2e.

The carbon cost of HDPE stormwater sewer is assumed to be 2.02 kgCO_2e/m. The carbon cost of excavation and laying the sewer is assumed to be 3.26 kgCO_2e/m (Hammond and Jones, 2011). The separation of stormwater flows therefore adds (2.02 x 300) + (3.26 x 300) = 1,584 kgCO_2e.

Carbon cost may be monetised at a rate of £5.91/tCO_2e (DECC, 2015), producing a financial carbon cost of £25.02.

Table 6-1 presents construction financial and carbon cost data for the Stage 1 interventions in Urquhart.

Table 6-1: Construction financial and carbon costs for Stage 1 interventions in Urquhart.

Intervention	**Financial Cost (£)**		**Carbon Cost (kgCO_2e)**		**Carbon Cost (£)**		**Total (£)**	
	High	**Low**	**High**	**Low**	**High**	**Low**	**High**	**Low**
Conventional Storage Tank	278,775	101,700	86,625	-	511.95	-	137,300	102,212
Regional Infiltration Basin	129,091	39,811	5,796	4,158	34.25	24.57	129,125	39,835
Source Bioretention Systems	11,792	7,551	3,062	1,560	18.10	9.22	11,810	7,560

6.2.2 Operational Costs and Benefits 2017 – 2050

The operational costs incurred and benefits received as a result of each intervention are calculated, and application the following equation gives the net cost (or benefit) of each intervention over the period 2017 to 2050 (i.e. post-Stage 1 intervention to immediately pre-Stage 3 intervention):

$$PV = \sum_{t=2016}^{t=2050} \frac{C_t - CA_t}{(1+r)^t}$$

C_t Costs incurred in year t
CA_t Costs avoided in year t
r Discount rate

The potential costs of each intervention are: the financial and carbon costs incurred through maintenance, and the cost of pumping water (if required).

The potential benefits of each intervention are: the costs of pumping and treating stormwater reduced, carbon sequestration, and social and urban community benefits.

The maintenance costs of an infiltration basin are (Table 4-7) are between 110.14 + 0.34/m^2 + 0.89/m^3 and 762 + 2.41/m^2 + 7.59/m^3. For an infiltration basin of 180 m^3 capacity and plan area 110 m, the annual maintenance costs are therefore between (110.14 + 0.34 x 110 + 0.89 x 180) = £308 and (762 + 2.41 x 110 + 7.59 x 180) = £2,393

The carbon cost of maintenance is estimated to range from 0.009 to 0.09kgCO_2e/m^2, which = 0.99 to 9.9 kgCO_2e.

The carbon sequestration achieved by an infiltration basin is 0.087 to 0.11kgCO_2/e, which = 9.57 to 12.1 kgCO_2e.

The net annual carbon cost associated with the infiltration basin is therefore (9.9 – 9.57) = 0.33 kgCO_2e or (0.99 – 12.1) = -11.11 kgCO_2e. The costs can be monetised to £0.002 and -£0.07 respectively.

The calculation of the pumping and treatment costs reduced through the use of an infiltration basin is assessed through the application of a typical year rainfall event group to the Urquhart model in the baseline state to understand the total flow volume from Urquhart (108,624 m^3). Of this event group, one storm was isolated and applied to the spreadsheet model of the infiltration basin. The difference between the flow volume from Urquhart in the baseline state (521 m^3) and the flow volume from Urquhart with an infiltration basin (493 m^3) for a single storm is considered representative of the typical year, and so the flow from Urquhart in the baseline was assumed reduced by the same percentage observed (493/521 x 108,624) = 102,786 m^3. This suggests that the total reduction in flow volume is 108,624 – 102,786 = 5, 838 m^3. The cost of pumping through a pumping station is considered to be £0.09/ML. The costs avoided are therefore (0.09 x 5.838) = £0.56. Flow from Urquhart passes through three pumping stations to treatment, so (3 x 0.56) = £1.68. The cost of treating wastewater is considered to be £1.89/ML, so the avoided costs are (1.89 x 5.838) = £11.03. This approach to identified costs avoided through pumping and treatment of stormwater was taken to reduce computational modelling time and is potentially inaccurate; however the magnitude of these costs compared to operational and social and urban community benefits is small and therefore any inaccuracy is unlikely to affect the results.

For an infiltration basin, the social and urban community benefits are £11.14 per household per month for amenity and £0.45/m^3 infiltrated to ground water, and biodiversity benefits of between £208 and £447/ha/yr. Assuming that 15 houses have access to the basin for amenity value, the annual benefits are (11.14 x 15 x 12) = £2005.20. It is known from the calculation of the costs associated with pumping and treatment that about 5,838 m^3 of stormwater is infiltrated per year. Therefore the benefit is (0.45 x 5,838) = £2,627.10. The infiltration basin area is 0.011ha, and so provides biodiversity benefit of between (208 x 0.011) = £2.29 and (4475 x 0.011) £49.23 per year.

The range of total annual costs incurred is therefore (308 + -0.07) = £307.93 and (2393 + 0.002) = £2393.

The range of total annual benefits achieved are (1.68 + 11.03 + 2005.2 + 2627.1 + 2.29) = £4,647 or (1.68 + 11.03 + 2005.2 + 2627.1 + 49.23) = £4,694.

The annual benefit of the infiltration basin is therefore estimated to range from (4694 - 307.93) = £4,386 to (4647 – 2393) = £2,254. The net present value of these benefits for the period 2017-2050 is £86,407 and £44,405 respectively.

Table 6-2 presents the operational cost benefit assessment data for the three Stage 1 interventions in Urquhart. Note: a positive net present value describes a net benefit accrued by the infrastructure.

Table 6-2: Operational costs and benefits of the stage 1 interventions in Urquhart (2017-2050).

Intervention	**Operational Costs (£)**		**Volumetric Reduction Benefits (£)**	**Social and Urban Community Benefits (£)**		**Net Present Value 2017-2050 (£)**	
	High	**Low**		**High**	**Low**	**High**	**Low**
Conventional Solution	0		0	0		0	
Infiltration Basin	2,393	308	13	4,694	4,647	86,407	44,405
Bioretention	2,524	160	16	1,785	1,785	32,329	-14,244

6.2.3 Adaptation Costs

Following application of the transient scenario framework to Urquhart, the Stage 1 infrastructure was required to be adapted in 2050. Transient scenario analysis allows the pressures, expectations, and the responses of the society to be matched with the overall scenario narrative. This produced twelve different adaptations in 2050. The financial and carbon costs associated with their construction are presented in Table 6-3.

6.2.4 Complete Cost Estimates

Table 6-4 presents the complete cost estimated data derived from the application of the transient scenarios to Urquhart. This has been generated by aggregating the cost of constructing the stage 1 interventions, the operational costs and benefits for the period 2017-2050, and the costs of using scenario-defined stormwater management infrastructure to remediate the network to the required standard in 2050. High and low cost estimates have continued to be used; the process of calculating these is as follows. To generate a high cost estimate, the high stage 1 intervention cost and the high stage 3 intervention cost have been summed. The lower net present value for the period 2017-2050 is then subtracted from this construction cost value. This is therefore the worst case financial estimate over the transient scenario. Similarly, the low stage 1 intervention costs and the low stage 3 intervention cost have been summed and the high net present value for the period 2017-2050 is subtracted from this construction cost value to provide the best case financial estimate over the transient scenario.

Table 6-3: Complete high and low estimates for Urquhart.

Stage 1 intervention	**Total Cost (£)**							
	Markets		**Innovation**		**Austerity**		**Lifestyles**	
	High	**Low**	**High**	**Low**	**High**	**Low**	**High**	**Low**
Conventional Solution	211,777	129,469	138,299	102,613	154,896	106,939	137,300	102,212
Infiltration Basin	177,816	-12,501	85,719	-46,171	92,112	-44,921	84,720	-46,572
Bioretention System	88,118	-12,680	27,053	-6,283	51,402	744	26,054	-6,684

Table 6-4: Construction financial and carbon costs for Stage 3 adaptations in Urquhart.

Stage 1 Intervention	Scenario & Stage 3 Adaptation	Financial Cost (£)		Carbon Cost ($kgCO_2e$)		Carbon Cost (£)		Total (£)	
		High	Low	High	Low	High	Low	High	Low
Conventional Solution: Storage Tank	**Markets** 60m^3 expansion of storage tank	74,340	27,120	23,100	-	137	-	74477	27257
	Austerity new 110m^3 infiltration basin, 25m sewer	17,580	4,717	2,706	1,705	16	10.1	17596	4727
	Innovation 10 linear metre tree planting	997	400	292	185	1.73	1.09	999	401
	Lifestyles no adaptation required	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a
Source-Control Retrofit SuDS: Bioretention systems	**Markets** new 50m^3 storage tank	61,950	22,600	19,250	-	114	-	62064	19364
	Austerity new 85m^3 infiltration basin, 50m sewer	25,335	7,419	2,253	1,480	13.3	8.75	25348	7428
	Innovation 7 linear metre tree planting	698	280	204	130	1.21	0.77	699	281
	Lifestyles no adaptation required	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a
Regional-Control Retrofit SuDS: Infiltration basin	**Markets** new 75m^3 storage tank	92,925	33,900	28,875	-	171	-	93096	34071
	Austerity infiltration basin expanded by 110m^3	7,277	1,642	2,574	1,573	15.2	9.3	7392	1651
	Innovation 10 linear metre tree planting	997	400	292	185	1.73	1.09	999	401
	Lifestyles no adaptation required	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a

6.3 Winterton Results

Tables 6-5 to 6-8 present the costs and benefits associated with the application of the transient scenario framework in the Winterton catchment.

Table 6-5: Construction financial and carbon costs for stage 1 interventions in Winterton.

Intervention	Financial Cost (£)		Carbon Cost ($kgCO_2e$)		Carbon Cost (£)		Total (£)	
	High	Low	High	Low	High	Low	High	Low
Conventional Storage Tank	99,120	36,160	30,800		182		99,302	36,342
Regional Infiltration Basin	4,730	970	1,521	930	8.98	5.5	4,739	976
Source Bioretention Systems	4,586	2,937	1,099	560	6.5	3.3	4,593	2,940

Table 6-6: Annual operational costs and benefits of the stage 1 interventions in Winterton.

Intervention	Operational Costs (£)		Volumetric Reduction Benefits (£)	Social and Urban Community Benefits (£)		Net Present Value 2017-2050 (£)	
	High	Low		High	Low	High	Low
Conventional Solution	0		0	0		0	
Infiltration Basin	1,340	180	8	3675	3588	69,012	44,445
Bioretention	6,668	113	7	1312	1312	23,759	-105,379

Table 6-7: Construction financial and carbon costs for Stage 3 adaptations in Winterton.

Stage 1 Intervention	Scenario & Stage 3 Adaptation	Financial Cost (£)		Carbon Cost ($kgCO_2e$)		Carbon Cost (£)		Total (£)	
		High	**Low**	**High**	**Low**	**High**	**Low**	**High**	**Low**
Conventional Solution: Storage Tank	**Markets** 70m^3 expansion of storage tank	86,730	31,640	26,950	-	159		86,889	31,799
	Austerity new 70m^3 infiltration basin	5,094	1,045	1,638	1,001	9.7	5.9	5,104	1,007
	Innovation 40 linear metres tree planting	3,986	1,597	1,168	740	6.9	4.4	3,993	1,601
	Lifestyles no adaptation required	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a
Source-Control Retrofit SuDS: Bioretention systems	**Markets** new 70m^3 storage tank	86,730	31,640	26,950	-	159		86,889	31,799
	Austerity new 70m^3 infiltration basin	5,094	1,045	1,638	1,001	9.7	5.9	5,104	1,007
	Innovation 40 linear metres tree planting	3,986	1,597	1,168	740	6.9	4.4	3,993	1,601
	Lifestyles no adaptation required	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a
Regional-Control Retrofit SuDS: Infiltration basin	**Markets** new 70m^3 storage tank	86,730	31,640	26,950	-	159		86,889	31,799
	Austerity new 70m^3 infiltration basin (25m sewer)	14,669	4,120	1,770	1,133	10.5	6.7	14,680	4,127
	Innovation 40 linear metres tree planting	3,986	1,597	1,168	740	6.9	4.4	3,993	1,601
	Lifestyles no adaptation required	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a

Table 6-8: Complete high and low estimates for Winterton.

Stage 1 intervention	Total Cost (£)							
	Markets		Innovation		Austerity		Lifestyles	
	High	Low	High	Low	High	Low	High	Low
Conventional Solution	186,191	68,141	103,295	37,943	104,406	37,349	99,302	36,342
Infiltration Basin	47,183	-36,257	-35,713	-66,435	-25,026	-63,909	-39,706	-68,036
Bioretention System	196,861	10,980	11,3965	-19,218	115,076	-16,692	109,972	-20,819

6.4 Regret under High Cost Estimates

Regret is the difference between the performance of a strategy, and the performance of the best performing strategy in the same future scenario (Lempert, 2003). Low regret is an indicator of high flexibility (MWH, 2014). Peters *et al.* (2011) recommend the use of minimax-regret principle for measuring the flexibility in urban drainage studies.

In Urquhart, the complete high cost estimate for the conventional solution stage 1 intervention is £211,777 under the Markets scenario. For the infiltration basin under the Markets scenario, the high cost estimate is £177,816. The high cost estimate for the bioretention option under the Markets scenario is £88,118. The regret incurred by selecting the conventional solution is therefore (211,777 – 88,118) = £123,659. The regret incurred by selecting the infiltration basin is therefore (177,816 – 88,118) = £89,698. The regret incurred by selecting the bioretention option is therefore (88,118 – 88,118) = £0. This process is repeated for each scenario. The maximum regret incurred by each stage 1 intervention is noted as the “maximum regret”. The minimax-regret principle dictates that the option with the smallest maximum regret is the preferred option; in this case it represents the most flexible option.

Tables 6-9 and 6-10 present the regret tables for the high cost estimates for Urquhart and Winterton respectively. For the high cost estimate in Urquhart, the bioretention option is the most flexible option. For the high cost estimate in Winterton, the infiltration basin option is the most flexible option.

Table 6-9: Regret table for high cost estimates – Urquhart.

Stage 1 intervention	**Total Cost (£)**				
	Markets	**Innovation**	**Austerity**	**Lifestyles**	**Maximum Regret**
	High	**High**	**High**	**High**	
Conventional Solution	123,659	111,246	103,494	111,246	123,659
Infiltration Basin	89,698	58,666	40,710	58,666	89,698
Bioretention System	0	0	0	0	**0**

Table 6-10: Regret table for high cost estimates – Winterton.

Stage 1 intervention	**Total Cost (£)**				
	Markets	**Innovation**	**Austerity**	**Lifestyles**	**Maximum Regret**
	High	**High**	**High**	**High**	
Conventional Solution	139,008	139,008	101,258	139,008	139,008
Infiltration Basin	0	0	0	0	**0**
Bioretention System	149,678	149,678	47,217	47,217	149,678

6.5 Regret under Low Cost Estimates

Tables 6-11 and 6-12 present the regret tables for the low cost estimates for Urquhart and Winterton respectively. Using the low cost estimates, the infiltration basin is identified as the more flexible option in both the Urquhart and Winterton catchments.

Table 6-11: Regret table for low cost estimates – Urquhart.

Stage 1 intervention	**Total Cost (£)**				
	Markets	**Innovation**	**Austerity**	**Lifestyles**	**Maximum Regret**
	Low	**Low**	**Low**	**Low**	
Conventional Solution	142,149	148,784	151,860	148,784	151,860
Infiltration Basin	179	0	0	0	**179**
Bioretention System	0	39,888	45,665	39,888	45,665

Table 6-12: Regret table for low cost estimates – Winterton.

Stage 1 intervention	**Total Cost (£)**				
	Markets	**Innovation**	**Austerity**	**Lifestyles**	**Maximum Regret**
	Low	**Low**	**Low**	**Low**	
Conventional Solution	104,398	104,378	101,258	104,378	104,378
Infiltration Basin	0	0	0	0	**0**
Bioretention System	47,237	47,217	80,601	47,217	80,601

6.6 Regret under Average Cost Estimate

Tables 6-13 and 6-14 present the average total cost and regret tables for the Urquhart. Using the average cost estimates (the mean of the low and high cost estimates), the bioretention option is identified as the most flexible option.

Table 6-13: Average cost estimates – Urquhart.

Stage 1 intervention	Total Cost (£)			
	Markets	Innovation	Austerity	Lifestyles
	Average	Average	Average	Average
Conventional Solution	170,628	120,456	130,918	119,756
Infiltration Basin	82,658	19,774	23,596	19,074
Bioretention System	37,719	20,770	26,073	9,685

Table 6-14: Regret table for average cost estimates – Urquhart.

Stage 1 intervention	Total Cost (£)				
	Markets	Innovation	Austerity	Lifestyles	Maximum Regret
	Average	Average	Average	Average	
Conventional Solution	132,909	100,682	107,332	110,071	132,909
Infiltration Basin	44,939	0	0	9,389	44,939
Bioretention System	0	996	2,477	0	**2,477**

Tables 6-15 and 6-16 present the average total cost and regret tables for Winterton. Using the average cost estimates (the mean of the low and high cost estimates), the infiltration basin option is identified as the most flexible option.

Table 6-15: Average cost estimates – Winterton.

Stage 1 intervention	**Total Cost (£)**			
	Markets	**Innovation**	**Austerity**	**Lifestyles**
	Average	**Average**	**Average**	**Average**
Conventional Solution	127,166	70,619	70,878	67,822
Infiltration Basin	5,463	-51,074	-44,468	-53,871
Bioretention System	103,921	47,374	49,192	44,576

Table 6-16: Regret table for average cost estimates – Winterton.

Stage 1 intervention	**Total Cost (£)**				
	Markets	**Innovation**	**Austerity**	**Lifestyles**	**Maximum Regret**
	Average	**Average**	**Average**	**Average**	
Conventional Solution	23,245	121,698	115,346	121,693	121,698
Infiltration Basin	0	0	0	0	**0**
Bioretention System	98,458	98,448	93,660	98,447	98,458

6.7 Discussion

The minimax-regret principle has been used to examine the relative flexibility of conventional solutions and source-control and regional-control SuDS in two real-world catchments, based on construction costs, operational costs and benefits, and the costs of adaptation to future pressures in 2050 in four transient scenarios. High, low and average cost calculations were used to inform the regret analysis.

In the Winterton catchment, the regional-control SuDS intervention was identified as the most flexible infrastructure across all cost estimates. In the Urquhart catchment, however, the source-control SuDS was identified as the most flexible in both the high and average cost estimates. In the low cost estimate assessment, the regional-control SuDS intervention was the most flexible. This work has verified the consensus that conventional solutions are inflexible and are therefore inappropriate for use in adaptive management approaches.

Figure 6-1 shows that the costs associated with the construction of the infiltration basin in Urquhart were significantly influenced by the cost of installing 300 m of storm sewer to direct flow into the basin. The low cost estimate for storm sewer installation is £123/m; the high cost estimate is £383/m. The use of swales, estimated low cost 14.93/m^2 and estimated high cost £74.66/m^2, as a conveyance structure would have reduced the cost of the infiltration basin by £32,421 in the low cost estimate, £92,502 in the high cost estimate, and £62,462 in the average cost estimate. This would have led to the identification of the regional-control SuDS as the most flexible intervention.

The modelling inputs that have been used to generate the results presented in this Chapter have been selected from values provided in the literature, or created through the use of Perspective theory. For each input, for example, asset deterioration, a value has been selected which has been assumed to be representative of the relative manifestation of that phenomenon within each scenario narrative. However, no sensitivity analysis has been conducted on these values; either local sensitivity analysis, whereby each input value may be perturbed while the rest are held constant, or global sensitivity analysis, whereby the range of input values are modified over their whole range simultaneously. As a result, this work does not provide insight into which are the most critical input data, which could be used by policy makers to ensure that these most important factors are preferably mitigated, or to guide the attention of wastewater undertakers to important developments. However, work such as that conducted by Mark *et al.* (2008), Kleidorfer *et al.* (2014), Urich and Rauch (2014), and Sriwastava *et al.* (2016) stress that model results are most sensitive to changes in rainfall input and urban landform characteristics, and therefore it is considered that there is a significant body of literature that can be relied on to provide this insight. The lack of sensitivity analysis of the presented results to uncertainty in the model input values is mitigated by the use of a scenario analysis framework, whereby the uncertainty is clustered into four conflicting but internally-consistent depictions of possible future states of the world to describe the uncertainty space. Scenario analysis, when coupled with the utilisation of the minimax regret criterion, identifies robust strategies, which are less sensitive to uncertainty and broken assumptions (Lempert *et al.*, 2006; Lempert and Collins, 2007).

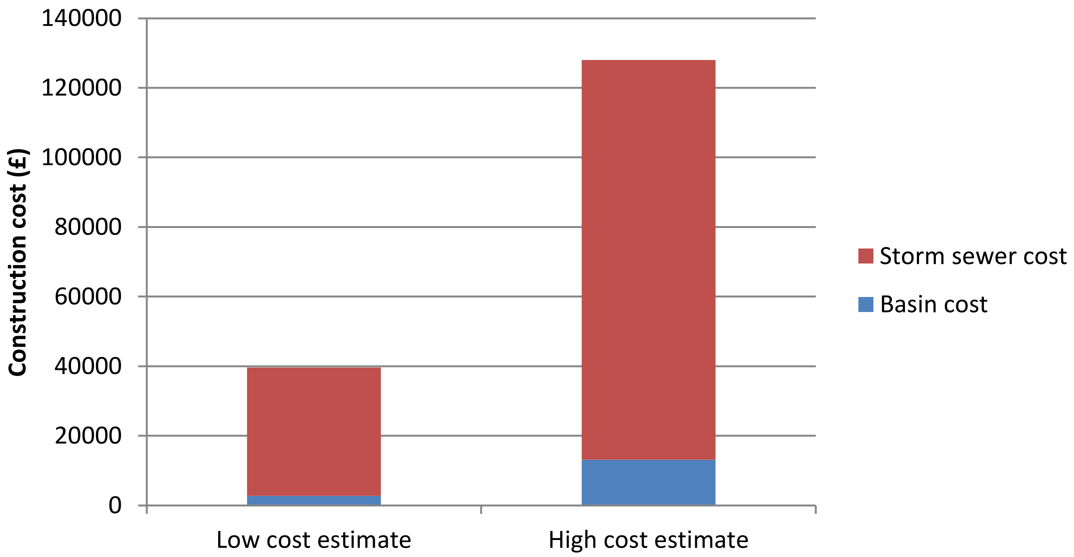


Figure 6-1: Breakdown of construction costs for the infiltration basin in Urquhart.

Complete transient scenario analysis allows for feedback between trends, events, societal values and preferred infrastructure (Haasnoot *et al.*, 2011), and is the only way to ensure the interplay between the unfolding scenario narrative and adaptations through time is explored (Kwakkel, Haasnoot and Walker, 2015). Throughout this study, a key assumption has been made, relating to the boundary of complete transient scenario analysis. This assumption has permeated many of the assessment methods used to generate the results presented in this chapter. Namely, it was assumed that there should be no "meta-feedback" between the unfolding states of the world depicted within the scenario narratives, and the assessment criteria and metrics used within this thesis to assess costs and benefits. As such, the assessment of dynamic scenario narratives has been made via a stationary assessment framework that reflects the values and economy of the present day.

A calculation of the cost of carbon generated recently, in 2015, was used in the assessment. This current carbon price is too low to make a significant impact on the outcome, as identified by Jowitt *et al.* (2012). It was assumed that the financial cost associated with the emission of carbon would remain constant across all scenario narrative, but it could be argued that it would, in fact, vary between scenario narratives. Those scenarios embodying a greater emphasis on environmental protection are likely to associate carbon emissions with a greater financial cost, and vice versa. By the same token, the financial value associated with such environmental benefits as improved air quality may change dependant on the sociological priorities of each scenario narrative. Such changes would reflect diverging societal values.

The financial data that has been used to inform the high and low construction and operation cost estimates has similarly been generated in the recent past, and as such reflects the current understanding of which construction and maintenance activities are required, the methods used to undertake the activities, and the cost of the activities in the present economy. This understanding may develop in the future, and vary across scenarios. For example, basins have been forecast to be the preferable infrastructure for the society characterised by the Fatalistic perspective, in part because of their relatively low construction cost. This low capital cost is a function of the current availability of heavy plant, and the present socio-economic conditions which enables their construction, transport and maintenance. However, for example, the availability of heavy plant in the Fatalistic perspective scenario may become restricted due to reduced availability of capital and suppressed entrepreneurial endeavour. This could cause the capital cost of basins to change from today’s estimations. Likewise, as discussed in Section 4.6.1, the majority of maintenance costs are incurred from inspection, reporting and information management services, and the cost associated with visits is high compared to the costs of most maintenance activities that are undertaken, due to labour rates which are used to calculate the costs of site visits. From the Egalitarian perspective, with its emphasis on community-based self-reliance, more localised management and inspection may cause maintenance costs to fall significantly; indeed, the relative ease with which one may inspect and maintain a single, large basin rather than multiple bioretention systems with sub-surface components could cause a shift in the preferred infrastructure compared to that which has been forecast in this thesis. Similarly, a consistent discount rate, generated in the present, and reflecting the current economic context, has been used across all scenarios. A discount rate represents the rate at which the current value of a unit cost declines through time, and this value could be different in different scenarios.

More generally, it has been assumed that each scenario narrative will continue to translate benefit and cost metrics to financial value. This was required by the SWITCH framework for the assessment of flexibility to enable a comparison between future states of the world, but it may be that case that, for example, the Egalitarian society comes to treasure biodiversity for its own sake, rather than its associated financial value. Additionally, the structure of the assessment framework itself, i.e. which costs and benefits are being measured, reflects the present understanding of important traits associated with stormwater infrastructure. Just as in recent years increased emphasis has been given to the contribution of stormwater infrastructure to carbon and biodiversity, for example, some currently unacknowledged traits may become relevant, or some traits may be neglected based on the values of the future societies.

The extent to which the results presented are sensitive to scenario-based alterations in the assessment framework and mechanisms is unknown. However, conceptually excluding the assessment framework from the development of the scenarios, and the subsequent use of a stationary assessment framework, is typical in flexibility literature (e.g. Eckart *et al.*, 2012; Gersonius *et al*., 2013), and the investigation of this sensitivity was considered outside the scope of this work.

6.8 Chapter Summary

Following application of the transient scenarios to the Urquhart and Winterton catchments in Chapter 5, this chapter has monetised the costs and benefits associated with three types of intervention, conventional solutions, regional-control SuDS, and source-control SuDS, for the period 2016-2050. The minimax-regret principle has been applied to identify which intervention is the most flexible. It was identified that the flexibility of the interventions varied between catchments; in some cases the regional-control SuDS was the most flexible, and in other the source-control SuDS was the most flexible. It was identified that this is due to the high cost associated with stormwater separation, used as part of the design of the regional-control SuDS option in Urquhart. Alternative options exist to direct stormwater to the regional, such as swales or rills; indeed the avoidance of this large cost should influence the distribution of retrofit SuDS.

7 Conclusions

7.1 Introduction

This Chapter summarises the conclusions reached in this thesis, presents the contributions to knowledge, and makes recommendations for future work.

The overall aim of this thesis was to present research that can contribute to the propagation of retrofit SuDS interventions for the improvement of urban drainage network performance in the UK.

The objectives to achieve this aim were:

1. The development of a transient scenario framework that can be used in urban drainage studies;

This objective has been accomplished through the adaptation of Casal-Campos' scenarios, which have previously been applied to urban drainage studies. Transient scenarios require feedback between the environment and the society. The adaptation to transient scenarios was achieved by forecasting the extent of climate change that could be expected in each scenario. Furthermore, each scenario narrative was associated with a "perspective", enabling the preference for stormwater management infrastructure in each scenario to be depicted.

2. The application of the transient scenario framework to examine the relative flexibility of source-control SuDS, regional-control SuDS, and conventional solutions;

The transient scenario framework has been applied in two real-world catchments, Urquhart in the Scottish Highlands, and Winterton in North Lincolnshire, as part of the SWITCH framework for the detailed measurement of flexibility. This was an assessment of the adjustability of the phased design of urban drainage networks. This work has verified the consensus that conventional solutions are inflexible and are therefore inappropriate for use in adaptive management approaches. The relative flexibility of source-control and regional-control SuDS was identified to be sensitive to the method of achieving the separation of stormwater flows. Site- and regional-control SuDS options are likely to provide greater flexibility than either exclusively source-control or regional-control options.

3. The creation of a method to increase the efficiency of retrofit SuDS interventions when the intervention is designed for the improvement of urban drainage network performance metrics.

A method to identify efficient locations to undertake stormwater disconnection has been developed. A technique to apply this method in InfoWorks CS hydraulic modelling software was developed. The application of this method in two real-world catchments led to the identification of more efficient distributions of stormwater disconnection than existing methods allow.

7.2 Chapter 2 Conclusions

Chapter 2 described the problems associated with traditional urban drainage networks and described how SuDS could be used in retrofit to offset these problems, and provided a literature review pertinent to the thesis. From Chapter 2, the following conclusions may be drawn:

1. There is a dichotomy in the design of retrofit SuDS in the UK; academic studies and guidance literature advocate for designs in accordance with the SuDS management train, inherited from the design of SuDS in new developments. However, retrofit SuDS may require the design of solution outside this hierarchy. Wastewater service providers prefer regional-control SuDS.

2. Future pressures are anticipated to degrade the performance of urban drainage networks in coming decades. Retrofit SuDS have been identified as a potential adaptation mechanism to protect legacy urban drainage networks. Adaptive management is an attractive way to handle uncertainty in predictions of future pressures. The fundamental premise of adaptive management is flexibility.

3. From (1.) and (2.), the comparison of conventional solutions, source-control and regional-control SuDS to understand their relative flexibility would be an interesting, useful and novel contribution to knowledge.

4. Existing methods to distribute retrofit SuDS within an urban drainage catchment are likely to lead to sub-optimal results if the SuDS are being installed to achieve the

improvement of some urban drainage performance metric. This is because they use the classical perception of suitability to distribute SuDS. It was identified that there is a clear need for a method to inform stormwater disconnection distribution for the purpose of improving the performance of urban drainage networks

7.3 Chapter 3 Conclusions

Chapter 3 presented the development of a method to distribute stormwater disconnection within an urban drainage network. From Chapter 3, the following conclusions may be drawn:

1. For a stormwater disconnection/retrofit SuDS project that aims to improve the performance of an urban drainage network, reduction of flow rate within the network is a desirable objective because network failure modes are associated with high flow rates

2. The generation of high flow rates within urban drainage networks can be assigned to the concept of areal co-contribution. Areal co-contribution is the concept of potentially disparate areas within an urban drainage catchment possessing similar times of concentration to some point of interest within the network. Stormwater run-off generated in these areas will arrive at the point of interest within the network simultaneously. Maximum areal co-contribution describes the largest collection of impermeable area within an urban drainage catchment that possesses a similar time of concentration to some point of interest within the drainage network.

3. The concept of stormwater disconnection efficiency is introduced. Stormwater disconnection for peak flow reduction is most efficiently undertaken within a location that contributes to the maximum areal co-contribution, where efficiency is measured by the improvement in performance objective metric per unit area disconnected.

4. Understanding the profile of areal-contribution in a catchment is simplified through the use of unit hydrograph methods. InfoWorks CS, hydraulic modelling software used commonly by the UK water industry, has no native capability to produce unit hydrographs. A method to construct disaggregated unit hydrographs within

InfoWorks CS has been developed, using the native pollutant-modelling mechanism as a proxy for a tracer.

5. Testing the maximum areal co-contribution method to distribute stormwater disconnection against a two real-world catchments, it was identified that this method identified priority locations. This resulted in urban drainage performance metrics being improved more efficiently than could have been expected using methods identified in the literature.

7.4 Chapter 4 Conclusions

Chapter 4 presented the development of transient scenarios from an existing scenario framework to enable the comparative assessment of flexibility. From Chapter 4, the following conclusions may be drawn:

1. Transient scenario generation was achieved by forecasting the qualitative extent of future pressures within the four scenario narratives by 2050. Realistic quantified rates of the future pressures identified within the literature were then associated with each scenario. Modelling methods for each pressure were identified.

2. A further aspect of transient scenario analysis is that there is a link between the external (e.g. environmental) pressures, the impact on society, and the actions a society takes. Each scenario depicts a contrasting society. It was therefore necessary to forecast the types of infrastructure that each society would be likely to prefer. This was achieved using perspective theory. The hydraulic and hydrological processes that need to be modelled to appropriately represent each intervention were identified.

3. To provide an objective assessment of the costs and benefits incurred, high and low cost estimates were identified from the literature for the pertinent criteria for retrofit SuDS assessment.

7.5 Chapter 5 Conclusions

Chapter 5 presented the application of transient scenarios to two real-world case study catchments. From Chapter 5, the following conclusions may be drawn:

1. A qualitative process for understanding the expansion of urban area in future scenarios was developed and applied.

7.6 Chapter 6 Conclusions

Chapter 6 presented quantified the costs and benefits accrued by stormwater infrastructure during the transient scenarios, and used a minimax-regret principle to identify the relative flexibility of conventional, retrofit source-control SuDS and retrofit regional-control SuDS. From Chapter 6, the following conclusions may be drawn:

1. Exclusively regional-control SuDS are likely to incur high costs to separate stormwater from combined urban drainage networks.

2. The cost of regional-control interventions can be significantly reduced where separate storm drainage is known to exist, verifying the findings of Singh *et al.*, (2004), or where a cheaper conveyance structures, such as swales, are used to direct flows.

3. This suggests that retrofit SuDS designs that use site- and regional-control SuDS may be more flexible than either exclusively source- or regional-control SuDS designs.

7.7 Contributions to Knowledge

The specific contributions to knowledge made in this thesis are:

1. The concept of stormwater disconnection efficiency, which may be used to define priority stormwater disconnection locations;

2. The development from first principles, testing and application of the maximum areal co-contribution method to identify priority locations for stormwater disconnection in order to improve the performance of an urban drainage network;

3. The generation of transient scenarios for use in urban drainage studies, including the forecasting of preferred stormwater management infrastructure in each scenario;
4. The identification of key criteria to form a multi-criteria assessment framework for interventions that have the objective of improving urban drainage network performance metrics;
5. The identification that the cost, and flexibility, of regional-control retrofit SuDS is highly sensitive to the techniques used to separate stormwater, and that site- and regional-control retrofit SuDS designs may present the most flexible intervention type that could be constructed presently.

7.8 Recommendations for Further Work

Based on the work conducted in this thesis, the following recommendations for further work are made:

1. This thesis isolated the performance of the urban drainage network as the primary motivating factor in the distribution and selection of retrofit SuDS. This may be broadened to encompass the urban water cycle to understand the impact on water provision within each scenario narrative. This may influence the choice of SuDS; for example in a water-stressed scenario rainwater harvesting would be more attractive.
2. Related to this, the scenarios could be expanded to comprise more facets of climate change than simply precipitation intensity; the effect on pipe deterioration as soil conditions change may increase the requirement for SuDS, and infiltration-based SuDS particularly.
3. The precipitation intensity uplift of 20% against which the 2016 interventions were designed resulted in over-designed infrastructure in the Lifestyles scenario narrative. An interesting amendment to work presented in this thesis would be to challenge the convention of using a +20% uplift storm, and to examine how adaptations to the urban drainage network may be required at shorter time intervals to 2050.

4. The scenario parameters used in both the Urquhart and Winterton catchments were identical; it is recommended for future work that the distinctions in local future pressures are assessed.

5. Where SuDS-based adaptations to retrofit SuDS were undertaken, these SuDS were not linked to form a management train. The possibility of developing a management train through periodic adaptation of infrastructure would potentially provide greater holistic benefits, especially where disconnected stormwater run-off is directed to a local water course.

6. The logic used in this thesis locked-in the SuDS used to a very limited range; basins, bioretention systems and tree-planting. A useful addition to this work would be to understand the inherent flexibility of more types of SuDS.

7. In this thesis, rainfall is assumed to fall uniformly across a catchment. The effect on the distribution of stormwater disconnection when considering spatially-variable rainfall profiles is an important piece of future work.

8. It was identified in Section 6.7 that current practice when applying a transient scenario analysis introduces a conceptual boundary, between dynamic scenario narratives and a stationary assessment framework, and the assumption that this assessment framework may be used across all scenarios may prejudice the results. Future studies should investigate the effect of considering “meta-feedback” between the scenario narratives and the assessment framework.

References

Ackers, J. C., Butler, D. and May, R. W. P. (1996) *CIRIA Report 141: Design of sewers to control sediment problems*.

Allan, C., Xia, J. and Pahl-Wostl, C. (2013) 'Climate change and water security: Challenges for adaptive water management', *Current Opinion in Environmental Sustainability*. Elsevier B.V., 5(6), pp. 625–632. doi: 10.1016/j.cosust.2013.09.004.

Allitt, M. and Tewkesbury, A. (2009) 'Investigations into Urban Creep at 5 Cities', in *WaPUG Autumn Conference*.

Anandhi, A., Frei, A., Pierson, D. C., Schneiderman, E. M., Zion, M. S., Lounsbury, D. and Matonse, A. H. (2011) 'Examination of change factor methodologies for climate change impact assessment', *Water Resources Research*, 47(3), pp. 1–10. doi: 10.1029/2010WR009104.

Arnbjerg-Nielsen, K. (2008) 'Quantification of climate change impacts on extreme precipitation used for design of sewer systems', in *11th International Conference on Urban Drainage, 2008*. Edinburgh, Scotland.

ASCE (2012) *Toward a Sustainable Water Future: Visions for 2050*. Edited by W. M. Grayman, D. P. Loucks, and L. Saito. Reston, VI: American Society of Civil Engineers. doi: 10.1061/9780784412077.

Ashley, R., Blanksby, J. and Cashman, A. (2008) *Adaptable Urban Drainage - Addressing Change In Intensity, Occurrence And Uncertainty of Stormwater (AUDACIOUS) Main summary report*.

Ashley, R. and Hopkinson, P. (2002) 'Sewer systems and performance indicators—into the 21st century', *Urban Water*, 4(2), pp. 123–135. doi: 10.1016/s1462-0758(02)00010-9.

Ashley, R. M., Digman, C., Stovin, V. R., Balmforth, D., Glerum, J. and Shaffer, P. (2011) 'Retrofitting surface water management measures : Delivering multiple value', in *12th International Conference on Urban Drainage, September 2011*. Porto Alegre, Brazil.

Ashley, R., Stovin, V., Moore, S., Hurley, L., Lewis, L. and Saul, A. (2010) *London Tideway Tunnels Programme: Thames Tunnel Project Needs Report. Potential source control and SUDS applications: Land use and retrofit options*.

Ashley, R. and Tait, S. (2012) *PREPARED Report No 2012.007: Use of scenarios in PREPARED*.

Van Asselt, M. (2000) *Perspectives on Uncertainty and Risk: The PRIMA Approach to Decision Support*. 1st edn. Springer Netherlands. doi: 10.1007/978-94-017-2583-5.

Atkins (2004) *Scottish Water SUDS Retrofit Research Project*.

Bach, P. M. (2014) *UrbanBEATS – A Virtual Urban Water System Tool for Exploring Strategic Planning Scenarios*. Monash University, Melbourne, Australia.

Bach, P. M., McCarthy, D. T. and Deletic, A. (2013) 'UrbanBEATS: A strategic planning tool for exploring water sensitive futures', in *8th International Water Sensitive Urban Design Conference*. Gold Coast, Australia, pp. 168–175.

Bach, P. M., McCarthy, D. T. and Deletic, A. (2015) 'Can we model the implementation of water sensitive urban design in evolving cities?', *Water Science and Technology*, 71(1), pp. 149–156. doi: 10.2166/wst.2014.464.

Backhaus, A. and Fryd, O. (2012) 'Analyzing the first loop design process for large-scale sustainable urban drainage system retrofits in Copenhagen, Denmark', *Environment and Planning B: Planning and Design*, 39(5), pp. 820–837. doi: 10.1068/b37088.

Badger, M., Stovin, V., Shucksmith, J., McCreath, K., Winter, D. and Brookes, A. (2014) 'Unanswered Questions When Optimising Retrofit SuDS', in *13th International Conference on Urban Drainage, Sarawak, Malaysia, September 2014*.

Bank of England (2017) *Inflation Calculator*. Available at: https://www.bankofengland.co.uk/monetary-policy/inflation (Accessed: 17 November 2017).

Bastien, N., Arthur, S., Scholz, M. and Wallace, S. (2009) 'Towards the best management of SuDS treatment trains', in *13th International Diffuse Pollution Conference*.

Bastien, N., Arthur, S., Wallis, S. and Scholz, M. (2010) 'The best management of SuDS treatment trains: A holistic approach', *Water Science and Technology*, 61(1), pp. 263–272.

doi: 10.2166/wst.2010.806.

Bastien, N. R. P., Arthur, S., Wallis, S. G. and Scholz, M. (2011) 'Runoff infiltration, a desktop case study', *Water Science and Technology*, 63(10), pp. 2300–2308. doi: 10.2166/wst.2011.181.

Berry, C. (2014) *The relationship between economic growth and population growth*, *SPERI British Political Economy*.

BGS (2013) *Suitability of the subsurface for infiltration SuDS in Great Britain*.

BRE (2003) *Digest 365: Soakaway Design*.

te Brömmelstroet, M. and Bertolini, L. (2008) 'Developing land use and transport PSS: Meaningful information through a dialogue between modelers and planners', *Transport Policy*, 15(4), pp. 251–259. doi: 10.1016/j.tranpol.2008.06.001.

Bunn, D. W. and Salo, A. A. (1993) 'Forecasting with scenarios', *European Journal of Operational Research*, 68(3), pp. 291–303. doi: 10.1016/0377-2217(93)90186-Q.

Butler, D. and Davies, J. (2009) *Urban Drainage 3rd Edition*. Spon Press.

Butler, D., Jowitt, P., Ashley, R., Blackwood, D., Davies, J., Oltean-Dumbrava, C., McIlkenny, G., Foxon, T., Gilmour, D., Smith, H., Cavill, S., Leach, M., Pearson, P., Gouda, H., Samson, W., Souter, N., Hendry, S., Moir, J. and Bouchart, F. (2003) 'SWARD: decision support processes for the UK water industry', *Management of Environmental Quality: An International Journal*, 14(4), pp. 444–459. doi: 10.1108/14777830310488676.

Butler, D. and McEntee, B. (2007) 'Impact of climate change on sewer storage tank performance', in *Novatech, June 2007*. Lyon, France, pp. 1131–1138.

Butler, D., Ward, S., Sweetapple, C., Astaraie-Imani, M., Diao, K., Farmani, R. and Fu, G. (2016) 'Reliable, resilient and sustainable water management: the Safe & SuRe approach', *Global Challenges*, 1(1), pp. 63–77. doi: 10.1002/gch2.1010.

Casal-Campos, A. (2016) *Integrated Management of Urban Wastewater Systems: Exploring Reliable, Resilient and Sustainable Strategies for an Uncertain Future*. The University of Exeter.

Casal-Campos, A., Fu, G., Butler, D. and Moore, A. (2015) 'An Integrated Environmental

Assessment of Green and Gray Infrastructure Strategies for Robust Decision Making', *Environmental Science and Technology*, 49(14), pp. 8307–8314. doi: 10.1021/es506144f.

Center for Neighborhood Technology (2010) *The Value of Green Infrastructure*. Chicago, IL. doi: 10.1002/9780470750506.

Chai, S. and Wildavsky, A. (1994) 'Culture, rationality and violence', in Coyle, D. and Ellis, R. (eds) *Politics, Policy and Culture: Applications of Cultural Theory*. Boulder, CO: Westview Press, pp. 159–174.

Chocat, B., Ashley, R., Marsalek, J., Matos, M. R., Rauch, W., Schilling, W. and Urbonas, B. (2004) 'Urban Drainage - Out-of-sight-out-of-mind?', in *Novatech 2004*. Lyon, France, pp. 1659–1690.

Chow, J., Savic, D., Fortune, D., Kapelan, Z. and Mebrate, N. (2013) 'Translating Legislative Requirements and Best Practice Guidance into a Systematic, Multi-Criteria Decision Support Framework for Effective Sustainable Drainage Design Evaluation', in *Proceedings of 2013 IAHR World Congress*.

CIRIA (2006) *CIRIA C635: Designing for exceedance in urban drainage - good practice*.

Colombo, A. F. and Byer, P. H. (2012) 'Adaptation, flexibility and project decision-making with climate change uncertainties', *Impact Assessment and Project Appraisal*, 30(4), pp. 229–241. doi: 10.1080/14615517.2012.731189.

Coombes, P. J., Kuczera, G., Kalma, J. D. and Argue, J. R. (2002) 'An evaluation of the benefits of source control measures at the regional scale', *Urban Water*, 4(4), pp. 307–320. doi: 10.1016/S1462-0758(02)00028-6.

DayWater (2005) *An Adaptive Decision Support System (ADSS) for the Integration of Stormwater Source Control into Sustainable Urban Water Management Strategies*. Available at: http://daywater.in2p3.fr/EN/ (Accessed: 20 August 2013).

DECC (2015) *Updated short-term traded carbon values used for modelling purposes*.

DEFRA (2006) *Flood and Coastal Defence Appraisal Guidance FCDPAG3 Economic Appraisal Supplementary Note to Operating Authorities – Climate Change Impacts*.

DEFRA (2008) *Future Water: The Government's water strategy for England*.

DEFRA (2009) *UK Climate Projections: Adapting to climate change*.

DEFRA (2011) *National Build Standards: Design and construction of new gravity foul sewers and lateral drains*.

DEFRA (2012) *Waste water treatment in the United Kingdom*.

DEFRA (2013) *Air Quality: Economic Analysis*.

Deletic, A., Dotto, C. B. S., McCarthy, D. T., Kleidorfer, M., Freni, G., Mannina, G., Uhl, M., Henrichs, M., Fletcher, T. D., Rauch, W., Bertrand-Krajewski, J. L. and Tait, S. (2012) ‘Assessing uncertainties in urban drainage models’, *Physics and Chemistry of the Earth*. Elsevier Ltd, 42–44, pp. 3–10. doi: 10.1016/j.pce.2011.04.007.

Demuzere, M., Faehnle, M. and Orru, K. (2014) ‘Evidence on the contribution of green urban infrastructure to climate change mitigation and adaptation’, in *Understanding Cities: Advances in integrated assessment of Urban Sustainability*. Newcastle, pp. 13–40.

deNeufville, R. (2003) ‘Real Options: Dealing With Uncertainty in Systems Planning and Design’, *Integrated Assessment*, 4(1), pp. 26–34. doi: 10.1076/iaij.4.1.26.16461.

Dessai, S., Hulme, M., Lempert, R. and Pielke Jr., R. (2009) ‘Climate prediction: a limit to adaptation?’, in Neil Adger, W., Lorenzoni, I., and O’Brien, K. (eds) *Adapting to Climate Change: Thresholds, Values, Governance*. Cambridge University Press, pp. 64–78. doi: http://dx.doi.org/10.1017/CBO9780511596667.006.

Dewar, J. A. (2002) *Assumption-Based Planning: A Tool for Reducting Avoidable Surprises*. Cambrige University Press.

Difrancesco, K. N. and Tullos, D. D. (2015) ‘Assessment of flood management systems’ flexibility with application to the Sacramento River basin, California, USA’, *International Journal of River Basin Management*, 13(3), pp. 271–284. doi: 10.1080/15715124.2014.917316.

Digman, C., Ashley, R., Balmforth, D., Stovin, V., Balmforth, D. and Glerum, J. (2012) *CIRIA C713: Retrofitting to manage surface water*.

Digman, C. J., Horton, B., Ashley, R. M. and Gill, E. (2015) *CIRIA Report W045c: BeST (Benefits of SuDS Tool) Technical Guidance*.

Dong, X., Guo, H. and Zeng, S. (2017) 'Enhancing future resilience in urban drainage system: Green versus grey infrastructure', *Water Research*. Elsevier Ltd, 124, pp. 280–289. doi: 10.1016/j.watres.2017.07.038.

Dotto, C. B. S., Mannina, G., Kleidorfer, M., Vezzaro, L., Henrichs, M., McCarthy, D. T., Freni, G., Rauch, W. and Deletic, A. (2012) 'Comparison of different uncertainty techniques in urban stormwater quantity and quality modelling', *Water Research*. Elsevier Ltd, 46(8), pp. 2545–2558. doi: 10.1016/j.watres.2012.02.009.

Douglas, M. (1970) *Natural Symbols: Explorations is Cosmology*. Barrie & Rockliff.

Douglas, M. (1982) *In the Active Voice*. Routledge.

Eckart, J., Sieker, H., Vairavamoorthy, K. and Alsharif, K. (2012) 'Flexible design of urban drainage systems: Demand led research for Hamburg-Wilhelmsburg', *Reviews in Environmental Science and Biotechnology*, 11(1), pp. 5–10. doi: 10.1007/s11157-011-9256-5.

Eckart, J., Tsegaye, S. and Vairavamoorthy, K. (2013) *SWITCH Report: Measuring the Flexibility of Urban Drainage Systems*.

eftec (2010) *Flood and Coastal Erosion Risk Management: Economic Valuation of Environment Effects. HANDBOOK for the Environment Agency for England and Wales*.

Ekström, M., Fowler, H. J., Kilsby, C. G. and Jones, P. D. (2005) 'New estimates of future changes in extreme rainfall across the UK using regional climate model integrations. 2. Future estimates and use in impact studies', *Journal of Hydrology*, 300(1–4), pp. 234–251. doi: 10.1016/j.jhydrol.2004.06.019.

Ellis, J. B., Deutsch, J.-C., Legret, M., Martin, C., Revitt, D. M., Scholes, L. N. L., Seiker, H. and Zimmerman, U. (2006) 'The DayWater decision support approach to the selection of sustainable drainage systems.', *Water Practice & Technology*, 1(1). doi: 10.2166/WPT.2006002.

Ellis, J. B., Deutsch, J.-C., Legret, M., Martin, C., Revitt, D. M., Scholes, L. N. L., Seiker, H. and Zimmerman, U. (2011) 'An Integrated Decision Support Approach to the Selection of Sustainable Urban Drainage Systems (SUDS)', in *SWITCH Conference: The Future of Urban Water; Solutions for Livable and Resilient Cities, Paris, 2011*. doi: 10.2166/WPT.2006002.

Ellis, J. B. and Viavattene, C. (2014) 'Sustainable urban drainage system modeling for managing urban surface water flood risk', *Clean - Soil, Air, Water*, 42(2), pp. 153–159. doi: 10.1002/clen.201300225.

Ellis, R. and Coyle, D. (1994) *Politics, Policy and Culture: Applications of Cutural Theory*. Boulder, CO: Westview Press.

Environment Agency (2010) *Water: Planning ahead for an uncertain future. Water in the 2100s. Briefing note GEHO0811BSMM-E-E*.

Environment Agency (2013) *Water Appraisal Guidance; Assessing Costs and Benefits for River Basin Management Planning*.

Environment Agency (2015) *Cost estimation for SUDS - summary of evidence*.

Environment Agency (2016) *EA 2016 Flood risk assessments climate change allowances*.

European Commission (1991) *Directive 91/271/EEC: Urban Waste Water Treatment Directive*.

European Commission (2000) *Directive 2000/60/EU: The Water Framework Directive*.

Evans, E. P., Ashley, R. and Hall, J. W. (2004) *Foresight Flood and Coastal Defence Project: Scientific Summary*. doi: 10.1117/2.5200408.0012.

Evans, E. P., Simm, J. D., Thorne, C. R., Arnell, N. W., Ashley, R. M., Hess, T. M., Lane, S. N., Morris, J., Nicholls, R. J., Penning-Rowsell, E. C., Reynard, N. S., Saul, A. J., Tapsell, S. M., Watkinson, A. R. and Wheater, H. S. (2004) *Appendix A, Foresight; Future Flooding. Scientific Summary: Volume I.*

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., Mikkelsen, P. S., Rivard, G., Uhl, M., Dagenais, D. and Viklander, M. (2015) 'SUDS, LID, BMPs, WSUD and more – The evolution and application of terminology surrounding urban drainage', *Urban Water Journal*, 12(7), pp. 525–542. doi: 10.1080/1573062X.2014.916314.

Foster, J., Lowe, A. and Winkelman, S. (2011) *The Value of Green Infrastructure for Urban Climate Adaptation*. Washington, D.C.

Foxon, T. J. (2000) 'A multi-criteria analysis approach to the assessment of sustainability of urban water systems', in *15th European junior scientist workshop, Stavoren, The Netherlands, 11–14 May 2000*.

Georges, K., Thornton, A. and Sadler, R. (2009) *Environment Agency Report: SC070010/R2: Transforming wastewater treatment to reduce carbon emissions*.

Gersonius, B., Ashley, R., Jeuken, A., Pathinara, A. and Zevenbergen, C. (2015) 'Accounting for uncertainty and flexibility in flood risk management: Comparing real-in-options optimisation and adaptation tipping points', *Journal of Flood Risk Management*, 8(2), pp. 135–144. doi: 10.1111/jfr3.12083.

Gersonius, B., Ashley, R., Pathirana, A. and Zevenbergen, C. (2013) 'Climate change uncertainty: Building flexibility into water and flood risk infrastructure', *Climatic Change*, 116(2), pp. 411–423. doi: 10.1007/s10584-012-0494-5.

Gersonius, B., Nasruddin, F., Ashley, R., Jeuken, A., Pathirana, A. and Zevenbergen, C. (2012) 'Developing the evidence base for mainstreaming adaptation of stormwater systems to climate change', *Water Research*. Elsevier Ltd, 46(20), pp. 6824–6835. doi: 10.1016/j.watres.2012.03.060.

Gordon-Walker, S., Harle, T. and Naismith, I. (2007) *Environment Agency Report SC060024: Cost-benefit of SUDS Retrofit in Urban Areas, Science Report – SC060024*.

Haasnoot, M., Kwakkel, J. H., Walker, W. E. and ter Maat, J. (2013) 'Dynamic adaptive policy pathways: A method for crafting robust decisions for a deeply uncertain world', *Global Environmental Change*. Elsevier Ltd, 23(2), pp. 485–498. doi: 10.1016/j.gloenvcha.2012.12.006.

Haasnoot, M., Middelkoop, H., Beek, E. Van and Deursen, W. P. A. Van (2011) 'A method to develop management strategies for an uncertain future', *Sustainable Development*, 381(November 2009), pp. 369–381. doi: 10.1002/sd.

Haasnoot, M., Middelkoop, H., Offermans, A., van Beek, E. and van Deursen, W. P. A. (2012) 'Exploring pathways for sustainable water management in river deltas in a changing environment', *Climatic Change*, 115(3–4), pp. 795–819. doi: 10.1007/s10584-012-0444-2.

Hall, J. W., Lempert, R. J., Keller, K., Hackbarth, A., Mijere, C. and Mcinerney, D. J. (2012) 'Robust Climate Policies Under Uncertainty: A Comparison of Robust Decision Making and

Info-Gap Methods', *Risk Analysis*, 32(10), pp. 1657–1672. doi: 10.1111/j.1539-6924.2012.01802.x.

Hammond, G. and Jones, C. (2011) *Embodied Carbon: The Inventory of Carbon and Energy (ICE)*. Edited by F. Lowrie and P. Tse. Bath. doi: 10.1680/ener.2011.164.4.206.

Hammond, G. P. and Jones, C. I. (2008) 'Embodied energy and carbon in construction materials', *Proceedings of the Institution of Civil Engineers*, Energy 161(EN2), pp. 87–98. doi: 10.1680/ener.2008.161.2.87.

Hocke, S. (2004) *Flexibilitätsmanagement in der Logistik – Systemtheoretische Fundierung und Simulation logistischer Gestaltungsparameter*. Bayreuth.

Holland, D. (1967) 'Rain intensity-frequency relationship in Britain', *British Rainfall 1961*. London, Part III, pp. 43–51.

Holling, C. S. (1973) 'Resilience and Stability of Ecological Systems', *Annu.Rev.Ecol.Syst.*, 4, pp. 1–23. doi: 10.1146/annurev.es.04.110173.000245.

Holling, C. S. (1978) *Adaptive Environmental Assessment and Management*. New York: John Wiley & Sons.

Horton, R. E. (1940) 'The infiltration-theory of surface-runoff', *Eos, Transactions American Geophysical Union*, 21(2), p. 541. doi: 10.1029/TR021i002p00541-1.

Houdeshel, C., Pomeroy, C., Hair, L. and Moeller, J. (2011) 'Cost-Estimating Tools for Low-Impact Development Best Management Practices: Challenges, Limitations, and Implications', *Journal of Irrigation and Drainage Engineering*, 3(4), pp. 183–189. doi: 10.1061/(ASCE)IR.1943-4774.0000179.

HR Wallingford (2004a) *Report SR 627: Whole Life Costing for Sustainable Drainage*.

HR Wallingford (2004b) *The Operation and Maintenance of Sustainable Drainage Systems (and Associated Costs), SR 626, HR Wallingford, UK*. doi: 10.1017/CBO9781107415324.004.

Hyder Consulting (2004) *Retrofitting Sustainable Urban Drainage Systems. Case Study - Dunfermline*.

IPCC (2007) *Climate Change 2007 Synthesis Report*, *Intergovernmental Panel on Climate*

Change [Core Writing Team IPCC. doi: 10.1256/004316502320517344.

IPCC (2014a) 'Climate Change, Adaptation, and Vulnerability', *Organization & Environment*, 24(March), pp. 1–44. doi: http://ipcc-wg2.gov/AR5/images/uploads/IPCC_WG2AR5_SPM_Approved.pdf.

IPCC (2014b) *Summary for Policymakers, Climate Change 2014: Synthesis Report. Contribution of Working Groups I, II and III to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change*. doi: 10.1017/CBO9781107415324.

Irish Water (2015) *National Paediatric Hospital Project: Development Impact Assessment*.

Jeuken, A. and Reeder, T. (2011) 'Short-term decision making and long-term strategies: How to adapt to uncertain climate change', *Water Governance,* 1, pp. 29–35.

Jowitt, P., Johnson, A., Moir, S. and Grenfell, R. (2012) 'A protocol for carbon emissions accounting in infrastructure decisions', *Proceedings of the Institution of Civil Engineers - Energy*, (165), pp. 89–95.

Juan-García, P., Butler, D., Comas, J., Darch, G., Sweetapple, C., Thornton, A. and Corominas, L. (2017) 'Resilience theory incorporated into urban wastewater systems management. State of the art', *Water Research*. Elsevier Ltd, 115, pp. 149–161. doi: 10.1016/j.watres.2017.02.047.

Kazmierczak, A. and Carter, J. (2010) *Adaptation to climate change using green and blue infrastructure. A database of case studies.*

Keefe, R. (2012) *Reconsidering California Transport Policies: Reducing Greenhouse Gas Emissions in an Uncertain Future*. Pardee RAND Graduate School.

Kellagher, R. (1981) *The Wallingford Procedure - for design and analysis of urban storm drainage*.

Kleidorfer, M., Mikovits, C., Jasper-Tönnies, A., Huttenlau, M., Einfalt, T. and Rauch, W. (2014) 'Impact of a changing environment on drainage system performance', *Procedia Engineering*. Elsevier B.V., 70(2009), pp. 943–950. doi: 10.1016/j.proeng.2014.02.105.

Kleidorfer, M., Möderl, M., Sitzenfrei, R., Urich, C. and Rauch, W. (2009) 'A case independent approach on the impact of climate change effects on combined sewer system performance', *Water Science and Technology*, 60(6), pp. 1555–1564. doi: 10.2166/wst.2009.520.

Koste, L. and Malhotra, M. (1999) 'A theoretical framework for analyzing the dimensions of manufacturing flexibility', *Jounral of Operation Management*, 18, pp. 75–93.

Kuller, M., Bach, P. M., Ramirez-Lovering, D. and Deletic, A. (2017) 'Framing water sensitive urban design as part of the urban form: A critical review of tools for best planning practice', *Environmental Modelling & Software*. Elsevier Ltd, 96, pp. 265–282. doi: 10.1016/j.envsoft.2017.07.003.

Kundzewicz, Z. (2002) 'Non-structural Flood Protection and Sustainability', in *Proceedings of the International Hydrological Programme International Workshop, London, Ontario, Canada, 18-20 October 2001*, pp. 3–13.

Kwadijk, J. C. J., Haasnoot, M., Mulder, J. P. M., Hoogvliet, M. M. C., Jeuken, A. B. M., van der Krogt, R. A. A., van Oostrom, N. G. C., Schelfhout, H. A., van Velzen, E. H., van Waveren, H. and de Wit, M. J. M. (2010) 'Using adaptation tipping points to prepare for climate change and sea level rise: a case study in the Netherlands', *Wiley Interdisciplinary Reviews: Climate Change*, 1(5), pp. 729–740. doi: 10.1002/wcc.64.

Kwakkel, J. H., Haasnoot, M. and Walker, W. E. (2015) 'Developing dynamic adaptive policy pathways: a computer-assisted approach for developing adaptive strategies for a deeply uncertain world', *Climatic Change*, 132(3), pp. 373–386. doi: 10.1007/s10584-014-1210-4.

Lee, J. G., Selvakumar, A., Alvi, K., Riverson, J., Zhen, J. X., Shoemaker, L. and Lai, F. hsiung (2012) 'A watershed-scale design optimization model for stormwater best management practices', *Environmental Modelling and Software*. Elsevier Ltd, 37, pp. 6–18. doi: 10.1016/j.envsoft.2012.04.011.

Lee, K. (1993) *Compass and Gyroscope: Integrating Science and Politics for the Environment*. Washington, D.C.: Island Press.

Lee, K. (1999) 'Appraising adaptive management', *Conservation Ecology*, 3(2).

Lempert, R. J. (2003) *Shaping the Next One Hundred Years: New Methods for Quantitative, Long-Term Policy Analysis*. 1st edn. RAND. doi: 10.1016/j.techfore.2003.09.006.

Lempert, R. J. and Collins, M. . (2007) ‘Managing the risk of uncertain threshold responses: Comparison of robust, optimum, and precautionary approaches’, *Risk Analysis*, 27(4), pp. 1009–1026.

Lempert, R. J., Groves, D. G., Popper, S. W. and Bankes, S. C. (2006) ‘A General, Analytic Method for Generating Robust Strategies and Narrative Scenarios’, *Management Science*, 52(4), pp. 514–528. doi: 10.1287/mnsc.1050.0472.

Makropoulos, C. K., Natsis, K., Liu, S., Mittas, K. and Butler, D. (2008) ‘Decision support for sustainable option selection in integrated urban water management’, *Environmental Modelling and Software*, 23(12), pp. 1448–1460. doi: 10.1016/j.envsoft.2008.04.010.

Manocha, N. and Babovic, V. (2017) ‘Development and valuation of adaptation pathways for storm water management infrastructure’, *Environmental Science and Policy*. Elsevier, 77(1), pp. 86–97. doi: 10.1016/j.envsci.2017.08.001.

Mark, O., Svensson, G., König, A. and Linde, J. J. (2008) ‘Analyses and Adaptation of Climate Change Impacts on Urban Drainage Systems’, *11th Conference on Urban Drainage*, pp. 1–10.

Marlow, D. R., Moglia, M., Cook, S. and Beale, D. J. (2013) ‘Towards sustainable urban water management: A critical reassessment’, *Water Research*. Elsevier Ltd, 47(20), pp. 7150–7161. doi: 10.1016/j.watres.2013.07.046.

Marsalek, J. and Chocat, B. (2002) ‘International report: Stormwater management’, *Water Science and Technology*, 46(6–7), pp. 1–17.

Martelli, A. (2014) *Models of Scenario Building and Planning: Facing Uncertainty and Complexity*. 1st edn. Palgrave Macmillan UK. doi: 10.1057/9781137293503.

McPherson, E. ., Maco, S. ., Simpson, J. ., Peper, P. ., Xiao, Q., van der Zanden, A. . and Bell, N. (2002) *Western Washington and Oregon Community Tree Guide: Benefits, Costs and Strategic Planting*.

Medellin-Azuara, J., Mendoza-Espinosa, L., Lund, J. and Ramirez-Acosta, R. (2007) ‘The application of economic-engineering optimisation for water management in Ensenada, Baja California, Mexico’, *Water Science & Technology*, 55(1–2), pp. 339–347. doi: 10.2166/wst.2007.038.

Mell, I. ., Henneberry, J., Hehl-Lange, S. and Keskin, B. (2013) ‘Promoting urban greening: Valuing the development of green infrastructure investments in the urban core of Manchester, UK’, *Urban Forestry & Urban Greening*, 12(3), pp. 296–306.

Milly, P. C. D., Betancourt, J., Falkenmark, M., Hirsch, R. M., Kundzewicz, Z. W., Lettenmaier, D. P. and Stouffer, R. J. (2008) ‘Climate change. Stationarity is dead: whither water management?’, *Science (New York, N.Y.)*, 319(5863), pp. 573–574. doi: 10.1126/science.1151915.

Moddemeyer, S. (2015) ‘Sustainability is dead; long live sustainability’, *Water 21*, pp. 12–14.

Moore, S. L., Stovin, V. R., Wall, M. and Ashley, R. M. (2012) ‘A GIS-based methodology for selecting stormwater disconnection opportunities’, *Water Science and Technology*, 66(2), pp. 275–283. doi: 10.2166/wst.2012.172.

Moore, T. L. C. and Hunt, W. F. (2013) ‘Predicting the carbon footprint of urban stormwater infrastructure’, *Ecological Engineering*. Elsevier B.V., 58, pp. 44–51. doi: 10.1016/j.ecoleng.2013.06.021.

Mugume, S. N., Gomez, D. E., Fu, G., Farmani, R. and Butler, D. (2015) ‘A global analysis approach for investigating structural resilience in urban drainage systems’, *Water Research*. Elsevier Ltd, 81, pp. 15–26. doi: 10.1016/j.watres.2015.05.030.

Murphy, J., Sexton, D., Jenkins, G., Boorman, P., Booth, B., Brown, K., Clark, R., Collins, M., Harris, G. and Kendon, L. (2010) *UK Climate Projections science report: Climate change projections*. doi: 10.1787/9789264086876-5-en.

MWH (2014) *Demonstrating the multiple benefits of SuDS - a business case*.

Myers, S. C. (1984) ‘Finance Theory and Financial Strategy’, *Interfaces*, 14(1), pp. 126–137. doi: 10.1287/inte.14.1.126.

Nakicenovic, N., Davidson, O., Davis, G., Grubler, A. and Kram, T. (2000) *Summary for Policymakers: Emissions Scenarios. A Special Report of Working Group III of the Intergovernmental Panel on Climate Change*, *IPCC*. doi: 92-9169-113-5.

NERC (1975) *Flood Studies Report*.

Offermans, A., Haasnoot, M. and Valkering, P. (2011) ‘A method to explore social response for sustainable water management strategies under changing conditions', *Sustainable Development*, 19(5), pp. 312–324. doi: 10.1002/sd.439.

OFWAT (2010) *Future water and sewerage charges 2010-15: final determinations*.

OFWAT (2011a) *Comparing the Arrangements for the Management of Surface Water in England and Wales to Arrangements in Other Countries Final Report*.

OFWAT (2011b) *Future Impacts on Sewer Systems in England and Wales Summary of a Hydraulic Modelling Exercise Reviewing the Impact of Climate Change, Population and Growth in Impermeable Areas up to Around 2040*.

Osborn, T. J., Hulme, M., Jones, P. D. and Basnett, T. A. (2000) 'Observed trends in the daily intensity of United Kingdom precipitation', *International Journal of Climatology*, 20(4), pp. 347–364. doi: 10.1002/(SICI)1097-0088(20000330)20:4<347::AID-JOC475>3.0.CO;2-C.

OST (2002) *Foresight Futures 2020: Revised Scenarios and Guidance. Environmental Futures.*

Pahl-Wostl, C. (2007) 'Transitions towards adaptive management of water facing climate and global change', *Water Resources Management*, 21(1), pp. 49–62. doi: 10.1007/s11269-006-9040-4.

Peat, M., Moon, K., Dyer, F., Johnson, W. and Nichols, S. J. (2017) 'Creating institutional flexibility for adaptive water management: insights from two management agencies', *Journal of Environmental Management*. Elsevier Ltd, 202, pp. 188–197. doi: 10.1016/j.jenvman.2017.06.059.

Peters, C., Sieker, H., Jin, Z. and Eckart, J. (2010) *Sustainable Water Management in the City of the Future - Deliverable 2.1.4: Assessing future uncertainties associated with urban drainage using flexible systems – the COFAS method and tool*.

Philadelphia Water Department (2017) *Green City Clean Waters: The City of Philadelphia's Program for Combined Sewer Overflow Control - A Long Term Control Plan Update*. Philadelphia.

Pittner, C. and Allerton, G. (2010) *SUDS for Roads Whole Life Cost and Carbon Tool*. Edinburgh, Scotland.

POST (2007) *Urban Flooding Postnote 289*.

Prudhomme, C., Jakob, D. and Svensson, C. (2003) ‘Uncertainty and climate change impact on the flood regime of small UK catchments’, *Journal of Hydrology*, 277(1–2), pp. 1–23. doi: 10.1016/S0022-1694(03)00065-9.

Reynolds, J. H. (2000) ‘Maintenance of sewers and water mains within the concept of serviceability to customers’, *Pipes and Pipelines International*, 45(2), pp. 5–10.

Robert Bray Associates (2012) *Ashby Grove Raingarden: SUDS Design Statement*.

Royal Haskoning (2012) *Costs and Benefits of Sustainable Drainage Systems*.

Salminen, P., Hokkanen, J. and Lahdelma, R. (1998) ‘Comparing multicriteria methods in the context of environmental problems’, *European Journal of Operational Research*, 104(3), pp. 485–496. doi: 10.1016/S0377-2217(96)00370-0.

Saltelli, A., Ratto, M., Tarantola, S. and Campolongo, F. (2006) ‘Sensitivity analysis practices: Strategies for model-based inference.’, *Reliability Engineering and System Safety*, 91((10-11)), pp. 1109–1125. doi: 10.1016/j.watres.2010.05.001.

Scholes, L., Revitt, D. M. and Ellis, J. B. (2008) ‘A systematic approach for the comparative assessment of stormwater pollutant removal potentials’, *Journal of Environmental Management*, 88(3), pp. 467–478. doi: 10.1016/j.jenvman.2007.03.003.

Scholz, M. (2006) ‘Decision-support tools for sustainable drainage’, *Proceedings of the Institution of Civil Engineers*, Engineerin(ES1), pp. 117–125. doi: 10.1680/ensu.2006.159.3.117.

Scholz, M. (2015) ‘Sustainable Drainage Systems’, *Water*, 7(5), pp. 2272–2274. doi: 10.3390/w7052272.

Schueler, T., Hirschman, D., Novotney, M. and Zielinski, J. (2014) *Center for Watershed Protection Manual 3: Urban Stormwater Retrofit Practices*. Washington, D.C.

Scottish Water (2015) *Sewers for Scotland*. 3rd edn.

Seattle Government (2009) *Street Edge Alternatives (SEA) Project*. Available at: http://www.seattle.gov/util/About_SPU/Drainage_&_Sewer_System/Natural_Drainage_Systems/Str eet_Edge_Alternatives/SPU_001805.asp (Accessed: 12 May 2013).

Semadeni-Davies, A., Hernebring, C., Svensson, G. and Gustafsson, L. G. (2008) 'The impacts of climate change and urbanisation on drainage in Helsingborg, Sweden: Combined sewer system', *Journal of Hydrology*, 350(1–2), pp. 100–113. doi: 10.1016/j.jhydrol.2007.05.028.

Shaw, E. M. (1998) 'Unit hydrograph method in UK flood studies', in *Encyclopedia of Hydrology and Lakes*, pp. 659–660.

Sieker, H., Helm, B., Krebs, P., Schlottmann, P. and Tränkner, J. (2008) 'Flexibility - a planning criterion for stormwater management', in *11th International Conference on Urban Drainage, Edinburgh, Scotland, 2008*.

Singh, R., Jefferies, C., Stovin, V., Morrison, G., Gillon, S. and Parks, C. (2005) 'Developing a planning and design framework for retrofit SUDS', in *10th International Conference on Urban Drainage, Copenhagen, Denmark, 2005*.

Smith, K. W. S., Jefferies, C. and Blackwood, D. J. (2011) 'Removal of surface water from combined sewers applying the SWITCH Transitioning approach', in *12th International Conference on Urban Drainage, Porto Alegre, Brazil, September 2011*.

SNIFFER (2006) *SNIFFER Project UE3(04)UW5: Retrofitting Sustainable Urban Water Solutions*.

Song, X., Zhang, J., Zhan, C., Xuan, Y., Ye, M. and Xu, C. (2015) 'Global sensitivity analysis in hydrological modelling: Review of concepts, methods, theoretical framwork, and applications.', *Journal of Hydrology*, 523(225), pp. 739–757. doi: 10.1016/j.hydrol.2015.02.013.

Spiller, M., Vreeburg, J. H. G., Leusbrock, I. and Zeeman, G. (2015) 'Flexible design in water and wastewater engineering - Definitions, literature and decision guide', *Journal of Environmental Management*, 149, pp. 271–281. doi: 10.1016/j.jenvman.2014.09.031.

Sriwastava, A., Tait, S., Schellart, A., Van Dorpe, M., Kroll, S. and Shucksmith, J. (2016) 'Simulation of sewer overflow volume in Flanders using a global sensitivity analysis', in *8th International Conference on Sewer Processes and Networks*, pp. 299–306.

Stapleton, C. M., Wyer, M. D., Crowther, J., McDonald, A. T., Kay, D., Greaves, J., Wither, A., Watkins, J., Francis, C., Humphrey, N. and Bradford, M. (2008) 'Quantitative catchment profiling to apportion faecal indicator organism budgets for the Ribble system, the UK's sentinel drainage basin for Water Framework Directive research', *Journal of Environmental Management*, 87, pp. 535–550.

Stovin, V. R., Moore, S. L., Wall, M. and Ashley, R. M. (2013) 'The potential to retrofit sustainable drainage systems to address combined sewer overflow discharges in the Thames Tideway catchment', *Water and Environment Journal*, 27(2), pp. 216–228. doi: 10.1111/j.1747-6593.2012.00353.x.

Stovin, V. R. and Swan, A. D. (2007) 'Retrofit SuDS—cost estimates and decision-support tools', *Proceedings of the Institution of Civil Engineers - Water Management*, 160(4), pp. 207–214. doi: 10.1680/wama.2007.160.4.207.

Stovin, V., Swan, A. and Moore, S. (2007) *Retrofit SUDS for Urban Water Quality Enhancement Final Report – May 2007*.

Swan, A. (2003) *A decision-support system for the design of retrofit sustainable urban drainage systems (SUDS)*. The University of Sheffield, Sheffield, UK.

Thompson, M., Ellis, R. and Wildavsky, A. (1990) *Cultural Theory*. Boulder, CO: Westview Press.

Thompson, M. and Wildavsky, A. (1986) 'A Cultural Theory of Information Bias in Organizations', *Journal of Management Studies*, 23(3), pp. 273–286. doi: 10.1111/j.1467-6486.1986.tb00954.x.

Tran, H. T., Balchanos, M., Domerçant, J. C. and Mavris, D. N. (2017) 'A framework for the quantitative assessment of performance-based system resilience', *Reliability Engineering and System Safety*. Elsevier, 158(October 2016), pp. 73–84. doi: 10.1016/j.ress.2016.10.014.

Trees & Design Action Group (2014) *Trees in Hard Landscapes A Guide for Delivery*.

Tscheikner-Gratl, F., Mikovits, C., Rauch, W. and Kleidorfer, M. (2014) 'Adaptation of sewer networks using integrated rehabilitation management', *Water Science and Technology*, 70(11), pp. 1847–1856. doi: 10.2166/wst.2014.353.

UK Government (2010) *The CRC Energy Efficiency Scheme Order*.

UK Government (2011) *Climate Resilient Infrastructure : Preparing for a Changing Climate*.

UK NEA (2011) *UK National Ecosystem Assessment*.

UKWIR (2009a) *UKWIR Report 09/WM/07/13: Exploring the Cost Benefit of Separating Direct Surface Water Inputs from the Combined Sewerage System*.

UKWIR (2009b) *UKWIR Report 10/WM/07/14: Impact of Urban Creep on Sewerage Systems*.

UKWIR (2010) *UKWIR Report 10/CL/11/3: Energy Efficiency in the Water Industry: A Compendium of Best Practices and Case Studies*.

Upton, D. M. (1994) 'The management of manufacturing flexibility', *California Management Review*, 36(2), pp. 72–89. doi: 10.2307/41165745.

Urich, C., Bach, P. M., Sitzenfrei, R., Kleidorfer, M., McCarthy, D. T., Deletic, A. and Rauch, W. (2013) 'Modelling cities and water infrastructure dynamics', *Proceedings of the Institution of Civil Engineers - Engineering Sustainability*, 166(5), pp. 301–308. doi: 10.1680/ensu.12.00037.

Urich, C. and Rauch, W. (2014) 'Exploring critical pathways for urban water management to identify robust strategies under deep uncertainties', *Water Research*. Elsevier Ltd, 66, pp. 374–389. doi: 10.1016/j.watres.2014.08.020.

Urrutiaguer, M., Lloyd, S. and Lamshed, S. (2008) 'Determining WSUD Project Benefits using a Multi-Criteria Assessment Tool', in *11th International Conference on Urban Drainage, Edinburgh, Scotland, UK, 2008*.

USEPA (2010) *Green Infrastructure Case Studies: Munipical Policies for Managing Stormwater with Green Infrastructure*.

Valkering, P., Cörvers, R., Offermans, A. and Haasnoot, M. (2008) 'Perspective method for long term water management', *H2O Tijdschrift voor Watervoorziening en Waterbeheer (In Dutch)*, 41(14/15), pp. 18–20.

Van De Ven, A. H. and Poole, M. S. (1995) 'Explaining Development and Change in Organizations', *Academy of Management Review*, 20(3), pp. 510–540. doi: 10.2307/258786.

Viavattene, C. and Ellis, J. B. (2013) 'The management of urban surface water flood risks: SUDS performance in flood reduction from extreme events', *Water Science and Technology*, 67(1), pp. 99–108. doi: 10.2166/wst.2012.537.

Viavattene, C., Scholes, L., Revitt, D. M. and Ellis, J. B. (2008) ‘A GIS based decision support system for the implementation of stormwater best management practices’, *11th International Conference on Urban Drainage, Edinburgh, Scotland, UK, 2008*, pp. 1–9.

Villarreal, E. L., Semadeni-Davies, A. and Bengtsson, L. (2004) ‘Inner city stormwater control using a combination of best management practices’, *Ecological Engineering*, 22(4–5), pp. 279–298. doi: 10.1016/j.ecoleng.2004.06.007.

van der Voorn, T., Quist, J., Pahl-Wostl, C. and Haasnoot, M. (2017) ‘Envisioning robust climate change adaptation futures for coastal regions: a comparative evaluation of cases in three continents’, *Mitigation and Adaptation Strategies for Global Change*, 22(3), pp. 519–546. doi: 10.1007/s11027-015-9686-4.

Voskamp, I. M. and Van de Ven, F. H. M. (2015) ‘Planning support system for climate adaptation: Composing effective sets of blue-green measures to reduce urban vulnerability to extreme weather events’, *Building and Environment*. Elsevier Ltd, 83, pp. 159–167. doi: 10.1016/j.buildenv.2014.07.018.

Walker, W. E., Haasnoot, M. and Kwakkel, J. H. (2013) ‘Adapt or perish: A review of planning approaches for adaptation under deep uncertainty’, *Sustainability*, 5(3), pp. 955–979. doi: 10.3390/su5030955.

Wang, T. and Neufville, R. De (2004) ‘Building Real Options into Physical Systems with Stochastic Mixed-Integer Programming’, in *8th Real Options Annual International Conference, Montreal, Canada, 2004*.

WaPUG (2002) *Wastewater Planning Users Group. Code of practice for the hydraulic modelling of sewer systems*. 3rd edn.

Ward, S., Farmani, R., Atkinson, S., Butler, D., Hargreaves, A., Cheng, V., Denman, S. and Echenique, M. (2012) ‘Towards an integrated modelling framework for sustainable urban development’, in *9th International Conference on Urban Drainage Modelling, Belgrade, 2012*.

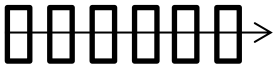
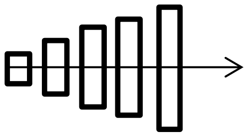
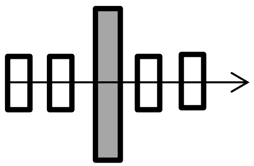
Willems, P., Arnbjerg-Nielsen, K., Olsson, J. and Nguyen, V. T. V. (2012) ‘Climate change impact assessment on urban rainfall extremes and urban drainage: Methods and shortcomings', *Atmospheric Research*. Elsevier B.V., 103, pp. 106–118. doi: 10.1016/j.atmosres.2011.04.003.

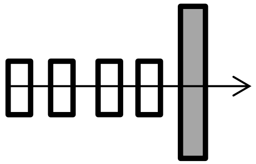
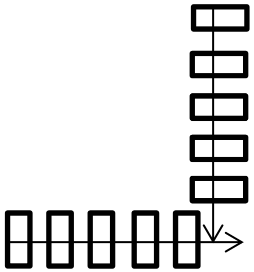
Woods-Ballard, B., Kellagher, R., Martin, P., Jefferies, C., Bray, R. and Shaffer, P. (2007) *CIRIA C697: The SUDS manual*.

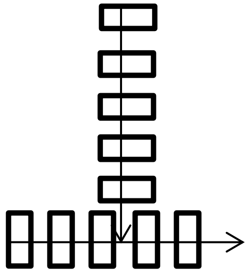
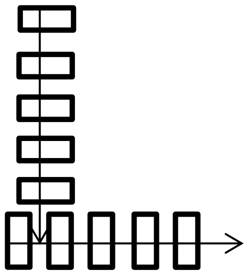
Woods Ballard, B., Wilson, S., Udale-Clarke, H., Illman, S., Scott, T., Ashley, R. and Kellagher, R. (2015) *CIRIA C753: The SUDS Manual*.

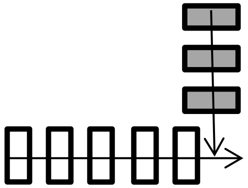
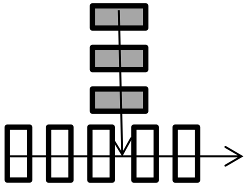
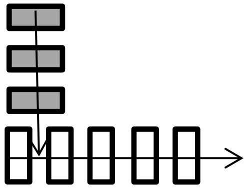
Yazdani, A., Appiah Otoo, R. and Jeffrey, P. (2011) 'Resilience enhancing expansion strategies for water distribution systems: a network theory approach', *Environ. Model. Softw.*, 26 (12), pp. 1574–1582.

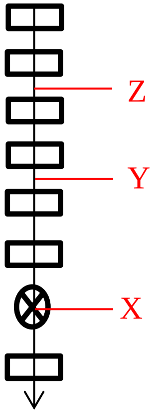
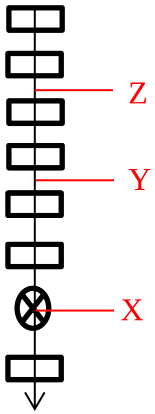
Appendices

Ref	Modification		Represents	Sensitivity	Peak Flow Rate (l/s)	Flow Rate after 20% Stormwater Disconnection	
	Test	Detail				Indiscriminate (l/s)	Unit-Hydrograph Targeted (l/s)
1	The influence of catchment length		Linear catchment	3 subcatchments	325.9	260.7	231.0
				5 subcatchments	381.8	305.4	265.5
				7 subcatchments	429.2	343.4	302.0
2	The influence of incremental changes in area		Centre of urban area	-	843.0	674.4	630.0
3	The influence of exceptional subcatchment area		Development in middle section of catchment	Shaded subcatchment is 3 x average	793.5	634.8	505.3
				Shaded subcatchment is 5 x average	1205.3	964.2	834.7
				Shaded subcatchment is 7 x average	1617.1	1298.6	1164.1

			Development in extremities of catchment	Shaded subcatchment is 3 x average	707.9	566.2	419.5
				Shaded subcatchment is 5 x average	1119.5	895.6	748.9
				Shaded subcatchment is 7 x average	1531.2	1225.0	1078.3
4	The influence of two equally sized catchments joining at a "junction"		Junction in downstream section of catchment	Two catchments of 3 subcatchments	651.8	521.5	462.1
				Two catchments of 5 subcatchments	763.5	610.8	531.1
				Two catchments of 7 subcatchments	858.5	686.6	604.1

			Junction in centre of catchment	Two catchments of 3 subcatchments	634.4	479.9	459.4
				Two catchments of 5 subcatchments	749.4	599.5	535.9
				Two catchments of 7 subcatchments	821.8	657.5	568.0
			Junction in upstream section of catchment	Two catchments of 3 subcatchments	587.6	470.1	340.6
				Two catchments of 5 subcatchments	631.2	505.0	369.6
				Two catchments of 7 subcatchments	656.1	524.1	415.1

5.	The influence of two non-equal catchments joining at a "junction		Junction in downstream section of catchment	Shaded catchment of 3 subcatchments	673.5	538.8	519.8
				Shaded catchment of 7 subcatchments	803.5	642.8	620.2
			Junction in centre of catchment	Shaded catchment of 3 subcatchments	677.7	542.1	498.4
				Shaded catchment of 7 subcatchments	759.8	607.8	507.7
			Junction in upstream section of catchment	Shaded catchment of 3 subcatchments	592.1	473.7	401.1
				Shaded catchment of 7 subcatchments	723.1	563.9	473.5

6.	CSO (Flow limit, excess volume lost from system)		CSO restricting pass forward flows to 20% of peak baseline flow at each location (Baseline [no CSO] flow 429.2)	CSO at Location X	267.3	230.2	81.2
				CSO at Location Y	375.2	274.8	221.5
				CSO at Location Z	406.9	343.4	303.8
7.	Flow Constrictions (Flow limit, excess volume retained in system)		Pumping stations, reduction in pipe capacity restricting pass forward flows to 20% of peak baseline flow at each location	Constriction at Location X	267.3	230.2	81.2
				Constriction at Location Y	384.1	300.1	228.5
				Constriction at Location Z	429.2	343.4	299.9