Reliable and Resilient Surface Water Management through Rapid Scenario Screening

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I certify that all material in this thesis which is not my own work has been identified and that no material has previously been submitted and approved for the award of a degree by this or any other University.

Signature: …………………………………………………………..

ABSTRACT

Surface water flooding causes significant damage, disruption and loss of life in cities, both in the UK and globally. These impacts have historically been managed through application of conventional urban drainage systems designed to meet specified design standards. Conventional strategies have performed well in the past, but are becoming increasingly unfit for purpose due to intensifying hazards caused by several emerging challenges, including climate change, urban growth and aging drainage infrastructure.

In response, an extensive range of alternative novel interventions has been developed. These have been successfully applied across many case studies and their performance to meet design standards on specific sites is now well understood. However, application is still limited and challenges exist regarding how to maximise performance at the urban catchment scale and incorporate resilience to extreme rainfall events within design.

This thesis addresses these challenges through evaluating intervention performance using a rapid scenario screening framework. This framework delivers insight into the complex permutations of intervention strategies at a catchment scale through evaluating alternatives, scales, spatial interactions and responses to a range of rainfall events. The study achieves novelty through developing a new modelling methodology which applies cell parameterisation to represent urban drainage systems and interventions using an existing cellular automata model. The framework is applied at a high level to screen intervention performance using easily accessible data and simplified intervention strategies, it is envisaged that this style of analysis is appropriate for initial catchment assessment to evidence and direct future flood management actions.

The research finds intervention scale, distribution and placement to be important factors in determining performance within the context of initial catchment screening using theoretical modelling parameters. Although localised interventions provide benefit at a smaller scale, catchment based strategies are required to substantially reduce estimated annual damage costs across urban areas. The most effective intervention was consistently found to be extensive application of decentralised rainfall capture, which reduced expected annual damage in a UK case study by up to 76%. Intervention distribution and placement

are also demonstrated to significantly influence cost effectiveness of strategies, with a wide range of ratios predicted, ranging from £0.10 to £26.0 saved per £1 spent. The most cost effective interventions across the case studies investigated were found to be high volume local drainage interventions targeted in areas of intense flooding.

Results demonstrate significant variation in strategy performance depending on rainfall intensity and duration. Analysis across events ranging from 2 to 1000 year return periods found many interventions which performed well during design standard events demonstrate substantial decreases in effectiveness during higher magnitude rainfall. Of particular note are interventions with finite storage capacities, which exhibit considerable decreases in performance at certain threshold levels. The implications of this finding are that designing interventions with resilient performance requires simulation of many rainfall scenarios, and that interventions with resilient properties, such as green infrastructure, do not necessarily achieve resilient performance.

The research also identifies that rapid screening frameworks contribute an adaptable and useful tool for stakeholder engagement, intervention design and scenario exploration. Case study application of the framework alongside catchment stakeholders in Melbourne, Australia, facilitated an efficient and collaborative design screening process which benefitted from enhanced communication across a wide range of expertise. The simplified development of intervention strategies provided a clear communication tool which supported the multi-disciplinary investigations required for urban planning in a complex environment. Analysis of many strategy permutations highlighted the advantage of multiple smaller intervention strategies accumulating towards catchment scale benefits, a possibility which is advantaged through stakeholder communication tools, such as this framework.

Overall, this thesis demonstrates that reliable and resilient surface water management can be achieved through decentralised catchment scale implementation of interventions, complemented by targeted and cost effective high volume measures. Complexity and variation of outcomes across a range of scenarios indicates the importance of encapsulating the complex permutations of options when evaluating interventions and provides justification for future application of rapid scenario screening frameworks.

‘DON’T PANIC’

Douglas Adams, The Hitchhiker’s Guide to the Galaxy

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LIST OF ABBREVIATIONS

AEP	Annual Exceedance Probability
AHP	Analytical Hierarchy Process
BMP	Best Management Practices
CADDIES	Cellular Automata Dual DraInagE
CIRIA	Construction Industry Research and Information Association
CSO	Combined Sewer Overflow
DEM	Digital Elevation Model
DTM	Digital Terrain Model
EAD	Expected Annual Damage
F/NF	Flood/ No Flood
FEH	Flood Estimation Handbook
GIS	Geographical Information System
GPU	Graphics Processing Unit
ICM	Infoworks Integrated Catchment Management
IDF	Intensity Duration Frequency
IWM	Integrated Water Management
LID	Low Impact Development
LIDAR	LIght Detection And Ranging
PFS	Priority Flood Spot
ReFH	Revitalised Flood Hydrograph
SD	Systems Dynamics
SuDS	Sustainable Drainage Systems
SWM	Surface Water Management
SWMM	Storm Water Management Model
SWMP	Surface Water Management Plan
WSUD	Water Sensitive Urban Design

1. INTRODUCTION AND SCOPE

This thesis develops a rapid scenario screening methodology in response to surface water flood hazards exacerbated by climate change, population growth and rapid urban development. The central argument of this thesis is that utilising rapid analysis to screen many permutations of future scenarios can evidence and direct reliable and resilient surface water flood management across urban catchments.

Cities are facing unprecedented shocks from natural hazards (Carter et al., 2009; Wong and Brown, 2009; Jabareen, 2013; Norton et al., 2015; Committee on Climate Change, 2017; Guerreiro et al., 2018). The convergence of people, economic activity and social function makes cities uniquely vulnerable to the challenges of a changing climate, growing population and urban expansion (Djordjević et al., 2011; Hallegatte et al., 2013). Managing environmental hazards is necessary and urgent to prevent major future disruption to social and economic functions in cities (Butler et al., 2017), but assessing and implementing management strategies at the urban catchment scale is complicated and expensive. New approaches and methodologies are required for the effective future management of urban environments (Pitt, 2008; Cabinet Office, 2011). In particular, recent studies emphasise a need to manage surface water flood hazards (EWA, 2009; Douglas et al., 2010; Ellis and Lundy, 2016; Committee on Climate Change, 2017; Löwe et al., 2017; Guerreiro et al., 2018; Wing et al., 2018).

This introductory chapter presents the motivation for research and outlines the structure of the thesis. Motivation is described through detailing the context of surface water management, future hazards, legislation and available management interventions. The thesis structure is presented through establishing aims and objectives for the research and linking these with subsequent chapters. The introduction is concluded through justifying the originality and contribution to knowledge delivered through this thesis.

1.1. Motivation for research

1.1.1. Surface water flood management

Flood management is an established discipline in UK environmental policy, however there is an emerging recognition that a historic focus on fluvial and coastal flooding has left a gap in managing urban surface water (Pitt, 2008). Recent reports highlight surface water flooding accounts for 50% of the properties at risk in the UK (DEFRA, 2012). Damage from surface water flooding is significant, with current annual damages estimated between 0.25 and 0.5 billion GBP in the UK alone (DEFRA, 2012; Committee on Climate Change, 2017). This is predicted to rise to between 0.5 and 1 billion GBP over the next 50 years (Committee on Climate Change, 2012). Some studies estimate current damage costs from surface water to already constitute up to 40% of UK annual flood losses (DEFRA and Environment Agency, 2007; Douglas et al., 2010).

Severe damage from recent extreme events has led governments, academics and communities to prioritise building resilience to future hazards (DEFRA and Environment Agency, 2011; Davoudi et al., 2012; Ofwat, 2012; Viavattene and Ellis, 2013; Aldunce et al., 2015; HM Government, 2016; Butler et al., 2017; Committee on Climate Change, 2017). Current action is insufficient to manage future levels of risk (HM Government, 2016; Committee on Climate Change, 2017), therefore future research must advance ‘business as usual’ design standards beyond a contemporary management approach focused on minimising routine disruptions, towards strategies which build resilience to extreme events.

Surface water flooding is a global issue, with many international government reports and academic studies emphasising the need for management strategies to be implemented (US EPA, 2002; Chocat et al., 2007; EWA, 2009; Wong and Brown, 2009; Barbosa et al., 2012; Burns et al., 2012; Leitão et al., 2013; Fletcher et al., 2015; Mguni et al., 2016; Wing et al., 2018). Need for action is evidenced through growth of international surface water management agendas such as sustainable drainage systems (UK), sponge cities (China), water sensitive urban design (Australia) and low impact development (USA), to name a few (Fletcher et al., 2015).

Although many potential management strategies exist, recent government reviews (Committee on Climate Change, 2015) indicate current implementation of management strategies is insufficient despite clear and established legislation presented in the 2010 Flood and Water Management Act (HM Government, 2010). Reviews call for enhanced evidence to support and increase implementation of new surface water management strategies, however spatial disaggregation of complex urban catchments, uncertainty regarding hazard characteristics and the multitude of management options results in a challenge evaluating the many permutations of potential scenarios. Delivering cost effective, reliable and resilient surface water management requires consideration of these interacting factors. A central argument of this thesis is that this challenge can be managed through enhanced screening of many scenarios using resource efficient frameworks.

1.1.2. Future hazards

It has long been understood that climate change, population growth and urban expansion are increasing future risk to cities (Djordjević et al., 2011). However, recent advances in research highlight that the magnitude of these hazards has been systematically underestimated (Wing et al., 2018). Even in low impact climate change scenarios, cities are likely to face far greater hazards from flooding than previously recognised (Guerreiro et al., 2018). This section will present the main challenges driving innovation in surface water flood management.

Climate change

The causes and effects of climate change are subject to extensive contemporary research (IPCC, 2014). It is apparent that the effects of a changing climate have manifested themselves globally on human and physical systems and that this change needs to be managed (Barker, 2007; Jones et al., 2012; IPCC, 2014; Committee on Climate Change, 2017).

A changing climate is predicted to increase the seasonality and variability of weather patterns, influencing the occurrence and characteristics of extreme weather events (Djordjević et al., 2011; DEFRA, 2012; Jones et al., 2012; Committee on Climate Change, 2017). The most relevant impact to surface water management is the increase in intensity and duration of extreme precipitation which may result in flooding which exceeds the capacity of existing drainage systems and design standards (Westra et al., 2014). This is of particular concern where changes to climate could also exacerbate other anthropogenic pressures such as urban sprawl and changes in land use (Quevauviller, 2011). A consistent prediction is that future extreme rainfall events will increase in frequency and in magnitude, thus increasing surface water flooding hazards (EWA, 2009; Wheater and Evans, 2009; Guerreiro et al., 2018). Many climate scientists highlight that it is extreme events, and not gradual change, which pose the most risk to humans

(Meehl et al., 2000). Therefore it is crucial that future hazard management accommodates mechanisms to plan for extreme events.

Building resilience to extreme events is prioritised in recent UK strategic policy, including the National Risk Register (Cabinet Office, 2017a), the 2010 Flood and Water Management Act (HM Government, 2010), Keeping the Country Running (Cabinet Office, 2011), National Flood Resilience Review (HM Government, 2016), the UK Climate Change Risk Assessment (Committee on Climate Change, 2017) and the public summary of sector security and resilience plans (Cabinet Office, 2017b). This thesis responds to this through enhancing analysis of novel engineering solutions to counter the negative effects forecast by UK and international climate change predictions.

Urban and demographic change

It is well recognised that expansion of urban areas can express negative effects on the water cycle (Weng, 2001; Chocat et al., 2007; Marlow et al., 2013; Butler et al., 2018). This is attributed to disruption of the natural processes which regulate water flow and quality within catchments (White, 2008). Disruption is most relevant when urban development has occurred rapidly and drainage infrastructure has been in place for long periods of time; typically where systems are designed for past climates, land use or out of date environmental standards (Johnson and Priest, 2008).

In relation to surface water flooding, the main impact from urbanisation is that the sealing of native soils with impervious surfaces greatly increases the volume of runoff during precipitation events (Goonetilleke et al., 2005; Chocat et al., 2007; Karvonen, 2011; Barbosa et al., 2012). Compounding challenges caused by urbanisation include a reduction in groundwater infiltration rates, increased sediment and soil erosion, competition for subterranean utilities space, increased sewerage requirements and smaller areas available above ground to store, capture or attenuate runoff (Wong and Eadie, 2000; Chocat et al., 2007).

Urbanisation is recognised as a global problem and is exacerbated by growing populations and a trend for an increasing proportion of people to cluster in cities. This issue is particularly pronounced in the developing world, where studies predict urban populations to double and city areas to triple by 2030 (Djordjević et al., 2011; Marlow et al., 2013). In the UK a similar, although less dramatic trend is expected, with evidence of continued migration into cities and a population size predicted to rise by 9 million by 2030 (Butler et al., 2014). The effect of an increasing population living in urban areas will be compounded by the trend towards a greater proportion of single occupancy households. Currently at 5%, this is expected to reach 18% of all households by 2030 (Office for National Statistics, 2009).

Complex infrastructure and an abundance of residential, commercial and governance structures results in surface water flooding having particularly damaging effects within urban environments. In addition to urban areas being highly vulnerable, their characteristics, in particular the predominance of impermeable surfaces, also means that these spaces can further increase surface water flows and exacerbate impacts. The combination of generating large quantities of runoff in the same spaces as potential for high density vulnerabilities prioritises cities as crucial areas to manage effectively (White, 2008; Wong and Brown, 2009; Chen et al., 2016). Future research should develop new mechanisms for planning surface water management interventions which take into account the complex spatial disaggregation of urban environments.

Limitations of current surface water management systems

Within this thesis, the term 'surface water management system' refers to the physical infrastructure installed to manage hazards, rather than a broader societal-infrastructure interaction described in other systems research (Babovic et al., 2018a).

Another emerging threat to urban surface water management is a reliance on legacy solutions and aging drainage systems (Ana and Bauwens, 2010). Historic application of conventional drainage infrastructure has successfully achieved a consistent level of performance relative to design standards (Butler et al., 2014). However, it is apparent that new approaches are required to address future challenges.

Conventional drainage infrastructure laid to service urban environments typically includes pipe and gully networks, storage tanks, combined sewer outfalls and other drainage features (Chocat et al., 2007; Butler et al., 2018). Over time these systems deteriorate. Ultimately, this may lead to system failures such as leaks and blockages which result in sub optimal operation of networks and an increased risk of flooding and contamination (Fenner, 2000; United States General Services Administration, 2011).

As the majority of urban assets are buried their condition is difficult and expensive to monitor, particularly where excavation is required to examine pipes. This is due to the long time period (often decades) between laying pipes and finding them again resulting in records being lost or damaged. This is a particular problem when pipes are laid before digital archiving was available or where regulatory changes mean responsibility for maintenance has changed (for example the privatisation of water and sewerage). This is often evident in historic cities, for example parts of London are still served by the sewerage system designed by Bazalgette in the late 1800's. Cost is further compounded by the price of managing other urban services, particularly where required excavation of networks may lead to disruption. A common example of this is a road closure to access and repair a collapsed or blocked subterranean pipe system. Consequently, it is difficult to ascertain the condition and investment requirements to manage future surface water flooding through relying on existing systems alone.

Differences in land use, design standards and planning regulations mean that urban drainage networks have often been designed to accommodate significantly smaller demands than for which they currently operate. Significant proportions of networks in major cities have been laid years before detailed current guidance and hydraulic modelling software has been available with which to accurately quantify the requirements for pipe capacities (Fenner, 2000).

Pipe networks are considered one of the most capital intensive infrastructures (Wirahadikusumah et al., 2001). Therefore the risk of deterioration and potential upgrades is an expensive threat to mitigate, requiring extensive investment and analysis (Fenner, 2000; Ana and Bauwens, 2010). In the context of global financial crisis and austerity, regulatory bodies are placing additional pressures on infrastructure operators to ensure customer costs are as low as possible (Ofwat, 2017). By this logic it can be said that the financial pressures of investment also act as an emerging threat to surface water management.

1.1.3. Legislation and governance

The previous section outlines that magnitude and likelihood of future flood damage is predicted to escalate as a result of increasing precipitation intensity, expanding urban areas and a reliance on aging urban drainage infrastructure (Barker, 2007; Wheater and Evans, 2009; Ana and Bauwens, 2010; IPCC, 2014). UK flood policy has identified this risk and legislated towards identifying and managing hazards (Pitt, 2008; DCLG, 2010; HM Government, 2010).

Flooding is amongst the top three hazards prioritised in the UK Climate Change Risk Assessment of the greatest emerging environmental challenges to the UK (Committee on Climate Change, 2017). Surface water flooding is specifically reinforced by legislation (HM Government, 2010), DEFRA guidance (DEFRA, 2012, 2018a; HM Government, 2016) and strategic assessments undertaken by the Committee on Climate Change (2015, 2017).

Severe damage from recent extreme events has emphasised the flood resilience agenda and led governments and academics to prioritise building resilience to future extreme events (Ofwat, 2012; HM Government, 2016; Committee on Climate Change, 2017). Strategic reviews indicate that current action is insufficient to manage future levels of risk (HM Government, 2016; Committee on Climate Change, 2017) and that new hazard management frameworks and novel interventions are required to manage future extreme flooding (Commitee on Climate Change, 2015).

One mechanism for achieving this is implementation of Surface Water Management Plans (SWMP) as set out in the PPS25 and detailed in DEFRA technical guidance (DCLG, 2010; DEFRA, 2010). SWMP's are investigations designed to outline long term solutions to manage surface water across local authority jurisdictions and develop a strategy for partnership working across organisations operating within this boundary. Application of plans typically involves large scale strategic risk assessments, followed by focused studies in areas defined as vulnerable to flood hazards. A key objective of this process is to identify possible interventions which can be applied to alleviate flood risks.

1.1.4. Challenges of implementing available interventions

The previous sections have outlined several significant challenges facing surface water management in cities. However, despite these issues, technical understanding and availability of a range of surface water flood management interventions exists.

Interventions include both tested and novel measures such as conventional piped drainage networks, sustainable drainage systems (SuDS), green infrastructure, property level resilience measures, nature based solutions and catchment management, to name a few (Fletcher et al., 2015; Woods Ballard et al., 2015; Ossa-Moreno et al., 2017; Schanze, 2017; Butler et al., 2018). Application of these interventions is supported by current legislation such as the aforementioned Flood and Water Management Act, which specifies for local flood risk strategies to be developed and implemented (HM Government, 2010).

Despite technical understanding, supportive legislation and a wide range of intervention options, recent studies indicate application of new intervention strategies still faces multiple challenges. Barriers for implementation include failure to accommodate new measures in institutional decision making frameworks, uncertainty regarding effectiveness of novel interventions in a heavily regulated and risk averse water industry and a lack of evidence regarding the hydrological performance and cost effectiveness of novel strategies at the catchment scale and during extreme rainfall events (Gersonius, 2008; Cettner, 2012; Gersonius et al., 2012; Ellis, 2013; Lamond et al., 2015; Woods Ballard et al., 2015; Fenner, 2017; O'Donnell et al., 2017; Ossa-Moreno et al., 2017; DEFRA, 2018b). Collecting evidence to mitigate these barriers is compounded by the computational expense of simulating the many possible intervention strategies across multiple rainfall scenarios at a detail which adequately represents the spatial disaggregation of urban catchments (Hunter et al., 2008a; Dottori and Todini, 2011; Dottori et al., 2013; Jayasooriya and Ng, 2014; Mikovits et al., 2015; Löwe et al., 2017). Government guidance supports academic findings through calling for new approaches to generate evidence and support implementation of future surface water management (Pitt, 2008; Commitee on Climate Change, 2015).

One way to address the computational expense of detailed performance analysis whilst maintaining adequate spatial resolution and representation of surface water dynamics and interventions is to adapt and structure application of decision support processes at different levels of complexity. Recognising and adapting to the trade-off between model detail and required accuracy for a range of decisions supports analysis undertaken in steps, from screening to detailed design. Novel analysis can be applied to initially screen scenarios using easily accessible data to inform and direct requirements for additional evidence, which may require subsequent application of enhanced model complexity and resources. This creates an opportunity to evaluate the many permutations of scenarios through developing high level screening processes which enhance understanding of options by quickly evaluating relative importance of available interventions, locations and hazards, before taking this information to refine and direct future management actions. This thesis responds to this opportunity through investigating the scale, interactions, distribution, performance and economics of novel interventions in urban catchments through developing a framework which facilitates high level insights and analysis regarding the complex permutations of scenarios in urban catchments.

1.2. Aims and objectives

The aim of this thesis is to **develop rapid scenario screening to investigate the performance of surface water management strategies in urban catchments across design standard and extreme events.**

In order to achieve this aim, a number of objectives have been identified:

1. Review literature regarding screening intervention performance under design standard and extreme rainfall events.
2. Develop a screening framework to enable assessment of many intervention scenarios at the urban catchment scale.
3. Validate the framework against industry best practice.
4. Investigate the flood reduction performance of strategic and specific interventions.
5. Evaluate intervention cost effectiveness across many rainfall scenarios.
6. Verify application of the framework through practical application with catchment stakeholders.
7. Investigate the relationship between resilience and reliability of interventions.
8. Develop recommendations for practical application of this methodology.

1.3. Thesis structure

This thesis contains eight chapters. The structure of these chapters and their connection to the thesis objectives are presented in Figure 1.1. The present chapter (Introduction) describes the motivation and scope of research.

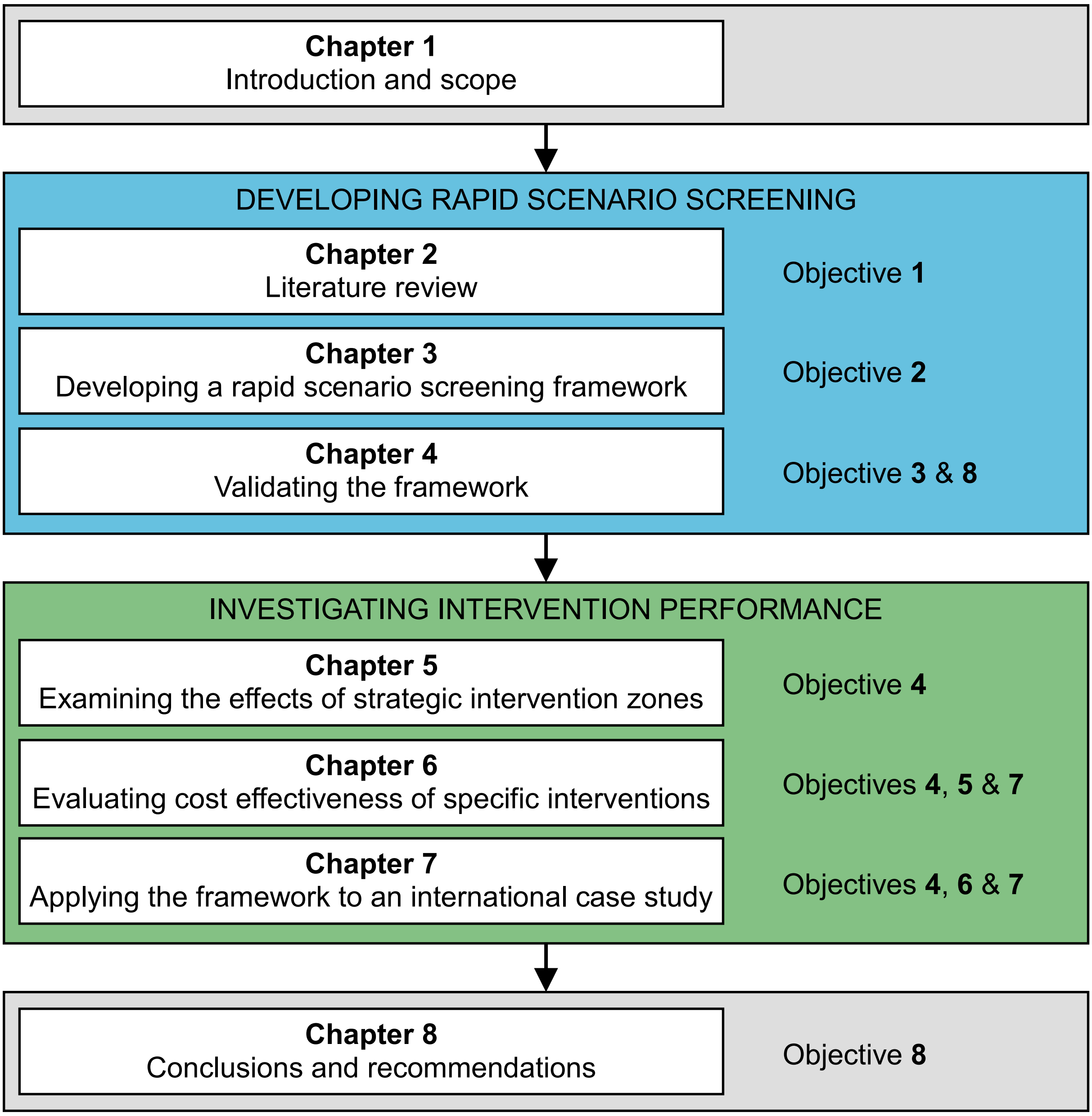

Figure 1.1: Thesis structure and objectives

Chapter Two responds to Objective One through reviewing current scientific literature regarding intervention strategies, selection processes and management of design standard and extreme surface water flood scenarios.

Chapter Three responds to Objective Two and the gaps in literature by presenting the development of a rapid scenario screening framework. This forms the basis of the methodology applied within the thesis.

The framework is published in the following peer reviewed journal publication: Webber, J.L., Gibson, M.J., Chen, A.S., Savic, D., Fu, G. and Butler, D. 2018. Rapid assessment of surface-water flood-management options in urban catchments. *Urban Water Journal* 15 (3) pp 210 – 217.

Chapter Four responds to Objective Three by validating the framework through comparing results versus results simulated using standard industry software. Validation is undertaken using a case study of St Neots, UK, with model outputs taken from a professional Surface Water Management Plan (SWMP), published by the engineering consultancy 'Arcadis' on behalf of Cambridgeshire County Council (Arcadis, 2012). Validation is made relative to a range of scenarios, representing incremental advances in complexity and realism of the catchment study area. The chapter also responds to Objective Eight by exploring practical application of the framework relative to the current approaches applied by Arcadis to develop the SWMP.

This research is published in: Webber, J.L., Booth, G., Gunasekara, R., Fu, G. and Butler, D. 2018. Validating a rapid assessment framework for screening surface water flood risk. *Water and Environment Journal* (accepted and pending publication).

Chapter Five responds to Objective Four. This chapter applies the framework to evaluate the flood reduction performance of strategic intervention zones. Strategic intervention zones are applied through changing catchment characteristics to represent effects possible using a range of interventions. This analysis is intended as a preliminary screening measure to identify the potential for subsequent analysis of specific interventions. Analysis is carried out across a case study in Exeter, UK.

The research presented in this chapter is published as part of: Webber, J.L., Gibson, M.J., Chen, A.S., Savic, D., Fu, G. and Butler, D. 2018. Rapid assessment of surface-water flood-management options in urban catchments. *Urban Water Journal* 15 (3) pp 210 – 217.

Chapter Six advances framework application towards assessing specific interventions and responds to Objectives Four, Five and Seven. The chapter describes how specific interventions are represented within the framework and then evaluates intervention performance across a range of rainfall and placement scenarios in a case study located in Exeter, UK. Performance analysis is split into two levels of detail. Stage one consists of 144 scenarios and assesses flood damage relative to variation in rainfall intensity, duration and frequency. Stage two is a more detailed analysis, in which 792 scenarios are evaluated through developing a cost effectiveness metric which includes estimated annual flood damage compared to intervention capital, operation and maintenance costs over a thirty year planning period. Analysis of reliability and resilience is made through evaluating intervention performance during a range of design standard and extreme rainfall events.

The research presented in the stage one analysis, screening intervention response to rainfall duration, is published as: Webber, J.L., Fu, G. and Butler, D. 2018. Rapid surface water intervention performance comparison for urban planning. *Water Science and Technology* 77 (8) 2084 – 2092.

The cost effectiveness research presented in this chapter is currently under review as: Webber, J.L., Fu, G. and Butler, D. 2019. Comparing cost effectiveness of surface water flood management interventions in a UK catchment. *Journal of Flood Risk Management*.

A discussion of reliability and resilience, using examples drawn from this chapter is also published as: Webber, J.L., Fu, G. and Butler, D. 2018. How can we build reliable and resilient surface water flood management? *Proc. 6th Joint EWA/JSWA/WEF Conference 2018, Munich*.

Chapter Seven responds to Objectives Four, Six and Seven through verifying the framework using a real world case study in Melbourne, Australia. This research represents work undertaken in collaboration with a range of catchment stakeholders to workshop, represent and assess the performance of 75 scenarios, representing a range of strategies and their response during design and extreme rainfall, applied across Melbourne City Centre.

This research is presented in the following publication under review: Webber, J.L., Fletcher, T.D., Cunningham L., Fu, G., Butler, D. and Burns, M.J. 2019. Is green infrastructure a viable strategy for managing urban flooding? *Urban Water Journal* (in review).

Preliminary findings from this research project are also published as: Webber, J.L., Fletcher, T.D., Fu, G., Butler, D. and Burns, M.J. 2018. Evaluating city scale surface water management using a rapid assessment framework in Melbourne, Australia. *Proceedings of the International Conference of Urban Drainage Modelling, 2018, Palermo*.

Chapter Eight, concludes the thesis through summarising the key findings from the work and providing recommendations for future research. The chapter also addresses Objective Eight by synthesising guidance for application of rapid scenario screening in practice.

1.4. Originality and contribution to knowledge

This thesis contributes a range of novel outcomes to the field of surface water management. This thesis has:

- Contributed a novel rapid scenario screening framework which delivers insight into how intervention performance can deliver maximum benefits given the many permutations of intervention type, scale and distribution possible within urban catchments. (*Chapter Three*)
- Developed and validated a methodology to represent urban drainage systems and interventions through parameterising cells within a 2D cellular automata modelling structure. Validation of this approach in a UK urban catchment demonstrates comparable accuracy (98.5%) versus outputs from industry standard modelling. (*Chapter Four*)
- Found that although centralised interventions provide benefit at smaller scales, catchment based strategies are required to substantially reduce flood extent and estimated annual damage costs across urban areas. The most effective intervention was consistently found to be extensive application of decentralised rainfall capture, which reduced estimated annual damage in a UK case study by 76% versus a business as usual baseline. (*Chapters Six and Seven*)
- Demonstrated the importance of intervention distribution and placement on strategy cost effectiveness. Analysis of hundreds of scenarios indicates a wide range of cost effectiveness ratios for interventions, ranging from

£0.10 to £26.0 damage reduction per £1 spent, with the most cost effective interventions identified as high volume localised drainage measures targeted in areas of intense flooding. The implications of spatially varying cost effectiveness are two-fold: Firstly, future intervention performance analysis should include spatial simulation of flood dynamics; and secondly, development of decentralised catchment scale strategies should be complemented by application of targeted cost effective interventions. (*Chapter Six*)

- Identified that intervention performance rankings vary in response to changing rainfall return periods, highlighting that performance during design standard events is not indicative of resilience to extreme intensities. In particular, interventions with defined storage capacities demonstrate tipping points at which a significant performance reduction is observed. The implications of this finding are that evaluating resilient performance requires simulation of many rainfall scenarios and that interventions with resilient properties, such as green infrastructure, do not necessarily achieve resilient performance. (*Chapters Six and Seven*)
- Developed practical guidance for screening catchment flood hazards and identifying cost effective, reliable and resilient interventions using rapid scenario screening as a decision support tool. (*Chapter Eight*)

2. LITERATURE REVIEW

This chapter responds to Objective One: ‘Review literature regarding screening intervention performance under design standard and extreme rainfall events’. The chapter is structured through identifying available interventions and decision support processes for strategy selection and then evaluating existing approaches to define, measure and manage surface water flooding in design standard and extreme scenarios.

2.1. Available interventions

Technical understanding and availability of a range of both tested and novel surface water flood management interventions already exists, however uptake of novel strategies remains below capacity (Mijic et al., 2016; O’Donnell et al., 2017; Thorne et al., 2018). This section outlines broad categorisation of interventions with the intention of introducing and contextualising a range of potential surface water management strategies and terminology. A detailed review of literature regarding specific intervention performance is undertaken in Section 6.1.

Conventional urban drainage strategies are often referred to as ‘grey’ solutions due to a basis of construction using concrete, metal and plastic (Hamill, 2001; Mitchell, 2006; Butler et al., 2018). These approaches focus on removing water from a catchment surface using sewers to convey flows to treatment or environmental discharge through combined sewer overflows (Chocat et al., 2007; Burns et al., 2012). Conventional solutions form the basis for the majority of contemporary and historic urban drainage systems and consequently there is extensive expertise regarding design, construction, maintenance and monitoring; leading to a high degree of confidence regarding effective system performance across infrastructure design life and standard loadings.

Although confidence in conventional systems is high, their limitations have long been recognised (Section 1.1.2). In particular the lack of flexibility due to finite design capacities leads to difficulty managing changing runoff volumes and conditions (DEFRA, 2010). Flexibility is an important consideration for future water management given the likely increases in precipitation volume, frequency and intensity associated with climate change (Wheater and Evans, 2009; IPCC, 2014; HM Government, 2016; Committee on Climate Change, 2017) and expanding impermeable urban environments caused by sprawling cities and increasing urban populations (Wong and Eadie, 2000; Chocat et al., 2007; Djordjević et al., 2011; Marlow et al., 2013). The long design life for drainage infrastructure means that many systems are aging and need repair or replacement (Ana and Bauwens, 2010). This can be prohibitively expensive due to complex subterranean infrastructure now present throughout urban catchments. Research has responded to the limitations of conventional drainage measures through developing an extensive range of alternative novel interventions.

A general consensus amongst urban drainage research and practice is the need to move towards more sustainable 'green' drainage solutions (Ellis, 2013; Ellis and Lundy, 2016; Mguni et al., 2016; O'Donnell et al., 2017; Thorne et al., 2018). This interest is reflected in an exponential growth of related terminology appearing in published scientific literature (Fletcher et al., 2015). However, this terminology is fragmented and inconsistent due to informal development of the science across a broad range of regional and institutional perspectives.

Fletcher et al (2015) conducted a comprehensive review of green drainage terminology, motivated through recognising this need for clarity, as emphasised by the Joint Committee on Urban Drainage, International Water Association and International Association of Hydro-Environmental Engineering and Research. The review found terminology to include: Low Impact Development (LID); Water Sensitive Urban Design (WSUD) (Wong and Eadie, 2000; Wong, 2006); Integrated Urban Water Management (IUWM) and Integrated Water Management (IWM) (Niemczynowicz, 1996; Harremoës, 1997); Sustainable Urban Drainage Systems (SUDS) (Woods Ballard et al., 2015); Best Management Practices (BMP) (United States Government, 2011); Stormwater Control Measures (SCM) (National Research Council, 2008); Alternative or Compensatory Techniques (AT/CT) (Fletcher et al., 2015); Source control and Green infrastructure (Walmsley, 1995). It should be noted that this list is not exhaustive. The unifying theme of these terms is the management of surface water through mimicking natural hydrological processes such as infiltration and retention (Fletcher et al., 2015).

Fletcher et al (2015) highlight the concept of 'green infrastructure' as encompassing the range of terminology used to describe 'green' urban drainage, with green infrastructure representing a remit which goes beyond surface water management (Tzoulas et al., 2007), in particular through incorporating multiple ecosystem services alongside drainage functionality, with the potential to provide urban communities a range of benefits through advancing public health, recreation and aesthetics (CIRIA, 2015; Jose et al., 2015; Mijic et al., 2016; Ossa-Moreno et al., 2017). The broad remit of this language supports the utility of rapid scenario screening sought through this thesis and encapsulates the range of terminology applied to represent interventions. As such, the term 'green infrastructure' will be applied throughout this document (Jayasooriya and Ng, 2014).

Despite established inclusion of green infrastructure within academic, government and commercial discussion, several gaps are apparent regarding application (Pitt, 2008; MWH, 2014; Burns et al., 2015c; Woods Ballard et al., 2015; Schubert et al., 2017). Barriers for implementation include failure to accommodate new measures in institutional decision support frameworks, uncertainty regarding effectiveness of novel interventions in a heavily regulated and risk averse water industry and a lack of evidence regarding hydrological performance and cost effectiveness (Harremöes, 2002; Elliott and Trowsdale, 2007; Mijic et al., 2016; O'Donnell et al., 2017). Effective future management requires a developed and enhanced understanding of how the scale, distribution and range of novel interventions can be best applied to achieve maximum performance at the catchment scale and during extreme rainfall events. A range of approaches are available with which to evaluate and evidence surface water flood management strategies, these are discussed in the following section.

2.2. Decision support for intervention selection

Evaluation and implementation of interventions requires a robust and transparent evidence base, including consideration of the many permutations of strategy type, distribution and scale across study areas (House of Commons, 2016). This section outlines current intervention evaluation methodologies from scientific literature and industry practice. The section is divided into 'qualitative' techniques, which provide descriptive assessment methodologies, and 'quantitative' approaches, which apply numeric metrics to measure intervention performance.

2.2.1. *Qualitative techniques for evaluating intervention performance*

This subsection will discuss qualitative intervention screening techniques. The scope of this section is to evaluate the strengths and limitations of each approach, with reference to specific examples from scientific literature.

Expert review, ranking and multi-criteria analysis

The most basic form of intervention screening can be achieved using expert judgement to compare intervention types and locations. Judgement may be informal, through selecting options for further design, or structured through workshops, engagement and questionnaires.

High level comparison of interventions can be undertaken using a SWOT (strength, weakness, opportunity, threat) analysis. For example, Mguni et al (2016) undertook a preliminary SWOT analysis of green infrastructure drainage measures in Sub-Saharan Africa through literature evaluation. The analysis supports green infrastructure as a viable flood risk management option, however indicates that further studies to quantify performance and co-ordinate work between multi-disciplinary stakeholders is required to achieve implementation. The paper provides a comprehensive analysis of broad scale suitability, however this form of SWOT analysis is unsuitable for comparing the more nuanced effects of intervention performance when applied in specific catchments.

Analysis can be enhanced through wider inclusion of stakeholders and expert organisations. Douglas et al (2010) undertook analysis of non-structural mitigation to pluvial flooding through stakeholder engagement (44 households with a history of internal flooding), discussion with local authorities and hosting catchment pluvial flood risk workshops in Heywood, Greater Manchester. The workshops were based on historic flood records, which provided a high degree of confidence in outputs amongst stakeholders. However, this approach is only possible where there is flood history, engaged local stakeholders and reliable flood records; the latter of which is rare due to difficulty obtaining high resolution measurements that coincide with the flood peaks in urban areas (Neal et al., 2009). This approach also relies on past events as an indicator of future performance, and so is unsuitable for analysis of future catchment changes or screening the performance of potential future interventions (Kjeldsen et al., 2014).

Ellis et al (2004) applied expert scoring analysis to support green infrastructure installed on highways. The approach applied criteria describing the performance, environmental, social and economic impacts of a detention/ retention basins, wetlands, infiltration features, porous paving, swales and filters applied across highways. Analysis was conducted using a spreadsheet tool, with inclusion of flooding through criteria relating to storage volume, flood incidents, flooded properties and disruption costs. Data was achieved through community and expert scoring, of which the study acknowledges the subjective bias this may lead to. The study found that although the procedure was adaptable, there was still a need to quantify long term performance and full life costs.

Assessment via expert judgement can be formalised through application of multi-criteria analysis. The UK Sustainable Water industry Asset Resource Decisions (SWARD) project (Ashley et al., 2002) applied decision mapping to determine how sustainability was included by water service providers. Findings were applied to select appropriate criteria to form a decision support tool which applied weightings to rank interventions. The work was not solely focused on urban flooding, and took a wider view of sustainability based on social, technical, economic and environmental criteria (Foxon et al., 2002). This was supported by the production of urban drainage case studies, including analysis of green infrastructure versus conventional drainage solutions. The project provided a structured analytical approach to evaluating a wide range of criteria; however, enumerating the criteria relied on supporting analysis and modelling, therefore requiring extensive additional analysis outside of the framework and restricting the number of options which could be assessed.

Makropoulos et al (2008) developed the multi-criteria analysis Urban Water Optioneering Tool (UWOT) for intervention selection decision support. The tool applied quantitative and qualitative criteria from the SWARD framework (Ashley et al., 2002) to set multiple objectives to solve using a genetic algorithm (Savic and Walters, 1997). The approach was focused on broad scale integrated water management rather than solely on surface water, and consequently did not represent the spatial analysis of flooding and flood damage; however, the framework did include water re-use and capture objectives and is adaptable to a variety of inputs. This adaptability enables consideration of an extensive range of issues, however, as with Ashley et al (2002), relies on pre-modelled performance data to set up. The study highlighted that ‘optioneering’ tools will increasingly be required to develop integrated and context specific solutions for urban water challenges.

Similarly, Young *et al* (2010) applied an analytical hierarchy process (AHP), a pair wise expert ranking system, to select surface water management techniques. Results recommended using the AHP system were then modelled using the Storm Water Management Model (SWMM), with the study finding potential advantages over alternative selection methods. This approach enabled screening of many options combined with subsequent detailed analysis, however a lack of modelling flood dynamics within the screening stage restricts users to judgement based on past performance, neglecting both novel interventions and unpredicted consequences or mechanisms regarding surface water runoff.

Martin et al (2007) applied the ELECTRE III (Roy, 1978) multi-criteria analysis to investigate structural and non-structural drainage measures. The study applied multi-criteria categorisation into a two stage process where initially unacceptable options are rejected (based on the French national drainage survey) with a subsequent option analysis based on performance ranking scores and quantified values. Application of this method was supported by Chitsaz et al (2015), who conducted a comparison of many multi-criteria decision making approaches including simple additive weighting, compromise programming, VIKOR, TOPSIS, AHP and ELECTRE, finding the latter the most effective at managing complex input criteria.

Expert judgement and multi-criteria approaches provide a fast method for screening potential flood interventions. However, fundamentally, these approaches can only prioritise preference based on a range of values and weightings reflecting current understanding and pre-generated analysis. Analysis of novel flood management interventions requires a robust evidence base grounded in understanding of intervention performance (Cettner, 2012; House of Commons, 2016). Spatial variation of performance is of particular importance when considering surface water flood management due to the complex spatial disaggregation of urban catchments.

GIS techniques

Geographic Information Systems (GIS) enable visual and spatial analysis of intervention performance. The systems are commonly applied in intervention assessments, either to calculate or present performance metrics.

GIS can integrate remote sensing data-sets to inform broad scale understanding of flood risk. Weng (2001) used this approach to model urban growth effects on runoff through a simplified distributed surface runoff model, first applied by the United States Soil Conservation Service and based on runoff curve numbers (Pyke et al., 2011). The approach was applied at a high level across the entire Zhujiang Delta, and so was not able to identify sites for interventions. The study found that highly urbanised areas are more prone to flooding, but was applied at too coarse a resolution to identify specific opportunities for interventions in any particular urban areas.

Finer scale spatial analysis for selecting intervention sites at an urban catchment scale can also be applied. Makropoulos et al (2007) applied high resolution GIS techniques to evaluate siting interventions in new developments. Similarly, Todorovic and Breton (2014) applied geospatial analysis to select intervention options based on the potential distribution of pollutants.

GIS are not designed for simulation of runoff and so are frequently applied for spatial analysis of outputs from other simulation approaches, such as 1D-2D models (Viavattene and Ellis, 2013). The SUDSLOC model developed by Viavattene et al (2008) is one such approach to amalgamate hydraulic model outputs to strategically select green infrastructure locations. However, authors indicate that the underlying flood simulation is computationally demanding, particularly for larger catchments. As such the approach is best suited to screening flood risk priority spots, rather than large numbers of interventions across multiple scenarios.

Spatial analysis also extends to economic assessment of flood damages through spatial calculation of flood damage using flood depth damage curves and building locations in a catchment. For example as the basis of the CORFU damage assessment tool (Hammond et al., 2015; Chen et al., 2016). As with other GIS approaches, this relies on previous modelling of flood dynamics associated with individual interventions.

GIS intervention selection is of particular benefit in spatial analysis of interventions. Fenner (2017) support earlier work of Jayasooriya and Ng (2014) in identifying that many tools now utilise GIS interfaces to assess surface water flood management. GIS in itself does not constitute a full analytical process for evaluating model performance, however it provides a powerful tool for analysing the outputs from other assessment methodologies (Viavattene and Ellis, 2013).

2.2.2. Quantitative techniques for evaluating intervention performance

Current standard techniques to quantify flood depth and extent typically apply deterministic computational models based on solution of hydrodynamic equations. For simplicity these can be classified into lumped, 1D and 2D models (Butler et al., 2018).

Lumped models

Lumped or semi distributed models aggregate catchment elements into larger sub-catchments (Pina et al., 2016). Calculations are then based on these sub-catchment units, resulting in a coarse resolution but fast simulation (Jamali et al., 2018). Units are typically identified using GIS processing with a variety of volume continuity approaches applied to identify runoff rates. These are formulated as a function of rainfall and land use categories (US EPA, 2002; Pyke et al., 2011). Outputs from these methodologies provide a fast screening utility, but only account for total runoff volume. As such they are suitable for identifying broad indicative trends which can be used for strategic analysis, but are unsuitable for modelling high spatial resolution of flood dynamics. This is of particular importance given the high resolution spatial variability of intervention effectiveness in urban catchments (Dottori et al., 2013).

1D hydraulic modelling

1D models represent cross sectional average flow conditions through channels in a study domain (Vojinovic and Tutulic, 2009). In the case of urban surface water flooding this typically includes the pipe network or a simplified study catchment, represented using channels.

Pipe surcharge and overland flooding can be approximated through a virtual ‘cone’ proposed over each node of a 1D system (Butler et al., 2018). The cone acts as a temporary store for surcharged flows and enables flood volumes to be approximated. This approach does not simulate movement of runoff across the surface and assumes all flood water originates from the sewer system; consequently, this is not a sufficient representation for overland flow caused during high intensity rainfall, the typical cause of surface water flooding (Westra et al., 2014), which exceeds the capture capacity of sewer systems.

Other studies apply rapid flood spreading models to distribute volume from cones across the catchment. This is achieved through splitting the surface domain into large sub-catchments and specifying flow directions between these (Band, 1986; Martz and de Jong, 1988). This is a computationally efficient approach for estimating surface volume, however has two main limitations: no simulation of runoff from outside of the sewer system (as described above) and no time element to represent flow across a high resolution domain. Studies frequently emphasise the significance of high resolution topography on surface water flood dynamics and so modelling the effect of interventions requires spatial and temporal analysis of runoff (Bates et al., 2006; Mignot et al., 2006; Yu and Lane, 2006; Hunter et al., 2008a; Neal et al., 2009; Fewtrell et al., 2011; Chen et al., 2012; Schubert and Sanders, 2012; Dottori et al., 2013).

A similar approach can also be applied to represent 1D flow through a catchment, with flooding considered as a series of interconnected channels and ponds (Heywood et al., 1997). This approach requires pre-processing or understanding of runoff routes across the catchment, and does not incorporate the spatial and temporal nuances of flow paths which may coalesce or diverge in response to different rainfall intensities. The method is also too simplistic to take into account the high resolution features which control runoff, as discussed above, although may be sufficient where flow remains within the channel profile delimited by street kerbs (Djordjević et al., 1999).

In summary, 1D models offer a fast and simplified analysis of urban flooding, however do so at the expense of representing a high spatial resolution of flood dynamics. Understanding intervention performance within the complex spatial disaggregation of urban catchments requires evaluation of intervention interactions with runoff at a fine spatial scale (Fewtrell et al., 2011; Dottori et al., 2013), therefore alternative modelling approaches are required.

2D hydraulic modelling

Simulation of surface water runoff across the catchment is undertaken using 2D models based on solution of St Vennant shallow water equations (Chow, 1959) or simplifications of these, such as a kinematic or diffusion wave model (Ponce et al., 1978; Elliott and Trowsdale, 2007; Hunter et al., 2008a; Butler et al., 2018). These partial differential equations are commonly calculated using explicit or implicit finite difference solvers where calculation is divided into discrete steps. An extensive range of commercial and academic models are available, each with variations on equations, numerical solvers, time-step controls and input mechanisms. Popular industry models include Infoworks ICM, TUFLOW, Flood Modeller Pro and MIKE, to name but a few.

When coupled with a 1D representation of the piped system, 2D models are considered to be the most accurate representations of urban surface flooding, however accurate simulation is achieved at a trade-off versus high computational and data expense (Bamford et al., 2008; Löwe et al., 2017). This can lead to extensive model setup times and force studies to focus analysis on a small number of options or scenarios. Restrictions on time, budget and data can lead to decision makers considering only tried and tested interventions, resulting in institutional inertia and stifling innovation (Cettner, 2012; O'Donnell et al., 2017). Where new interventions are considered, the model specialism required to accurately simulate these can also constrain analysis to specific measures, which may result in the need to apply several models to assess a range of options (Zhou, 2014).

The trade-off between model complexity and simulation time can force practitioners to simplify aspects of modelling in order to simulate adequate numbers of strategies or scenarios in an acceptable timeframe. One area where this can easily be achieved is through simplification of input elevation models; high resolution inputs can lead to significant increases in simulation time due to an exponential increase in required calculations relative to the change in model cell size. However, maintaining high resolution data is important, Fewtrell (2011) found that errors caused by coarse representations of topography were significantly larger than the differences between a range of numerical approximation schemes across a range of simplifications in 2D models. This work is supported by a body of literature highlighting the need to include high resolution data in 2D flood modelling (Fewtrell et al., 2008; Schubert et al., 2008).

Fewtrell (2011) also highlights that simplified models can provide a viable alternative to St Vennant based simulations (Mikovits et al., 2015; Löwe et al., 2017). This has the advantages of increasing simulation speed (Yu and Lane, 2006; Hunter et al., 2008a; Schubert et al., 2008; Néelz and Pender, 2013), which in turn enables simulation of more scenarios using higher resolution elevation data. Dottori (2013) does however council that modellers should be wary of a false confidence associated with high resolution inputs, due to a range of uncertainties which can propagate throughout the modelling process. Precision is not an indicator of accuracy, and all models should be applied as tools for specific applications (Box, 1976).

Hydraulic modelling to quantify intervention performance

Several review articles specifically focus on assessing the tools available to quantitatively assess green infrastructure interventions. Elliot and Trowsdale (2007) conducted a review of a broad spectrum of urban stormwater models specialised to include green infrastructure techniques. The authors developed research from previous reviews (Zoppou, 2001) and identified 40 models, which they reduced to ten current available approaches. The study found the majority of models were not well suited to modelling high spatial resolution of individual interventions or their effects on catchment surface flood dynamics. Instead, the majority of models divided analysis into sub-catchments which limited the resolution of spatial analysis. The authors identified considerable scope for improving current approaches through adding cost modules, visualisation of individual measures and the increasing the spatial resolution of runoff generation.

Jayasooriya and Ng (2014) conducted an updated review of current surface water flood models and responded to previous studies by advancing the scope to assess incorporating economic analysis within models (Zoppou, 2001; Elliott and Trowsdale, 2007; Ahiablame et al., 2013). The study reviewed 20 models, of which the ten most popular were reviewed in detail. The study identified several key challenges in developing green infrastructure modelling, in particular finding that models need to include a wider range of green infrastructure practices, be applicable to a range of regions using easily obtainable input data, facilitate stakeholder engagement and use new technologies to advance capabilities of decision support systems.

Rapid modelling using cellular automata

A rise in availability of high resolution data has encouraged the development of a new series of models which apply novel cellular automata systems for modelling surface water flooding. Cellular automata are grid based systems which apply water routing rules based on simplified hydraulic equations to achieve speed increases versus traditional 2D models. The simplicity of this model structure suited to computational parallelisation, providing the possibility of further speed increases (Dottori and Todini, 2011).

A range of cellular automata flood models have been developed. Caviedes-Voullième et al (2018) carried out a review of current cellular automata flood models, identifying the relatively new development of the methodology, with the majority of advances being made in the past ten years. Early examples of these models were applied to simulate fluvial dynamics (Murray and Paola, 1994). More recently the approach has been applied to pluvial flood dynamics across flood plains (Parsons and Fonstad, 2007; Rinaldi et al., 2007; Douvinet et al., 2015; Li et al., 2015; Kassogué et al., 2017), however this application is typically undertaken across extensive 100 km^2 (and larger) catchments using coarse resolution elevation modelling typically above a 25 m x 25 m cell size.

Other studies have focussed on development of the underpinning cellular automata mechanisms against case studies or synthetic test catchments (Dottori and Todini, 2011; Guidolin et al., 2012; Ghimire et al., 2013; Liu et al., 2015; Gibson et al., 2016; Abbasizadeh et al., 2018; Caviedes-Voullième et al., 2018). A common finding from these studies is the high computational efficiency and an increase in simulation speed, versus traditional 2D models whilst maintaining comparable accuracy. Gibson et al (2016) found a 98-99% correlation in surface water flood extent relative to Infoworks ICM, a common industry model, at a speed increase of five to twenty times.

A small number of recent studies have applied cellular automata models to urban pluvial flooding. Abbasizadeh et al (2018) coupled the SWMM 1D drainage simulation and a cellular automata model to investigate high resolution runoff in an urban setting using a 4 m x 4 m resolution grid, finding a good comparison with the 2D model TUFLOW. Liu et al (2015) performed a similar analysis using a 5 m x 5 m grid, finding a high computational efficiency but recommending future studies apply finer resolution elevation models.

There is a current gap in literature regarding analysis of high resolution urban flood modelling and investigating intervention performance using these measures despite well documented high computational efficiency, suitable for analysing many scenarios, and documented accuracy of cellular automata methods, relative to existing modelling approaches (Gibson et al., 2016). Only one intervention assessment study was identified, where Lu et al (2018) applied the cellular automata model 'CADDIES' to assess high surface water flood management options in a London catchment. The study was able to utilise fast assessment to examine the response of a range of strategies to three rainfall events. The limitation of this study was the application of relatively coarse (5 m x 5 m) elevation model and only assessing three types of interventions (green roofs, permeable paving and bio-retention systems).

Fast analysis using cellular automata is a promising and novel approach to include many simulations within intervention selection, whilst retaining application of high spatial and temporal resolution essential for understanding of urban flood dynamics (Fewtrell et al., 2011).

2.2.3. General guidance on intervention selection

Some studies provide general guidance on selecting intervention measures. These are intended to support other forms of analysis and include detailing availability and technical information for green infrastructure implementation.

Bowker (2007) evaluated suitability of flood resistance and resilience measures at the property level. The study provides itemised cost estimates for a range of permanent and temporary solutions applied to domestic properties. The source provides a comprehensive breakdown of costs for measures suitable for protecting properties, but functions as a list and does not provide guidance on a process to select and measure performance of measures.

The Environment Agency have produced a similar report detailing cost estimation for SUDS (Environment Agency, 2015), building on previous work from 2007 (Environment Agency, 2007a). The report details a comprehensive analysis of the capital and operational costs of implementing SUDS, and includes a wide range of measures such as green roofs, rainwater capture tanks, permeable paving and infiltration measures. As with Bowker (2007), the study does not detail a process for selecting or measuring the performance of measures.

The Construction Industry Research and Information Association (CIRIA) have published several extensive best practice guides for application of conventional and green infrastructure drainage solutions in the UK. The SuDS manuals C753 (Woods Ballard et al., 2015) and C697 (Woods Ballard et al., 2007) provide detail on design, application and technical performance of many interventions, however does not outline a methodology for scenario or strategy performance analysis. The Benefits of SuDS tool (CIRIA, 2015) furthers analysis through providing sustainability assessment indicators with similarities to the SWARD framework (Ashley et al., 2002); however, as with the SWARD framework, the approach requires extensive prior analysis to develop suitable input data, and as such is better suited to detail design rather than strategic screening.

2.3. Reliable surface water management to meet specified design standards

Contemporary surface water management has been underpinned by the concept of 'reliability', defined as "the degree to which the system minimizes level of service failure frequency over its design life when subject to standard loading" (Butler et al., 2014, 2017). Simply put, systems are designed to minimise the likelihood of failure under a predicted stress. Stresses, in this case typically intense rainfall, are defined by a probability specified through design standards. Performance is assessed through evaluating the likelihood of system failure up to specified rainfall intensities and durations. Sub-standard system performance, in this case - urban flooding, is then mitigated through identifying and implementing interventions (Linkov et al., 2014).

In practice, reducing failure probability to zero is not possible due to inherent uncertainties associated with unknown future rainfall (Kjeldsen et al., 2014). Consequently, performance assessment takes this into account through risk management. This is the process of specifying and testing intervention performance up to a specific design standard. Typical risk management approaches identify the vulnerabilities of a system and quantify potential losses (Linkov et al., 2014). Adjustments are then made to reduce the probability or consequences of failure (Vis et al., 2003). Acceptable failure probabilities are commonly specified as part of a legislated or agreed design standard.

The concept of risk as a tool for identifying requirements for flood management strategies is well established and is laid out as a standard approach (both explicitly and implicitly) in the mainstream of governance guidelines (DEFRA and Environment Agency, 2007, 2011; DCLG, 2010; HM Government, 2010; Defra, 2012; Environment Agency, 2013; House of Commons, 2014; Committee on Climate Change, 2017; DEFRA, 2018b), commercial methodologies (Conroy and Webber, 2013)and academic research (Merz et al., 2006; Johnson and Priest, 2008; Schelfaut et al., 2011; Hammond et al., 2015; Schanze, 2017; Shah et al., 2018).

The main benefit of a risk management methodology is the ability to clearly measure and evidence decisions using quantifiable metrics. Risk metrics are typically a function of the event probability and consequence (Vis et al., 2003). Mathematically, risk is typically calculated as a function of the probability of an event multiplied by a quantified measurement of its impact (Dawson et al., 2011). This is expressed by Dawson and Hall (2006) as:

$$R = \int_{x} p(x)c(x)dx \qquad \text{Equation 2.1}$$

Where R is risk, p(x) is probability of event x and c(x) is the consequence resulting from event x.

Slight variations on this calculation are expressed throughout literature, however this represents a typical calculation technique, reflective of the common themes. Typically probability is represented as a return period or annual exceedance probability (AEP) percentage. Consequence is often expressed as a monetised value. The units used within this formula vary by application. Provided units are consistent across any comparisons made this does not represent a problem; however, inconsistency can lead to confusion when comparing risk using a variety of methods. This is of particular note where studies adopt a relative scoring system to represent risk, rather than an absolute measure (such as economic costs) which can be transferred to a wider comparison.

Other studies represent risk using an estimated annual damage (EAD) (Wheater and Evans, 2009; Hammond et al., 2015). The main advantage of this calculation method is that it provides an absolute value which can be applied relative to annual flood management costs as part of cost benefit analysis.

Risk management is able to offer a decision maker a clear understanding of where a system is most vulnerable for specified failure mechanisms under circumstances for which appropriate amounts of data are available (Stirling, 2010; Bond et al., 2015). This is often used in combination with cost data to provide the evidential basis for decision making and management (Meyer et al., 2013; Hammond et al., 2015; Chen et al., 2016); however, this approach alone does not provide the whole picture. Despite risk management constituting the standard method for managing performance in systems, studies recognise several distinct limitations in its application (Howard et al., 2010; Stirling, 2010; Crisis and Risk Network and Center for Security Studies, 2012; Linkov et al., 2014; Aldunce et al., 2015; Bond et al., 2015).

One of the main limitations of the current risk based paradigm is that the narrow focus on quantifying specific risks does not consider uncertainty effectively, as in order to calculate the risk all probabilities and impacts need to be known, understood and quantified (Stirling, 2010). Where significant areas of uncertainty exist, these are often omitted from analysis. An example of this is failing to examine a flood management strategies performance during high magnitude events because the probability cannot be accurately ascertained or consequences modelled. This unpredictability and lack of knowledge impedes risk management, and means not all risks can be accurately accounted for (Linkov et al., 2014).

Another limitation of a risk management approach is that by conducting analysis relative to a guideline design standard, the assessment only considers a snapshot of the system performance state. Behaviour of a system across a range of events is missed in favour of highly specialised protection for a particular scenario. When facing uncertain and highly dynamic risks such as rain storms it is crucial to consider a more adaptive and flexible approach for planning intervention strategies. This is most evident when considered in the context of unprecedented or unlikely extreme events (Bond et al., 2015). Current studies predict an increasing likelihood of future extreme rainfall and runoff in response to climate change and urban growth, and highlight the need for approaches to manage a range of possible future scenarios (Chocat et al., 2007; EWA, 2009;

Howard et al., 2010; IPCC, 2014; HM Government, 2016; Committee on Climate Change, 2017). Analysis which includes a range of scenarios can identify the adaptability of interventions to future uncertainties, in particular identifying how incremental changes in rainfall characteristics may affect strategy performance.

Other studies further this by suggesting that probabilistic risk management methodologies are not suitable to manage non-linear and highly dynamic risks which exhibit significant uncertainties over the long term (Crisis and Risk Network and Center for Security Studies, 2012). This is of particular importance in developing urban areas and installing flood protection due to the requirement for infrastructure to remain effective over long planning horizons and future legislation. Surface water flood management must accommodate and manage major uncertainties regarding future climatic, social and economic conditions (Brown and Farrelly, 2009; Howard et al., 2010). During analysis, risk probability is extrapolated from past experiences, which are unlikely to be representative of future scenarios. Studies argue that analysis using historical data as a predictor for future events may vastly underestimate the likelihood of high magnitude events occurring due to selection bias originating from relatively short measurement periods (Kjeldsen et al., 2014; Guerreiro et al., 2018). Consequences can also be underestimated due the high degree of complexity in urban systems, this of particular concern where qualitative assessment methods are applied.

Consequently, some studies argue that risk management can also contribute to the vulnerability of a system by enhancing a feeling of safety which may not be merited (Vis et al., 2003). This is often the case where stakeholders misunderstand probabilistic design standards and the significant hydrological, modelling and data uncertainties intertwined with calculations (Merz et al., 2008; Dottori et al., 2013).

The widely recognised limitations of planning based on design standards does not discredit current analysis techniques, as a strong understanding of likely scenarios should also form a fundamental basis of management; however, given the severe consequences of flooding, there is a strong justification to plan for events outside of current understanding. This problem is widely recognised in scientific literature and alternative methods are currently being proposed, a pre-eminent narrative in current methodologies is the potential for ‘resilience’ based planning, which is described in the next section of this review (Cabinet Office, 2011; Ofwat, 2012; Aldunce et al., 2015; Pizzo, 2015; HM Government, 2016; Butler et al., 2017).

2.4. Resilient surface water management to manage extreme rainfall events

The concept of resilience is widely used across a variety of disciplines. Contemporary application has increased in popularity in recognition of a need to manage system functionality beyond design conditions (Aldunce et al., 2015). The concept has been present in engineering literature for over 200 years, where it was used by Tredgold in 1818 to describe a property of timbers which could withstand sudden loads, and by Mallet in 1856, where a 'modulus of resilience' was used as a measure of a materials ability to withstand severe conditions (Hollnagel, 2014). 'Resilience' is also applied within mechanics to refer to "the ability of a body or material to return to its original state after being altered, due to the potential energy that has been stored through modification from a previous state" (Pizzo, 2015).

The term rose to contemporary prominence in hazard management after discussion by C.S. Holling in a 1973 paper on the resilience of ecosystems (Holling, 1973). In this paper, Holling defines resilience as "a measure of the persistence of systems and their ability to absorb change and disturbance and still maintain the same relationships between populations or state variables". Since this point the term resilience has been adopted by a wide range of disciplines, from business planning to social science, and the precise nature of what resilience is and how it should be applied is now subject to extensive contemporary academic debate (Aldunce et al., 2015).

Each discipline has adopted the term with slight variations from the original definition, resulting in a noted lack of consistency and confusion in the application (Klein et al., 2003; Gallopín, 2006; Folke et al., 2010; Schelfaut et al., 2011). This range of applications and diverging development has led to an extensive set of definitions which is often characterised as diluting the meaning of the term towards that of a buzzword (Muller, 2007; Lhomme et al., 2013; Linkov et al., 2014). This results in difficulty operationalising the ideas behind the term (Schelfaut et al., 2011). However, this wide range of approaches also provides an opportunity for lessons learnt and experiences gained across various fields to be translated into the field of engineering.

This section of the literature review will explore the pre-eminent narratives regarding resilience found in the contemporary cross-disciplinary academic literature. The objective of this is to synthesise theoretical discussion, identify key messages and ascertain a workable definition which can be used to operationalise the term in surface water management.

2.4.1. Defining the term 'resilience' in water engineering

As with many other disciplines, there is ongoing debate within the field of engineering as to the definition of resilience and how this can be operationalised to form a useful, actionable and measurable outcome (Aldunce et al., 2015; Pizzo, 2015).

Hashimoto et al (1982) identified that traditional performance metrics for water systems, such as reliability, typically relied on measures of the mean and standard variance of a systems operational behaviour which would obscure the impacts of extreme events. The study recognised that in many cases the most important aspects of water system operation were during infrequent extreme events where failure could lead to significant negative consequences for populations. Hashimoto proposed a three component measure named 'RRV' which included reliability (likelihood of failure), resiliency (speed of recovery) and vulnerability (magnitude of consequences). RRV has principally been applied in the analysis of water distribution reservoirs but is currently limited to application assessing relatively simple systems (Fowler et al., 2003; Kjeldsen and Rosbjerg, 2004; Wang and Blackmore, 2009).

A similar approach was adopted by Moy et al (1986), where the maximum time duration of failure was used as resilience. These studies were predominantly focused on reservoir operation where failure duration was of paramount importance. The focus on failure duration implies that short failures are insignificant, however in practice a short failure may still have a large magnitude and therefore a large consequence. Surface water management studies indicate that flood damage costs tend to be linked to peak depth rather than peak duration, as inundation is typically fast acting but short lived (Penning-Rowsell et al., 2010);

However, the concept of assessment based on peak rather than mean flood depth remains applicable.

One prominent characterisation of resilience is presented by Fiksel (2003, 2006), in which resilient systems are shown as to retain functionality over a wide range of possible system states, therefore meaning that the system is likely to operate more effectively during a disturbance. Resilience is contextualised as an alternative to traditional resistant systems, which are designed to recover quickly from a perturbation within a narrow band of tolerance, but cannot to operate under a wide range of conditions (Figure 2.1). Fiksel also presents a resilient system with multiple equilibrium points, a concept related to ecological systems where the balance between components can shift, this will be further explored in Section 2.4.3.

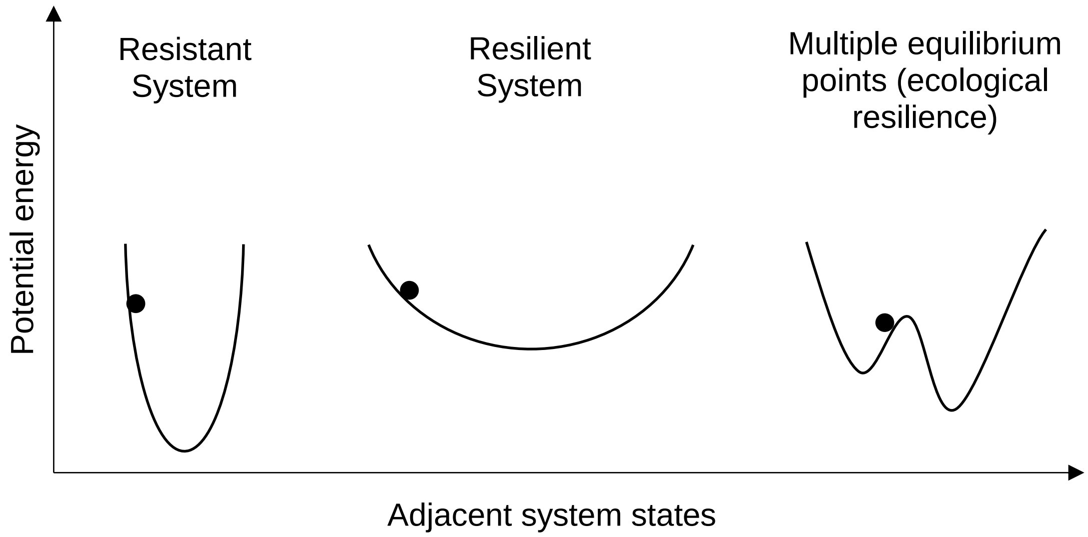

Figure 2.1: Examples of system behaviour (Fiksel, 2003)

Operation across a range of conditions is supported by Butler et al (2014, 2016) who indicate the role of resilience as a methodology to reduce the impacts from unexpected events, beyond an everyday level of service. This relationship is presented in Figure 2.2 which develops a conceptual model indicating that resilience accommodates unexpected high magnitude/ low probability events falling outside of normal planning policy, which is covered by reliable system design (Section 2.3).

The ‘Safe and SuRe’ (Safe, Sustainable & Resilient) project provides a definition of resilience which includes the concept of managing exceptional conditions by defining the term as: "*The degree to which the system minimises level of service failure magnitude and duration over its design life when subject to exceptional conditions*" (Butler et al., 2014). The definition is also expressed within Butler et al (2014) as:

Resilience = min (**failure:** magnitude, duration) Equation 2.2

This creates a quantitative measure of resilience which encapsulates the concepts of managing extremes through reducing magnitude and bouncing back by minimising the duration of failures. The Safe and SuRe project specifies resilience as general or specified. General resilience is the ability of a system to as minimise failure to all threats. Specified resilience is the ability of a system to minimise failure to a particular threat based on an operational goal. Specified resilience can be represented graphically through its relationship between consequences and level of service (Figure 2.3).

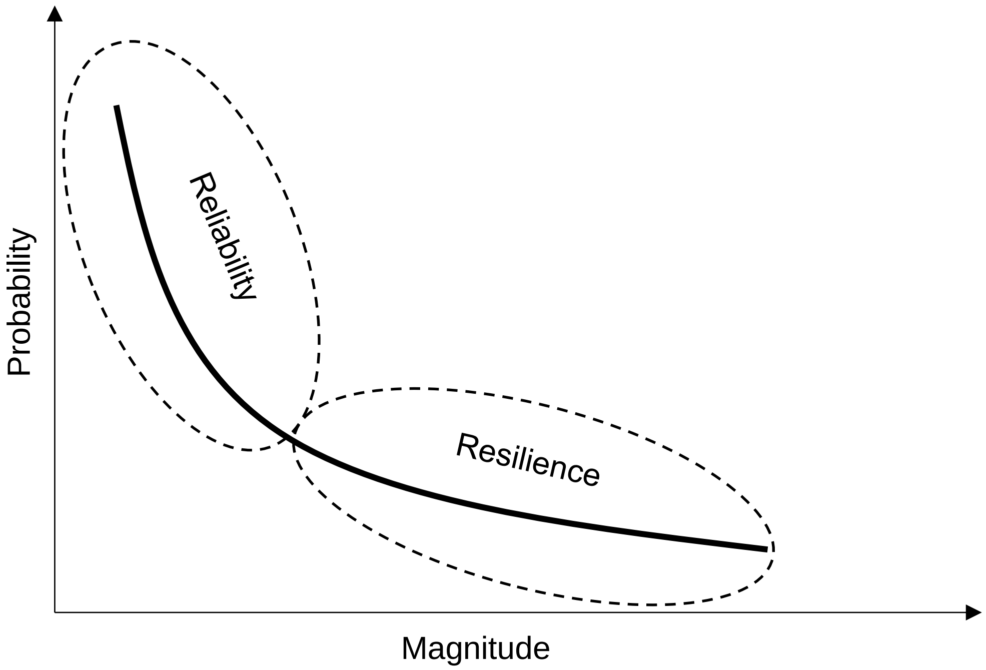

Figure 2.2: The relationship between reliability and resilience (Butler et al., 2014)

The Safe and SuRe project also discusses resilience in terms of properties and performance. A property may contribute towards resilience, but does not fully determine whether a system is resilient. Properties may include factors such as connectivity, reliability, resistance, redundancy and adaptability (Cabinet Office, 2011). Resilient performance is the emerging behaviour of the system as a result of a combination of properties. Confusion regarding this concept is common within literature (Butler et al., 2017). To understand resilience it is crucial to measure performance, rather than summarising properties which may not accumulate to a desired outcome.

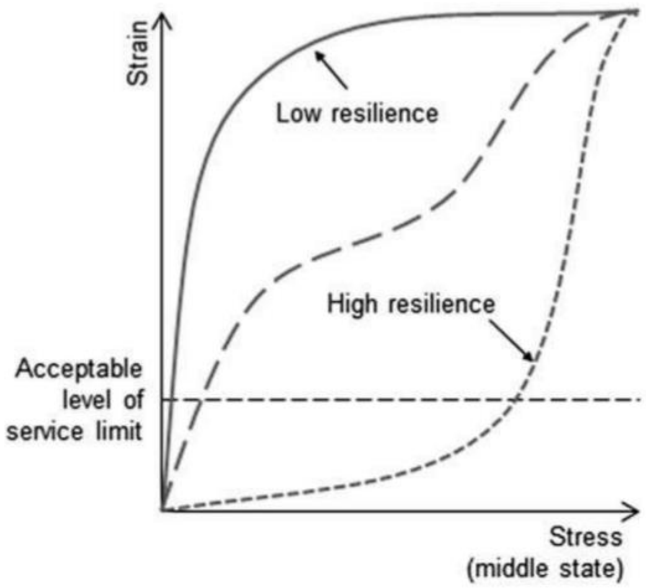

Figure 2.3: Specified resilience to a determined service level (Butler et al., 2017)

2.4.2. Measuring resilience in water engineering

A long-standing critique of resilience science has been a lack of operational and quantitative application of theories (Aldunce et al., 2015). This is of particular note in complex systems, such as surface water management in cities. This section will outline flood resilience measurement techniques from academic literature.

Several studies have compared stormwater interventions, although rarely in the context of resilience (Section 2.2). Lamond et al (2015) conducted a systematic literature review, describing a range of SUDS in detail and summarising applications where each was likely to be most effective. The study concluded that the application of SUDS attributes increased urban resilience, but the research did not expand analysis into a quantitative assessment.

The CORFU project (COllaborative Research on Flood resilience in Urban areas) (Djordjević et al., 2011) quantifies the cost effectiveness of flood reliability and resilience interventions across eight case study cities including Barcelona, Beijing, Dhaka, Hamburg, Mumbai, Nice, Seoul and Taipei. The study incorporated a range of flood hazards, and was not focused on surface water. Cost analysis was achieved using hydraulic models coupled with GIS and flood resilience was measured using an index (Batica et al., 2013). Assessment was conducted at a strategic city scale, with flood hazards simulated using a range of industry standard 2D models. Flood resilience assessment was made through assigning scores to natural, physical, social, economic and institutional dimensions of resilience at different scales. Scores ranged from zero (very low, not available) to five (high, requirements fully provided). Scores were assigned at building and city scales, repeated across a range of events and weighted to accumulate an index score. The study acknowledged the subjectivity of the scoring and weighting system, leaving a gap in research for a quantified and operational measure of resilience, such as economic cost, across events.

Related research was undertaken by Hu and Khan (2013), who developed a 'five layer framework' to assess resilience. This framework was created to assess city level resilience by including 'five dimensions of resilience': reflect, resist, relief, response and recovery. Each dimension was linked to a time period before, during or after an event and measured through a range of indicators. This approach relies on scoring each indicator, rather than on an objective measure (i.e. economic cost of failure). The framework has been used in conjunction with city growth models applied to a qualitative assessment of Dhaka's resilience and planning processes.

The Safe and SuRe project (Butler et al., 2014, 2017) has developed a framework which links threats through to the consequences (Figure 2.4). This splits analysis of a system into four components and identifies opportunities for actions between each component as: mitigation, adaption, coping and learning.

The project has proposed a definition for resilience which allows a quantification of resilience in a practical setting by measuring the failure magnitude and duration during extreme events (Section 2.4.1). This approach has been applied to a range of challenges, including: wastewater treatment (Sweetapple et al., 2014, 2017), water distribution (Diao et al., 2016), urban drainage (Mugume et al., 2015) and flooding (Casal-Campos et al., 2015).

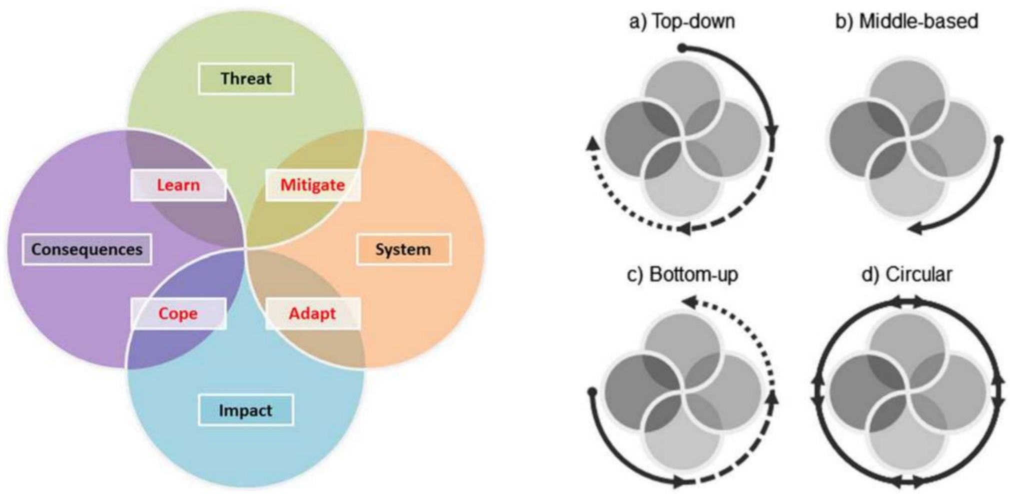

Figure 2.4: The Safe and SuRe Framework (Butler et al., 2014, 2017)

As of yet this research has not been applied to assessing surface water flooding interventions. The closest this assessment style has come to investigating urban drainage is the global resilience assessment method applied by Mugume et al (2015). This approach studied a 1D sewer network and used flood cones to represent surface flood volume (Section 2.2.2). This approach accommodates a wide range of threats which lead to urban flooding by examining pipes (links) in an urban drainage system. The method moves away from hydraulic engineering approaches which rely on computational modelling to predict causes of threats by instead focusing assessment on a 'middle state' analysis which examines the effects, rather than cause and probability, of system failure. Middle state analysis accommodates uncertainties and unknown causes of failure which cannot always be quantified (Stirling, 2010; Bond et al., 2015). Link failure permutations are applied to identify changes to system performance when one or multiple pipes fail, irrespective of probability, allowing analysis to build up an understanding of functionality for all hazards. From this analysis a resilience index describing residual functionality can be constructed. Similar approaches have been undertaken in highways emergency planning and are referred to as multiple centrality assessment (Porta et al., 2008).

Recent developments of this approach by Sweetapple et al (2018) have developed this into a standalone analysis tool, however this is only applicable to simplified systems such as pipe networks, and cannot accommodate the spatial complexity and computational resources required to examine surface water flooding. A similar middle state analysis was undertaken by Lhomme et al (2013) who applied GIS methods to assess flooding on road networks in New Orleans. These approaches focus on failure of a network (middle state analysis) rather than the reasons for failure and as such are able to accommodate unknown failure mechanisms within an assessment (Bond et al., 2015). However, global resilience analysis approaches have two common limitations: they are only suitable for application across a well-defined network (i.e. pipes, roads); and methods output resilience indicators such as connectivity and centrality which are not directly communicable to a wide range of stakeholders. Furthermore, by focusing analysis on a network and gauging resilience through connectivity measures, the most significant determinant of resilience is the structure of the network itself. Restricting the scope of analysis to the network means that interventions which may have an impact outside of a network cannot be included within analysis. Spatial disaggregation of complex urban surfaces are difficult to represent using this approach, so alternative approaches are best suited to modelling surface water flooding.

Although not applied to storm water, Hashimoto et al (1982) developed a three component measure which included resilience (RRV, Section 2.4.1). This method has predominantly been applied in the measurement of water supply systems (Fowler et al., 2003; Kjeldsen and Rosbjerg, 2004). The RRV (reliability, resiliency and vulnerability) method considers both failure duration and consequence, therefore shows parallels to the resilience formulation derived as part of the Safe and SuRe project (Butler et al., 2014, 2017).

The FREEMAN (Flood Resilience Enhancement and Management) project (Schelfaut et al., 2011) is a general flood resilience project aiming to bring flood resilience into practice. The study focuses on governance requirements to achieve resilient systems, highlighting the need for institutional interplay, flood management tools and improved risk communication. The paper outlines these factors in relation to several flood management case studies, but does not expand to discussing how resilience can be implemented or improved in practice. This is similar to many other studies which adopt similar high level and qualitative assessments of resilience (Hu and Khan, 2013; Restemeyer et al., 2013; Hollnagel, 2014; Kaklauskas et al., 2014). This subset of literature provides a general discussion of the properties of resilient systems, but does not connect these through to the performance of specific intervention strategies.

System dynamics (SD) modelling has also been used to quantify resilience to general flooding. Gotangco et al (2015) used SD to quantify household resilience to flooding in Manila in response to a gap in literature regarding the quantification of resilience (Cutter et al., 2010; Bruneau et al., 2012). A systems dynamic model developed by Simonovic and Peck (2013) is proposed and applied as a solution to this. Relationships are visualised though a SD model template and resilience is measured through a subjective scoring based framework. The main advantage of this approach is system dynamics ability to test leverage points within a system. This work shares similarities with previous SD resilience models in that it is intended to be descriptive rather than predictive, therefore applicable to exploring system behaviour rather than simulating the effects of specific interventions.

The challenge of predicting and optimising scenarios to develop resilient water infrastructure is addressed by contemporary literature regarding decision making under deep uncertainty (DMDU). This work measures resilience through evaluating strategy behaviour in response to changes, rather than a historic paradigm of predicting and intensively modelling system response to a likely outcome (Babovic et al., 2018b). DMDU achieves this through focusing analysis on many possible scenarios and interventions to characterise a systems response to an increasing stress, for example increasing rainfall intensity. A key element of a robust DMDU analysis is inclusion of a wide range of scenarios through many simulations (Lempert et al., 2013; RAND, 2013), which requires consideration of the trade-off between computational complexities of modelling versus the detail required for effective decision support. The appropriate level of this trade-off is subject to current academic discussion and is context specific, but literature suggests that in practical terms it is best achieved through ongoing stakeholder engagement.

Resilience is also evaluated through exploring indicators representing desired system properties. For example, Kaźmierczak and Cavan (2011) apply analysis of 26 indicators to assess the vulnerability of populations to flooding. The study only identifies broad interventions, such as promoting property level flood protection, but highlights the need for future analysis to tailor interventions to spatial variation in land use and housing. Other studies such as the Cabinet Office (2011) report on critical infrastructure resilience, the UKWIR (2013) good practice guide for resilience planning and Bahadur et al's (2013) study characterising resilience, all follow a similar approach by measuring resilience based on system characteristics. These studies prioritise installing measures which promote properties such as ‘high intervention diversity’, ‘recovery’ and ‘inclusion of local knowledge’ However, methodologies focused on system properties only assess proxy measures of resilience through subjective scoring, rather than evaluating the emerging performance of specific strategies (Butler et al., 2014). As such, there remains a need for new quantitative approaches capable of measuring system and intervention performance.

Many other studies propose flood resilience assessment through frameworks. These are typically generalist and are therefore are attributable to a range of contexts, including surface water management. Examples of frameworks include: Resilience of the built environment (Hollnagel and Fujita, 2013; Hollnagel, 2014); reliability, resistance, recovery and response framework (Cabinet Office, 2011; Ofwat, 2012); flood resilience assessment using the service risk framework (Conroy and Webber, 2013); regional resilience framework (Foster, 2006); resilience-vulnerability of urban areas (Romero Lankao and Tribbia, 2009); flood resilience of cities (Restemeyer et al., 2013); the resilience thermometer (Kaklauskas et al., 2014); and, resilience of critical infrastructure (Labaka et al., 2016). None of these frameworks have been applied to surface water management or intervention evaluation.

Evaluation of existing literature regarding measuring resilience in the field of water engineering indicates that current approaches deliver strong methodologies for measuring simple water systems, for example global resilience analysis for pipe networks, however there is currently a lack of operational assessment for more complex surface water management. Where studies do assess surface water management these tend to evaluate resilience through describing intervention properties, rather than performance. This is typically undertaken through qualitative descriptions or indices aimed at a limited niche application. Consequently, a gap exists regarding quantitative assessment and comparison of intervention performance during extreme events. Work in this area has been undertaken by the Safe & SuRe project (Butler et al., 2014, 2017), however this has only limited application to surface water flooding (Casal-Campos et al., 2015; Mugume et al., 2015).

2.4.3. Lessons learnt from resilience applied across other fields

Resilience is a broad term applied across many disciplines (Aldunce et al., 2015; Pizzo, 2015). Although these are not the focus of this project it is important to frame resilience in regard to its wider application, highlighting where additional benefit can be achieved through transferring knowledge to the field of engineering and surface water management. This section of the review will explore the pre-eminent resilience narratives from the fields of ecology, organisational planning and urban planning. It should be noted that resilience is also commonly cited in many other fields, however the disciplines discussed here are most relevant to a thesis evaluating surface water management.

Ecological resilience

Much of the discussion of resilience within academic literature is influenced by Holling's 1973 paper, 'resilience and stability of ecological systems' (Holling, 1973). Holling frames resilience in relation to two types of system behaviour, stability and resilience. Stability is defined as the "*ability of a system to return to equilibrium after a temporary disturbance"* and emphasises minimising fluctuation and enhancing a rapid return to a desired system state following system failure. Maintenance of a pre-determined equilibrium in a predictable world has similarities to engineering resilience, which is grounded in minimising failure by preventing movement from a desired fixed state or performance goal (Vis et al., 2003; Butler et al., 2014; Linkov et al., 2014; Pizzo, 2015). Ecological resilience instead concentrates on maintaining relationships and basic system states during change. Holling defines resilience as a "*measure of persistence of systems and of their ability to absorb change and disturbance and still maintain the same relationships between populations or state variables*".

The concept of 'maintaining relationships between populations or state variables' is easily applicable to an ecological context due to the constantly shifting dynamic equilibrium in which populations and ecological systems exist. It is straight forward to imagine one population changing in response to a disturbance and this cascading across other variables within a system, ultimately leading to a shift in equilibrium but a relatively unchanged system function. An example of this is Kolmogorov's predator prey relationship model (Hoppensteadt, 2006): Simply explained, growth in the prey population will cause a spike in the predatory population, resulting in a subsequent decline in the prey population and a corresponding drop in the predatory population until the lack of predators allows the cycle to repeat. The population of both predators and prey are therefore in dynamic equilibrium. Expanding the scope of the equilibrium by including relationships across a whole ecosystem demonstrates how ecological systems with adaptable populations can fluctuate in a dynamic equilibrium. It is also possible for sudden shifts to dramatically change an established trend of fluctuations and shift the equilibrium into a new regime, for example due to the inclusion of a new predator (Scheffer et al., 2001), representing a threshold over which the system loses stability (Steffen et al., 2007; Rockström et al., 2009). Systems can be robust to certain frequent disturbances, and very fragile to infrequent threats, this is referred to as ‘highly optimised tolerance’, and draws parallels with design standard planning in engineering practice (Folke et al., 2010).

This description of a system works well in the context of ecological systems consisting of fluctuating populations, however the change in relationship is hard to apply to an engineering system where components generally operate in one of two states: function or fail. One way of conceptualising this would be to imagine a fixed ‘digital’ level of function in an engineering system (the system functions or fails) compared to the ecological ‘analogue’ diversity in system components. Consequently there remains a distinction between ecological resilience (adapting function to accommodate disturbance) and engineering resilience (rapidly recovering from a disturbance) (Walker et al., 2004; Butler et al., 2014). This difference is emphasised by literature which highlights that ecological systems are in a state of constant flux and do not return to the stable equilibrium, which is the goal of engineering resilience (Pickett et al., 2004). This is characterised by Davoudi et al (2012) as bouncing back (engineering) versus bouncing forth (ecology). Recent research has applied bouncing fourth within engineering through implementing anti-fragility in urban water systems, representing a paradigm whereas systems evolve following disturbances (Babovic et al., 2018a).

Despite the delimitation between definitions it is arguable that water engineering should adopt aspects from both engineering and ecological approaches in order to maximise the benefits of adopting a resilience based approach. Holling emphasises that ecological resilience can be achieved through recognising ignorance, ambiguity and uncertainty in decision making and ensuring the ability to facilitate adaptive management through keeping options open (Holling, 1996). This concept is applicable to surface water management through designing interventions which can be gradually added to a system, in a way analogous to the adaption in ecological systems. Surface based green infrastructure can provide this adaptive capacity through facilitating an incremental surface water management pathway whilst avoiding costs of predicting future infrastructure requirements for increasing the size of subterranean sewer networks (Kunapo et al., 2018; Lu et al., 2018). This level of analysis requires modelling capable of simulating performance of many future interventions and scenarios.

Another key message from ecological literature is the inherent resilient properties of decentralised systems, capable of managing failure of an individual component through heterogeneity (Holling, 1996). From an ecological perspective, this results in strategies aimed at maintaining many populations which can fulfil similar function, for example a rich species diversity. From an engineering perspective this can be translated to many interventions which operate independently across a catchment, so system function can be continued in the case of individual components (interventions) failing.

The ecological approach offers important lessons which can be adapted into an engineering approach: Namely evaluating system response to infrequent hazards, encouraging heterogeneity through evaluating novel interventions and investigating the effect of distributed solutions. A limitation of ecological resilience is that application is broadly theoretical, with little possibility of a general practical application. Over time this has been compounded with a fractured narrative, which diverges from Holling (1973) by subtly altering definitions and approaches. Multiple sub-meanings are embedded into definitions of resilience, making them unworkable and unclear (Cutter et al., 2010; Prashar et al., 2012; Pizzo, 2015). Where frameworks are developed they are typically qualitative and based on properties or a question-answer process with no consideration of enumerating option scores to assist decision making (Walker et al., 2004; Gallopín, 2006; Folke et al., 2010). This trend is similar to that observed in other disciplines and as such adds weight to the argument that actionable and quantifiable resilience approaches need to be implemented.

Organisational resilience

Although not a specific 'discipline', there is a wide body of literature which links resilience to organisational and governance strategies. This body of literature focuses on the importance of establishing management processes to embed resilience within decision making (Adger, 2000). Organisational resilience occupies an extensive body of literature, not all of which is relevant for this project, therefore this section contains relevant highlights and trends from research where useful lessons can be applied to surface water flood management.

The Resilience Alliance, an organisation of academic institutions which promotes resilience in socio-ecological systems suggests four steps to include a resilience approach within management (Walker et al., 2002, 2004; Bond et al., 2015):

- Establish key system attributes through stakeholder engagement.
- Identify drivers through stakeholder engagement and expert vision.
- Undertake quantitative resilience analysis.
- Evaluate management and policy implications of findings.

The Resilience Alliance process emphasises the need to bring stakeholders together and clearly communicate options (and associated uncertainties) to develop resilience enhancement strategies. This integrated vision focuses on keeping options open for participatory management in light of uncertainties associated with long term planning. This approach can be translated into surface water management through assessment of many scenarios and engagement regarding a range of intervention options.

Brown and Farrelly (2009) conducted an analysis of 53 studies to the barriers of delivering sustainable urban water management. The authors reinforce the messages from the Resilience Alliance through highlighting a requirement for future urban water management policies to include adaptive, co-ordinated and participatory approaches to overcome socio-institutional barriers in water management. In particular, the study calls for co-ordination between multiple stakeholder organisations, which can be achieved through transparent communication of uncertainties and management options (Jabeen et al., 2010; Lopez-Marrero and Tschakert, 2011).

Resilience assessment within organisational management is typically qualitative, with studies structured through discussion of the merits of multiple scenarios (Romero Lankao and Tribbia, 2009; Wardekker et al., 2010; Gómez-Baggethun et al., 2012; Watts et al., 2012; Kaklauskas et al., 2014). Scenarios allow analysis of possibilities through stakeholder and expert engagement and can accommodate uncertainty in long term decision making through discussing a wide range of possible scenarios. In most instances scenarios are built up through expert discussion and tailored to specific situations (Gómez-Baggethun et al., 2012; Watts et al., 2012). However more general scenario building has also been undertaken where constructed instances are applicable to a broad range of themes (Kaklauskas et al., 2014; Casal-Campos et al., 2015).

A conclusion that can be drawn from this literature is that communication of interventions is a crucial part of decision support towards resilience. Effective communication strategies are found to consist of a variety of methods including geospatial representation and stakeholder engagement (Foster, 2006; Jabeen et al., 2010; Wardekker et al., 2010; Lopez-Marrero and Tschakert, 2011; Scolobig et al., 2015). A common thread from this body of work is that studies endorse the concept of increasing resilience but are largely theoretical. Studies typically present broad, non-specific and qualitative frameworks which are unsuitable for detailed decision support. Developing specific actions using qualitative or semi-quantitative subjective scoring methods can propagate institutional inertia or bias, rather than developing new and effective interventions (Marlow et al., 2013). As such it is crucial that resilience frameworks enable stakeholder participation and communication through quantitative analysis of interventions.

2.5. Chapter conclusions

This chapter has responded to Objective Two through evaluating the techniques applied for comparing surface water flood management intervention performance across design and extreme events in urban catchments. The main message from this chapter is that a current gap exists regarding methodologies to screen the many possible interventions and scenarios which should be considered for reliable and resilient surface water management. This message can be broken into four key findings:

- Comparison and selection of interventions can be undertaken using a wide range of qualitative and quantitative approaches, however a trade-off exists between fast but low resolution methods, which are not suited to support decisions requiring a spatial understanding of intervention hydrology, and high resolution but computationally intensive flood simulations, which are not suited to support decisions requiring analysis of many scenarios.
- New approaches, such as cellular automata flood models, provide an opportunity to enhance consideration of potential strategies through evaluating the many permutations of intervention type, scale and distribution possible in urban catchments. Despite documented speed and accuracy of these techniques, relatively recent development of the technology means application to surface water management is currently limited to coarse resolution modelling over large catchments, rather than applying efficiency towards analysis of many scenarios and interventions.
- Design standard planning cannot accommodate residual risks caused by extreme events, particularly given future threats of climate change, urbanisation and population growth; therefore future resilient surface water management is required.
- Current inclusion of resilience within flood management (and wider literature) tends to be applied through qualitative frameworks or specific niche applications. Although current research is addressing quantitative resilience measurement, for example deep uncertainty frameworks and global resilience analysis, gaps remain regarding application of actionable and quantitative resilience planning encompassing failure magnitude and duration during extreme events in ways which are easily communicable to stakeholders.

The next chapter in this thesis will develop a framework to evaluate interventions in response to the gaps identified within current literature. The framework is based around a capability of quantitatively assessing flood dynamics of intervention strategies across a wide range of scenarios, encompassing design standard and extreme events.

3. DEVELOPING A RAPID SCENARIO SCREENING FRAMEWORK

This chapter responds to Objective Two: 'Develop a screening framework to enable assessment of many intervention scenarios at the urban catchment scale'. In response to the research gaps identified within the literature review, this chapter develops a framework to screen many surface water flood management scenarios and develop strategic evidence which can later be applied to steer detailed design. The novelty of the framework lies in its capacity to quantitatively assess hundreds of intervention scenarios whilst retaining a simulation of high resolution flood dynamics.

This chapter presents an overview of the framework, the fundamental science of the underlying flood models and the requirements for implementing each step in the process. This framework forms the methodology for intervention assessment applied later in this thesis.

Research presented within this chapter is published in: 'Rapid assessment of surface water flood management options in urban catchments' (Webber et al., 2018a) and 'Rapid surface water intervention performance comparison for urban planning' (Webber et al., 2018d).

3.1. Framework structure

The framework prioritises easily accessible data and utilises a computationally efficient surface water routing model, capable of generating results to steer further investigations at a low resource cost. Fast data entry and processing speeds are achieved through simplification of land use and intervention characteristics, alongside clear performance metrics.

The framework (Figure 3.1) is split into four steps: characterise study area, represent intervention scenarios, simulate scenarios and assess intervention performance. The data requirements and actions within each of these steps are detailed in subsequent sections of this chapter.

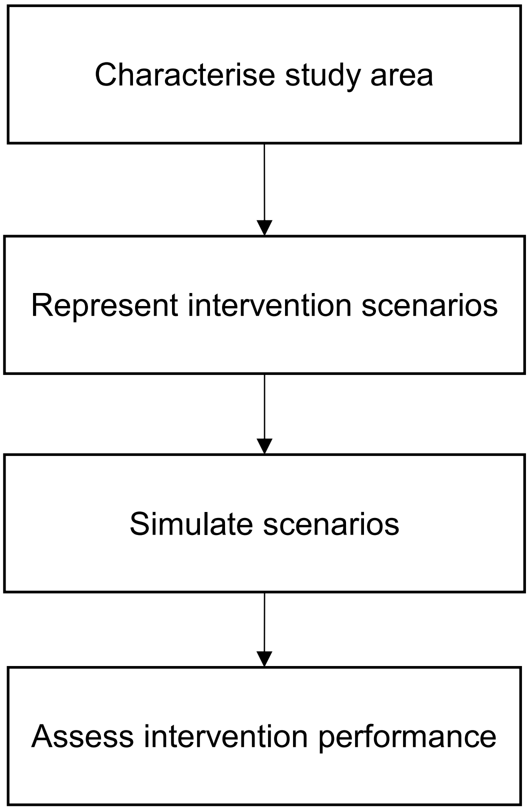

Figure 3.1: Framework for surface water intervention assessment

3.1.1. Scope of framework

Engineering research has the potential for direct translation from science to real world application and benefits. A framework responding to the need for enhanced catchment screening requires implementation using data likely to be available at the early stages of a design project, where screening takes place; therefore, processes and data sources applied in this research are intended to be applicable using accessible data and low processing requirements (Mikovits et al., 2015; Löwe et al., 2017).

Accessible data are classified as sources which would commonly be freely available to a UK researcher or consultant. These formats are typically available through open source products, project partner databases or educational licenses. Where processing is required, it is intended that this should be applicable quickly and using methods possible through commonly available software, such as GIS. Catchment and intervention screening may be undertaken in a variety of contexts, therefore it is important that each stage of the framework is adaptable to data sources applied at a range of resolutions, depending on the purpose of each study. Specific data requirements and processing steps are described in detail in corresponding sections of this chapter.

It should be noted that the intention of this research is to generate a screening process to support, and not replace, requirements for detailed hydro-dynamic modelling. Fast implementation of the framework requires assumptions and simplifications in representing several physical processes commonly required for detailed design. This is a fact with all modelling tools, which can only ever provide a simplified version of reality to generate answers for specific purposes, supported by the frequently cited aphorism "*all models are wrong, but some are useful*" (Box, 1976). All assumptions are detailed within this chapter, and their effects are examined in Chapter Four, which validates the framework described here.

3.1.2. Framework modelling architecture using the CADDIES flood simulation model

Requirements for the framework are informed by the input data types of the underlying flood model applied, therefore it is important to introduce this model before outlining each stage of the setup. Section 2.2.2 outlines the potential advantages of urban flood modelling using cellular automata. In response, the framework utilises the 'Cellular Automata Dual-DrainagE Simulation' (CADDIES) model for flood simulation (University of Exeter, 2017). This section explains the underlying assumptions and mathematical basis of this novel modelling approach to enable the reader to understand how the framework has been structured.

CADDIES is a cellular automata based surface water modelling tool developed at the Centre for Water Systems, University of Exeter (Ghimire et al., 2013; Guidolin et al., 2016; University of Exeter, 2017). The model uses a regular square grid based cellular automata system to rapidly simulate overland flow. This avoids the computational cost of solving complex hydrodynamic equations via application of simplified cellular automata transition rules, resulting in increased computational speed versus traditional modelling techniques (Gibson et al., 2016; Guidolin et al., 2016). Utilisation of regular grids is well suited to execution across parallel and multiple core systems, leading to potential further increases in computational speed when applied using a GPU. When combined with high resolution 1D LiDAR the model is able to accurately simulate flow within an urban catchment (Gibson et al., 2016).

It should be noted that the CADDIES model has been developed prior to this PhD, however it has not previously been applied for rapid scenario screening in surface water flood management. The novel contributions made within this thesis arise from application of the existing CADDIES model as a component within a novel screening framework. Specifically, novelty is achieved through developing and applying a methodology to represent interventions through parameterising cells within the existing 2D cellular automata modelling structure; and through novel application of the framework as an option screening tool to evaluate many simulations and develop new insight into intervention performance across design standard and extreme rainfall.

The next section of the thesis describes the underlying science which has previously been published in order to provide context and background to support the novel developments described throughout the remainder of the thesis (Ghimire et al., 2013; Guidolin et al., 2016; University of Exeter, 2017).

Governing equations applied in CADDIES

Figure 3.2 presents the modelling architecture applied in CADDIES. The model utilises a cellular automata across a regular rectangular grid. Movement is controlled by routing water between neighbouring cells using a Von Neumann neighbourhood (Von Neumann and Burks, 1966; Guidolin et al., 2012). A Von Neumann neighbourhood allows water to travel between linked cells in four directions, north, east, south and west (Figure 3.2).

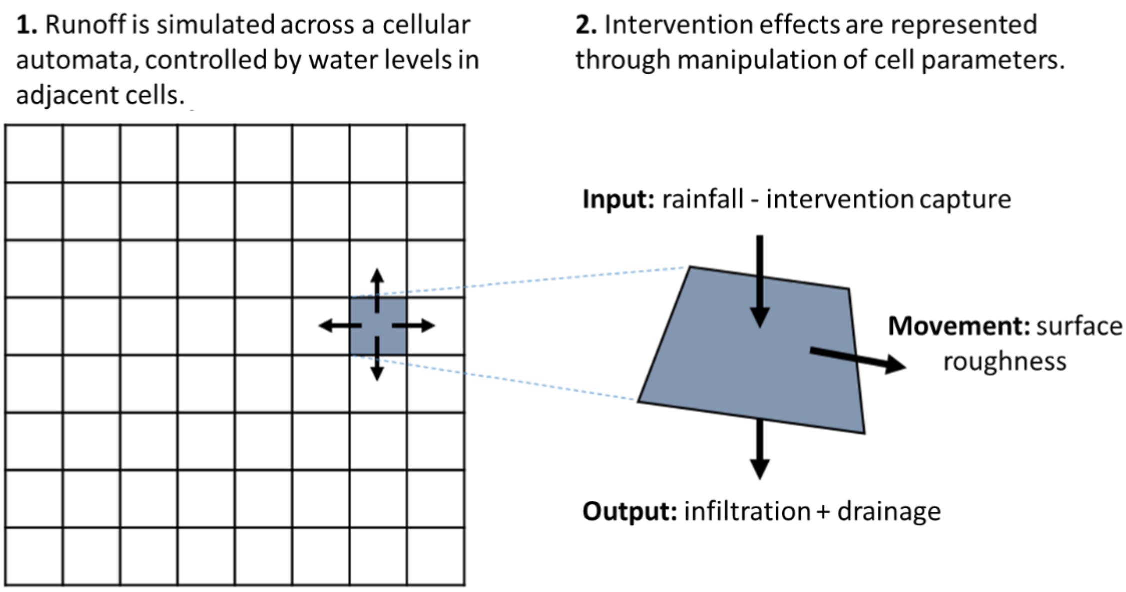

Figure 3.2: CADDIES model architecture (Webber et al., 2018d)

The model is simulated through a series of time-steps. At each model time-step a transition rule is applied to specify water movement between cells in each neighbourhood. The transition rule is applied in series of steps, outlined below.

The first calculation at each time step determines the water level in each cell by adding input and subtracting output volumes from the current cell volume (Figure 3.2).

The direction of movement is then determined by ranking water levels within the neighbourhood on a cell by cell basis. Only the outflow from the central cell is considered, this provides the advantage of being able to calculate each cell independently, thus saving time versus traditional shallow water (St Vennant) equations which require solution of partial differential equations. Water can move in multiple directions where the water volume in the source cell is larger than the free space in multiple neighbouring cells (Figure 3.3). Every cell is evaluated to calculate flow directions across the entire model domain. Analysis on a cell by cell basis also presents the opportunity to efficiently parallelise computational implementation on a GPU (Gibson et al., 2016).

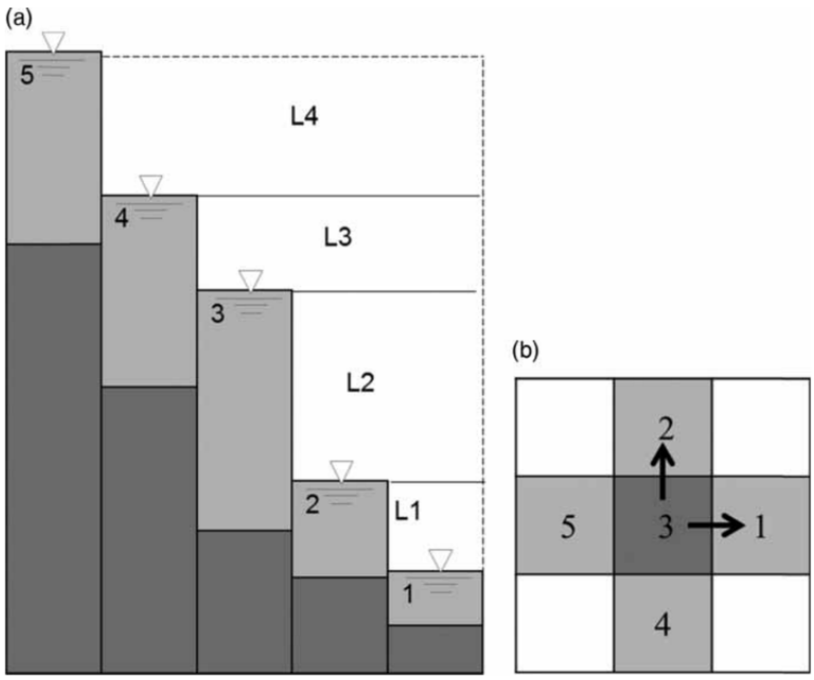

Figure 3.3: Ranking cells and calculating 'free volume' to determine the direction of water movement (Ghimire et al., 2013)

Water volume flux is controlled by water volume in the source cell distributed according to the ranking identified in Figure 3.3. This assumes that cells will only receive water from neighbours with a higher water level and that cells will reach equivalent water levels within the surrounding neighbourhood.

Simulation resolution is controlled via a time step, specified as a function of water velocity. The simulation time step specifies the frequency of calculations within the simulation. Time steps are controlled by an equation which relates the length of a cell and maximum velocity of runoff to the required number of calculations per second.

$$\Delta t = \frac{\Delta l}{v_{max}} \times \alpha \qquad \text{Equation 3.1}$$

Where Δt is time-step (s), v_{max} is maximum velocity (m/s) and α is dimensionless scaling factor.

Smaller time steps capture flow movement more frequently and are required when calculating fast moving flows. Small time steps increase the number of calculations required within a simulation and consequently lead to a decrease in model speed. Larger time steps increase the speed of simulation through reducing the number of calculations and are appropriate where velocity is low relative to the length of the cell for which it has to travel.

Water velocity is controlled within each cell by Manning's and critical flow equations, i.e., Equations 3.2 and 3.3 respectively (Butler et al., 2018). Water movement is limited by a transferrable volume, calculated as the minimum of the total volume in the source cell and the total free volume available in a receiving cell (Ghimire et al., 2013).

$$v = \frac{1}{n} R^{\frac{2}{3}} S_0^{\frac{1}{2}} \qquad \text{Equation 3.2}$$

Where v is velocity (m/s), n is Manning's roughness coefficient ($s/m^{1/3}$), R is hydraulic radius (m) and S is hydraulic slope (-).

$$v_c = \sqrt{gd} \qquad \text{Equation 3.3}$$

Where v_c is critical velocity (m/s), g is gravitational acceleration (m/s^2) and d is water depth (m).

Combining Equations 3.2 and 3.3, the CADDIES velocity is calculated as (Ghimire et al., 2013):

$$v = min\left\{\frac{1}{n}\ R^{\frac{2}{3}}\,S_0^{\frac{1}{2}}, \sqrt{gd}\right\} \qquad \text{Equation 3.4}$$

In the CADDIES implementation, the hydraulic radius is equal to the water depth and the slope is equal to the water surface elevation slope.

Computational implementation of the CADDIES model

Simulation is controlled using an input file which communicates the parameters and global settings required to run each simulation. Discussion of the framework structure requires an understanding of how inputs are communicated to the model. This section outlines the procedure and format of specifying parameters.

The CADDIES model is implemented using four parameters which are used to calculate the water slope and time step, which in turn control the movement of water from and to cells. These parameters are elevation, input, output and roughness. Each of these parameters is specified on a cell by cell basis through parameter matrices. To save computational storage space, each parameter matrix contains codes which are indexed to a parameter value table; therefore implementation of each parameter requires a matrix, which specifies intervention type and location, and a value table, which specifies the exact value of each intervention.

Parameter matrices are formatted as '.asc' files. These files are a matrix composed of square cells. Each file specifies the co-ordinates (x and y), number of cells (x and y), cell size, a no data value (typically -9999) and a parameter index code for each cell in the matrix. Parameter value tables are formatted as '.txt' files which index parameter codes to values. Values can be fixed (elevation, output and roughness) or vary at defined time steps (input).

Cell elevation is specified as a fixed value in m. The cell elevation is used to represent the surface of the model and is explained fully in Section 3.2.1. This parameter is used alongside cell input, output and water level to calculate water depth at the start of each time step. The elevation matrix is populated with values and, unlike other parameters, is not linked to a value table.

Cell roughness is specified as a fixed value in terms of Manning's 'n' coefficient. This is used to represent different surface types and is applied within the CADDIES velocity equation to control the movement speed of water through the time step (Equation 3.4). Specifying cell roughness is explained fully in Section 3.2.2.

Cell output is specified as a fixed value in mm/hour. The cell output parameter represents the water leaving a cell in each time step and is used alongside cell elevation, water level and input to calculate water depth. Cell output is a sum of water removed through infiltration, the sub-surface drainage system, evapotranspiration and removal through interventions. Specifying components of the cell output rate are fully described in Section 3.2.2 (infiltration and evapotranspiration), Section 3.2.3 (drainage), and Section 3.3.1 (intervention effects).

Cell input is specified as a value which can be manipulated temporally to provide a changing input rate across a simulation. This value is used alongside cell elevation, water level and output to calculate water depth. Cell input is primarily used to represent catchment rainfall, although the parameter can also be applied to represent the effects from watercourses and other inputs, such as burst pipes and pumping. Procedures used to specify the components of the cell input rate are described in Section 3.2.4 (rainfall), and Section 3.3.1 (intervention effects).

Other parameters can be manipulated to control the speed, accuracy and outputs of the simulation. These are specified within the input file and are described in Section 3.4 of this chapter.

Application of CADDIES within the framework

The main advantage of the CADDIES model is fast assessment through efficient simulation which avoids the computational resource cost of solving shallow water hydro-dynamic equations. In turn this speed presents an opportunity to examine significantly more simulations than current standard techniques; therefore providing utility as an option screening tool, capable of evaluating many flood management scenarios.

Simplified representation of parameters into four user specified values also presents a flood model which can be quickly set up through specifying a matrix and value table for each parameter. Parametrisation also provides possibility of simplified representation of interventions to examine many effects within a single framework.

Sections 3.2, 3.3 and 3.4 describe how the CADDIES model architecture is implemented for flood risk management through characterising the study area, representing interventions and running many simulations.

3.2. Characterising study area

The first stage of analysis is to setup a representation of the study area. The study area constitutes a baseline scenario which can be adapted in subsequent steps of the framework by adding rainfall, interventions and adapting catchment parameters develop additional scenarios.

The study area consists of a computational representation of the key physical parameters which control surface water runoff across a catchment. Figure 3.4 shows how the study area is built from four key components, including the macro-topographical elevation profile of the landscape, the micro-topographical features such as buildings and roads and the characteristics of land use types. Representing the study area also includes identifying and specifying rainfall events to examine within analysis.

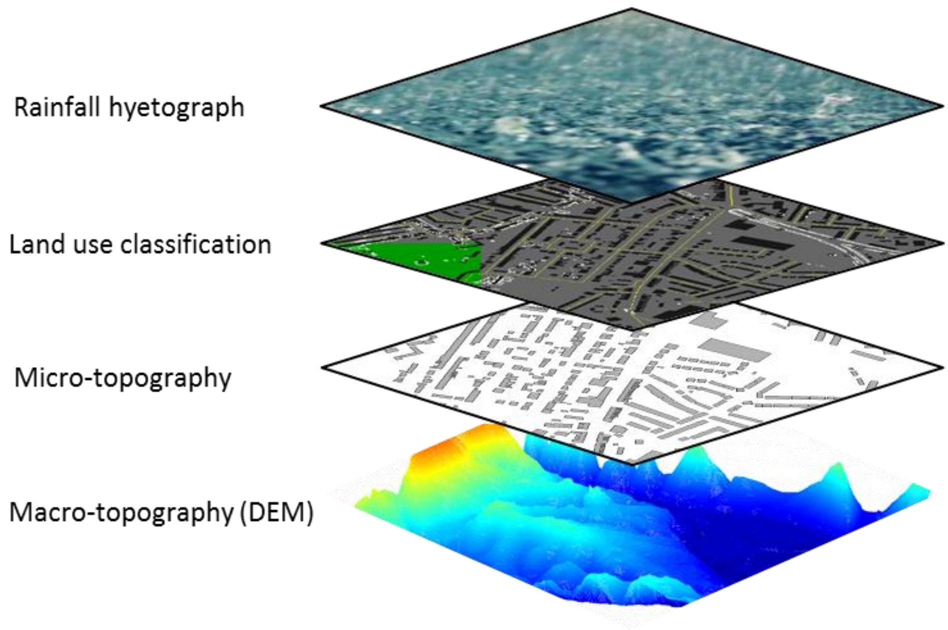

Figure 3.4: Required steps to characterise the study area within the framework

3.2.1. Representing the catchment elevation

The principal controlling factor in surface water runoff is the accumulation of runoff across a catchment driven by gravity and controlled by catchment topography. The topography is constituted of macro and micro features which channel flows across the surface (Figure 3.5). Macro-topographical features include the slope and elevation profile which control the broad scale of flow across landscape. In this study, macro-topography is defined as landscape features which are captured in an unprocessed input elevation model. Micro-topographical features are constituted of smaller elements which may have significant local influence by acting as channels for runoff to concentrate and coalesce, in this study defined as features which are below the spatial resolution or not represented within an unprocessed input elevation model. The resolution of input models is changeable, depending on data availability and model purpose, therefore these terms are relative to the specific contexts of each particular model application or study.

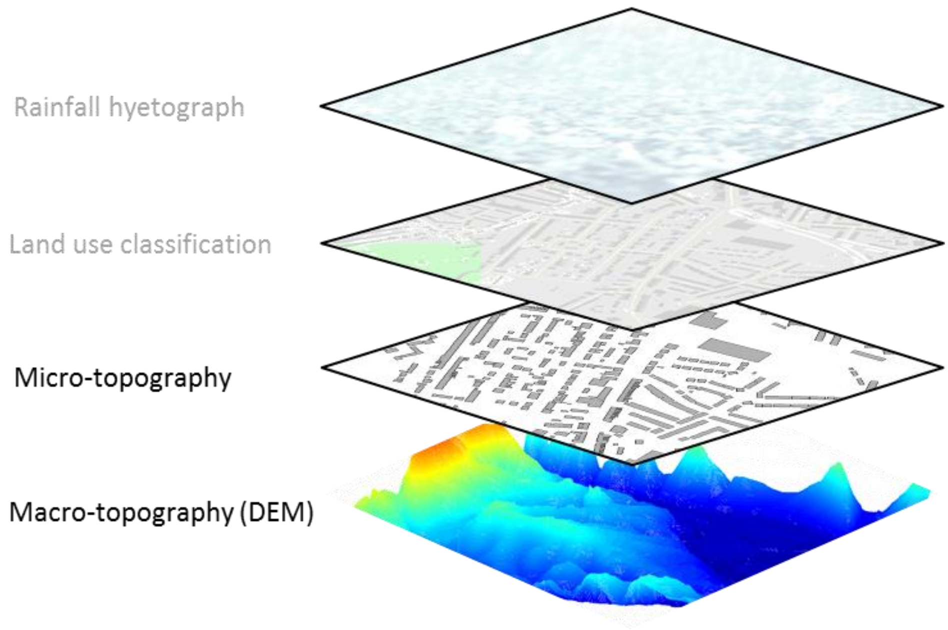

Figure 3.5: Representation of surface elevation using macro and micro topographical features

Catchment macro-topography

2D models specify macro-topography through digital elevation models (DEM's) which describe the elevation of surfaces within a catchment. The surface is specified into components which can be reported at varying shapes and resolutions. The majority of commonly available DEM products specify elevation using a regular grid format divided into square cells, where the elevation within each cell is considered an equal and level surface. Data constituting elevation models is typically captured using aerial survey techniques such as LiDAR. Other data capture methods such as manual surveys, photogrammetry or digitising existing mapping can also be applied in areas where specific small scale features need to be included or aerial surveys are unavailable. Accurate DEM's from LiDAR surveys are available for 70% of England through the Environment Agency's data archive and building footprints are available for the entirety of the UK through Ordinance Survey mapping (UK Government, 2017). High quality elevation data is commonly available from scientific or national data collection agencies in other countries (Hunter et al., 2007).

An important distinction exists regarding DEM and digital terrain models (DTM's). DEM's are processed to only include the elevation of the bare earth surface, whilst DTM's also include the elevation of features such as tree canopies (Podobnikar et al., 2000). When modelling runoff it is important to apply a DEM representing the ground level topography interacting with flows.

Resolution of the DEM is controlled by the number and scale of cells used to represent a particular area. Large rural flood plains are often simulated using coarse resolution grids, defined as above 20 m by 20 m per cell (Dottori et al., 2013) and may approach upwards of 100 m by 100 m per cell (Hunter et al., 2007), whereas modelling of surface runoff in urban areas requires a much finer resolution to adequately represent the influence of surface features. Studies use a range of cell sizes, although any cells less than 2 m by 2 m are typically considered high resolution, this is also referred to as 'very fine' resolution (Dottori et al., 2013). It should be noted that, despite this definition, the application of these terms is inconsistent amongst the literature. When using high resolution data, vertical errors can be considered approximately 10 – 20 cm (Fewtrell et al., 2011; Chen et al., 2012).

Elevation for this framework is represented using a regular grid with a high resolution DEM containing cells less than 2 m by 2 m. This data is commonly available as a direct download and as such application of this format minimises required processing times. Data of this resolution is typically only available for developed countries, however the framework can still function using coarser resolution macro-topography, provided the trade-off between cell resolution and representation of urban flow paths is acknowledged.

Catchment micro-topography

Recent studies indicate the high influence of small scale features on the movement of water across urban environments. Micro-topographical features include a range of items which will influence flow paths on a local scales, including drainage ditches (Bates et al., 2006), walls (Yu and Lane, 2006), fences (Mignot et al., 2006), road camber and kerbs (Fewtrell et al., 2011), buildings (Syme et al., 2004; Chen et al., 2012; Schubert and Sanders, 2012) and vegetation (Dottori et al., 2013). Changes in local flow conditions can have a significant impact on catchment flow dynamics so it is important to include these features within analysis.

Micro-topographical features which have a particular significance in urban environments include buildings and the road network (Syme et al., 2004; Fewtrell et al., 2011; Chen et al., 2012; Schubert and Sanders, 2012; Dottori et al., 2013).

Buildings can direct runoff around thresholds and so their effect can become significant across densely populated cities (Chen et al., 2012; Schubert and Sanders, 2012). Structures are typically the primary receptor of damage within a catchment and so need to be included within models. Buildings generate complex flow dynamics which vary with water depth and velocity depending on structural thresholds, integrities and internal composition (Mignot et al., 2006; Dottori et al., 2013). Building threshold levels mean that shallow water is likely to flow around the edges of structures. Deeper flooding may enter a structure and either pond or flow through to another exit. The scope of a fast assessment methodology means it is impractical to individually survey each building, particularly over a large urban catchment, therefore complex flow dynamics associated with buildings have been included through raising the building threshold level or changing parameters within the structure to slow the flow of water (Syme, 2008).

Roads also represent a significant conveyance mechanism for urban surface water (Fewtrell et al., 2011). During intense rainfall these are likely to act as channels for shallow flows which remain below kerb height. Other features such as ditches, railway embankments and walls may have a significant effect in particular urban areas (Dottori et al., 2013), therefore these items should be considered during the catchment screening process.

Processing steps

Macro and micro-topographical inputs are processed to generate one file representative of the catchment surface. Processing is achieved using GIS to combine several layers representing macro and micro-topography. Literature highlights that accurate simulation of catchment runoff can be enhanced by combining these features (Hunter et al., 2008a).

The DEM is used as the basis for the study area elevation model. The resolution of this format will vary depending on data availability. In the UK, DEM's representing urban areas are typically available at a cell resolution of 1 m by 1 m. Significant micro-topographical features are then incorporated into the DEM through overlaying layers containing these features, typically shape files representing building and road outlines. Buildings are included through raising the elevation of cells within building boundaries. Roads are included through dropping levels in the DEM by the kerb height to create a slight channel along the road system. The exact level of elevation change depends on the characteristics of areas being evaluated. In certain cases elevations will not be changed at all, and instead land use characteristics within boundaries of micro-topographical features will be altered to reduce or increase flow velocities and generate preferential flow paths.

Limitations

Incorporating features smaller than high resolution modelling grids is a frequently reported challenge in 2D modelling (Dottori et al., 2013). Inclusion of features through DEM processing will capture feature effects and enable simulation to capture flood dynamics associated with them. However, this approach cannot take into account the full dynamics of the urban environment during intense rainfall, when micro-topographical features may change in response to high energy flows. Aspects of the urban environment which may be moved by high energy flows include vegetation, fences, soils and, in extreme cases, cars (Mignot et al., 2006). It is impractical to collect data or simulate these features within screening models, nevertheless this limitation should be recognised when assessing simulations.

Practically, the most likely issue resulting from high velocity flows shifting features will be blockages to drainage system inlets. However, during flows of this magnitude these inlets are likely to exceed capacity and so this limitation has to be accepted, yet acknowledged (Dottori et al., 2013).

3.2.2. Land use classification

Land use is specified as one of four components of representing a study area (Figure 3.6). Differences in land use characteristics across an urban catchment will influence water velocities, infiltration, drainage and evapotranspiration rates. Application of the CADDIES flood model includes representation of these influences by changing input, output and roughness parameters in model cells (Figure 3.2). The roughness parameter controls the velocity of flow across a cell, output removes water from a cell at a set rate, whilst input adds water to a cell surface at a set rate. Spatial adjustment of these parameters is used to simply represent the physical characteristics of land use and interventions.

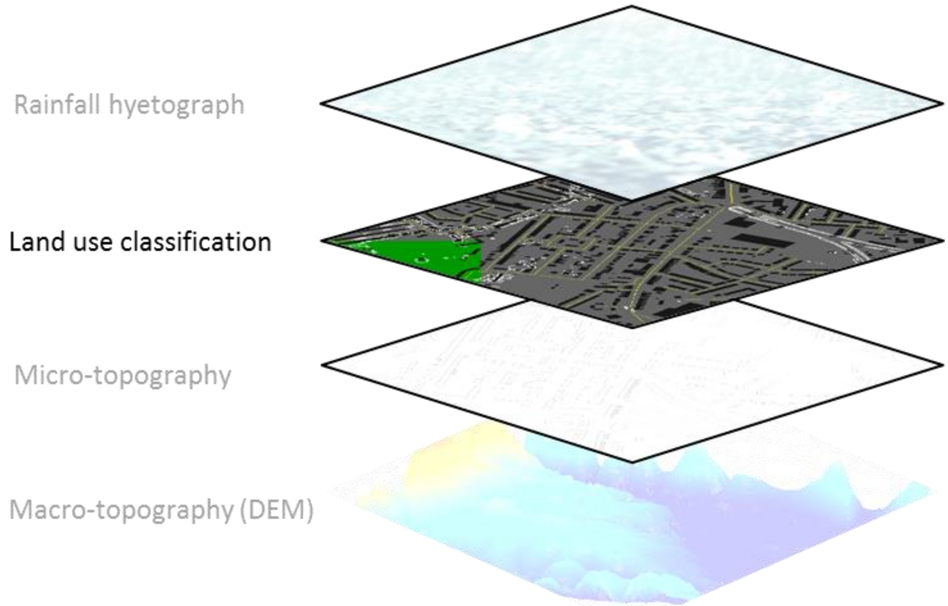

Figure 3.6: Representation of land use within the catchment

Classification

Land use is included within this framework through classifying each cell within a land use matrix and then indexing this matrix to a series of parameters describing the land use effect. The land use classification matrix is stored at the same resolution as the catchment elevation model.

Although in practice many land use types can be assigned to a cell, a framework utility focused towards a fast setup and simulation time is developed through assigning a relatively small number of land use types across a catchment. This creates a fast and simple classification using broad categories.

Land use classification is achieved using GIS to categorise cells into a matrix which codes each cell depending on the predominant land use type. Land use formats are typically available as shape files, which can be converted into a grid based format, suitable for CADDIES, using a GIS raster creation function. This can be achieved at a variety of resolutions. Where shape files do not match with a desired raster grid cell during conversion, the dominant land use by area is applied to that cell. Land use for the UK is specified using OS master map data to define land use zones within the study area. Any other land use classification is suitable for input into the model, provided the format can be transformed onto a regular grid. For very broad classification schemes it is possible to do this manually using satellite imagery (i.e. Google Maps) or catchment imagery.

Parameters associated with each land use type are indexed to communicate the roughness, input and output values to be applied to each cell. The exact number of categories will vary on a project by project basis, depending on the level of detail required for analysis. The following sub sections outline specification of each parameter.

Roughness

Cell roughness is used to calculate the maximum velocity of runoff through Manning's equation (Equation 3.2), this is in turn used by CADDIES to set the required time-step, as described in Section 3.1.2. Roughness is represented using Manning's coefficient 'n'. Values for this coefficient can be attributed to land use types based on commonly accepted specifications found in the literature (Arcement Jr and Schneider, 1989; Hunter et al., 2007; Woods Ballard et al., 2015; Butler et al., 2018).

Parameter values are indexed computationally through a roughness value table, an example of which is shown in Figure 3.7. This figure contains each code used within the roughness matrix and its associated values.

```
Name, Example roughness value index

Number sequences, 2

Value 1, [value]

Value 2, [value]

...
```

Figure 3.7*:* Example roughness value index file from CADDIES

Infiltration

Infiltration rates of land surfaces are included within the framework through adjusting the output rates in cells. Infiltration is specified based on permeability of underlying soil types. Soil infiltration rates can be captured through field measurements, existing survey data and soil classifications. Soil classification is commonly available in the UK through resources such as the UK Soil Observatory (UKSO, 2017) and Soilscape (Cranfield Soil and Agrifood Institute, 2018) databases. Soil classification can be linked to infiltration rates through literature such as UNFAO guidance (United Nations Food and Agriculture Organisation, 2017). Similar data sources are available in other countries.

In the case of soils, it should be noted that CADDIES applies a set infiltration rate which does not simulate the underlying physical processes controlling water movement through a substrate or contributions from ground water flow (Hunter et al., 2007; Beven and Germann, 2013). This is accommodated within modelling through application of conservative infiltration values, which are more likely to represent longer term rates. The simplicity of this approach is deemed acceptable for high level analysis of surface water runoff due to: the resource restrictions of measuring detailed soil porosity data versus the proposed fast utility of the framework; and, for the majority of soil types, the likely limited effect infiltration will have at removing water during high intensity rainfall typically responsible for surface water flooding (Mark et al., 2004).

Infiltration rates form a component of the cell output rate. The cell output rate is indexed against values stored in input tables in a similar way to cell roughness (Figure 3.8).

```
Name, Example output value index

Number sequences, 2

Value 1, [value]

Value 2, [value]

...
```

Figure 3.8: Example infiltration value index file from CADDIES

Evapotranspiration

Surface water flood events are typically caused during short duration, high intensity rainfall. Therefore it is assumed that the evapotranspiration potential of plants would have negligible impact on flood level and are not included within the cell output rate (Mark et al., 2004). It is however possible to accommodate these rates within the framework through increasing the output value for each cell (Figure 3.8).

3.2.3. *Representing sub-surface drainage systems*

CADDIES is designed for rapid simulation of 2D runoff and does not include a direct representation of the 1D piped system. Surface water and combined sewerage is represented within the framework though manipulation of the water output parameter in each cell. This approach is consistent with recent practices developed for UK surface water flood mapping as applied by the Environment Agency (2013).

Applying an output rate to represent urban drainage systems

Surface water drainage systems are represented through adjusting cell output parameters to remove water from the surface at a rate similar to the pipe network running at full capacity. Parameterisation requires understanding of the pipe system layout and drainage sub-catchments within the study area.

Where the layout of the surface water drainage system is available, a peak flow rate per drainage sub-catchment can be calculated through assuming that the pipe full flow in the trunk sewer acts as the limiting factor on flow rate within each sub-catchment. Figure 3.9 shows the method for converting 1D pipe schematics into a 2D output rate per cell.

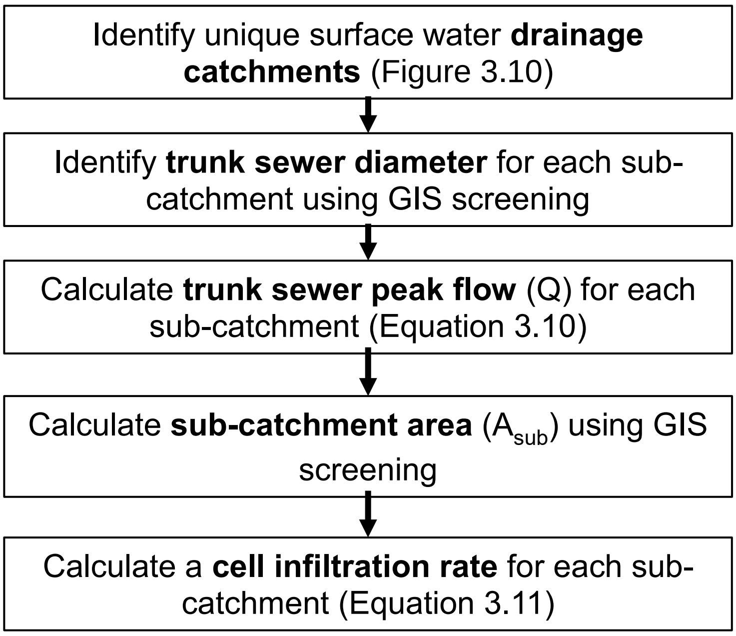

Figure 3.9: Process for representing the surface water network in CADDIES through adjusting the cell output rate

Surface water drainage sub-catchments are generated from existing pipe layout data (Figure 3.10). Required data includes pipe invert levels, diameters, locations and sub-catchments. Unique sub-catchments are identified by determining which areas flow to a single output pipe, referred to as the trunk sewer. The flow capacity in the trunk sewer is assumed to be the limiting factor on flow rate for each sub-catchment. In practice specific pipes in the network may have a lower flow rate than this, however this captures the maximum possible peak flow rate from the sub-catchment and enables a fast screening to determine network flow.

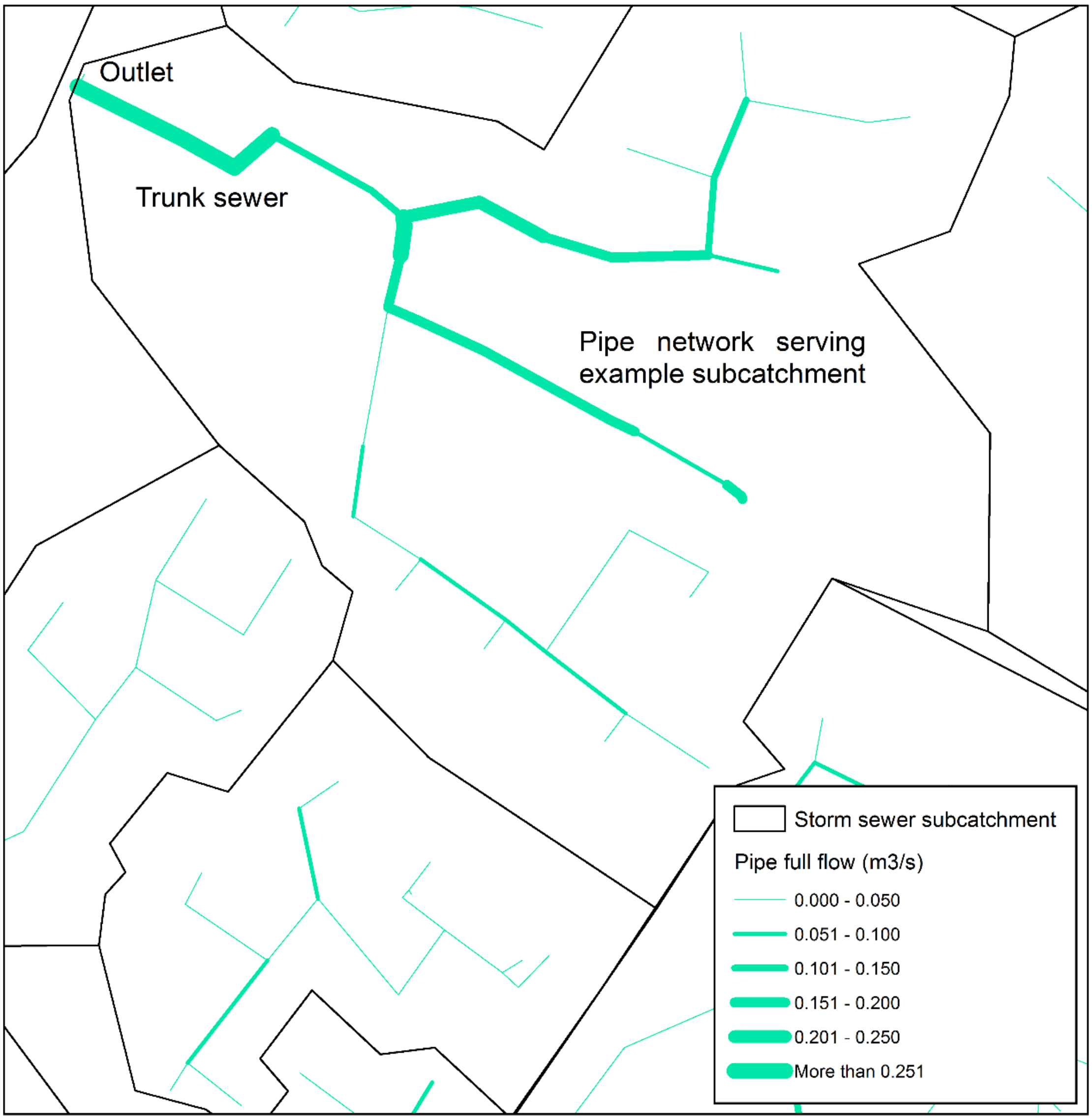

Figure 3.10: Example identifying a unique sub catchment and associated trunk sewer

Once the trunk main has been identified the peak flow rate is calculated through determining the pipe area and pipe full flow velocity. In the case of a circular pipe section, the area can be calculated simply using:

$$A = \pi r^2 \qquad \text{Equation 3.5}$$

Where ‘A’ is area (m) and ‘r’ is pipe radius (m).

Pipe full flow velocity is calculated through application of the Colebrook-White equation:

$$\frac{1}{\sqrt{\lambda}} = -2\log_{10}\left(\frac{k_s}{3.7D} + \frac{2.51}{R_e\sqrt{\lambda}}\right) \qquad \text{Equation 3.5}$$

Where 'λ' is the dimensionless friction factor, 'k_s' is pipe roughness (m), 'D' is pipe diameter (m) and 'R_e' is the dimensionless Reynolds number.

Equation 3.5 can be rearranged to Equation 3.8 in order to provide an explicit expression for velocity through substituting λ using the Darcy-Weisbach equation (Equation 3.6) and R_e using the pipe full Reynolds number calculation (Equation 3.7).

$$h_f = \frac{\lambda L}{D} \cdot \frac{v^2}{2g} \qquad \text{Equation 3.6}$$

$$R_e = \frac{vD}{\mathrm{v}} \qquad \text{Equation 3.7}$$

$$\mathrm{v} = -2\sqrt{2gS_f D}\log_{10}\left(\frac{k_s}{3.7D} + \frac{2.51v}{D\sqrt{2gS_f D}}\right) \qquad \text{Equation 3.8}$$

Where 'h_f' is head loss due to friction (m), 'L' is pipe length (m) '*v*' is kinematic viscosity (m^2/s), 'g' is gravitational acceleration (m/s^2), 'S_f' is the dimensionless hydraulic gradient, and 'v' is velocity (m/s).

Pipe full velocity can then be substituted into Equation 3.10 to calculate the pipe full flow rate 'Q' in m^3/s.

$$Q = \mathrm{v}A \qquad \text{Equation 3.9}$$

Once a peak flow rate per sub-catchment trunk sewer has been calculated, Q is averaged across all cells within the sub-catchment to generate a drainage rate per cell (m^3/s).

$$Drainage\ rate\ per\ cell = A_{cell} \times \frac{Q}{A_{sub}} \qquad \text{Equation 3.10}$$

Where 'A_{cell}' is the area of each cell (m^2) and 'A_{sub}' is area of the sub-catchment (m^2).

The drainage rate is converted to mm/hour and added to the existing output rate per cell specified during the land use classification (Figure 3.8). This then creates one value which represents outputs due to water losses in each cell, constituting drainage, infiltration and evapotranspiration (if included).

Technical details of surface water drainage networks (pipe size, location, inverts, lengths) are typically available in the UK, however ageing drainage infrastructure, data confidentiality, and the difficulty of surveying sub-surface assets means that sometimes details are unavailable or have gaps regarding pipe locations, sizes and conditions (Ana and Bauwens, 2010). Where data might be available at the later stages of a project, the expense of surveying networks or data confidentialities may also mean that the data is unavailable for initial stages of strategic design. Where this is the case, screening methods such as this may be applied to prioritise areas data collection in support of future detailed modelling. In this instance it is possible to apply high level assumptions regarding the capacity of the drainage network, such as those applied by the Environment Agency in broad scale surface water flood mapping (Environment Agency, 2013). This process represents the existing combined sewer system through an infiltration value of 12 mm $hour^{-1}$.

Limitations

This method creates a simple method for representing the sub-surface drainage system, however a model architecture aimed at speed creates several simplifications which effect the representation of physical processes.

The predominant simplification removes the 1D system in favour of an output rate added to the 2D runoff routing mechanism on a sub-catchment basis. This generates a uniform value across each unique sub-catchment using several assumptions. These assumptions relate to the piped system, contributing sub-catchment, network maintenance and destination of flows.

Regarding the pipe system, the assumption is made that the upstream pipe network is able to transfer flows up to the pipe-full capacity of the trunk sewer. Although it is anticipated that pipes are typically designed to achieve this, the irregularities of an aging drainage network and designing with partial knowledge gaps mean that narrower upstream pipes may throttle flow and prevent the outflow discharging at full capacity (Hamill, 2001; Butler et al., 2018). Similarly, a uniform drainage rate across all cells assumes that all areas of the sub-catchment will contribute to the piped system equally. A modern designed network may achieve this, however it is likely that historic design and iterative retrofit and replacement of pipes will lead to a range of upstream pipe sizes throughout the network, creating a variable drainage rate across areas of the sub-catchment.

Representing drainage using a steady output rate also neglects the simulation of sub surface drainage interactions and hydraulic flow phenomenon such as surcharge, backflow and narrow pipes throttling flows in certain catchment locations (Dottori et al., 2013). This may be a significant hazard in specific catchments and so should be screened prior to applying a 2D representation of flows.

The sub-catchment drainage rate assumes that all areas of the sub-catchment are able to contribute to the pipe network equally and instantly. This does not simulate the need for flows to enter the network through a defined inlet. In practice, certain areas within the catchment may not be able to flow into network inlets as a result of catchment topography ponding runoff, features blocking flow paths and poorly designed or maintained inlets (Dottori et al., 2013).

The framework assumes that the pipe network is properly maintained and operating without blockages. In reality blockages and maintenance issues are frequently present in pipe networks, however typically remain unseen until flooding occurs (Schmitt et al., 2004; Ana and Bauwens, 2010; Butler et al., 2018). It is possible to adjust flow rates to account for sedimentation and partial blockages by restricting diameters or reducing the peak flow rate to include a safety margin. This approach is straight forward and can easily be applied within this framework. However, the effects of a full pipe failure are incredibly variable and difficult to predict, depending on failure cause, location and timing (Ana and Bauwens, 2010). It is therefore difficult to accommodate this analysis in 1D or 2D pipe models without having a computationally expensive systematic failure of certain pipes, inlets and outlets (Mugume et al., 2015; Diao et al., 2016). Simulation using this framework can include a 'worst case scenario', representing a situation where the pipe network has failed totally (DEFRA, 2018b). This is be achieved by removing the drainage component of the cell output rate. It should be noted that situations similar to this 'worst case' treatment can also occur during times of high intensity rainfall which falls at a rate faster than the network can capture runoff.

Removal of water from the model assumes that all rainfall captured within the surface water drainage network is transferred to a destination where it will not cause disruption. This assumption reflects removing rainfall where it falls, but once this is removed from the model it does not represent the possibility of surface water re-emerging at another point in the network or outfall. High intensity short term rainfall, responsible for the majority of urban surface water flooding, is unlikely to contribute significant amounts of volume to cause flooding in major watercourses, but this limitation should be considered carefully as the approach may not be suitable where small water courses, culverts or pipe full flow phenomenon such as surcharge are expected to contribute to surface water flood risk. This risk can be mitigated through initial analysis of flood risk such as evaluating flood histories, interviewing catchment stakeholders and reviewing previous studies in the area of investigation. These actions are typically recommended as part of strategic flood risk assessments (DEFRA, 2010).

3.2.4. Rainfall generation

Rainfall is the final component required to characterise the study area (Figure 3.11). Rainfall is represented in the model through a spatially and temporally variable inputs, specified on a cell by cell basis through a range of indexed hyetographs.

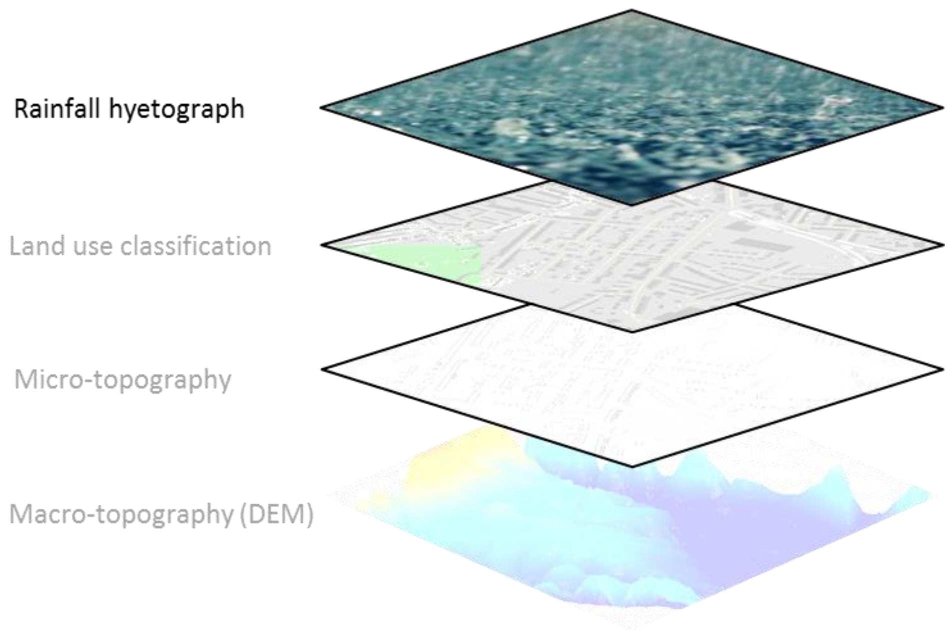

Figure 3.11: Characterising rainfall events within the framework

Input hyetographs

Rainfall for the study area is specified using an input hyetograph describing the intensity of an event at user specified time steps. This approach can accommodate simple block rainfall, design storms and time series rainfall. Applicability of rainfall depends on the purpose of investigation and available setup time.

Data availability

Any rainfall data can be applied within the framework, provided it is transferred into a hyetograph format (see processing steps section). A wide variety of data sources are available depending on catchment location and the context of analysis. Rainfall data will typically be presented as design or historic rainfall patterns in an event or time series format (Ward and Robinson, 1990).

Design rainfall is constituted of a synthetic and statistically developed rainfall pattern which reflects characteristics specified to each catchment. Rainfall may be supplied within a statistically constructed distribution or as a single block of constant rainfall. Statistical relationships are developed using an intensity, duration and frequency (IDF) relationship. Applying design rainfall to assess flood hazard in catchment provides the advantage of providing a large range of rainfall events which constitutes many rainfall probabilities and magnitudes. When combined with fast assessment methodologies this enables research to investigate intervention performance across many events. The drawback from studying flooding using design rainfall is that it is difficult to validate model performance versus real world measurements. UK design storms are available through the Centre for Ecology and Hydrology Flood Estimation Handbook (Centre for Ecology and Hydrology, 1999, 2013). This resource specifies an IDF relationship at a km^2 scale across the UK. Similar organisations can provide this information in other countries.

Historic rainfall represents records of previous rainfall events within a catchment. Records are measured using a variety of techniques including rain gauges and weather radar. Different data collection methodologies constitute a range of advantages and limitations regarding temporal and spatial accuracy and precision of records (Ward and Robinson, 1990). The advantage of studying intervention performance using historic rainfall is the opportunity for practical validation of model results versus real life measurements enabling an enhanced confidence in model results. However, limits on the events which have been recorded within a catchment typically constrain analysis to a small range of events, particularly when considering extreme rainfall, and consequently restrict analysis of resilience (Neal et al., 2009; Kjeldsen et al., 2014). Extreme rainfall is rare, thus limiting data collection. Rainfall measurement techniques are often calibrated to everyday rainfall, and as such can provide erroneous readings for high intensity storms (Westra et al., 2014). For example tipping bucket rainfall gauges can fail to action at a fast enough rate to capture the most extreme high intensities. The short duration and potentially highly localised nature of high intensity rainfall also results in rainfall measurement techniques failing to collect full records of rainfall. Surface water flooding during these events is also typically of short duration therefore, even when accurate rainfall measurements can be captured, records may not accurately record catchment flood conditions.

Event rainfall represents the intensity and duration of a single rainfall occurrence. Time series rainfall is constituted of a longer term record of several rainfall events and intervening periods. The duration of the series can be adapted to capture antecedent periods. This format of rainfall provides useful information regarding the conditions within a catchment prior to intense rainfall, which can be an important controlling factor in determining catchment characteristics such as soil saturation, watercourse levels, groundwater flows and remaining capacities for rainfall capture or storage interventions (Ward and Robinson, 1990). The longer time period recorded increases simulation time and the storage space required for output times. In the case of surface water flooding, which is predominantly associated with high intensity rainfall, this increase in simulation requirement does not provide sufficiently enhanced analysis of the catchment and so, for this framework, analysis will focus on applying rainfall events whilst accommodating antecedent conditions through conservative parameter assumptions and sensitivity analysis.

Processing steps

Rainfall hyetographs are converted into a comma delimited format, readable by the CADDIES model (Figure 3.12). Multiple hyetographs can be added into each model to represent spatially variable rainfall or the effects of interventions capturing incoming precipitation (see Section 3.3.1).

Figure 3.12 shows an example input file describing two block rainfall hyetographs. The file specifies the rainfall input rate (intensity) which changes in steps for defined blocks of time. The sequence of blocks can be expanded to multiple steps to represent temporally complex rainfall patterns.

```
Name, Example input hyetograph

Number sequences, 2

Value 1 (mm/hr), [intensity at start], [intensity at end]

Time 1 (seconds), [time at start], [time at end]

Value 2 (mm/hr), [intensity at start], [intensity at end]

Time 2 (seconds), [time at start], [time at end]
```

Figure 3.12: Example block rainfall input file from CADDIES

Hyetograph effects are referenced to cells through indexing. Each cell is linked to a specified hyetograph to represent the spatial distribution of rainfall. The number of hyetographs can also be increased to include many different rainfall patterns within the same simulation.

Catchment critical rainfall

Engineers typically base designs for surface water management systems on a critical duration event where all upstream areas are contributing rainfall to a specific location. This identifies conditions which are likely to lead to the most significant damage.

Identifying critical rainfall characteristics is straight forward when designing linear systems (for example pipe networks), however is a challenging concept when considering flood hazard across an entire catchment, exacerbated by spatial complexities of disaggregated catchment surfaces and differing intensities generated using a range of rainfall profiles.

Consequently, the characteristics leading to the most significant damage may not be readily predictable, and may change in response to different intervention strategies. Therefore rapid analysis using the framework enables investigations to incorporate a range of rainfall profiles and compare maximum flood depths across many scenarios to identify the catchment's critical event. This may be undertaken to guide rainfall selection at the initial stages of project design, or as part of a detailed intervention assessment undertaken within the main evaluation of an analysis.

3.3. Representing intervention strategies

This section outlines how interventions are represented within the modelling framework. At this stage, discussion outlines the general methodology and mechanisms for modelling interventions within the framework. A detailed discussion of modelling specific interventions is presented in Section 6.1.

Interventions are applied using an input matrix which overlays the land use setup defined in the previous section, this allows many intervention strategies to be stored and added to the model efficiently. Figure 3.13 identifies how this overlay relates to the land use setup discussed in the previous section.

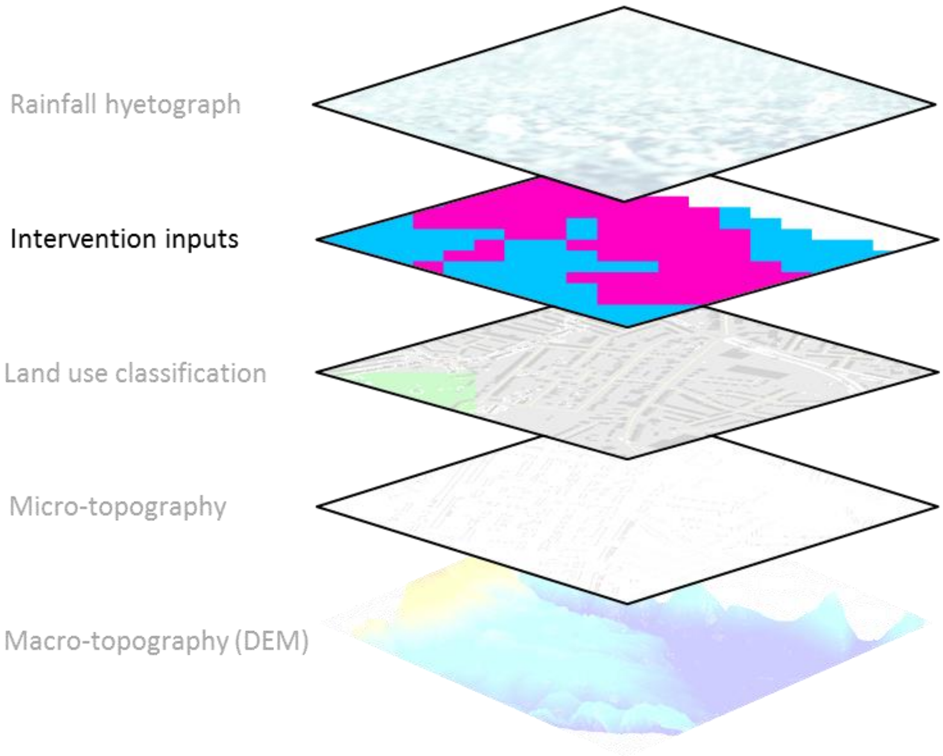

Figure 3.13: Representation of interventions within the modelling framework

3.3.1. Representing interventions through the model architecture

General approach

Intervention strategies are included within the model through a simplified representation of measures using spatial manipulation of elevation, input, output and movement speed parameters (Figure 3.2). All intervention effects are applied on a cell by cell basis, controlled by the resolution of the input land use. Simple representation using these parameters is unsuitable for detailed design, but has the advantage of enabling multiple, fast simulations to determine performance of intervention types and locations in a particular catchment.

Representing interventions effects on the catchment surface

Interventions which change surface types and land use are represented through adjusting a roughness parameter in corresponding cells. The roughness value is changed across the entire footprint of the intervention, rounded to the nearest minimum cell size. Examples of interventions likely to change surface roughness include measures such as permeable paving, swales, urban green space, etc (Woods Ballard et al., 2015).

Representing interventions effects on rainfall capture

Rainfall is specified through an input value in each cell which can be programmed to include a temporally variable rainfall rate through specifying an input hyetograph for each cell. Interventions which capture incoming rainfall are represented in the framework through adjusting input hyetographs to reduce water input to selected cells. Adjustments vary depending on the storage capacity, attenuation rate and capture efficiency specific to each intervention.

Hyetographs are adjusted across all areas which constitute the capture footprint of an intervention. This area is specified down to the resolution of the input model cells. The input volume removed per cell is estimated through dividing the total storage volume of an intervention by the size of the area on which it is situated, typically the roof of a building. For example, an average roof size in the UK is 45.5 m^2 (DCLG, 2015). Therefore, a 100 litre water butt collecting from this surface would capture approximately 2.2 l of rainfall per 1 m^2 cell. Figure 3.14 shows how the hyetograph is manipulated to achieve this. Figure 3.14a shows an unedited example rainfall profile. Figure 3.14b, c, and d show edited profiles representing capturing rainfall using different capture volumes. This example assumes a 100% efficiency of rainfall capture interventions.

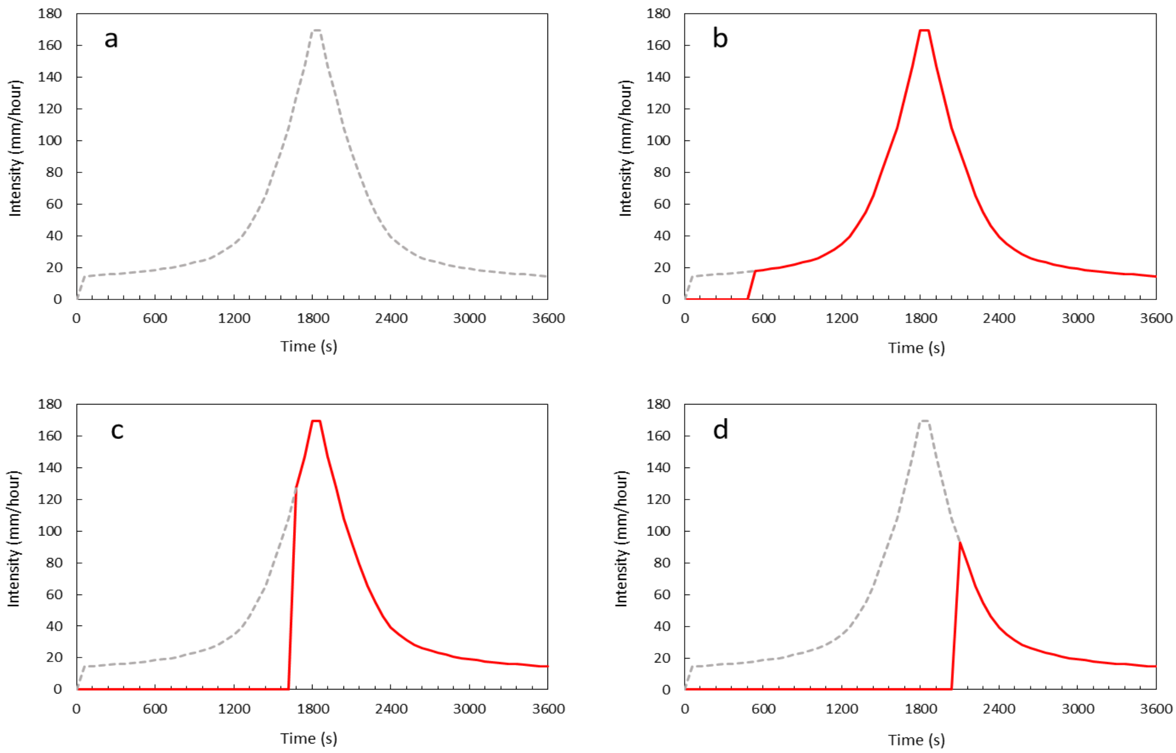

Figure 3.14: Example representing rainfall capture through cell hyetograph manipulation

Representing interventions effects on infiltration

Infiltration is included within the model through increasing the outflow rate in cells. The infiltration rate specified is controlled by the porosity of the underlying soil or drainage medium, which can be attributed based on literature and field studies specific to each intervention.

Representing surface water drainage interventions

Surface water drainage interventions are specified through adjusting cell output rates, as described in Section 3.2.3.

Representing changes to elevation profile

Interventions which change catchment topography are represented through adjusting elevation levels in the underlying elevation model, as described in Section 3.2.1. These interventions are likely to include measures such as swales, attenuation ponds, wetlands, flood walls, embankments, landscaping and other micro and macro topographic features. Elevation in a cell can also be adjusted to account for flood resilience interventions such as raising building thresholds.

Interventions with multiple effects

Many interventions are represented using multiple parameters, for example swales and infiltration trenches will slow runoff in addition to infiltrating water. Intervention effects are described along with each case study later in the thesis.

Interventions modelling summary

Table 3.1 presents a summary to describe parameter changes required to represent interventions within CADDIES. This table provides reference regarding interventions which is developed to describe representation of specific interventions in detail within Section 6.1. It should be noted that this table shows the predominant general parameter change requirements for representing groups of interventions, but that application to represent specific interventions and contexts may vary and should be developed with stakeholders on a site by site basis.

Table 3.1: General summary of predominant parameter changes used to represent intervention types when using the rapid scenario screening framework

Intervention	**Elevation**	**Input**	**Output**	**Roughness**
Rainwater capture	✗	✓	✗	✗
Surface capture	✓	✓	✓	✓
Catchment drainage	✗	✗	✓	✗
Catchment infiltration	✗	✗	✓	✗
Walls/ embankments	✓	✗	✗	✗
Landscaping	✓	✗	✓	✓
Surface ponding	✓	✗	✗	✗
Runoff speed	✗	✗	✗	✓
Raising thresholds	✓	✗	✗	✗

3.3.2. Storing intervention scenarios using input maps

Parametrisation of CADDIES across a regular grid enables intervention scenarios to be stored as numeric matrices, in which the value in each cell is indexed to a set of parameters which describe the effects of each intervention. This enables a computationally efficient storage of interventions through a series of matrices and an associated parameter table.

This representation of strategies is fast to set up and simple to modify. By inputting the intervention matrix on top of the land use matrix it is easy to run simulations without having to set up multiple models. This has the further advantage of enabling code to apply new intervention maps on top of the land use data to automatically simulate many scenarios.

In many circumstances the intervention will not alter the underlying land use parameters, for example a drainage system upgrade would not change the cell surface roughness parameter. Where this the case, a ‘n/a’ effect can be specified which retains the land use parametrisation for that particular value. This provides the utility of enabling the same intervention to have a range of effects based on the land use it occupies.

3.4. Simulation using cellular automata

This section outlines the model set up conditions applied to run the CADDIES flood model. These conditions are specified using a simulation input file. This section will describe settings controlling trade-offs between speed and accuracy, and how these are implemented within the framework. Full description of the CADDIES modelling process is found in Section 3.1.2.

Time control and simulation time steps

Simulation duration is specified in seconds and controlled using the ‘time start’ and ‘time end’ settings. Times are specified in seconds and generally extend for the duration of the rainfall event plus an allocated runoff time to capture dynamics of surface water movement following an event.

Time step settings

CADDIES speeds processing times through application of an adaptive time step controlled by the length of a cell and the water velocity. The time step is automatically reduced as runoff velocity increases. It is necessary to define a minimum and maximum time step to prevent the calculation requirements approaching infinity where velocity is zero, or very small time steps affecting model performance where velocity is high. The software recommended minimum and maximum time steps are 0.01 and 60 seconds, respectively (University of Exeter, 2015).

CADDIES automatically adapts the time step and updates values stored as peak outputs at set intervals specified by the user. The developer recommends setting this to 60 seconds to provide good accuracy at a level which will not affect performance.

Roughness, rainfall and infiltration setup

Roughness, rainfall and infiltration parameters are either specified as a uniform global value or as a spatially and temporally varying value. Global parameters are unsuitable to define complex urban environments where multiple land use types and intervention configurations are present. This research applies variation in parameters. Parameter values defining these are specified as by the user as part of the input file.

Initial conditions

Initial conditions for each simulation are set on a context specific basis. Initial losses can be represented through manipulation of effective rainfall through the input hyetograph. This is achieved by subtracting rainfall at the beginning of a simulation to represent interception, depression storage and initial soil wetting. Depression storage can also be accommodated through application of high resolution LiDAR when developing the catchment DEM.

Initial conditions should not be neglected when evaluating low intensity rainfall or non-urban catchments, however literature indicates that initial losses are not significant in determining the effect of high intensity storms in urban areas, the focus of this study (Butler et al., 2018).

Uncertainties regarding initial conditions can be accommodated through simulating many different scenarios including a range of potential parameters. For example, simulating multiple tank sizes to represent previous rainfall events reducing effective capacity.

Boundary characteristics

Boundary conditions are set by the user to specify water movement across the model boundary. Low elevations allow water to be lost through flow out of the domain, whereas high elevations act as a barrier which prevents runoff from leaving the model. This will typically only affect the water level at the model boundary, and so is only of importance when considering flood impacts at the very edge of the domain. Speed of CADDIES enables large areas to be simulated, therefore it is recommended that the model area encapsulates a total surface water catchment relevant to the study area examined. Where this is the case water flowing away from the catchment will not impact the investigation.

3.5. Intervention performance assessment

Performance assessment and the strategic direction of flood management should be supported through a robust and quantifiable evidence base (House of Commons, 2016). Many fast option assessment techniques are available, however as identified in Chapter Two, common methodologies typically achieve speed through qualitative or simplified metrics which do not adequately represent flood dynamics. The main novelty from this framework is the ability to quickly simulate flood dynamics, which then enables intervention performance assessment to be undertaken using quantified metrics detailing flood extents, depths and costs.

To identify intervention performance the model is run to outputs representing baseline and intervention scenarios. Baseline scenarios describe the catchment ‘as is’, without any intervention strategies (Section 3.2). Intervention scenarios apply new measures onto this baseline (Section 3.3). Comparison between these two sets of scenarios enables intervention performance effects to be measured.

This section outlines the approach taken to analyse intervention performance using the CADDIES model outputs.

3.5.1. CADDIES outputs

The CADDIES simulation outputs ‘.asc’ files identifying water depth and velocity for each cell within the study area. Outputs are provided at user specified intervals and as a peak value across the entire simulation. These outputs directly provide absolute flood extent and depths, and can be processed to provide flood damage costs (Section 3.5.4).

3.5.2. Flood extent

The simplest assessment of flood hazards is achieved through analysis of flood extent. Movement of water across the catchment during the simulation means that analysis at a particular simulation time step will not necessarily adequately represent flooding across the whole catchment, therefore a snapshot of a worst case flood extent is achieved by analysing the peak flood extent across sampled from an entire flood event. This enables one flood map to visualise hazards across the entire catchment, simplifying decision support.

3.5.3. Flood depth

Spatial analysis of flood extent is enhanced through assessment of flood depths across the catchment. As with flood extent, peak flood depths are analysed in order to present a worst case scenario across the whole catchment in a single output.

Maximum flood depth is a useful metric for identifying the peak impact caused by surface water flooding and provides data for application using damage cost assessment techniques. Limiting simulation outputs to one maximum depth file saves computational space where many model runs are required and provides decision support with simple visualisation of an interventions effects.

Absolute flood depth

Analysis of absolute flood depth for an intervention scenario includes application of the depth in each cell and provides analysis of the overall worst case impact in a particular scenario.

Relative flood depth

Intervention strategies typically aim to reduce flooding in a catchment. Absolute flood depth maps are descriptive of an individual scenario, but the large amounts of information presented in a single map can make it difficult to easily identify intervention performance. Relative flood depths present a metric within a single map to visualise the surface water depths before and after an intervention strategy is applied. A relative flood depth is calculated on a cell by cell basis using Equation 3.11, which equates the difference between a baseline depth matrix and an intervention depth matrix. A negative value within a cell indicates a reduction in flood depth due to an intervention, whilst a positive value indicates an intervention strategy increases flood depth in a cell.

$$Relative\ depth\ matrix = \ Intervention\ depth\ matrix - Baseline\ depth\ matrix$$

Equation 3.11

Relative depth matrices can be transformed into flood maps which visualise the effects of strategies and provide a simple tool for informing decision support. Maps can provide utility for developing intervention scenarios and identifying complex spatial variation in flood dynamics attributed to certain strategies. This is of particular relevance where certain interventions may generate new flow paths through changing the timing of flooding. Changes to flow paths may inadvertently create additional flooding even when water is captured upstream. Figure 3.15 presents an example of calculating and generating a relative flood map using a simple 2 x 2 cell matrix.

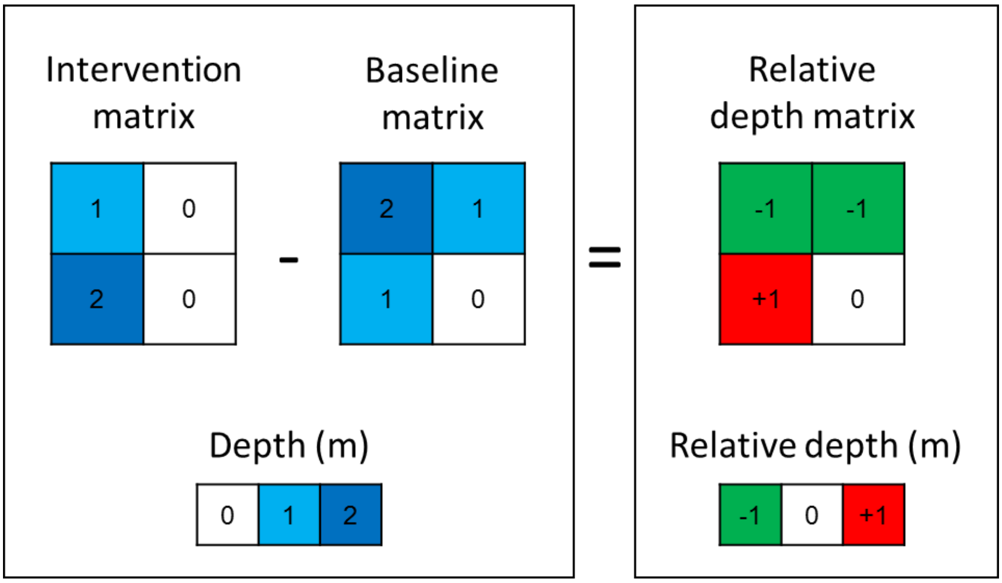

Figure 3.15: Example presenting calculation of a simple relative depth flood maps

Limitations of peak and relative flood depths

Maximum flood depth does not represent the total volume or extent of flooding at any particular moment. This metric is unsuitable for uses where a snapshot of flooding is required at a specific time step as it does not represent the propagation of flooding through the catchment. An example of an application where timing is required is emergency evacuation planning. It is possible to apply CADDIES to present information at user defined time steps to include temporal flood dynamics, however it is not required for the purposes of screening intervention performance to reducing flood damage.

Relative flood depth maps do not represent the change in timing for a flood event. For example, two strategies may reach the same peak magnitude, however one may reduce the duration of this peak. Timing has not been included within this assessment due to an assumption that the impact of short duration surface water flooding is linked to depth, not duration. This is supported by industry flood assessment methodologies which indicate flood costs are controlled by depth at up to twelve hour duration (Penning-Rowsell et al., 2010). Longer duration flood events associated with fluvial and groundwater mechanisms will also be influenced by the duration of flooding, which can significantly alter the impacts of damage and disruption.

3.5.4. Flood damage costs

Understanding flood depth and extent alone does not always provide a reliable assessment of the disruption associated with an event. Certain locations within a study area may be more susceptible to damage and disruption due to the presence of structures; therefore, a scenario with the greatest flood depth may not equate to the largest flood damage cost or impact. Assessment using flood damage cost metrics provide additional detail incorporating the spatial distribution of flooding across a catchment.

This section outlines calculating damage costs for specific events, translating this into an annual cost, projecting this cost into future calculations and the limitations associated with this screening methodology.

Depth damage model

Flood damage costs for each scenario are calculated through application of GIS based flood damage analysis (University of Exeter, 2014; Chen et al., 2016). This analysis estimates the costs of damage through assigning depth damage profiles to polygons within a catchment and then calculating a cost based on the peak water depth within each polygon. Damage is only related to depth, without consideration of velocity or other damaging factors such as contamination (Merz et al., 2010). Potential limitations of this approach and its application for strategic screening are discussed later within the section.

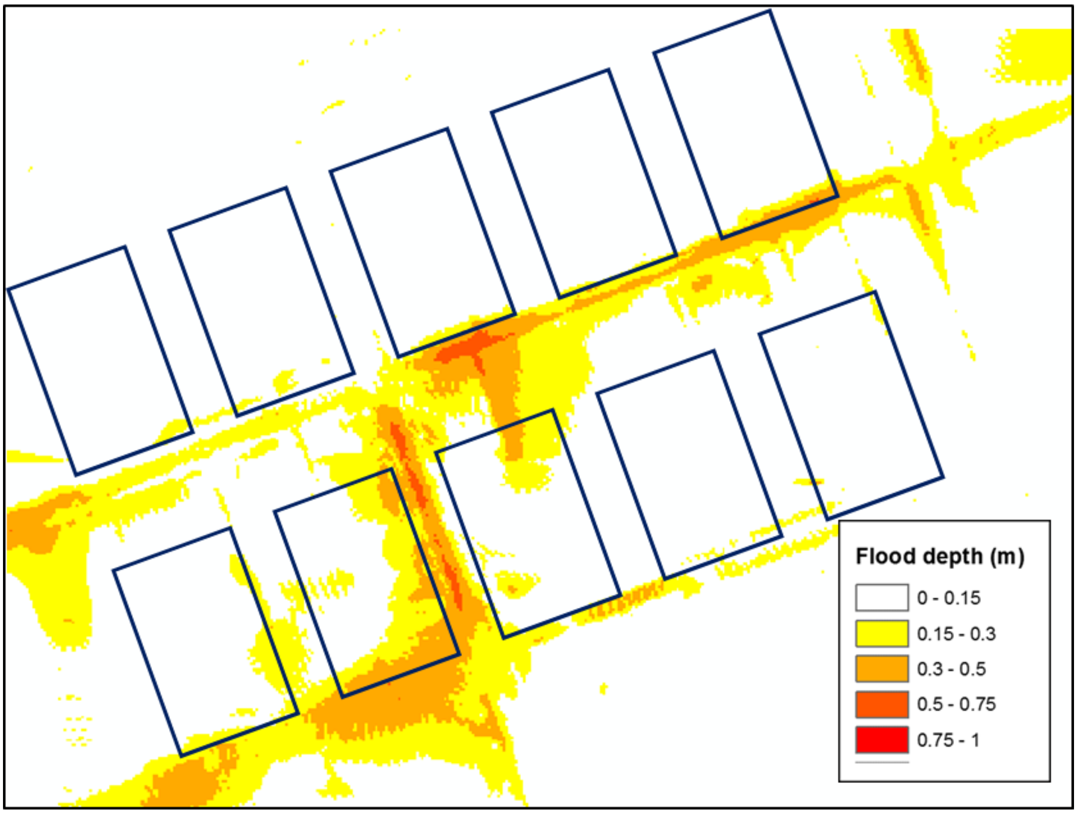

Figure 3.16: GIS analysis of flood damage costs through assessing peak depth across polygons representing buildings

Damage costs for average UK properties are specified in the multi-coloured handbook (Penning-Rowsell et al., 2010). This is an industry standard document which relates the direct and tangible costs of short duration inundation (<12 hours), typical of surface water flooding, to the building fabric and household inventory. Components of building fabric include the exterior and surrounding features, interior materials and finishing, floors, plumbing and electrical damage. Components of household inventory include appliances, furniture, audio visual equipment, personal items and the costs of domestic clean up. Figure 3.17 shows the flood damage curve for an average UK property using data supplied from the multi-coloured manual.

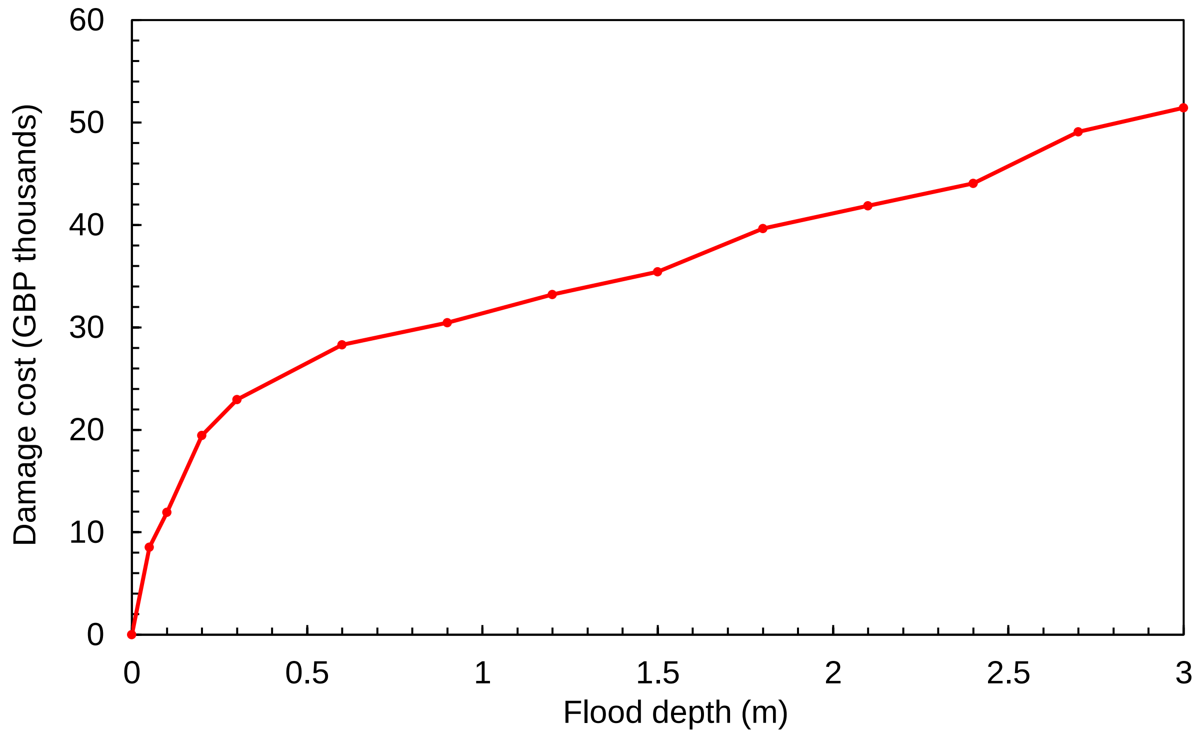

Figure 3.17: Depth damage curve for short duration flooding in an average UK property (Penning-Rowsell et al., 2010)

Estimated annual damage

Damage costs per scenario only present a limited view of intervention performance across a specific event. The life expectancy of intervention assets alongside unpredictable future hazards means that decision support should include assessment of costs over a longer time period whilst considering the likelihood of impacts (University of Exeter, 2014).

Estimated annual damage (EAD) represents an average expected damage per year when averaged over a long time period and represents a useful metric to describe the damage avoidance of intervention strategies. EAD is calculated through sampling cost damage across a range of different probability events to generate a curve representing damage versus annual exceedance probability (Figure 3.18). The EAD is equal to the area under the flood damage curve (Arnell, 1989). As intense local precipitation is the controlling factor in creating surface flooding it is reasonable to assume the return period of the rainfall can be applied as the return period for the flood (University of Exeter, 2014).

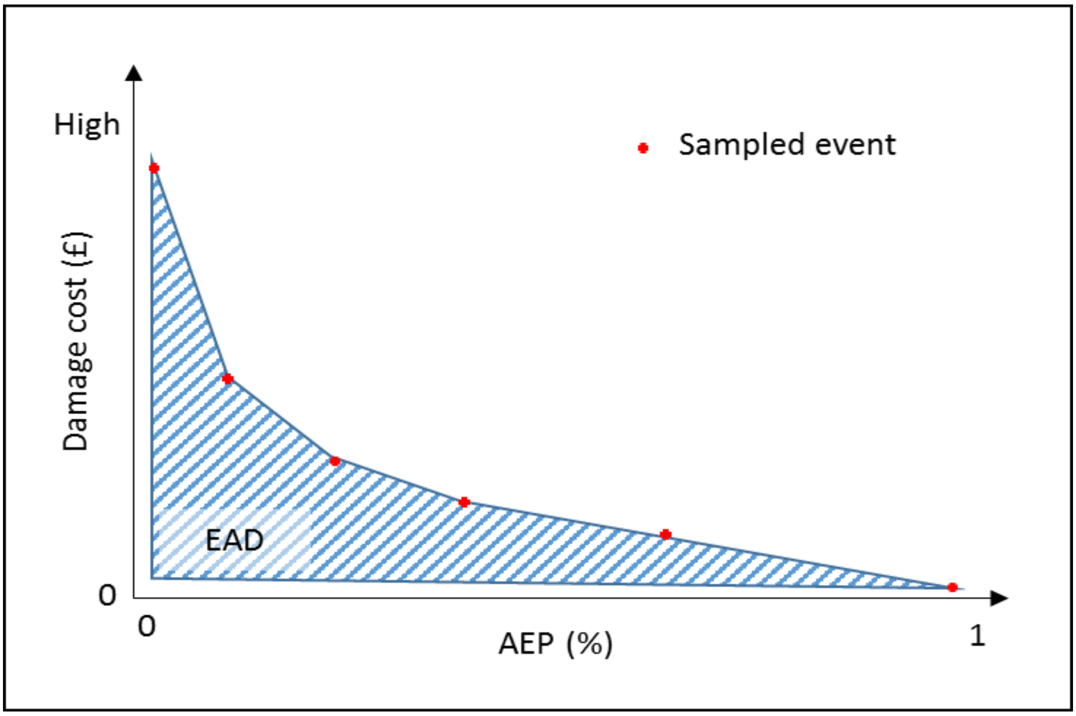

Figure 3.18: Graphical representation of sampling events to develop an EAD calculation

The more sampling points available result in a higher reliability of the EAD calculation (University of Exeter, 2014). For an effective representation of the EAD the curve should represent damage costs across a range of events including both low probability high magnitude as well as high probability, low magnitude occurrences. These curves will not meet the axes as practically, a certain or impossible event cannot be included within analysis. A limitation of many current assessment approaches is that modelling adequate numbers of simulations to build reliable sample sizes can be restricted by the computational expense of hydro-dynamic modelling, therefore fast assessment models and frameworks provide an opportunity to reliably include EAD within intervention assessments.

EAD calculation is expressed mathematically in Equation 3.12, which describes estimating the area underneath the curve from Figure 3.18 (University of Exeter, 2014).

$$EAD = \int_0^1 D(F)\, df \qquad \text{Equation 3.12}$$

Where *D(F)* is damage as a function of annual exceedance probability, *F.*

Projecting future costs using discounting

The time value of money, a core principle of economic theory, indicates that money at the present time is worth more than an identical sum in the future due to the potential interest growth on the value (Wong, 2015). Discounting is applied to calculate the present value of a benefit to be received in the future.

Interventions are typically designed to operate over a prolonged service life, and therefore strategy performance should be assessed relative to future benefits, which need to be discounted. Future costs can be calculated using a discount rate, In the UK, this is currently specified at 3.5% per year (HM Treasury, 2013). It should be noted that discounting adjusts net present value for future economic costs, and does not adjust costs in relation to potential future changes to probabilities of events.

Limitations regarding tangible surface water flood damage

This method provides a fast technique for generating flood damage estimates at the urban catchment scale. However, fast implementation provides limitations which should be acknowledged. These limitations are primarily caused by a lack of resources available for a screening process, either through cost of data, measurements required to collect the required data or the computational time to analyse and complex interactions between subsystems.

The cost assessment applied in this framework is focused on analysis of direct and tangible damages to properties. Direct tangible damage refers to the structural and contents damage incurred due to direct contact with flood water (Hammond et al., 2015).

With regard to the limitations of a direct and tangible cost assessment of property damage. Treating all properties with a single flood damage curve neglects the potential cost differences attributable to the wide range of different structures present within an urban catchment. Application of an ‘average building damage cost’ can be indicative and applicable across a wide area, but will not account for spatial differences in property types, function, sizes, layouts and construction materials or techniques. Certain buildings may also have specific structural vulnerabilities such as basements, sub-standard electrical wiring and damaged masonry which will also lead to increased damage versus regular properties. Alternatively, other structures may have flood resilient design features or coping mechanisms which limit property damage (Douglas et al., 2010; Kaklauskas et al., 2014; Gotangco et al., 2015; HM Government, 2016). Differences are further enhanced when considering commercial properties, which will potentially contain a large amount of stock of varying value. Nuances in insurance policies may also affect the damage costs and vulnerability of these structures. It is not practical, or possible, to survey all of these parameters when screening or estimating damage costs across a large area.

The simplified flood analysis also neglects the impact of critical infrastructure. Critical infrastructure in urban catchments is likely to include buildings such as hospitals, power relays, water treatment and distribution assets and administrative buildings. These structures may have important functionality, which if disrupted could result in significant damage and disruption costs (Crisis and Risk Network and Center for Security Studies, 2012).

Other direct and tangible impacts are not included within the assessment (Hammond et al., 2015). Damage and disruption to the road and transport network is not included (Pregnolato et al., 2017). In urban areas this can lead to major costs, however the survey and network analysis required to understand the importance of the transport link, density of traffic and potential alternative routes are beyond the scope of initial option screening. Surface water flood impacts are also created by intense and short duration events, which are unlikely to persist for long durations.

Disruption and damage to critical infrastructure and the road network may lead to cascading damage which can be challenging to predict and manage (Little, 2002). Several studies have investigated measuring permutations of cascading impacts, however interconnections across systems and scales requires a detailed understanding of asset functions within networks and renders this level of detailed analysis beyond the scope of an initial screening tool (Kinzig et al., 2006; Labaka et al., 2016).

It is challenging to include high levels of detail without extensive and high resolution surveys, particularly in the case of assessing individual structures or large scale networks for which bespoke surveys could become disproportionately expensive, essentially becoming full research projects within their own rights and significantly exceeding the scope of a screening tool. As such it is deemed appropriate to apply industry standard flood depth-damage curves for average residential properties within the screening tool. It is envisioned that outputs from this process will steer further detailed analysis and highlight areas where surveys and additional site investigations are required.

Limitations regarding intangible surface water flood damage

Other categories of impacts are also excluded from analysis as a result of challenging measurement requirements. Figure 3.19 presents a simple classification based on a direct versus indirect and tangible versus intangible impacts. Direct and tangible impacts, as discussed above, occur due to contact with flood water, whereas indirect impacts are secondary occurrences which are triggered by knock-on effects from hazards (Hammond et al., 2015). Tangible impacts can be measured through attributing damage costs, whereas intangible impacts are subjective and difficult to assign a robust and objectively cost.

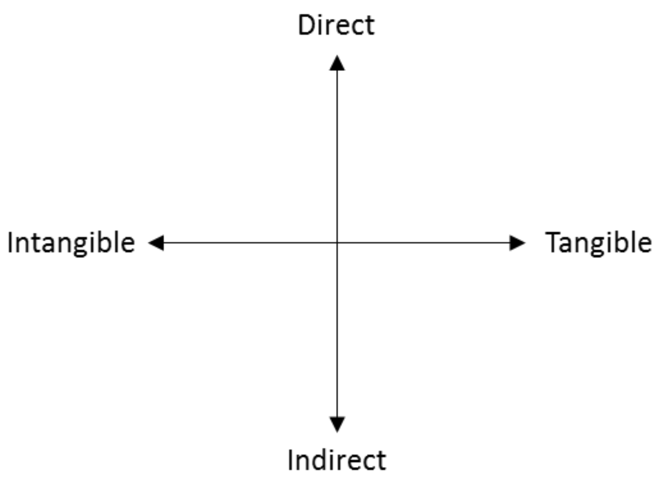

Figure 3.19: Classification of flood impact categories

In terms of surface water flooding, the largest intangible impact is likely to be human health impacts, with fatalities being the most significant, and hardest to quantify, outcome. Health impacts include physical and mental health effects (Hajat et al., 2005, 2014). Physical effects can include diseases or injuries caused during flooding, evacuation or clean-up operations. Secondary impacts, such as disruption to infrastructure or displacement of populations can also increase this hazard. Studies have reviewed epidemiological evidence regarding loss of life, disease and injury impacts (Ahern et al., 2005; Jonkman and Kelman, 2005; Jonkman et al., 2008), however impacts are subjective and influenced by a wide range of socio economic factors which are impractical to include within this style of screening process. Psychological impacts of flooding are even harder to identify and value within impact assessments (Hammond et al., 2015).

3.6. Chapter conclusions

This chapter has outlined the development of a rapid scenario screening framework. This framework delivers novelty through responding to gaps identified in literature regarding a fast and quantitative methodology for assessing many permutations of interventions, rainfall events and scenarios. The framework delivers a streamlined method, intended for application as a screening tool to complement and direct, rather than replace, detailed modelling.

Key conclusions from this chapter are:

- A framework for rapid scenario screening has been developed. The scope of the framework is aimed at generating evidence for decision support using fast preliminary option screening, and therefore is designed to use data requirements and assumptions commensurate with this utility.
- Land use, rainfall and interventions are represented using a simplified system of adjusting elevation, input, output and roughness parameters in cells across a study area.
- Simulation is undertaken using the cellular automata flood model CADDIES. This model applies simplified simulation of flooding based on cell states, which previous studies have demonstrated leads to higher simulation speed relative to industry standard hydro-dynamic modelling.
- Intervention performance is assessed using quantitative metrics, including flood depth, extent and damage costs. Analysis is achieved using simulation outputs, depth-damage curves and GIS processing.

Subsequent chapters in the thesis will validate this approach and then apply findings to investigate framework utility and intervention performance across multiple case studies.

4. VALIDATING THE FRAMEWORK

This chapter responds to Objectives Three, ‘validate the framework against industry best practice’ and Eight ‘develop recommendations for practical application of this methodology’. Validation of the framework is a crucial step in establishing suitability of the rapid scenario screening framework for application as catchment screening and intervention assessment methodology. To support utility towards screening flood management actions it is important to understand how the results from the framework compare to current approaches applied in industry.

Framework validation is evaluated though comparison of framework outputs with an industry standard integrated flood model which has been applied as part of a published surface water management plan (SWMP), representing established professional engineering practice. Past research has compared the underlying flood model used in the framework, ‘CADDIES’, with ‘Infoworks Integrated Catchment Management’ (ICM) to compare performance routing 2D runoff, but a gap remains regarding validating the approach against a model including a 1D pipe network and interventions (Gibson et al., 2016).

This chapter describes a case study in St Neots, Cambridge, outlines the modelling approaches used in the SWMP and framework and then evaluates framework performance through assessing variations between the two methods.

It is important to note the distinction between CADDIES, which has been developed and rigorously tested through previous studies (Guidolin et al., 2012, 2016; Ghimire et al., 2013; Gibson et al., 2016) and the rapid scenario screening framework (Chapter Three), a novel contribution taking advantage of CADDIES fast processing speed, alongside other research developments, to create utility towards intervention assessment.

The work presented in this chapter is published in: ‘Validating a rapid assessment framework for screening surface water flood risk’ (Webber et al., 2018b), which has been accepted for publication in the Journal of Water and Environmental Management.

4.1. Method

Framework validation is examined through three questions, representing scenarios with increasing levels of detail: Can the framework consistently prioritise areas of flood risk during a worst case scenario, with no functioning surface water drainage system? Can the framework consistently prioritise areas of flood risk taking into account the existing sub-surface drainage system? Is the approach suitable for modelling interventions in an urban catchment? These questions are answered through comparing the framework with a published SWMP, produced by Arcadis on behalf of Cambridgeshire County Council and simulated using the industry standard hydrodynamic model ICM (Arcadis, 2012).

It is important to understand the scope and limitations of the framework in respect to different levels of detail, therefore analysis is structured using three scenarios. These scenarios facilitate a performance comparison across a range of conditions, linked to the questions from the previous paragraph, which gradually increase in complexity. The full detail of these scenarios is described later in the chapter.

- **Scenario One, 'worst case'**, represents the catchment with no functioning surface water drainage system.
- **Scenario Two, 'surface water drainage'**, includes the existing surface water drainage system, with pipe locations and sizes provided by Cambridge County Council.
- **Scenario Three, 'intervention'**, includes the existing surface water system plus additional flood management interventions.

This section outlines the data, processes and assumptions required to setup both models. In certain circumstances differences between model architectures has prevented an identical application between both approaches, where this is the case it is specified within the methodology.

4.1.1. Characterising study area

Study area

St Neots is the largest town in Cambridgeshire, UK, with a population of 28 000. The town is situated on flat terrain which acts as the flood plain for the Great Ouse River and its tributaries. The study area is approximately 9.5 km^2 and is defined by the urban extent of the town, which includes suburbs and the surrounding road system (Figure 4.1). The area has a recorded flood history, including fluvial flooding adjacent to the river and surface water flooding in the urban area. St Neots is prioritised in the Cambridgeshire SWMP due to the number of properties and critical infrastructure at risk from surface water flooding, identified using multi-criteria analysis (Arcadis, 2012). The SWMP identifies several Priority Flood Spots (PFS) where flooding is of particular concern. PFS are Eaton Ford, Eynesbury, Town Centre and Riverside, as identified in Figure 4.1.

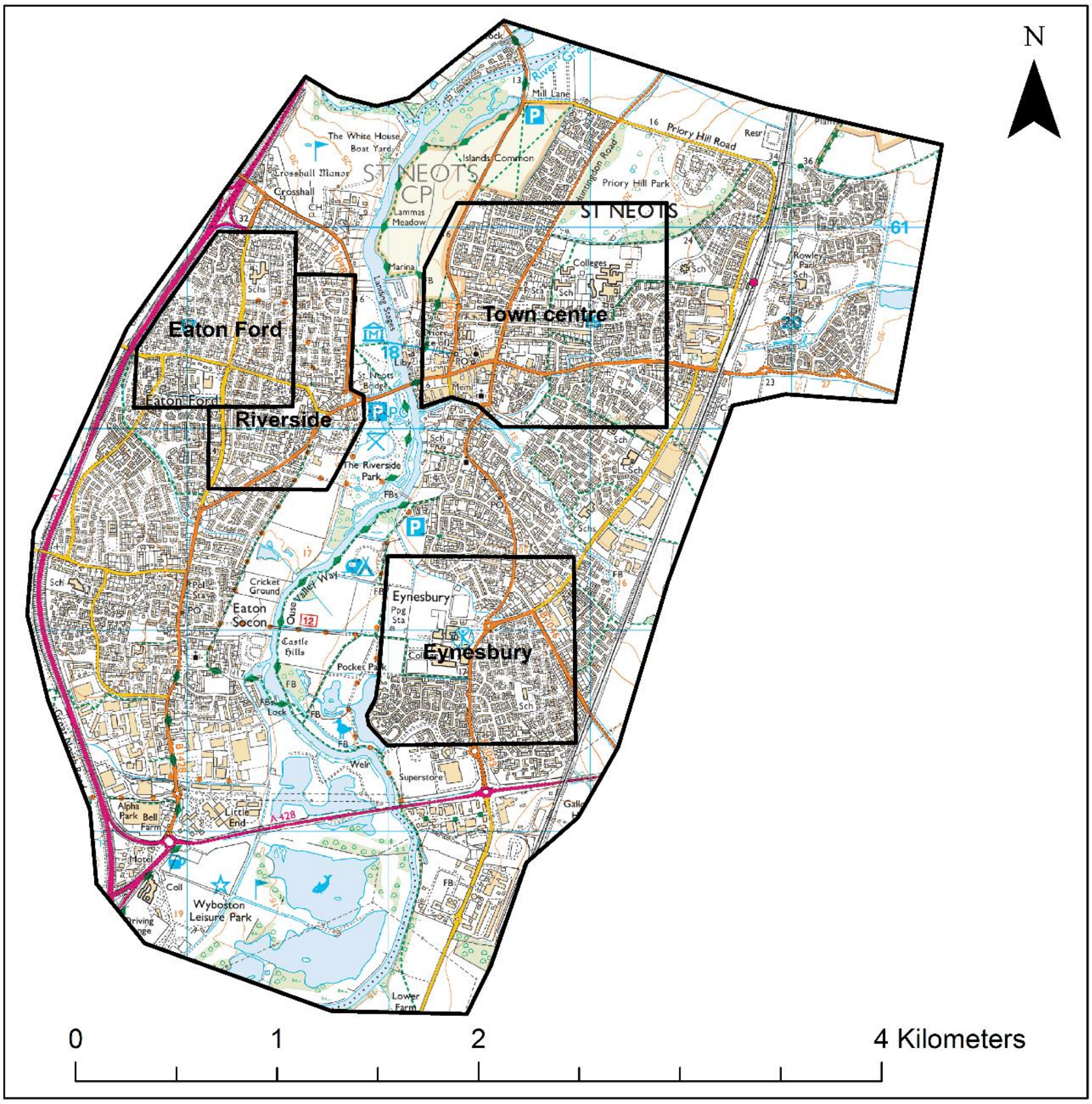

Figure 4.1: St Neots model extent, with priority flood spots highlighted

Characterising the catchment using ICM

This section details the approach used to represent St Neots within this model structure. Full details from this modelling study are published as part of the St Neots SWMP (Arcadis, 2012). ICM is an industry standard flood modelling software package which provides an integrated simulation of rainfall, overland runoff, the pipe network and watercourses (Innovyze, 2018).

The catchment area was specified by a polygon delimiting the surface water catchment. This area is contained by the A1 highway to the west and a railway line to the east (Arcadis, 2012). Elevation was represented using an irregular triangular mesh generated using 2 m resolution LiDAR data. The elevation of each triangle is set as the mean of the levels at each corner of the feature. The mesh was generated using the ICM mesh building function. Buildings were included within the landscape as voids within this mesh. This approach forces runoff to flow around the building thresholds. All rainfall landing on voids was specified directly into the surface water drainage system. Roads were included in the elevation model through a 100 mm reduction in elevation to account for kerb heights. This method was intended to ensure runoff would follow the road network before spilling onto other urban areas.

Land use was classified through application of a uniform roughness coefficient applied across the entire domain. The SWMP describes sensitivity analysis and determines a suitable surface Manning's roughness coefficient of 0.045 (Arcadis, 2012). The SWMP initially aimed to use variable roughness based on OS Mastermap land use types, however initial studies indicated a significant increase in processing and simulation time. Separation into urban and rural land use values was also discarded due to the "minimal impact on the flood extent" (Arcadis, 2012).

An infiltration rate of 2.5 mm/hour was applied across the entire domain, based on available local information (Arcadis, 2012). As with variable roughness, sensitivity analysis regarding this value is described in the SWMP.

Design rainfall was derived using IDF rainfall catchment descriptors from the Flood Estimation Handbook (FEH) (Centre for Ecology and Hydrology, 1999, 2013). Rainfall was represented using a series of design rainfall hyetographs representing rainfall in a 5.0%, 3.3%, 2.5%, 1.0% and 0.5% annual exceedance probability (AEP) events. A rainfall duration of two hours was applied for assessment due to previous screening identifying this event causing the most extensive flooding in the catchment across all exceedance probabilities (Arcadis, 2012).

The Revitalised Flood Hydrograph (ReFH) method was used to estimate the fluvial flows and levels for the modelled watercourses. On the River Great Ouse, the estimated 20 % annual probability fluvial flood flows and levels were applied as the upstream and downstream boundary conditions respectively for all the ICM simulations. However for the remaining tributaries in the ICM model, the estimated flood flow hydrographs (with a 2 hour storm duration) were applied as the upstream boundary condition for the respective annual probability flood event.

Characterising the catchment in CADDIES

The CADDIES model was set up to replicate as closely as possible the assumptions and approach applied using the ICM model. Elevation was included using the same 2 m resolution LiDAR DEM which underpinned the ICM approach. CADDIES applies runoff routing across a regular grid mesh and so the irregular triangular mesh applied in ICM could not be included within the model. Instead the elevation was input directly using the input DEM, reducing the pre-processing time required to generate the 2D mesh. Buildings were included within the elevation input file through application of a 1 m threshold level for all structures in the catchment. Thresholds were defined using the same OS Mastermap land use layer used to specify building locations in ICM. Raising the threshold of the structure replicated the ICM approach through forcing runoff to flow around the structure. Roads were included using the same DEM applied in the ICM approach.

The effects of land use were replicated through application of the same assumption to apply a constant uniform infiltration and roughness parameter across the entire catchment. Rainfall was also applied using the same input hyetographs applied in the ICM model.

The scope of CADDIES is limited to surface water flooding within the urban areas, and as such the watercourses were not included in the model.

4.1.2. Representing intervention scenarios

Representing the 'worst case' scenario

DEFRA guidance indicates that flood management should evaluate the effect of a 'plausible worst case scenario' (DEFRA, 2018b). The worst case scenario represents a total failure of the surface water drainage system. For this scenario the catchment was represented as described above, with no additional interventions applied. This scenario responds to recent UK government guidance which highlights a requirement for surface water management planning to develop a robust assessment of a ‘plausible worst case scenario’ (DEFRA, 2018b).

Representing the ‘surface drainage’ scenario

In ICM the urban surface water network was simulated using a detailed 1D model which represented pipe layout, diameters and invert levels. Runoff enters the surface water system through model nodes specified to each pipe and leaves the system at outfalls located along the watercourses running through the urban area.

The largest difference between the CADDIES framework and ICM was in the representation of the surface water sewer network. CADDIES does not include a 1D pipe system and so runoff captured by the surface water system was represented through adjusting the outflow rate within cells, effectively removing water from the simulation at a set rate. (Figure 3.1). Adjustments to cell outflow rates were made on a sub-catchment basis, defined using the sewer sub-catchments applied in the ICM model. It was assumed that the peak flow rate in each sub-catchment was set by the flow rate in the trunk sewer. The trunk sewer for each surface water sub-catchment was identified through evaluating the pipe diameters using a GIS database. The peak flow rate for each trunk sewer was calculated using the Colebrook White module in ICM. This rate was then averaged and applied across each cell in the associated sub-catchment as described in Section 3.2.3. The outflow drainage rate was capped at 300 mm/hour to avoid model instabilities generated by very high rates, typically generated where small catchments fed into culverts.

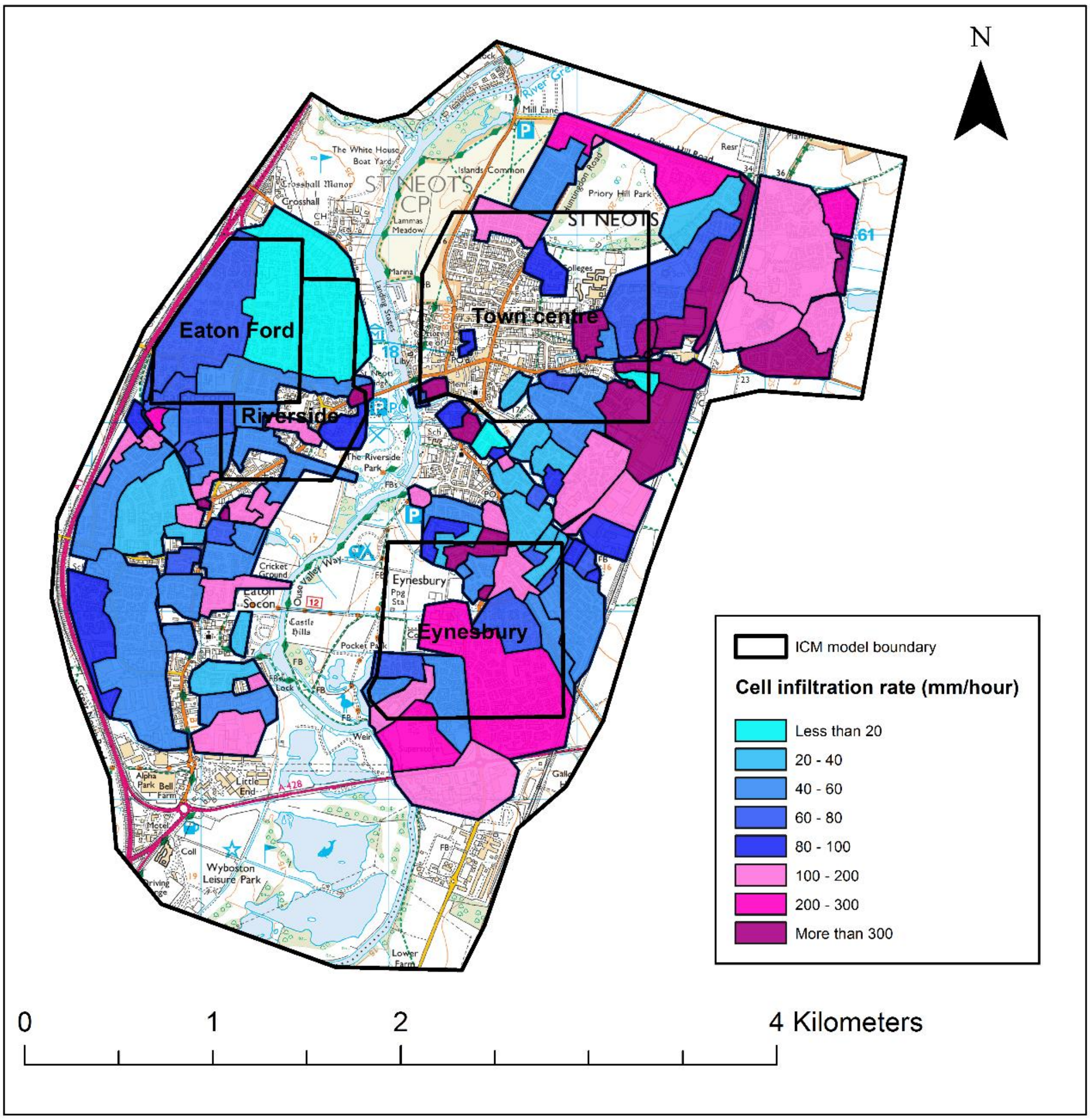

Figure 4.2: Surface water sub-catchments and corresponding outflow rates used in CADDIES

Representing the 'intervention scenario'

The intervention scenario corresponds to 'Option Combination C2' as specified in the Cambridge SWMP (2012). This option includes small scale engineering options applied at strategic locations within the catchment. Interventions included installing soakaways across the catchment, constructing swales at all four PFS, changing kerb heights and road elevation in Eaton Ford and Eynesbury, and adding a flood bund surrounding a property in Eaton Ford (Figure 4.3). These interventions were represented in ICM and CADDIES through changes to elevation models and parameters to reflect the planned interventions.

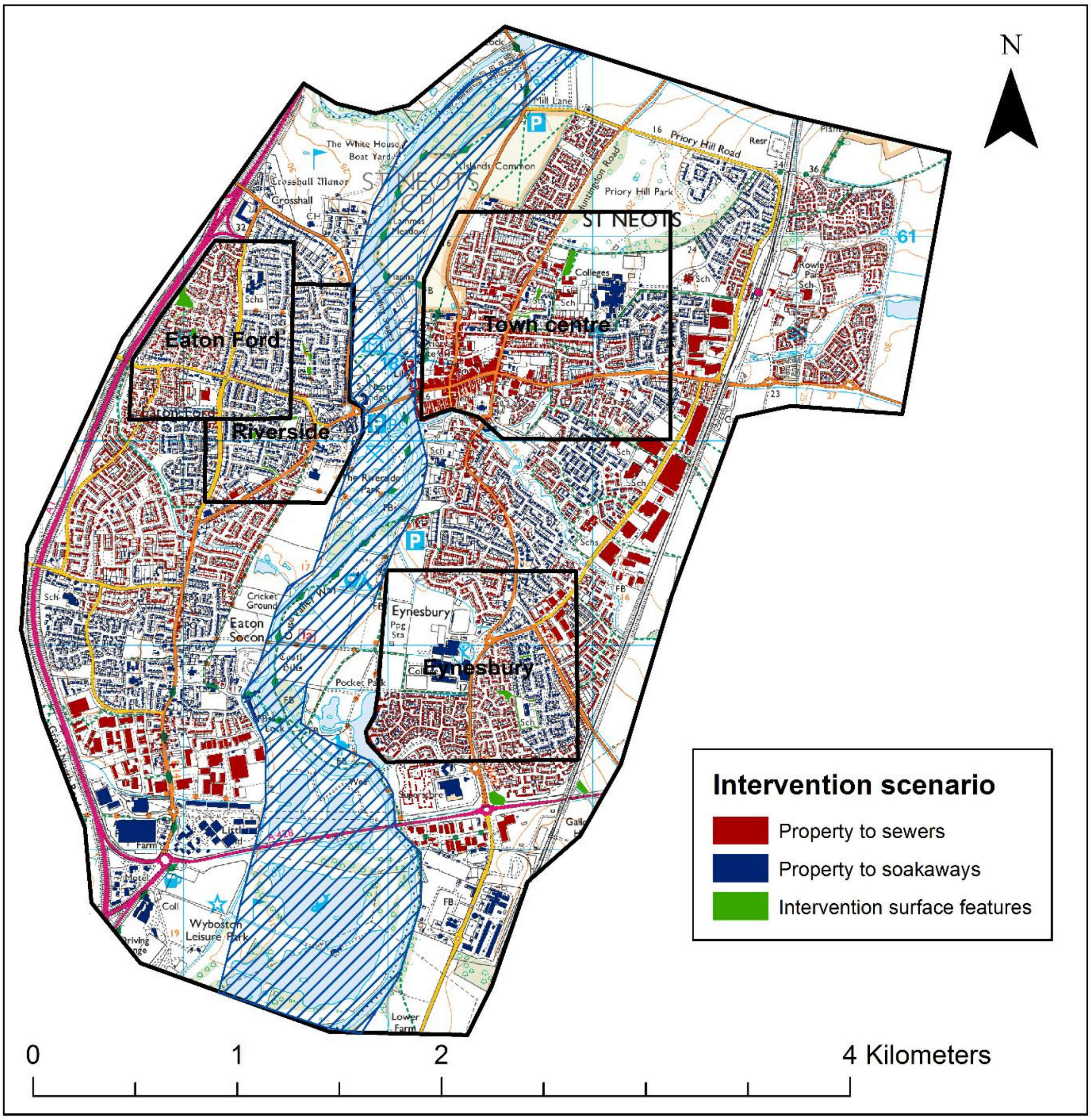

Figure 4.3: St Neots Surface Water Management Plan intervention scenario (Arcadis, 2012)

4.1.3. Simulation

CADDIES increases simulation speed whilst maintaining accuracy through application of an adaptive time step. The time step decreases towards a minimum as velocity increases, thus enabling the simulation to capture flow dynamics of fast moving runoff whilst stepping quickly through periods of low flow. Smaller time steps are more accurate, but result in a trade-off with simulation speed (Gibson et al., 2016). This simulation applied a minimum step of 0.01s, representing a very small time step capable of modelling fast moving flows. The modelled duration of each simulation included the rainfall event plus one hour for additional ponding and runoff.

4.1.4. Assessing model performance

Mean difference in peak depth per cell

Model performance was assessed through analysis of variation in flood depths between both modelling approaches. Performance was assessed in relation to the entire catchment (this included areas of fluvial flooding) and to PFS identified within the SWMP (Arcadis, 2012). Peak flood depth outputs from both models were transformed into an identical '.tif' format (a regular grid) using GIS and then variation was examined on a cell by cell basis. Evaluation of differences between model outputs was evaluated through analysis of the mean depth and standard deviation between corresponding model cells, constituting a comparison of absolute differences between cells.

Flood/ no flood correlation

In addition to assessing the absolute difference in flood depth it is also important for screening tools to reach similar conclusions, therefore a further metric, described as 'flood/ no flood' correlation (F/NF) was applied. F/NF correlation classifies the flood depth in each cell as either a flood or no flood, based on a flood threshold level of 30 cm (Environment Agency, 2013). All cells over this threshold are classified as a flood, all cells below it are classified as a no flood outcome. All cells in both models were classified and then compared to generate a percentage agreement ('F/NF correlation') between model outcomes for each scenario.

4.2. Results

4.2.1. Mean difference in peak depth per cell

Table 4.1 presents mean differences in peak flood depth per cell between CADDIES and ICM. Positive values indicate that CADDIES was on average shallower than ICM and negative values indicate CADDIES output a deeper peak depth per cell. The mean difference in peak depth per cell for the entire study area and across all AEP's was between 5 and 6 cm (with a standard deviation between 22 and 24 cm – Table 4.2). It should be noted that the 'entire study area' includes the watercourses, which are not currently included in CADDIES model. Focusing analysis on the PFS (Figure 4.1) demonstrated mean cell differences of less than 2 cm, with standard deviations between 5 and 12 cm across all AEP's.

Table 4.1: Model comparison for mean difference in peak depth per cell between CADDIES and ICM (m)

AEP	Entire study area	Eaton Ford PFS	Eynesbury PFS	Riverside PFS	Town Centre PFS	All PFS
Worst case scenario						
5.0 %	0.05	-0.00	-0.01	-0.01	-0.01	-0.01
3.3 %	0.05	-0.00	-0.01	-0.01	-0.01	-0.01
2.5 %	0.05	-0.00	-0.01	-0.01	-0.01	-0.01
1.0 %	0.04	-0.00	-0.01	-0.01	-0.01	-0.01
0.5 %	0.04	-0.00	-0.02	-0.01	-0.01	-0.01
Average	0.05	0.00	-0.01	-0.01	-0.01	-0.01
Surface water drainage scenario						
5.0 %	0.07	0.01	0.01	0.01	0.00	0.01
3.3 %	0.07	0.01	0.01	0.01	0.00	0.01
2.5 %	0.07	0.01	0.02	0.01	0.00	0.01
1.0 %	0.06	0.02	0.02	0.01	0.00	0.01
0.5 %	0.06	0.02	0.02	0.01	0.00	0.01
Average	0.06	0.02	0.02	0.01	0.00	0.01
Intervention scenario						
5.0 %	0.07	0.01	0.01	0.01	0.00	0.01
3.3 %	0.07	0.01	0.01	0.01	0.00	0.01
2.5 %	0.06	0.02	0.02	0.01	0.00	0.01
1.0 %	0.07	0.02	0.02	0.01	0.00	0.01
0.5 %	0.06	0.02	0.02	0.01	0.00	0.01
Average	0.06	0.02	0.02	0.01	0.00	0.01

All scenarios demonstrated consistent peak depths per cells between models. Model variance is approximately 1 to 2 cm with consistent performance across AEP’s. The distribution of variation in mean flood depth shows a consistent trend where differences in flooding are predominantly observed around the River Great Ouse flood plain and tributaries, with other smaller differences observed around building outlines and topographical features (Figure 4.4).

Table 4.2: Model standard deviation for mean difference in peak depth per cell between CADDIES and ICM (m)

AEP	Entire study area	Eaton Ford PFS	Eynesbury PFS	Riverside PFS	Town Centre PFS	All PFS
Worst case scenario						
5.0 %	0.22	0.06	0.06	0.06	0.09	0.07
3.3 %	0.23	0.06	0.07	0.07	0.09	0.07
2.5 %	0.23	0.06	0.08	0.07	0.10	0.08
1.0 %	0.24	0.07	0.09	0.08	0.11	0.09
0.5 %	0.24	0.08	0.10	0.10	0.12	0.10
Average	0.23	0.06	0.08	0.08	0.10	0.08
Surface water drainage scenario						
5.0 %	0.22	0.05	0.06	0.06	0.06	0.06
3.3 %	0.22	0.05	0.06	0.06	0.07	0.06
2.5 %	0.22	0.06	0.06	0.06	0.07	0.06
1.0 %	0.22	0.08	0.07	0.07	0.08	0.07
0.5 %	0.23	0.09	0.08	0.07	0.09	0.08
Average	0.22	0.07	0.07	0.06	0.07	0.07
Intervention scenario						
5.0 %	0.22	0.05	0.06	0.07	0.07	0.06
3.3 %	0.22	0.06	0.06	0.07	0.07	0.07
2.5 %	0.22	0.07	0.07	0.08	0.08	0.07
1.0 %	0.23	0.09	0.08	0.08	0.09	0.09
0.5 %	0.23	0.10	0.09	0.08	0.09	0.09
Average	0.22	0.08	0.07	0.08	0.08	0.08

Differences across the floodplain are attributed to the framework not representing the fluvial system which is included within the ICM model. This creates model variation through three key mechanisms. Firstly, input hydrographs add more water to the channel and tributaries, therefore increasing water depth on the flood plain, the differential is shown as red / orange in Figure 4.4. Secondly, ICM classifies the channels separately to the urban domain, meaning that water located here is not registered as a flood output. CADDIES does register this as a flood output, observed through CADDIES showing deeper flooding in the middle of channels (shown as green in Figure 4.4). Thirdly, the surface water drainage system in ICM outflows to the river and floodplain, increasing depth in these areas relative to the simplified mechanism in CADDIES which removes water from the model, rather than transferring it.

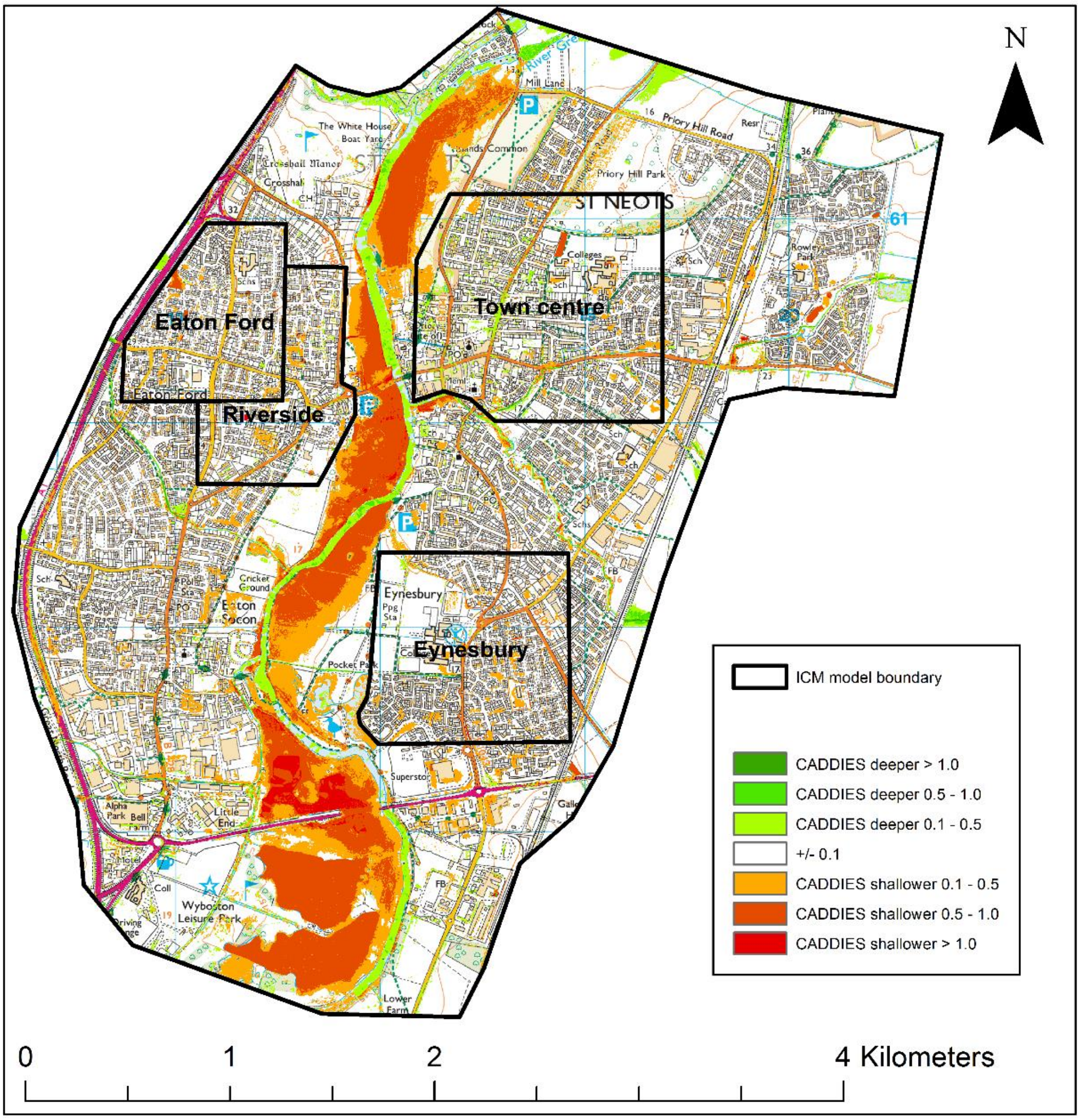

Figure 4.4: Mean peak flood depth difference per cell for the intervention scenario during a 1% AEP, 2 hour rainfall event

Variation at the edge of buildings is attributed to differences representing structures between the two methods. In ICM the elevation mesh technique represents buildings as a void, whereas the rapid screening framework applies an elevation uplift to represent structures. The elevation uplift can create areas of local ponding, and in the case of very deep water, also registers flood depths within a building. Representation as a void in ICM cannot register flood depth within the structure itself.

Closest model correlation is observed in the worst case scenario, in which the sewer system is not modelled. The mean difference in peak flood depth is 0.05 ± 0.23 m for the entire catchment and between 0.00 ± 0.06 m (Eaton Ford) and -

0.01 ± 0.10 m (Town Centre) for individual PFS. This supports application of the framework for modelling 2D runoff in circumstances with non-functioning drainage systems or extreme intensities which overwhelm system capacity; this type of short duration, high intensity rainfall is characteristic of the events typically leading to surface water flooding (Wheater and Evans, 2009; Douglas et al., 2010; Environment Agency, 2013; Committee on Climate Change, 2017). This finding is supported by previous evaluation of CADDIES, where close correlation was found when assessing the 2D runoff mechanism versus existing hydraulic models (Gibson et al., 2016) and the Environment Agency benchmarking tests (Néelz and Pender, 2013; Guidolin et al., 2016).

4.2.2. Flood/ no flood correlation

Table 4.3 presents analysis of F/NF correlation across the entire study area and individual PFS. The table indicates that models correlated at an average between 88% and 89% across the entire study area. This includes the fluvial system, the limitations of which are discussed in the previous section. Analysis of PFS, where fluvial input is minimised, indicates model correlation between 93% and 99%. PFS with no watercourses, such as Eynesbury, demonstrated the highest average correlation.

Models correlated more closely during the lower magnitude event in all cases. This is attributed to higher magnitude events exacerbating the differences between modelling approaches through two main mechanisms.

Firstly, more intense rainfall increases flood depths in areas not fully represented within the CADDIES approach, namely watercourses (where ICM includes additional inputs, as discussed in Section 4.2.1) and specific sub-surface features, such as culverts (presented in in Figure 4.8). Deeper flooding within these areas in ICM is driven by this increased flow, whereas CADDIES removes this water from the domain entirely. This effect is more prominent in scenarios where additional flow reaches watercourses in ICM through the sub-surface drainage system, evidenced through CADDIES outputs having a shallower mean peak flood depth than ICM during the surface water drainage and intervention scenarios (Table 4.1).

Secondly, the CADDIES model registers flooding across the full domain including buildings, which are represented using an uplift to the DEM, whereas the ICM approach used in the SWMP represents structures using voids which do not register flood depths. Therefore, as deeper and more extensive flooding occurs across the urban area, the CADDIES approach will inundate a greater area due to waters entering buildings and registering flooding within these cells. This effect is more prominent where deeper flooding is present across the catchment, as evidenced by CADDIES registering deeper mean flood depths for PFS during the worst case scenario (Table 4.1).

Application of the model subject to these limitations is further discussed in Section 4.3

Table 4.3: Model comparison for F/NF correlation (% of cells with the same F/NF classification using a 30 cm threshold)

AEP	Entire study area	Eaton Ford PFS	Eynesbury PFS	Riverside PFS	Town Centre PFS	All PFS
Worst case scenario						
5.0 %	89.3	99.1	98.8	97.9	98.2	98.5
3.3 %	88.9	99.0	98.5	97.4	97.9	98.2
2.5 %	88.5	98.9	98.2	96.8	97.6	97.9
1.0 %	87.5	98.4	97.3	94.5	96.6	96.7
0.5 %	87.3	98.0	96.4	93.1	95.7	95.8
Average	88.3	98.7	97.8	95.9	97.2	97.4
Surface water drainage scenario						
5.0 %	89.6	99.4	99.6	99.1	98.7	99.2
3.3 %	89.1	99.1	99.5	99.0	98.4	99.0
2.5 %	88.9	98.8	99.5	98.8	98.2	98.8
1.0 %	88.3	98.2	98.9	98.0	97.5	98.2
0.5 %	87.6	97.4	98.2	96.8	96.6	97.2
Average	88.7	98.6	99.1	98.3	97.9	98.5
Intervention scenario						
5.0 %	89.6	99.2	99.5	98.7	98.7	99.0
3.3 %	89.2	98.9	99.4	98.5	98.4	98.8
2.5 %	89.2	98.7	99.3	98.4	98.4	98.7
1.0 %	88.1	98.1	98.9	98.3	97.7	98.2
0.5 %	87.7	98.1	98.2	97.7	96.9	97.7
Average	88.8	98.6	99.1	98.3	98.0	98.5

Figure 4.5 presents F/NF correlation between the models across the worst case scenario in a 1% AEP event. F/NF correlation is indicated in green and variation in red. The figure presents a similar distribution to Figure 4.4, where variation is focused around the river channels and several topographical features, including buildings and embankments. To illustrate this point, watercourses and the fluvial flood zones have been identified in the figure.

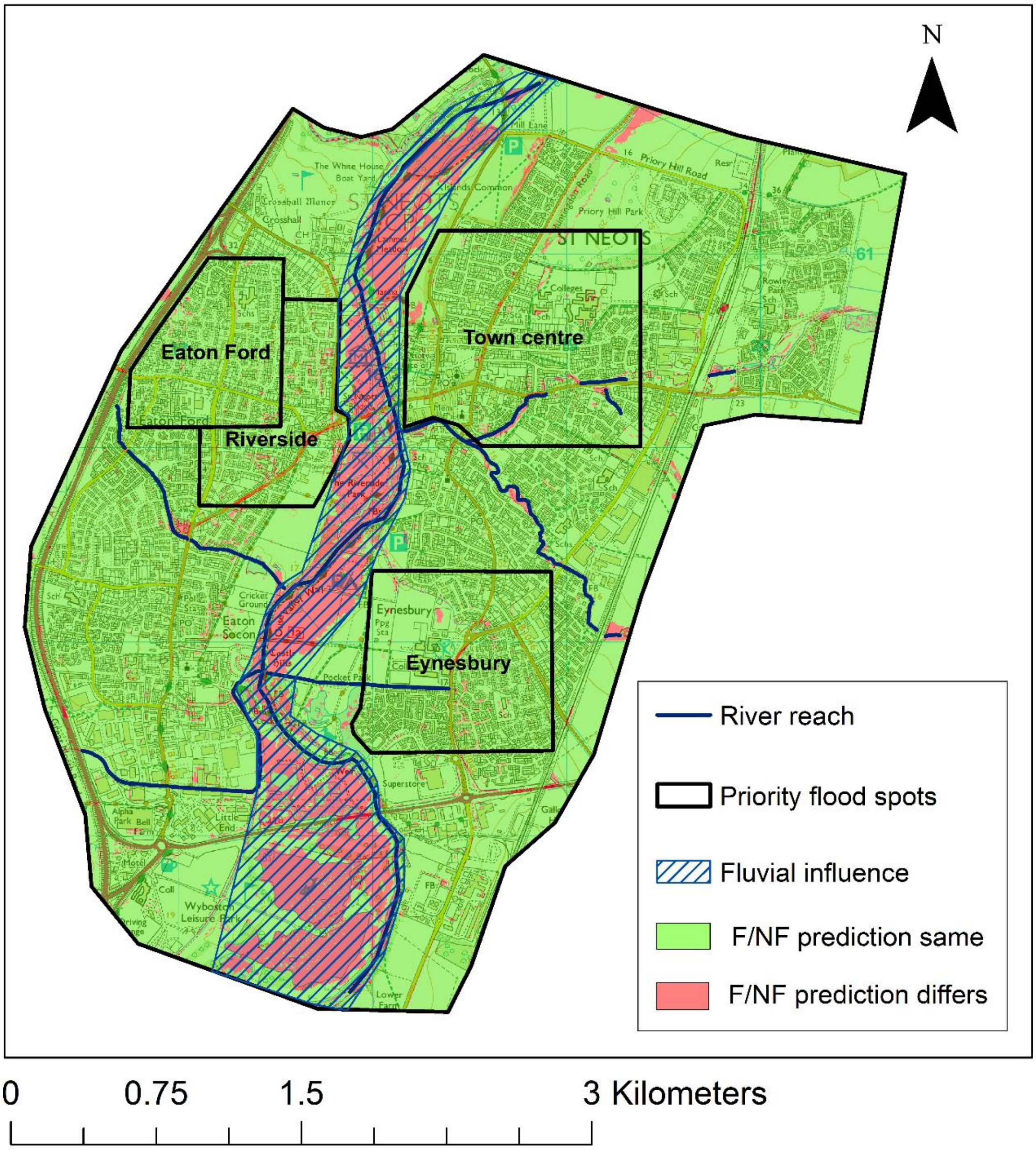

Figure 4.5: F/NF correlation overlain with areas of fluvial interaction for the worst case scenario, 1% AEP 2 hour event

Figure 4.6 and Figure 4.7 present F/NF correlation for the drainage system and worst case scenarios during a 1% AEP event. These figures show a similar distribution as found in the worst case scenario (Figure 4.6), indicating that model performance remains consistent across multiple levels of domain complexity.

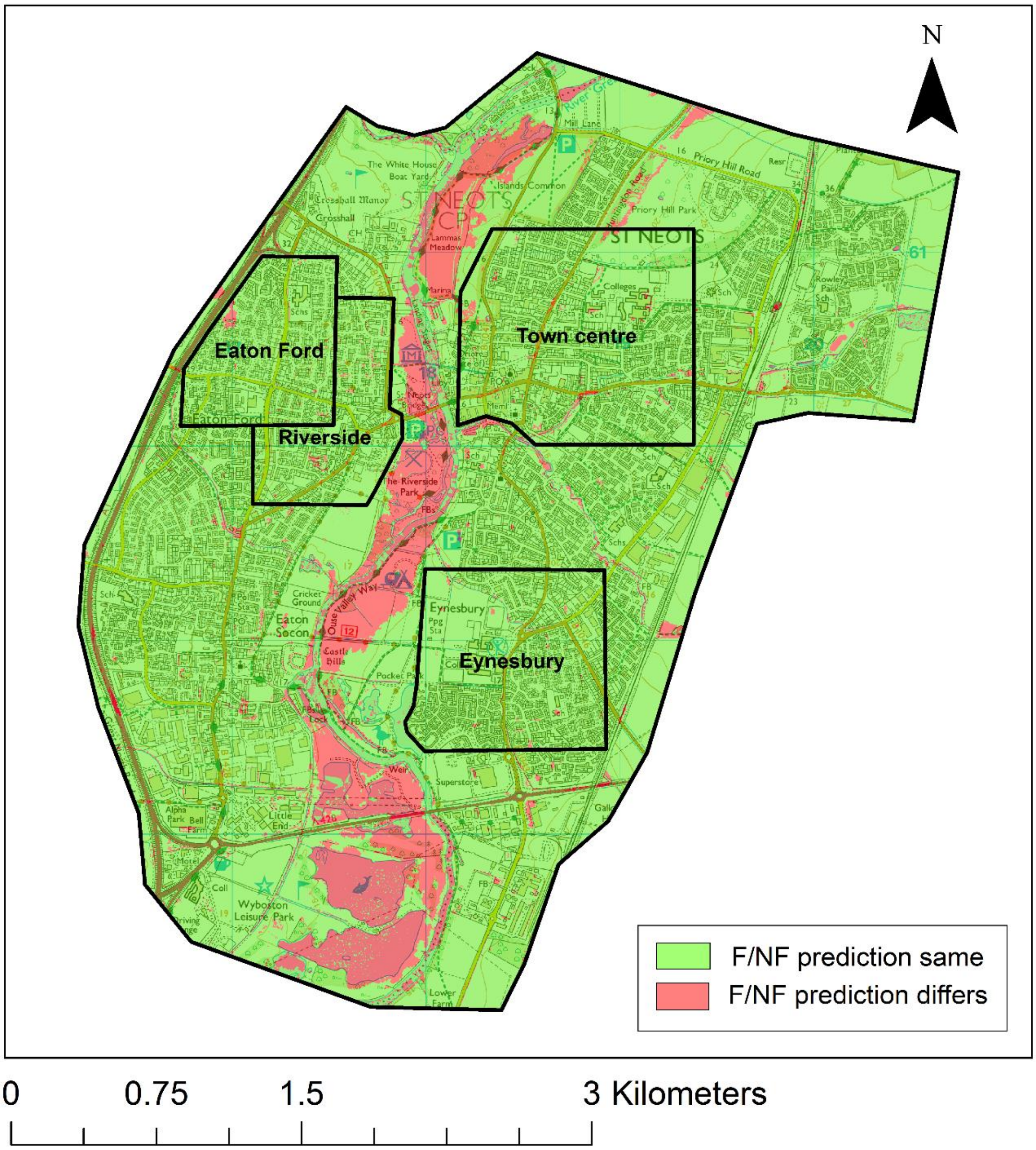

Figure 4.6: F/NF correlation for the ‘drainage system’ scenario during a 1% AEP, 2 hour rainfall event

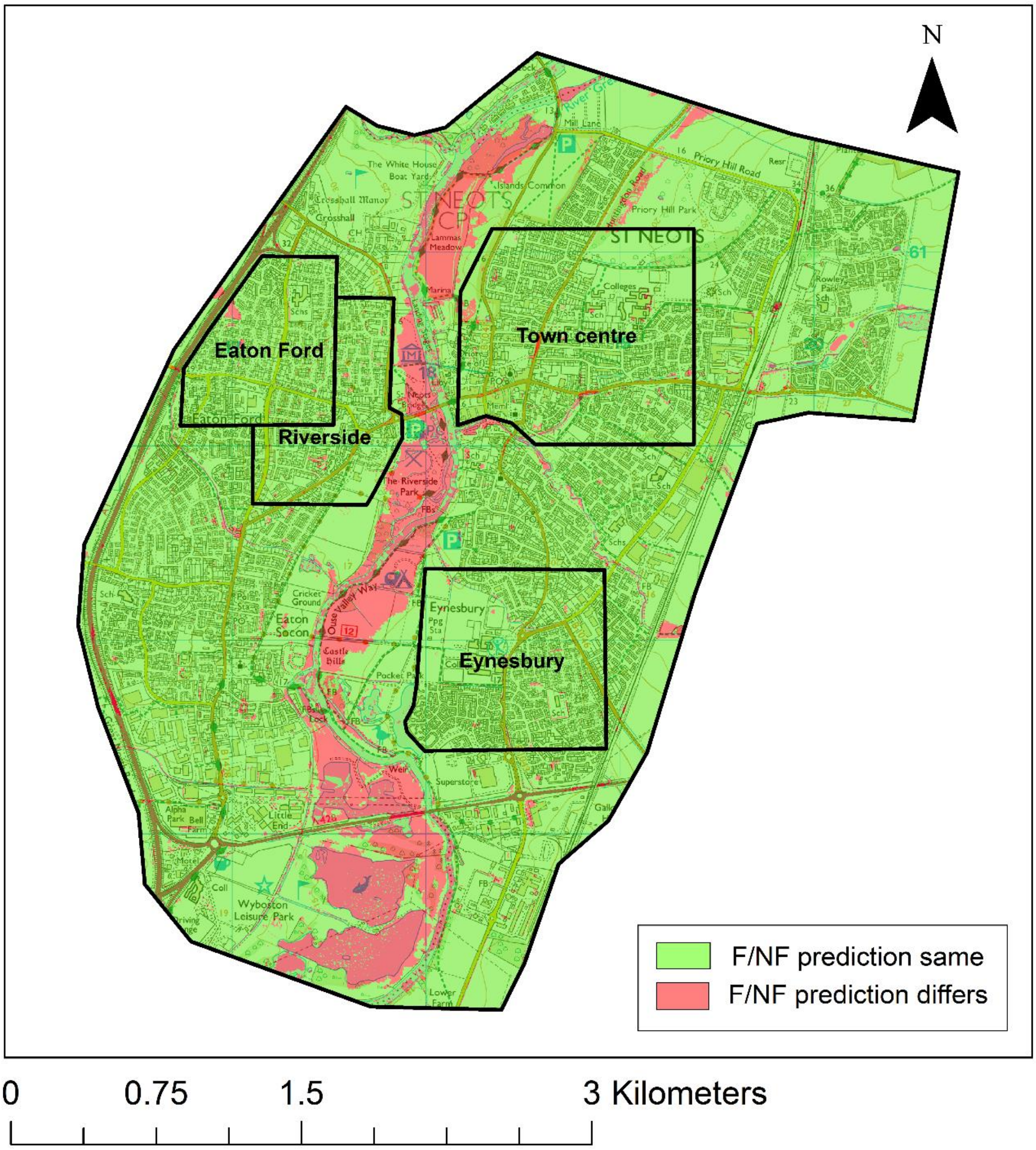

Figure 4.7: F/NF correlation for the ‘intervention’ scenario during a 1% AEP, 2 hour rainfall event

Embankments which demonstrate variation in F/NF prediction are those which are served by culverts, represented by a 1D system, and therefore not included within the CADDIES model. An example of this can be seen to the east of Eynesbury where the road embankment ponds water, resulting in a localised area of F/NF variation (Figure 4.8). This indicates a limitation of rapid scenario screening framework.

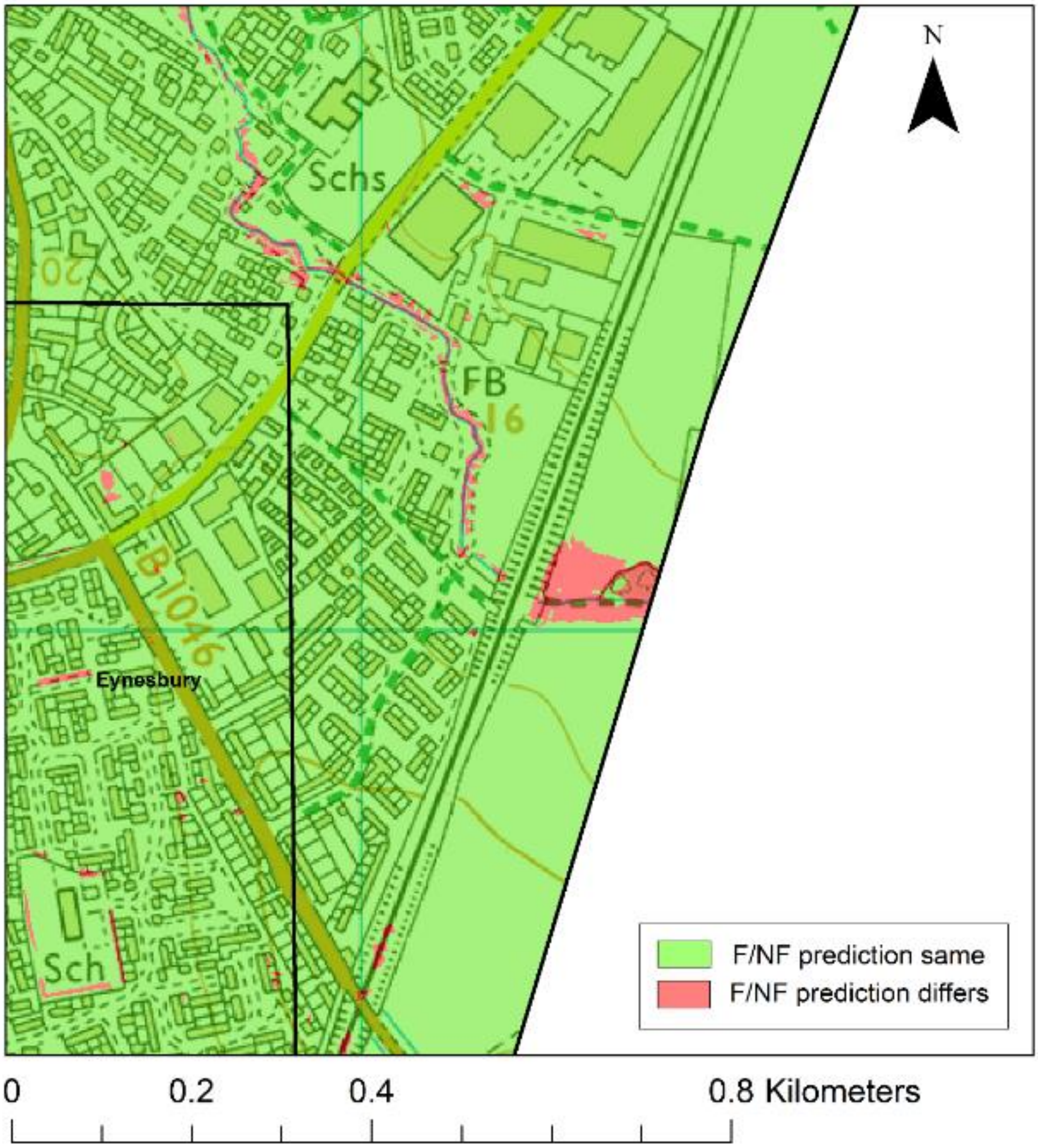

Figure 4.8: F/NF correlation at Eynesbury for the ‘intervention’ scenario during a 1% AEP 2 hour rainfall event

4.3. Discussion

4.3.1. Screening a worst case scenario using the framework

The degree of utility of the framework to screen catchments in the worst case scenario is evaluated relative to three questions, each progressing to a more nuanced level of application: Firstly, can the framework broadly replicate flood dynamics and identify PFS in the urban catchment? Secondly, does the framework correlate flood depths with industry standard techniques? And thirdly, are outcomes from application of the rapid screening technique comparable to analysis using the industry standard approach?

During the worst case scenario the rapid screening framework replicates identification of the four PFS identified in St Neots as part of the Cambridgeshire SWMP (2012). These regions are Eaton Ford, Eynesbury, Town Centre and Riverside. Only a minor variation in peak depths per cell of 0 to 2 cm ± standard deviation of 5 to 12 cm is observed in measurements across each PFS during all AEP events. Outcomes from both approaches are likely to be very similar due to the 97.4% average F/NF correlation across all PFS and each of the AEP events. The 88% F/NF correlation observed across the whole catchment is likely to be mitigated in practice through initial catchment assessment to discount areas of fluvial interaction or where complex subsurface drainage features create localised anomalies.

The models demonstrate similar results and outcomes, providing evidence that the rapid screening framework is acceptable for the purpose of screening flood hazards during worst case scenarios. However, it is emphasised that this comparison is between two models, and not recorded observations. Simplifications required for all models mean that neither approach should be considered a fully accurate representation of real life. In practice, models will always trade off simplifications in representation and limitations in data with accuracy, and should therefore be considered as tools for a specific application (Box, 1976), in this case screening using readily available data.

Model utility should be considered relative to the simplifications necessary within all flood models. Flood model accuracy in highly complex urban environments is likely to be affected by many factors. Variation between models and urban flood findings can commonly be attributed to uncertainties including: inaccuracies in topographic surveys (Dottori et al., 2013); spatial resolution of elevation models missing permanent micro-topographical features such as kerbs (Fewtrell et al., 2011), walls (Yu and Lane, 2006), ditches (Bates et al., 2006) and fences (Mignot et al., 2006); temporary micro-topographical features such as cars (Dottori et al., 2013); landscapes altered by high energy flows (Dottori et al., 2013); flow interactions with buildings, which vary with height and inundation duration (Chen et al., 2012; Schubert and Sanders, 2012); uncertainties in statistical construction of temporal and spatial patterns of design rainfall (particularly for low probability events); changes to boundary conditions during storms (Bates, 2004); and local short term irregularities such as blocked or damaged drainage features (Neal et al., 2009).

4.3.2. Including urban drainage systems within the framework

The primary limitation of the rapid scenario screening framework is considered to be the trade-off between representation of the 1D pipe system with a model architecture aimed at speed (Webber et al. 2018a; 2018b). This chapter has identified that representing the pipe system using spatial variation in cell outflow rates across model sub-catchments demonstrates an average PFS F/NF correlation of 98.5% with the 1D network included in the ICM model. Both models screen the catchment and identify the four areas at risk from surface water flooding. Within these PFS, models demonstrate a mean variation per cell of 0 to 2 cm with a standard deviation of 5 to 9 cm, alongside an average F/NF correlation of 98.5%. Correlation is similar across all return periods. The result of this correlation is that both modelling approaches are likely to result in similar outcomes for recommending further detailed modelling and prioritising areas of the catchment where interventions should be evaluated.

Representing sub-surface drainage using a simplified cell output rate appears an effective trade-off in areas where the water is removed; however, carries the limitation that water is not transferred to other regions where it may influence flooding, for example outflows to watercourses. High intensity short term rainfall, responsible for the majority of urban surface water flooding, is unlikely to contribute significant amounts of volume to cause flooding in major watercourses. However, this limitation should be considered carefully as the approach may not be suitable where small water courses, culverts or pipe full flow phenomenon such as surcharge are expected to contribute to surface water flood risk. This can be mitigated using initial analysis of flood hazards through taking actions such as evaluating flood histories, interviewing catchment stakeholders and reviewing previous studies in the area of investigation. These actions are typically recommended as part of strategic flood risk assessments (DEFRA, 2010).

As with the worst case scenario, finding that the rapid scenario screening framework correlates with existing methodologies is caveated with the need to examine the spatial distribution of results to ensure action taken reflects the strengths of the framework; namely, that the model is used to support further study in areas not influenced by fluvial flooding and that allowance has been made for significant sub-surface features.

4.3.3. Modelling interventions using the framework

The most complicated scenario within this analysis is the inclusion of additional interventions alongside the existing drainage network. This scenario involves representing the land use, sub surface drainage and additional flood protection measures modelled using ICM. Both models identify PFS and correlate closely on mean peak flood depth (average 1 cm ± 8cm) in cells and F/NF correlation (average 98.5%) within these regions. As discussed in previous sections, spatial analysis of differences attributes variation to watercourses and significant sub surface features such as culverts.

Close correlation between the two approaches supports application of rapid screening as a tool for examining an initial assessment of interventions in urban catchments (Webber et al 2018a, 2018b). Complexities modelling runoff in urban catchments (Dottori et al., 2013) alongside the high computational cost of 2D modelling (Elliott and Trowsdale, 2007; Hunter et al., 2008b; Mikovits et al., 2015) have traditionally restricted the number of interventions which can be screened during design. Speed of analysis using this framework lends the utility of screening many interventions in a short space of time. Utility is supported through the application of simple data, such as elevation, land use mapping and rainfall events. This data is likely to be available in the initial stages of engineering projects and therefore provides an opportunity for decision makers to examine catchments during preliminary analysis and to generate evidence to support the strategic direction and requirements for further detailed design.

4.3.4. Model speed

Due to licensing restrictions and data confidentiality it was not possible to run both approaches using the same computer, necessary for a robust comparison of simulation speed. However, it should be noted that existing published studies have investigated CADDIES speed, and demonstrated a five to twenty fold speed increase of the model versus ICM (Gibson et al., 2016). Additionally, there is a large body of supporting literature detailing the computational efficiencies achieved through implementing cellular automata flood models relative to traditional 2D modelling (Dottori and Todini, 2011; Ghimire et al., 2013; Li et al., 2015; Liu et al., 2015; Caviedes-Voullième et al., 2018; Lu et al., 2018). This is described in detail within Section 2.2.2. Application of rapid models facilitates analysis of many different scenarios and enables decision support to generate an evidence base which can include a large range of return periods, possible interventions and study area assumptions.

4.3.5. Sensitivity to changes in the cell output rate

The suitability of representing drainage through adjusting cell output rates was subject to a preliminary analysis where potential adjustments to the rate were evaluated. Preliminary analysis was made across the entire catchment area modelled under the assumptions outlined for the surface water drainage scenario (Figure 3.1) and standard cell output rate calculations, as described in Sections 3.2.3 and 4.1.2.

Table 4.4 presents the mean difference in peak depth per cell between ICM and CADDIES for a cell output rate calculation ± 50%. This indicates low sensitivity to changes in input value, with a slightly lower variation attributed to a reduced output rate. This may be attributable to the CADDIES method over estimating the drainage efficiency through removing runoff from each cell, rather than through specific inlets to the surface water system.

Table 4.4: Mean difference in peak depth per cell (m) between CADDIES and ICM whilst varying cell drainage output rates across the entire catchment in the ‘surface water drainage’ scenario

AEP	**Rate -50%**	**Standard calculation**	**Rate +50%**
5.0 %	0.06 ± 0.22	0.07 ± 0.22	0.07 ± 0.22
3.3 %	0.06 ± 0.22	0.07 ± 0.22	0.07 ± 0.22
2.5 %	0.06 ± 0.22	0.07 ± 0.22	0.07 ± 0.22
1.0 %	0.06 ± 0.22	0.06 ± 0.22	0.06 ± 0.22
0.5 %	0.05 ± 0.23	0.06 ± 0.23	0.06 ± 0.23
Average	0.06 ± 0.22	0.06 ± 0.22	0.07 ± 0.22

When this variation is evaluated in a specific catchment this difference is clearer. Table 4.5 presents the same analysis undertaken in the Eynesbury PFS. This indicates lower variation when the cell output rate is restricted and higher variation when the rate is increased, supporting the previous argument.

Table 4.5: Mean difference in peak depth per cell (m) between CADDIES and ICM whilst varying cell drainage output rates across the Eynesbury PFS in the 'surface water drainage' scenario

AEP	**Rate -50%**	**Standard calculation**	**Rate +50%**
5.0 %	0.01 ± 0.06	0.01 ± 0.06	0.01 ± 0.06
3.3 %	0.01 ± 0.06	0.01 ± 0.06	0.02 ± 0.06
2.5 %	0.01 ± 0.06	0.02 ± 0.06	0.02 ± 0.06
1.0 %	0.01 ± 0.07	0.02 ± 0.07	0.02 ± 0.07
0.5 %	0.01 ± 0.08	0.02 ± 0.08	0.03 ± 0.08
Average	0.01 ± 0.07	0.02 ± 0.07	0.02 ± 0.07

Table 4.6 applies the F/NF classification to determine how this variation effects outcomes from the rapid screening framework versus ICM. This indicates low variability in the division of outcomes when evaluating a flood or no flood across the entire catchment. A more detailed breakdown of this classification presented in Table 4.7 identifies that modifying the cell output rate creates a trade-off within correlation. Breaking down the classification into model variation in F/NF classification identifies a pattern in which decreasing the output rate will lead to higher flood match outcomes at the expense of lower no flood match outcomes. Increasing the drainage rate has the opposite effect, where no flood matches are more frequent at the expense of flood matches.

Table 4.6: F/NF correlation (%) per cell between CADDIES and ICM whilst varying cell drainage output rates across the entire catchment in the 'surface water drainage' scenario

AEP	**Rate -50%**	**Standard calculation**	**Rate +50%**
5.0 %	89.5	89.6	89.6
3.3 %	89.1	89.1	89.1
2.5 %	88.9	88.9	88.9
1.0 %	88.2	88.3	88.3
0.5 %	87.7	87.6	87.6
Average	88.7	88.7	88.7

Table 4.7: Comparison of F/NF classification outcomes between models (averaged across all AEP events) whilst varying cell drainage output rates across the entire catchment in the 'surface water drainage' scenario

Outcome		**Rate -50%**	**Standard calculation**	**Rate +50%**
ICM F	CAD F	1.0	0.8	0.8
ICM NF	CAD NF	87.7	87.9	88.0
ICM F	CAD NF	9.8	10.0	10.0
ICM NF	CAD F	1.5	1.3	1.3

Although similar, it should be noted that variation in F/NF correlation does not represent a 'true'/ 'false' 'positive'/'negative' metric, as would be the case if compared versus observed data. Instead this breakdown indicates the divisions of classification that each model outputs. Model variation is discussed extensively earlier in the chapter and is predominantly attributed to the influence of fluvial flood mechanics (ICM F, CAD NF) and subsurface drainage features (ICM NF, CAD F). This is of particular relevance when assessing the entire catchment, of which a large area is impacted by fluvial flooding (Figure 4.5).

Low sensitivity to the change in drainage parameters is attributed to analysis focusing on extreme short duration rainfall events responsible for surface water flooding, where the predominant controlling factor in flooding is overland flow rather than the drainage system, which has been developed to manage design standard events across the urban area. This results in a water balance with relatively low losses attributable to the output value (drainage rate) compared to the extreme rainfall input.

Water balance can be evaluated through a ratio of total output to input volumes (Pina et al., 2016). This provides a high level summary of water movement across a catchment during an entire event. This analysis is focused on evaluating peak surface water flood depths, the controlling factor in flood damages (Penning-Orwsell et al., 2010). The timing of peak depths varies from cell to cell based on the time of concentration, therefore a water balance summary across a whole simulation will not adequately capture the controlling conditions which lead to this snapshot within the simulation. This is predominantly due to ongoing infiltration rates continuing beyond the timing of the peak depth.

Instead the water balance and relative weighting of losses can be evaluated through examining the scale of drainage systems and likely input and output values during conditions leading up to peak flood depths. 44% of the model domain is specified to drain to a sewer sub-catchment, the average drainage rate per sq.m for each catchment is approximately 30 mm/ hour. Drainage catchment locations and capacities are shown in Figures 4.2 and 4.9, note each cell is 4 sq.m, with F/NF for individual cells detailed in Figures 4.5 to 4.7.

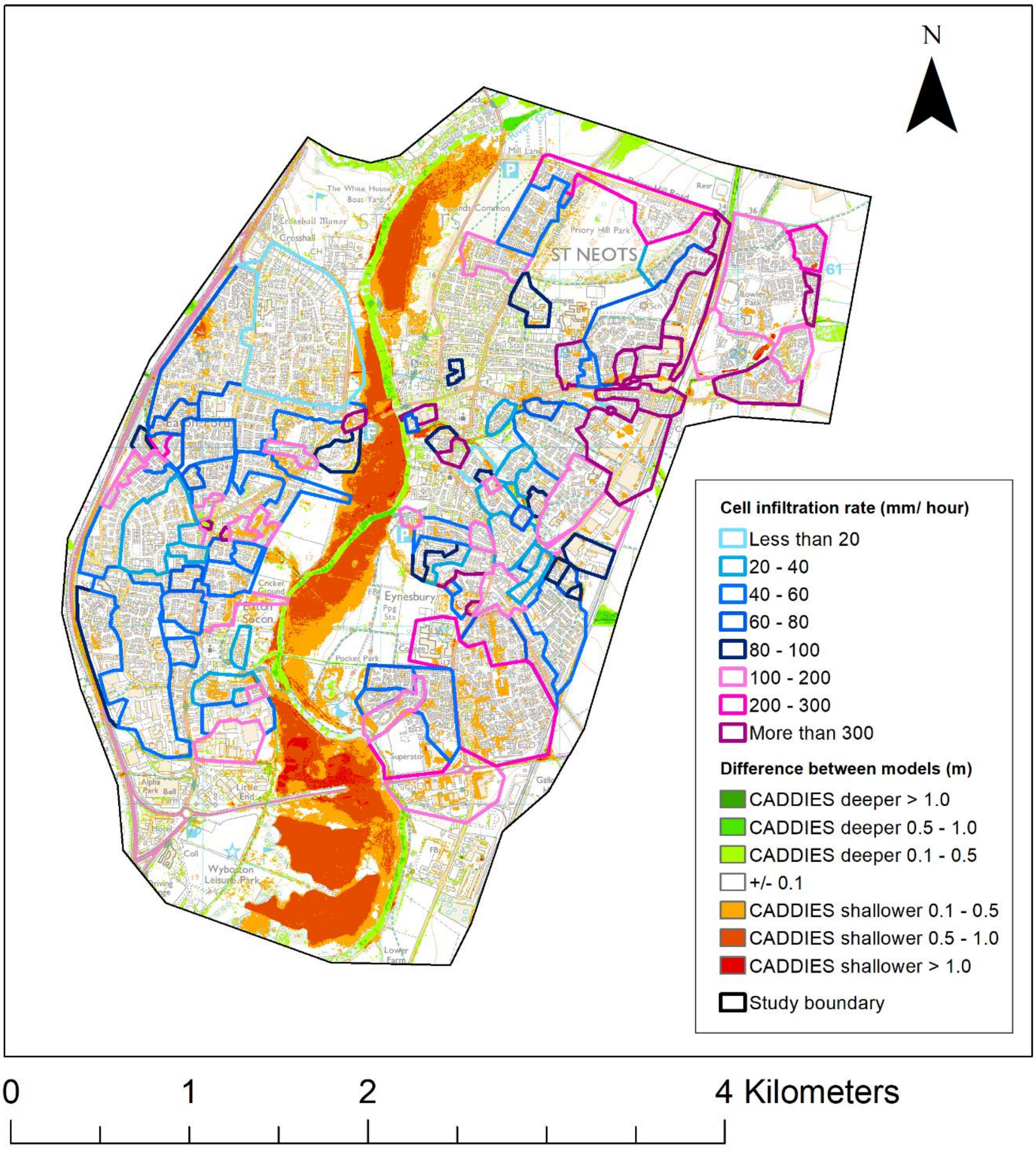

Figure 4.9: Absolute difference between models in 0.5% AEP, 2 hour surface water drainage scenario, with sub-catchments highlighted.

Peak rainfall inputs, which are likely to be linked to peak flood depths, reach a maximum intensity of 140 mm/ hour across the entire catchment. Therefore an increase or decrease of 50% in cell output rate makes a marginal difference relative to the overall flood dynamics, explaining a low sensitivity to this parameter. This emphasises the predominance of overland flow, rather than the drainage rate, in predicting surface water flooding during extreme events, which supports the application of rapid 2D modelling to screen flooding during extreme intensity events.

4.3.6. Considerations for future modelling in response to Objective Eight

The purpose of validating the framework is to support application towards utility as a catchment screening methodology, intended to support and direct further data collection, detailed modelling and management actions. Analysis of the two models identifies that the advantages of the rapid screening framework enable simulation of many scenarios and potential intervention strategies at a low computational and setup resource cost and with comparable accuracy relative to other contemporary 2D simulation approaches. Automation of the approach can generate hundreds of simulations and build an extensive set of 'what if' scenarios for preliminary decision support (Webber et al 2018a, 2018b). Simplification of several physical parameters, such as the sub-surface drainage system and watercourses, mean that this model should be applied only as an initial screening tool to direct and inform, rather than replace, detailed design models. Future application of the framework should be subject to recognition of several key considerations identified during the validation process.

The application of the framework should be subject to a preliminary analysis of catchment flood mechanisms. The 2D model applied is designed specifically to model surface water runoff, and as such is unsuitable in its current state for modelling other types of flooding, such as fluvial and groundwater floods. It is recommended that different approaches are applied where a preliminary analysis identifies these mechanisms as the cause of flooding within a particular study catchment. Preliminary analysis can be facilitated through investigating published flood reports, incident logs, flood histories and discussion with catchment stakeholders. Flood reports in the UK are available through catchment flood management partnerships which regularly update surface water flood management plans. Elsewhere, interaction with catchment stakeholders or reviewing local authority reporting is likely to contribute flood histories.

The model does not include a 1D representation of the piped system. Including an allowance for pipes through parametrisation of cell output values demonstrates a close correlation in urban areas, however will not represent complex pipe flow dynamics such as surcharge or throttled flows. The assumption of uniform rate across each sub-catchment, applied based on the trunk sewer diameter is also a simplification which does not necessarily represent water movement, capture from the surface into the pipes and spatial variation in contribution rates across the sub-catchment. The framework is therefore unsuitable in areas where sewer flooding is highlighted as the predominant cause of flooding. As with fluvial and ground water mechanisms, preliminary analysis of flooding is likely to be a sufficient process to mitigate this limitation, and enable a different approach, such as a 1D sewer network model, to be applied in areas where this is the primary issue.

In certain areas data may be limited or subject to restricted access. This can include elevation models, rainfall data, land use characteristics and sewer schematics. In particular, sewer networks often consist of legacy assets which may not be accurately mapped or may be classified as commercially sensitive and subject to data protection, therefore unavailable to the model user. These restrictions can be overcome through a framework design adaptable to a wide range of data types and resolutions in which a range of data sources can be converted into the appropriate formats. As with all models, the accuracy of analysis will be controlled by the resolution of data (Dottori et al., 2013). However, depending on the purpose of application, ranging from screening approximated catchment flood dynamics through to assessing high resolution data, the tool may still be suitable and will output useful analysis of removing runoff across the catchment. Particularly in the context of high intensity rainfall which exceeds drainage system capacity and soil infiltration rates.

Where land use data is unknown, the model can be run using uniform global parameters, sourced from academic literature. The same principle applies to the piped system, which can be approximated using a uniform drainage rate, a concept similar to that applied within the Environment Agency Surface Water Flood Mapping studies (Environment Agency, 2013).

Practical utility of the rapid screening framework can be summarised as suitable for initial catchment screening. This has application as part of developing evidence or enhancing scenario exploration and stakeholder communication to aid decision support. As with all models, this provides a tool for a specific purpose and it’s uncertainties and limitations should be evaluated on a case by case basis (Blöschl, 2006; Dottori et al., 2013).

4.4. Chapter conclusions

This chapter validates the application of the framework for screening catchment flood dynamics by comparing framework outputs with those from a published SWMP. Key conclusions from this chapter are:

- The rapid scenario screening framework is a suitable tool for screening surface water PFS and high level flood dynamics in urban catchments.
- The framework demonstrates close correlation with ICM when evaluating surface water flood hazards within priority flood spots. This finding applies to models constructed to multiple levels of detail, including a worst case overland flow (97.4%), inclusion of the sub-surface drainage system (98.5%) and addition of interventions to the catchment surface (98.5%).
- Parameterisation of cell output rates to represent the sub surface drainage system demonstrated high correlation with 2D-1D modelling however data confidentiality, record uncertainties and legacy assets mean that detailed schematics of surface sewers may not always be available, particularly at the initial stages of intervention screening. Low sensitivity of cell output parameters during extreme rainfall indicates that broad scale parameterisation, such as that undertaken as part of Environment Agency (2013) surface water flood mapping, is suitable for preliminary screening where this is the case.
- Application of the framework should be supported through preliminary analysis to ensure surface water flood hazards are not caused by interactions with local sub-surface drainage or river systems.

Comparison indicates that the rapid scenario screening framework is a promising tool for screening flood hazards and evaluating intervention options across urban catchments. This validation supports using the framework for initial catchment screening as part of scenario exploration to aid decision support. Subsequent chapters in the thesis will apply the methodology from Chapter Three and lessons learnt validating the approach in this chapter to evaluate reliability and resilience of surface water flood management strategies.

5. EXAMINING THE EFFECTS OF STRATEGIC INTERVENTION ZONES

This chapter responds to Objective Four: 'Investigate the flood reduction performance of strategic and specific interventions'. This is achieved through examining the effects of strategic intervention zones across an urban catchment. Understanding the effects of strategic intervention zones forms the initial stage of surface water flood management through developing evidence regarding the type and scale of action required to address hazards in a catchment. This is achieved through identifying catchment flood dynamics to prioritise areas where management is required and scoping the scale of intervention effects required to manage flooding.

It is envisaged that this method of analysis will form the basis of an initial assessment, which will support and direct further detailed modelling. Therefore, the scope of this chapter is to provide a broad scale screening of potential strategies using data which would be available at the beginning of a flood management project. This application is intended to demonstrate the potential for the framework to output useful results for pragmatically steering further investigation whilst using minimal data and computational requirements (Mikovits et al., 2015).

The chapter is structured by introducing the case study of an urban region in the UK. Model set up is described using the structure of the framework specified in Chapter Three. This set up is then modelled as a base case scenario to understand catchment flood hazards and prioritise regions for further analysis. The effects of a range of 'strategic intervention zones', each represented via applying intervention effects across a large area of the catchment, are then evaluated across the prioritised region. Analysis is undertaken to identify flood dynamics and estimate damage costs associated with each strategy.

The work presented in this chapter is published in 'Rapid assessment of surface water flood management options in urban catchments' (Webber et al., 2018a).

5.1. Defining ‘strategic intervention zones’ and ‘specific interventions’

This chapter focuses on the flood reduction effects of strategic intervention zones across an urban catchment. Before zones are evaluated it is first important to define the terminology applied in the thesis.

Strategic intervention zones are regions of the catchment in which parameters are modified to investigate the scale and scope of potential effects achievable through broad changes to landscape characteristics. Zones are typically evaluated using coarse resolution analysis as part of preliminary catchment investigations. Therefore, analysis using strategic zones is intended to examine how modifying catchment characteristics will influence flood dynamics, and is not designed to replicate the effects of any one particular intervention type.

A more detailed representation of interventions can be achieved through modelling **specific interventions**. These are representations of particular measures, parameterised and placed in a defined location across a study area. Specific interventions will be evaluated in more detail in Chapters Six and Seven.

Figure 5.1 presents the difference between modelling a strategic intervention zone (left) versus the specific interventions which may contribute towards a desired strategic outcome (right).

Strategic intervention zone

Specific intervention placement

Area of parameter changes

Figure 5.1: Indicative example presenting the theoretical differences between a strategic intervention zone (left) versus siting specific interventions (right)

Modelling strategic intervention zones is a simpler process than defining specific interventions as characteristics and siting of individual measures is not required. This fast approach provides the utility of evaluating how broad scale changes to a catchment will manifest themselves towards flood management potential. This is of particular application to providing an initial indication to the level of change required to manage flooding and can provide the basis towards identifying the requirements for specific interventions to achieve the desired effects.

5.2. Study catchment

The study catchment examined is in the city of Exeter, located in South West England. The city has a population of 120 000 and functions as the administrative and economic hub for the county of Devon (Devon County Council, 2011).

The 5 km by 4km study area investigated in this study includes both the urban extent of the city to the south of its major river, the River Exe, and surrounding rural areas which contribute runoff to the city (Figure 5.2). Land use in the city consists of a densely populated urban area, predominantly consisting of residential terraces with some light commercial and industrial units.

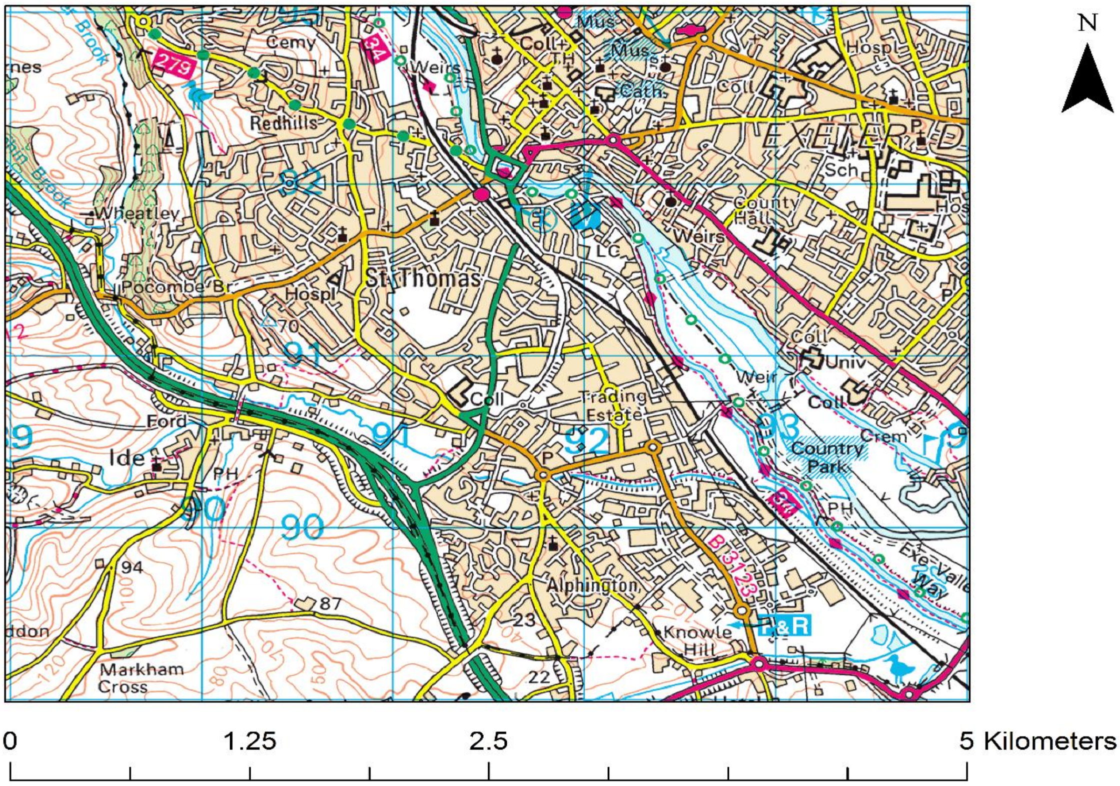

Figure 5.2: Map of the Exeter study catchment model extent

5.3. Method

This section outlines how the study area was set up using the modelling framework, following the process described in Chapter Three. The scope of analysis is restricted to data which would be available to a practitioner for catchment screening at the outset of a study. This includes elevation models, coarse land use mapping derived from open sources, national or regional rainfall profiles and land use parameter values from literature and industry standard guidance. Analysis is split into two distinct stages: Stage one, identifying flood dynamics and prioritising flood zones for further investigation; and stage two, examining the effects of strategic intervention zones across a PFS.

5.3.1. Characterising the study area

The catchment surface was specified using 1 m resolution DEM LiDAR with building thresholds and road locations added using shapefiles (Ordnance Survey, 2018). Building thresholds were specified at 0.15 m, representative of the level at which the level of flooding would typically exceed a damp proof course (Environment Agency, 2013).

Land uses were specified using Ordnance Survey Mastermap products and satellite imagery, and simplified into four categories: Urban, green space, buildings and roads. For the purposes of simplification, urban and green space areas were classified based on 250 m x 250 m grid cells, in which the predominant land use was attributed to the entire block. Road and building land use was specified at a 1 m x 1 m resolution overlaying this (Figure 5.3).

Parameter values associated with each land use type are identified in Table 5.1. Roughness values were attributed based on commonly accepted specifications found in the literature (Arcement Jr and Schneider, 1989; Woods Ballard et al., 2015; Butler et al., 2018). Buildings were attributed an artificially high Manning's 'n' value of 0.300 to account for water being held up within a structure during flooding (Syme, 2008). Infiltration for green spaces was based on standard values for the sandy loam soil found in the area (United Nations Food and Agriculture Organisation, 2017; Cranfield Soil and Agrifood Institute, 2018). It was assumed that buildings would have no capacity for infiltration.

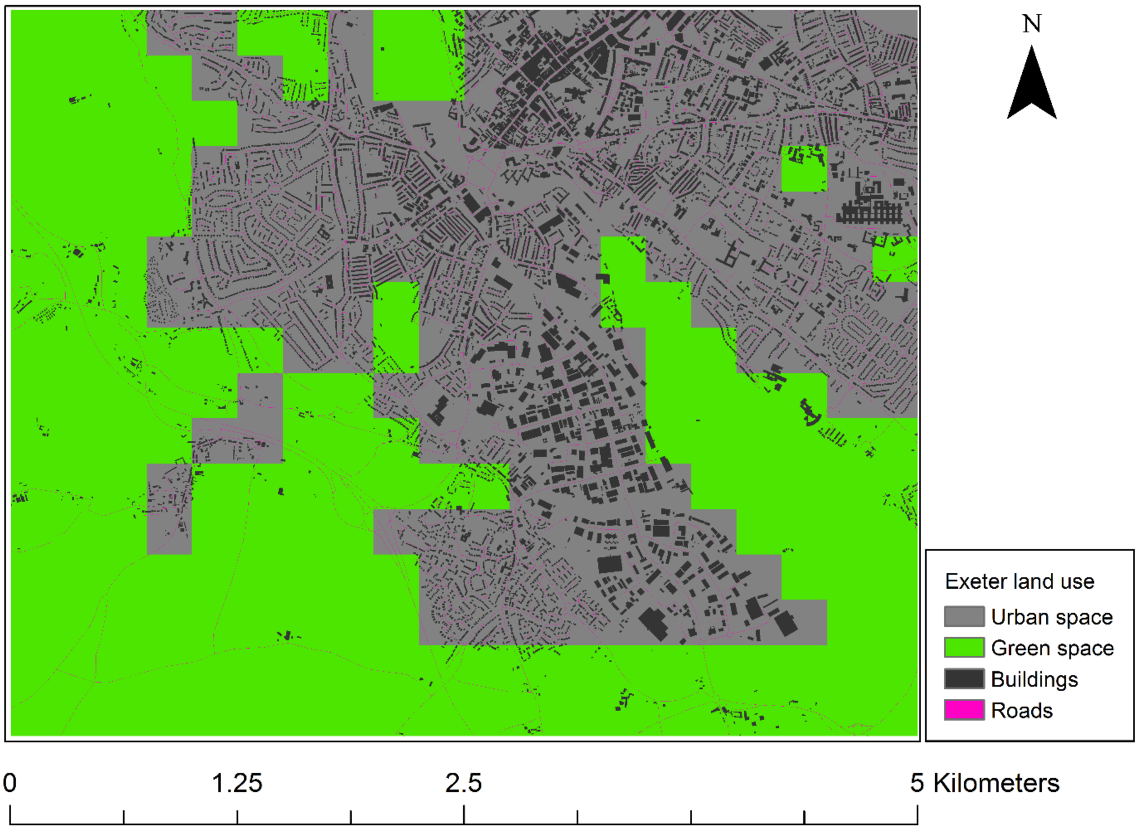

Figure 5.3: Land use classification for the Exeter case study area

Table 5.1: Parameter values for Exeter case study land use

Type	Roughness (Manning's n)	Infiltration (mm/ hour)
Urban	0.065	12
Green space	0.110	15
Building	0.300	0
Road	0.015	12

No data was available to define the underlying sewer system, therefore the areas served by the existing combined sewer system, including roads and urban land use, were represented using an infiltration value of 12 mm/ hour in line with the Environment Agency approach for surface water flood mapping (Environment Agency, 2013).

It was assumed that as an intervention screening exercise, limited data for the catchment would be available and analysis was therefore based on typical catchment profiles in England and Wales from the Environment Agency surface water flood mapping methodology (Environment Agency, 2013). To generate extensive flooding, rainfall generation was based on a high magnitude 200 year return period event using rainfall IDF relationships for the area of LiDAR coverage (Centre for Ecology and Hydrology, 2013). This equates to a constant intensity design storm of 47 mm/ hour to represent an assumed time of concentration of 1 hour (Environment Agency, 2013). The simulation was set to run for five hours (model time) so that water ponding after precipitation could be examined.

5.3.2. Representing intervention scenarios

The strategic intervention zones examined are detailed in Table 5.2. This outlines the associated roughness, infiltration and rainfall capture parameters applied to each zone. The characteristics of each zone investigated are intended to represent a broad range of possible parameters which subsequently explore a variety of effects likely to be achievable using specific interventions.

Examining this range of parameters also functions as a high level sensitivity analysis which indicates the relative importance of parameters within this catchment. Roughness parameters range from a smooth channel (0.010) to grasses (0.110) (Arcement Jr and Schneider, 1989; Woods Ballard et al., 2015; Butler et al., 2018). An infiltration rate of 20 mm/ hour and a rainfall capture rate of 20 mm/ hour were examined. ‘No change’ is specified in cases where an intervention did not affect an underlying land use parameter. Interventions were placed across urban areas in the catchment, with their effects assumed to apply to every cell within this extent Figure 5.4.

Table 5.2: Parameter values for Exeter case study land use

Type	Roughness (Manning's n)	Infiltration (mm/ hour)	Rainfall capture (mm/ hour)
Do Nothing	no change	no change	no change
Intervention A	no change	no change	20
Intervention B	no change	20	no change
Intervention C	no change	20	20
Intervention D	0.010	no change	no change
Intervention E	0.010	no change	20
Intervention F	0.010	20	no change
Intervention G	0.010	20	20
Intervention H	0.110	no change	no change
Intervention I	0.110	no change	20
Intervention J	0.110	20	no change
Intervention K	0.110	20	20

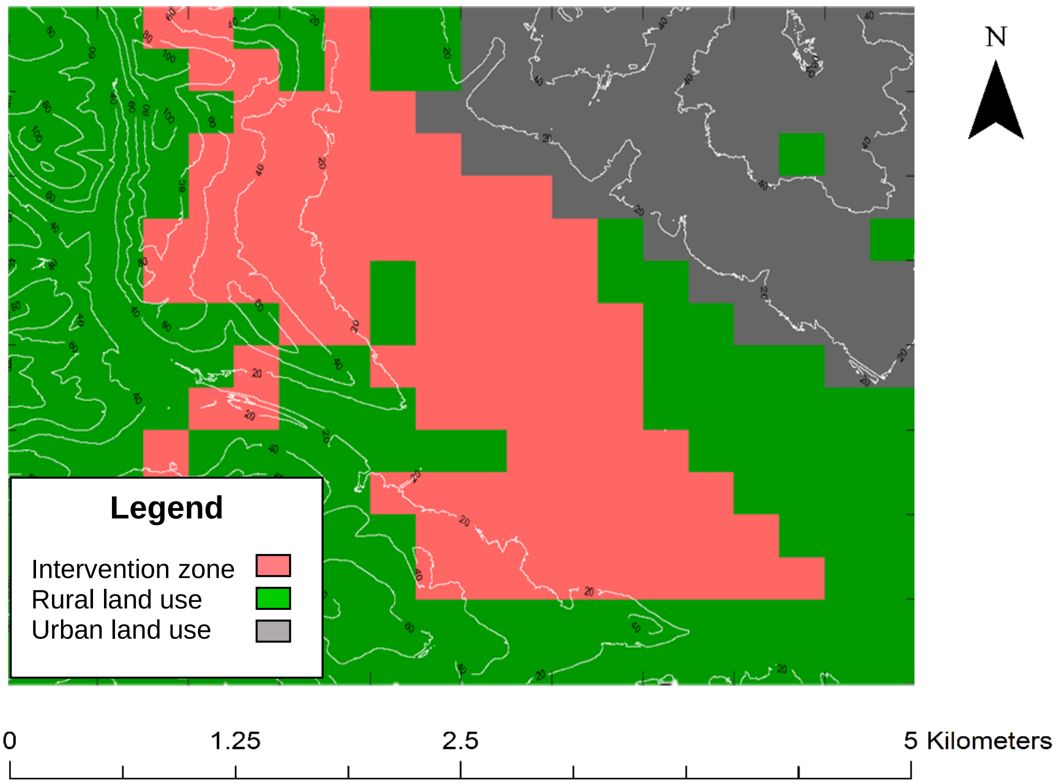

Figure 5.4: Distribution of a strategic intervention zone across the Exeter case study

5.3.3. Simulating scenarios

Simulation was undertaken using a minimum model time step of 0.01 s (Guidolin et al., 2016). Small time steps such as this have been demonstrated to deliver model accuracy of flood extents at 98 – 99% correlation with industry standard hydrodynamic flood models (Gibson et al., 2016). This was corroborated in Chapter Four, where correlation between 97 – 99% was identified across a city scale analysis.

5.3.4. Assessing intervention performance

Performance of intervention strategies was assessed through analysis of peak flood depth, flood extent, and damage costs to buildings, as outlined in Section 3.5. Comparison of intervention scenarios is made relative to a ‘do nothing’ approach.

Damage costs are calculated by applying a depth-damage function to building polygons within flood extents using GIS tools (Chen et al., 2016). Peak depths are used to ensure a worst case scenario is recorded. The depth-damage function applied for this case study was based on costs for an average three bedroom semi-detached property (Penning-Rowsell et al., 2010). Costs specified as GBP per depth per household were converted to GBP per depth per m^2 using average household sizes in England (DCLG, 2015).

5.4. Results

5.4.1. Screening flood dynamics and identifying a PFS

As presented in Chapter Four, the role of a PFS is to identify a region, or regions, of flood hazard which are prioritised for further analysis. Identification of a PFS enables subsequent stages of analysis to refine the scope and requirements for modelling by focusing on specific locations within a catchment.

Figure 5.5 presents peak flood depths across the study area in the ‘do nothing’ scenario. These depths represent the worst case scenario for all points in the catchment across the entire simulation. Flooding across the catchment is focused along two river channels, low points in topography and in the urban area highlighted within the figure.

This highlighted region identifies a PFS within the study catchment. This constitutes a region of localised flooding within the residential urban area, where depths of approximately 0.5 m are observed. Flooding in this area is observed adjacent to buildings and across roads and consequently this region is prioritised for further analysis.

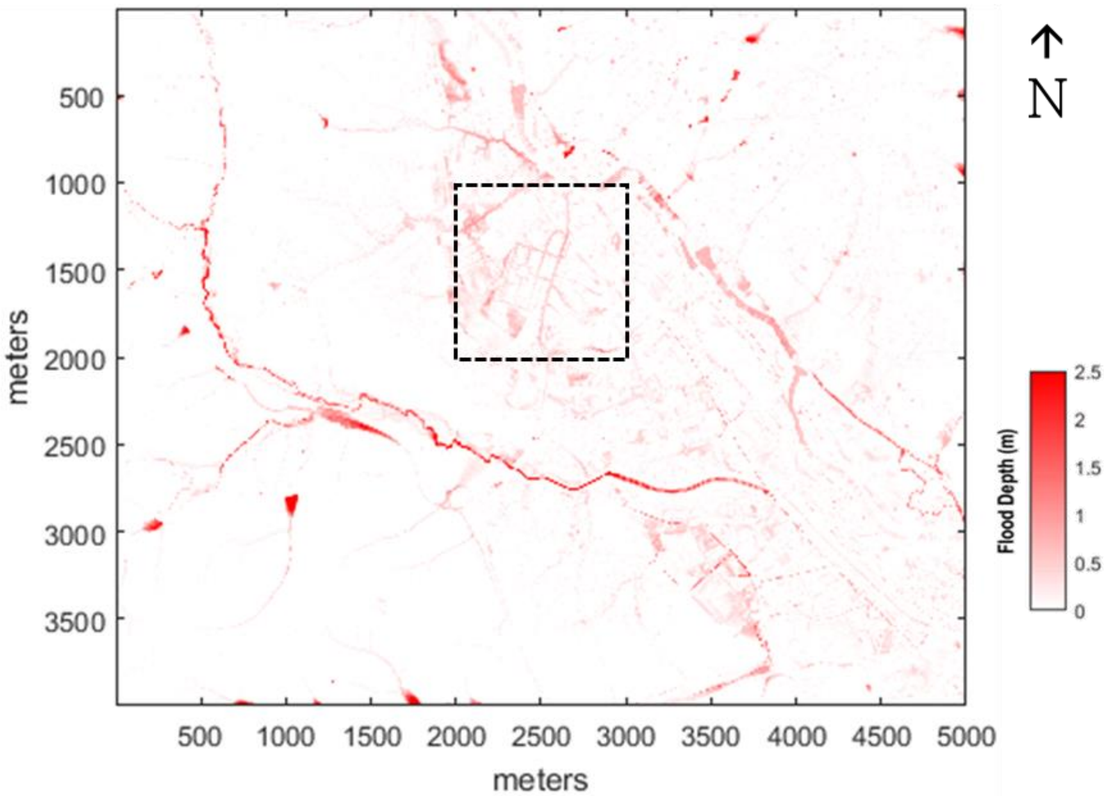

Figure 5.5: Peak flood depth for the 'do nothing' simulation during the one hour, 200 year rainfall event with a PFS highlighted

Isolated ponding is also seen distributed across the catchment. However, it is assumed that disruption associated with this flooding in rural regions will be minimal in comparison to the impact on the densely populated urban area, and so further analysis is focused on the aforementioned PFS.

Validity of the PFS and application of results

The results presented in this chapter are based on a high level screening using readily available data and a simplified representation of land use and interventions, intended to facilitate a rapid analysis towards evaluating many different intervention scenarios. The simplifications within this approach have been favourably compared to current detailed flood models in Chapter Four, however outputs from this modelling methodology should still be considered in the context of an initial and relative assessment between scenarios, intended to inform and direct future management actions.

Validity of these results has been considered at a high level through comparing flood outlines with the Environment Agency's Risk of Flooding from Surface Water flood mapping (Figure 5.6).

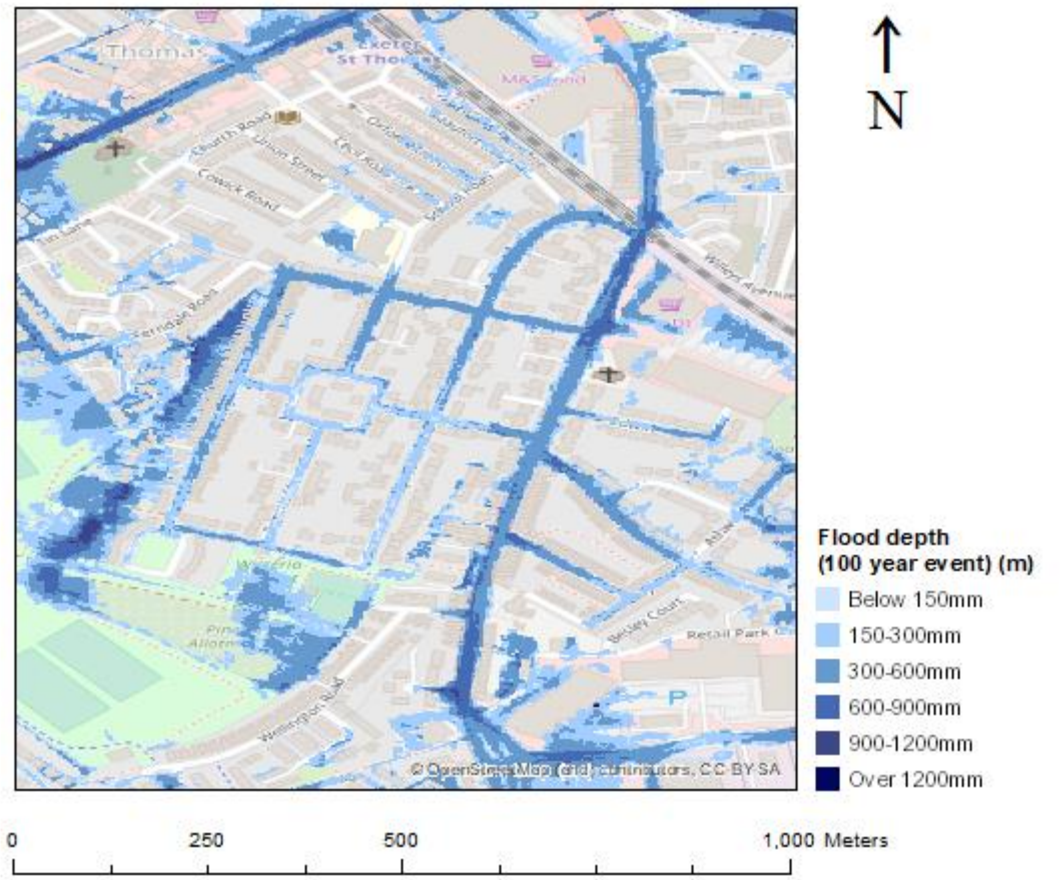

Figure 5.6: Peak flood depths for catchment during 100 year flood event (Environment Agency, 2013).

It should be noted that due to limitations regarding available return periods and unknown model inputs regarding the Environment Agency mapping, this is only intended as an indicative comparison to identify that the PFS identified in Figure 5.5 aligns with a region of surface water flooding highlighted by the Environment Agency Surface Water Flood Mapping. Full comparison between the two approaches is not possible due to unknown input data and limited return periods available using the Environment Agency mapping, however correlation regarding high level flood dynamics between sources supports application of this approach for identifying priority flood spots, aligning with the conclusions presented in Chapter Four and benchmarking of CADDIES versus other flood models undertaken in previous research (Néelz and Pender, 2013; Gibson et al., 2016; Guidolin et al., 2016).

Further details on how results should be applied are presented in Section 5.5.5.

5.4.2. Comparison of intervention effects on peak flooding in the PFS

Figure 5.7 is focused on the urban area identified in Figure 5.5. Intervention subplots highlight the difference in peak flood depth relative to the ‘do nothing’ depth. The absolute flood depth of the ‘do nothing’ scenario is shown in blue, differences relative to this for each intervention are shown in green (reduction) and red (increase).

Flood depth and extent differ between each intervention. Maximum depth and extent of flooding occur in Intervention D, Intervention H and the ‘do nothing’ scenarios. Minimum flooding is observed in Interventions C, G, I and K.

Interventions C and K lead to the largest reduction in peak flood depth across the catchment. These interventions capture 20 mm of rainfall and set the infiltration rate to 20 mm/ hour, whilst maintaining (C) or increasing (K) the surface roughness. Intervention G also leads to a large reduction in peak flood depth, however this is less pronounced than Interventions C and K, likely due to the decreased roughness (0.010) increasing the runoff speed and allowing water to pond.

Nine of the eleven interventions show a consistent increase or reduction in depth across the majority of the urban area. The two remaining interventions (F and H) show a spatial trade-off in changes to peak depth, with some areas benefitting from interventions and others showing deeper flooding. This is attributed to the uniformity across a large spatial extent of all the intervention strategies examined. More spatial variation may be observed where interventions are applied in small, discrete schemes.

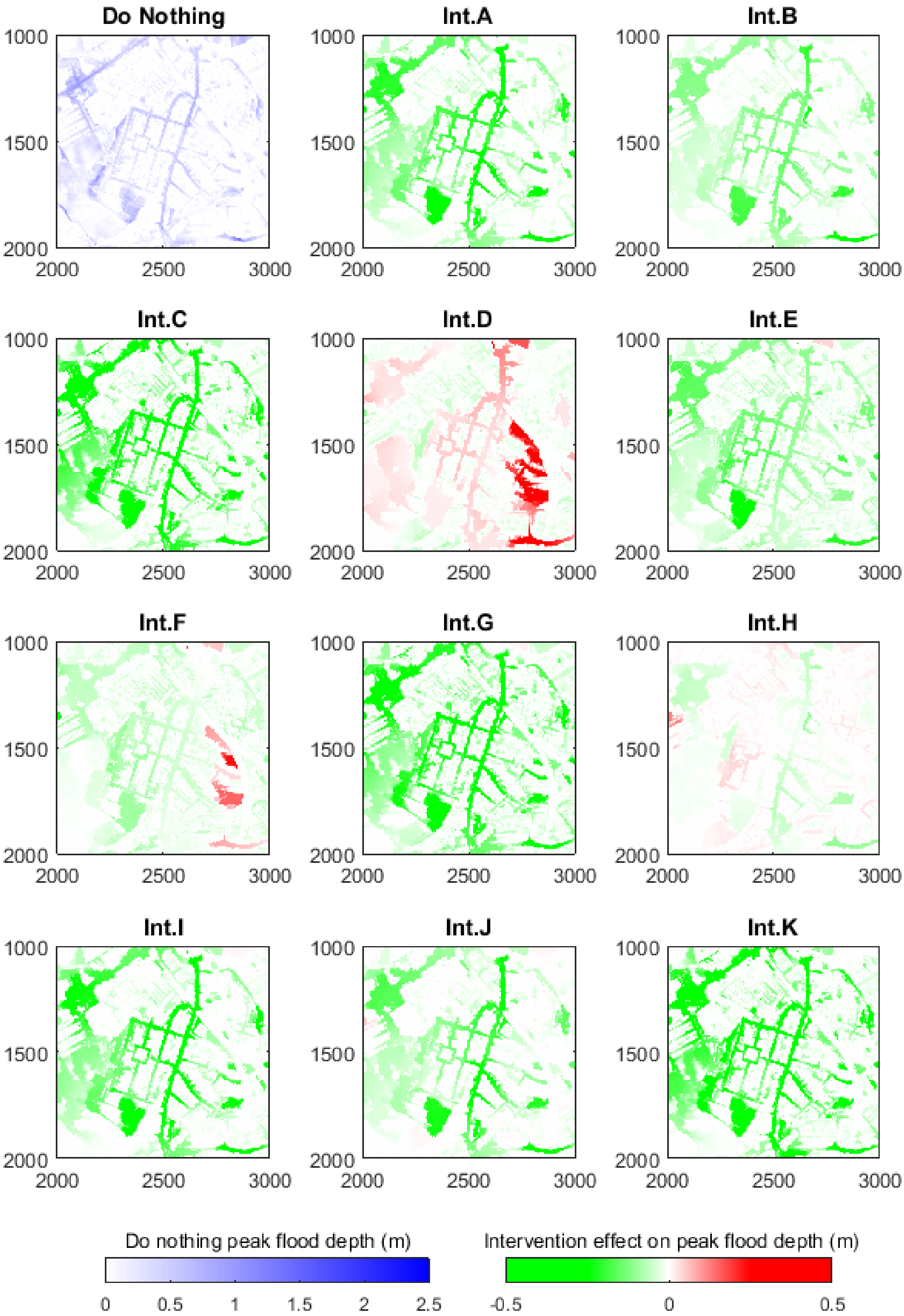

Figure 5.7: Relative maximum flood depth for intervention scenarios applied across the PFS in the Exeter case study during the one hour, 200 year rainfall event

5.4.3. Flood damage cost comparison

Spatial differences between flood depths associated with each intervention and locations of buildings mean that interventions which have the greatest impact on flood depth reduction may not cause the greatest reduction in flood damage costs. Figure 5.8 presents damage costs for each of the intervention strategies, highlighting the effects of changing catchment infiltration and rainfall capture rates. Analysis of flood damage costs indicates a clear hierarchy of intervention effect performance, highlighting that rainfall capture reduces flood damage more than infiltration.

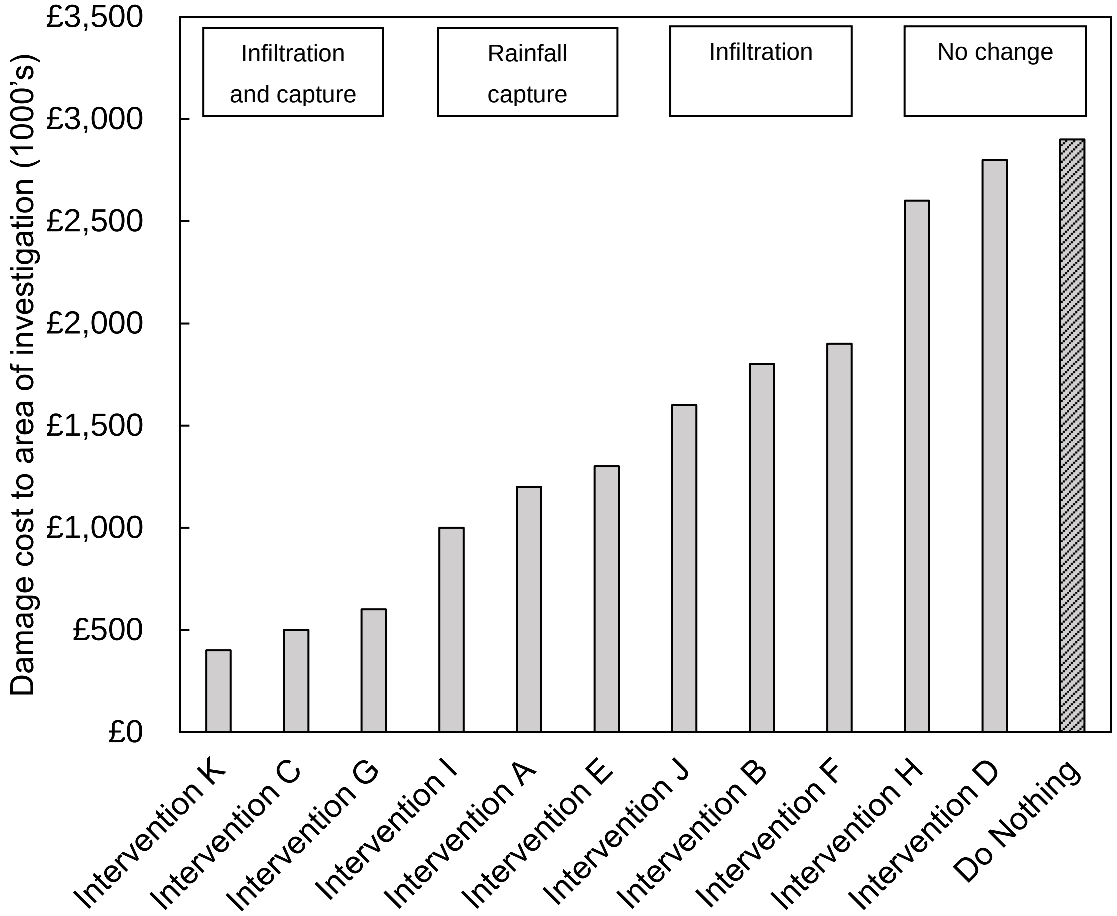

Figure 5.8: Comparing the effects of changing infiltration and rainwater capture rates across the Exeter study area during the one hour, 200 year rainfall event

Intervention strategies resulting in the least damage across the study area are those which include simultaneous infiltration and rainfall capture effects (K, C and G). These interventions generate damage costs between £0.4 and £0.6 million. The next best performing set are interventions which capture rainfall (I, A and E) which lead to between £1.0 to £1.3 million of damage, followed by interventions which increase infiltration (J, B and F) which result in £1.6 to £1.9 million of damage. The worst performing interventions are those which do not change infiltration and rainfall capture rates (H, D and Do Nothing) which lead to damage costs of £2.6 to £2.9 million.

Figure 5.8 clearly indicates that adjusting the rainfall capture parameter reduces damage further than the infiltration parameter, and that changing the roughness parameter demonstrates the least impact relative to other parameters.

Figure 5.9 further evaluates effects of changing surface roughness on the flood damage cost of each scenario. A trend is visible indicating that increasing the roughness parameter value tends to generate lower damage costs in all parameter sets. When the effect of increasing the roughness value is isolated across each set of interventions it appears to create a reduction of £0.1 to £0.3 million in damage, relative to the unchanged scenario. It is suggested that higher roughness leads to slower runoff, which enables an opportunity for more infiltration and also slows the ponding.

Conversely, lowering roughness tends to generate higher flood damage costs in this catchment. The negative effect of reducing roughness is typically around £0.1 million. This is not the case in the scenarios where infiltration and rainwater capture are unchanged, in which the 'Do Nothing' scenario generates the highest total damage costs.

Table 5.3 presents more detail regarding intervention ranking, as based on damage cost reduction relative to the Do Nothing scenario. Interventions K, C and G generate the largest reduction in damage costs of between 79 to 86%. Each of these interventions consists of rain capture and increasing infiltration rates. Interventions which capture rain and change surface roughness (I, E, A) perform better than those which increase the infiltration rate and change surface roughness (J, B, F).

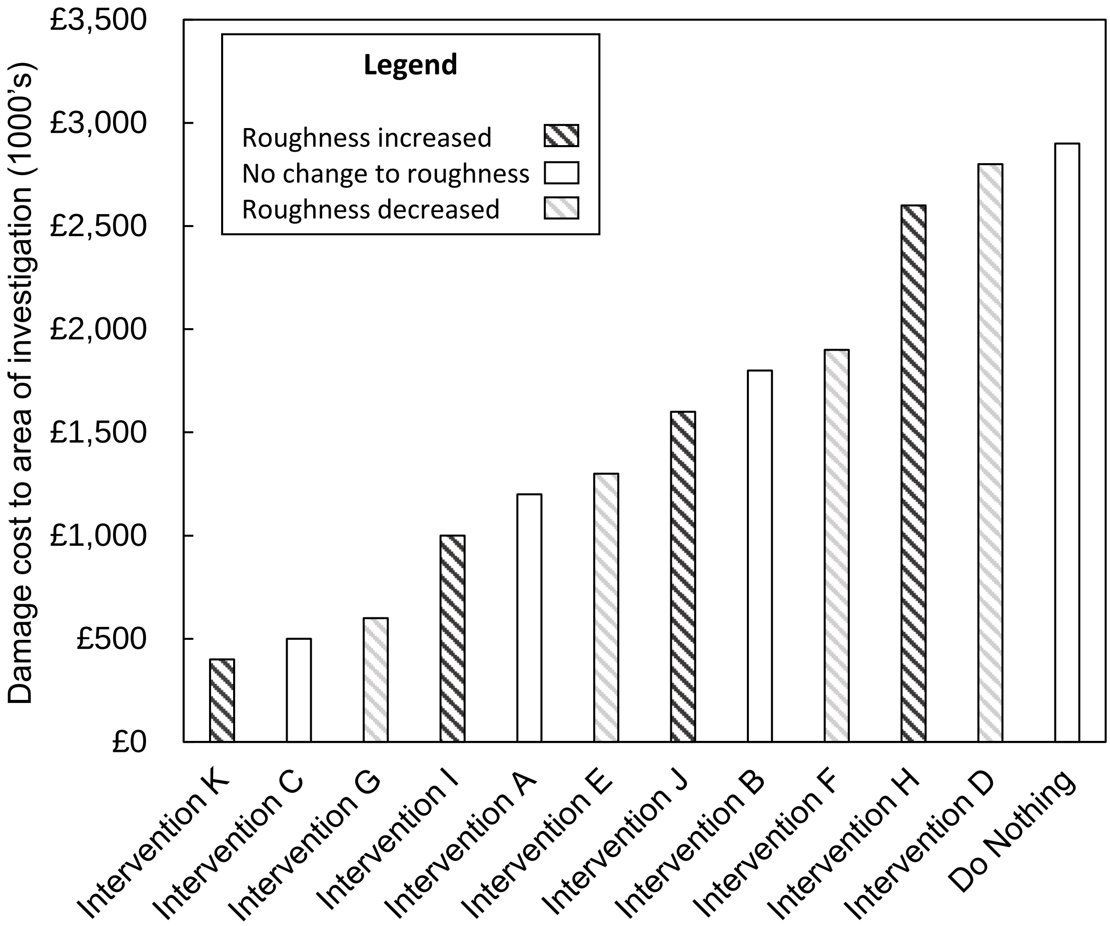

Figure 5.9: Comparing the effects of changing surface roughness across the Exeter study area during the one hour, 200 year rainfall event

All interventions where rainfall and infiltration rates are altered perform better with a higher surface roughness, evident through a 9.5% cost difference between Intervention J (roughness of 0.11) and Intervention F (roughness 0.01).

Interventions which just change roughness parameters (H) demonstrate a 10% reduction in cost versus the do nothing scenario, in comparison just increasing infiltration (B) realises a 38% saving and reducing rainfall (A) realises a 58% relative reduction.

Table 5.3: Percentage change in damage cost associated with each strategy relative to a baseline (do nothing) scenario

Rank	Intervention	Total damage (10^3) (GBP)	Damage cost reduction (%)
1	K	411	86
2	C	470	84
3	G	609	79
4	I	1,040	64
5	A	1,226	58
6	E	1,301	55
7	J	1,623	44
8	B	1,818	38
9	F	1,890	35
10	H	2,625	10
11	D	2,829	3
12	Do Nothing	2,920	-

Intervention D is the only intervention which shows a different performance when assessed by depth versus a cost based comparison. The intervention causes deeper peak flood depths than the ‘do nothing’ scenario (Figure 5.7), but damage costs are 3% less (Table 5.3). This is due to fast conveyance caused by a reduced runoff parameter preventing deep flooding against buildings in the northwest quadrant of the catchment, indicating spatial complexities regarding the relationship between runoff speed controlled by roughness and damage costs. Deeper flooding in the rest of the catchment does not coincide with building locations. This variation between evaluating the flood extent versus the flood damage indicates the importance of including the spatial value of areas which are flooded, and highlights the need to assess intervention performance ranking using context specific analysis and applicable metrics.

5.5. Discussion

5.5.1. Effects of strategic intervention zones within the catchment

Both the damage cost and flood extent analysis output the same results when ranking the best performing interventions. A small difference is observed in the worst performing interventions (‘Do Nothing’ and Intervention D), due to the location of deeper flooding not coinciding with buildings in the study area. Visualisation of flooding using the maps allows a quick overview of each intervention, however does not capture the significance of flood locations (Merz et al., 2004; Hammond et al., 2015).

Interventions which captured rainfall and increased infiltration showed the largest reduction in peak flood depth and damage costs relative to the ‘Do Nothing’ scenario. Interventions with these traits exhibited a catchment wide flood depth reduction alongside a 79% to 86% decrease in damage costs.

The case study demonstrated that, when applied across an identical area, rainfall capture reduced flood damage more than increasing infiltration (Figure 5.8). Rainfall capture strategies accounted for a 55% to 64% decrease in damage costs, whereas infiltration strategies accounted for 35% to 44%. However, in practice the available area for application, storage volume and costs for each intervention are very different. Strong performance of interventions based on rainfall capture is exaggerated by no limit being placed on storage in this case study. In reality, storage capacity for captured rainfall will limit the effectiveness of an intervention, particularly during prolonged rainfall or following wet antecedent conditions (Mentens et al., 2006; Stovin et al., 2012). Further over-estimation occurs due to the same surface area being specified for capture and infiltration. Roofs feature as the primary site for rainfall capture, however not all roofs are suitable for construction, and so assuming equal area for rainfall capture as infiltration will exaggerate the reduction effect (Viavattene and Ellis, 2013). Future studies should incorporate this finding by limiting the available storage potential for rainwater capture interventions.

Adjusting catchment roughness demonstrated the smallest reduction in flood damage when considered in isolation, relative to the effects of changing the other parameters across the study area. However, a clear trend is visible where higher roughness parameters are consistently associated with lower flood damages when applied with any other permutation of parameters (Figure 5.9). This supports strategies which slow runoff in a catchment and indicates a synergistic relationship with measures which simultaneously remove or capture runoff. This finding is supported by other green infrastructure literature which indicates that slowing runoff and returning catchments to a more natural hydrological cycle is likely to present an opportunity to improve flood management (Balmforth et al., 2006; Environment Agency, 2007b; Duffy et al., 2008; Wong and Brown, 2009; Woods Ballard et al., 2015; Bowen and Lynch, 2017). This finding is caveated with the need for a context and location specific analysis of roughness parameters. In many cases slow runoff across the catchment will extend the runoff hydrograph and consequently reduce flood depths; however, it is also conceivable that fast runoff away from buildings and areas of risk, for example through designing for exceedance, would also reduce damage costs (Balmforth et al., 2006). This supports the need for fast screening approaches which include spatial simulation of surface water flood flooding, and highlights that 'one size fits all' solutions are not sufficient to respond to complex spatial disaggregation in urban catchments.

5.5.2. Applying strategic intervention zones to identify opportunities for specific interventions

One utility of applying a screening approach to evaluate strategic intervention zones is to identify promising strategies which show potential for managing surface water flooding in a complex urban catchment. Findings regarding these promising strategies can then inform a prioritisation for investigating the application of specific interventions to achieve desired strategic effects. A critical step in this process is understanding the relationship between conceptual strategic intervention zones and the specific measures which may achieve a required performance.

In practice, specific interventions which may contribute to a rainfall capture strategy will include those which intercept incoming precipitation and store this for re-use or attenuation. These measures include interventions such as rainwater harvesting, attenuation tanks, water butts and green roofs (Stovin et al., 2007; Environment Agency, 2015; Woods Ballard et al., 2015). Rainwater capture strategies are likely to be situated on the roofs of buildings across the catchment and will be subject to several limitations on capacity, notably including the storage volume, antecedent conditions and installation costs (Mentens et al., 2006; Viavattene and Ellis, 2013). Surface based interventions with a finite capacity may also contribute to rainfall capture through storing water in ponds or surface features.

Interventions which can be implemented to increase the infiltration rate per cell are typically those which operate on the catchment surface to remove volume from surface flows. This mechanism includes both infiltration and surface drainage based interventions. Infiltration measures include options such as increasing green space within a catchment, installing porous surfaces such as permeable paving and installing green infrastructure such as tree pits, raingardens and swales. Surface drainage includes measures such as increasing the capacity of surface or combined sewers or installing green drainage infrastructure such as filter drains and soakaways (Woods Ballard et al., 2015; Butler et al., 2018).

Interventions corresponding to the increased roughness parameter include those which alter the catchment surface. This can be achieved as the primary aim of an intervention, for example slowing runoff using nature based solutions and green infrastructure (Burns et al., 2015d; Schanze, 2017), or as a secondary effect of installing another measure, for example a change in surface roughness attributed to installing permeable paving, swales, filter drains and other surface based strategies (Woods Ballard et al., 2015). In this regard, many interventions will effect multiple parameters. Parameterisation of specific interventions will be outlined in detail in Chapter Six of this thesis.

5.5.3. Utility of the framework for initial catchment screening.

The study has demonstrated two main utilities as an initial screening tool for flood risk management: screening flood risk and scoping required intervention effects.

Screening catchment flood dynamics has identified the predominant regions of flood hazards (PFS) across the catchment and forms the basis for prioritising the location of flood management strategies. The advantage of the framework is that assessment is evaluated through modelling flood depths and extents which are not typically associated with standard industry screening approaches, which typically rely on flood histories or previously conducted studies (DEFRA, 2010). Flood histories should be used with caution due to the potential for small sample sizes, inconsistencies or inaccuracies in data collection (for example classification of a surface water sewer blockage leading to flooding as a capacity rather than maintenance issue), bias towards regions with active and vocal reporting and basing decisions on records limited by the time period or technology of reporting (Kjeldsen et al., 2014). A reliance on the outputs of previously conducted flood models should also be treated with caution due to the time requirement for due diligence of methodologies in the context of crucial but subtle nuances of different modelling decisions, which can be missed by new parties adopting old projects (Dottori et al., 2013). A simple but significant example of this could be using results from a model which does not include subsequent changes to catchment land use, which would then divert flow paths in a catchment. Planning future strategies using a method reliant on historic events can also negate the importance of looking forward at future hazards, which are likely to be exacerbated by climate change and urban growth (Wheater and Evans, 2009; Howard et al., 2010). The clear identification of flood patterns which can be modelled using accessible data is evidenced through framework outputs such as Figure 5.5, which clearly identifies areas exposed to hazards during intense rainfall events. Application of data such as DEM's, rainfall descriptors and coarse land use mapping enables a fast and simple screening which can be undertaken at a minimal resource cost to inform next steps.

The framework has also evaluated potential effects of a range of strategies whilst estimating a relative flood damage cost for each scenario. As discussed in Section 5.5.2, this enables a method to steer preliminary design by identifying the scope, scale and effects of the specific measures required to manage flooding. Outputs such as relative flood depth (Figure 5.7) and avoided flood damage (Table 5.3) are easily accessible as decision support tools and provide a clear evidence structure for directing decision makers. The UK Government stipulates all investment decisions to be supported by transparent and accessible evidence bases such as this (Pitt, 2008; House of Commons, 2016).

The utility of this framework establishes preliminary understanding of catchment flood dynamics which can be applied as part of a suite of flood management tools to support and inform further analysis (Sayers et al., 2002). The logical next step towards application of the framework is developing understanding towards modelling specific interventions, of which the connections are discussed in Section 5.5.2. Investigation of specific interventions can be supported through advancing the simplified representations of interventions used in this example through application of finer resolution categorisation, both in terms of the number of intervention types and the scale of implementation. This is of particular importance when assessing the impacts of numerous small and dispersed interventions, such nature based solutions or dispersed green infrastructure (Schanze, 2017).

Development of the framework towards specific interventions will introduce a novel methodology with applications as an enhanced catchment screening tool which can utilise the computational efficiency of this approach for investigating intervention effects across multiple scales and events. This will respond to knowledge gaps regarding the application of a fast and quantitative screening methodology to evaluate suitable interventions for a given context, and the ability of interventions to manage a range of rainfall events, from design standard rainfall through to examining intervention resilience to infrequent, high magnitude events (Pitt, 2008; MWH, 2014; Burns et al., 2015c; Woods Ballard et al., 2015; Schubert et al., 2017). This is of particular significance given the need to build resilience to future extreme events (Ofwat, 2015; Butler et al., 2017) in combination with a limited understanding of how novel interventions, particularly green infrastructure, will perform during these events (Wheater and Evans, 2009).

5.5.4. Applying suitable metrics for assessing intervention performance

Peak flood depth per cell

Maximum flood depth was a useful metric for identifying the peak impact caused by surface water flooding and provided adequate data for a damage cost assessment. Limiting simulation outputs to one maximum depth file saves computational space where many model runs are required and provides decision makers with simple visualisation of each interventions effects. However, it should be noted that a maximum flood depth map does not represent the total volume or extent of flooding at any particular moment. This metric is therefore unsuitable for uses where accurate representations are required at a specific time step, such as in the case of emergency evacuation planning. If this use is required then time step and recording requirements can be adjusted in the simulation.

Damage costs

This study screened flood damage cost associated with a single event as a preliminary catchment analysis tool to examine flood dynamics and the relative performance of implementing strategic intervention zones to manage extreme rainfall. For subsequent stages of evaluating specific interventions, this approach could be expanded to assess damage across multiple events to calculate an expected annual damage cost, as described in Chapter Three.

The focus on direct flood damage also neglects the costs of implementing each strategy. In practice, decision makers will be constrained by budgets and application of different intervention strategies are likely to constitute a range of capital, operational and maintenance costs (Bowker, 2007; Environment Agency, 2007a). Future research regarding specific interventions should take these costs into account and compare these against the expected flood damage savings to evaluate the cost effectiveness of each strategy.

PFS

Chapter Four recommends application of rapid scenario screening to evaluate surface water flooding across a catchment and identify PFS. These are regions of hazard within the catchment which can be prioritised for further analysis and investigation.

Once a PFS has been highlighted upstream catchments can be identified by tracing contributing areas using GIS. Identifying a PFS and its contributing area facilitates a focused approach where resources can be targeted at prioritised regions. This benefits strategic design through focusing stakeholder attention towards developing and implementing strategies to manage hazards within these areas. Managing flooding across upstream areas enables the approach to consider the flood offsetting effect of multiple interventions mitigating downstream impact through a cohesive and decentralised management strategy. The effects of decentralised management to offset downstream risk are further explored within Chapter Seven, where flood hazards in Melbourne are managed through managing runoff from the upper catchment.

Terminology regarding PFS and how this relates to strategic intervention zones is shown in Figure 5.10. This indicates that the priority flood zone is used to define a surface water catchment in which hazards are identified. These hazards can be managed through a variety of interventions, which can be modelled through a strategic intervention zone, where intervention effects are applied on average using parameter changes across a wider region, or through defining specific interventions, as discussed in Chapter Six. The difference between strategic intervention zones and specific interventions is discussed in Section 5.1.

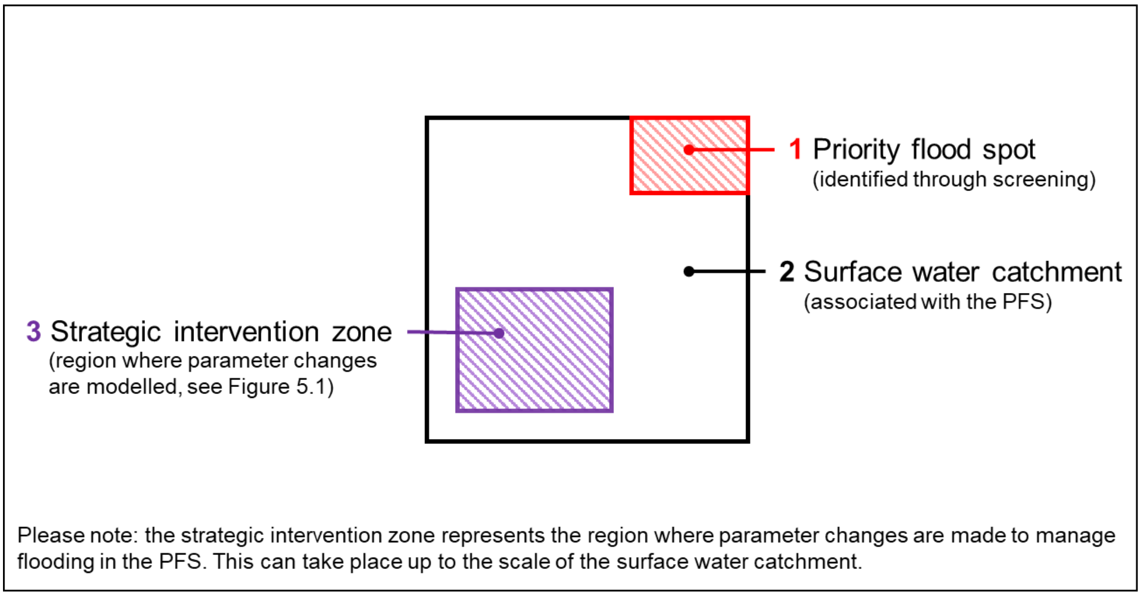

Figure 5.10: Conceptual relationship between PFS, a surface water catchment and a strategic intervention zone

The scale and location of a strategic intervention zone varies depending of the context, type and scale of parameter changes evaluated. The zone can extend up to the scale of the surface water catchment and include regions within a PFS.

5.5.5. Applying the results from catchment screening

As discussed whilst evaluating the utilities of the approach (Sections 5.5.3), the primary application of the framework is developing preliminary understanding of catchment flood dynamics. This forms one component of a suite of flood management tools to support and inform further analysis (Sayers et al., 2002). It is important to understand how results from the framework should be applied and which decisions they should inform.

Results from this style of initial catchment screening should primarily be applied to support understanding and direct future management actions using other complementary management tools. Useful decisions from screening include:

- Re-focusing analysis on areas of high flood hazard, or discounting areas of low hazards.
- Developing evidence to support investigating the performance of specific interventions (Chapter Six).
- Identifying data gaps which restrict modelling in specific areas.
- Highlighting key stakeholders to engage in subsequent steps of a flood management project.

- Informing the types of scenarios and strategies which show promise for further analysis using detailed modelling tools. For example rainfall events, interventions or catchment regions which would benefit from further analysis.

The assumptions associated with rapid scenario screening are reflective of the simplifications required to develop novel utility of evaluating many surface water flood scenarios during the initial stages of a management project, and are in line with similar approaches recommending simplified models to facilitate this style of analysis (RAND, 2013; Babovic et al., 2018b). The strength of the methodology lies in an ability to explore a wide range of scenarios using readily available data. Scenario exploration supports further actions through developing a robust evidence base which enhances understanding to direct a chain of subsequent analysis which incrementally guides subsequent flood management actions. Outputs and decisions from the approach are most useful when compared both relatively and the context of the limitations discussed in Chapters Three and Four.

5.6. Chapter conclusions

This chapter has demonstrated application of the framework to screen catchment flood hazards and compare the effects of twelve high level intervention strategies during a high magnitude flood event in an urban catchment. Analysis has focused on assessment of strategic intervention zones, represented through changing land use parameters across large areas of the catchment. It is envisaged that this style of analysis can generate understanding of the scope and scale of effects required to manage flood risk in an urban catchment. This knowledge can then form the basis for decision support regarding the direction of further investigations evaluating the specific interventions required to achieve these effects.

Key findings from this chapter are:

- The framework is applicable to identify flood dynamics and screen surface water hazards across an urban area. Analysis was achieved at a low resource cost whilst utilising data sources likely to be available at the inception of a flood management project.
- Ranking interventions based on flood extent and damage costs outputs similar results, however spatial differences between intervention effects versus building locations results in a slight variation in rankings between

the two metrics. Future application should include context discussing how metrics accommodate spatial variation of effects in surface water flood management.

- When intervention effects were evaluated independently, rainfall capture based strategic intervention zones demonstrated the lowest estimated damage costs (£1.0 M) relative to interventions which only infiltrated runoff (£1.6 M) or slowed runoff (£2.6 M). However, it is noted that this is based on an assumption of equal areas and capacities available for all intervention types, which in practice may not be the case.
- Intervention zones with multiple effects were the most effective strategies. Combined interventions generated the lowest damage cost estimates of £0.4 M. This supports future development of synergistic intervention strategies capable of applying a range of mechanisms to manage surface water management.

Conclusions are made specific to the context of the case study and are subject to several limitations associated with simplification of physical processes made as a result of a model architecture aimed at speed. Speed and computational efficiency are necessary to respond to gaps in traditional modelling regarding the ability to assess and evidence many strategies (Chapter Two). The limitations support the intention of this framework to guide and evidence optioneering, rather than conducting detailed design. Full discussion of these limitations is available in Chapters Three and Four.

The chapter has identified several recommendations for future research which will be developed in subsequent chapters of this thesis. These recommendations include: developing the assessment presented by refining parameters to represent specific interventions applied across urban catchments at a high resolution; investigating reliability and resilience of interventions through evaluating performance across multiple rainfall events; and, expanding the economic analysis to include the annual expected damages and cost effectiveness of each strategy.

6. EVALUATING COST EFFECTIVENESS OF SPECIFIC INTERVENTIONS

This chapter responds to Objectives Four, 'investigate the flood reduction performance of strategic and specific interventions', Five, 'evaluate intervention cost effectiveness over many rainfall scenarios' and Seven, 'investigate the relationship between resilience and reliability of interventions'. This is achieved through advancing the methodology introduced in Chapter Three to develop representation of specific interventions, and then assessing performance of these measures across a range of scenarios.

Despite established inclusion of novel surface water management strategies within academic, government and commercial discussion, several gaps are apparent in application (Pitt, 2008; MWH, 2014; Burns et al., 2015c; Woods Ballard et al., 2015; Schubert et al., 2017). This chapter principally responds to two of these gaps, namely, generating evidence regarding novel interventions through the application of a fast and quantitative screening framework to select suitable interventions for a given context, and evaluating the ability of interventions to manage a range of rainfall events, including resilience to extreme rainfall (Butler et al., 2017; Löwe et al., 2017).

This chapter is divided into three sections. Section 6.1 outlines representing specific interventions within the framework in order to address barriers for implementation of novel measures, including developing evidence for institutional decision making frameworks, uncertainty regarding effectiveness of novel interventions in a heavily regulated and risk averse water industry and a lack of evidence regarding the hydrological performance of novel interventions (Cettner, 2012; O'Donnell et al., 2017; Ossa-Moreno et al., 2017).

Section 6.2 assesses performance of specific interventions across a range of rainfall durations and return periods. This responds to a gap in literature regarding performance variation of flood management strategies across design standard and extreme rainfall events through analysis of 144 scenarios which represent a range of rainfall intensities, frequencies and durations (Pitt, 2008; Wheater and Evans, 2009).

Section 6.3 advances analysis towards investigating the resilience of interventions up to a 1000 year return period event and evaluates the effect of intervention placement on performance in an urban catchment. This section also responds to recommendations from Chapter Five by developing a cost effectiveness metric which enhances decision support through screening intervention economics over a thirty year planning horizon.

The work presented in this chapter draws from the papers ‘Rapid surface water intervention performance comparison for urban planning’ (Section 6.2), which is published in Water Science and Technology (Webber et al., 2018d), ‘Comparing cost effectiveness of surface water flood management interventions in a UK catchment’ (Section 6.3), which is published in the Journal of Flood Risk Management (Webber et al., 2019) and ‘How can we build reliable and resilient surface water management’ (Section 6.3), which is published in the proceedings of the Resilience of the Water Sector conference held in Munich, 2018 (Webber et al., 2018c).

6.1. Interventions

Green infrastructure is frequently cited as a desirable method with which to manage surface water and build resilience in urban environments (Balmforth et al., 2006; Environment Agency, 2007b; Duffy et al., 2008; Wong and Brown, 2009; Woods Ballard et al., 2015; Bowen and Lynch, 2017). Terminology describing such approaches varies, including a range of synonyms such as Water Sensitive Urban Design (WSUD), Low Impact Development (LID), Sustainable Drainage Systems (SUDS) and Best Management Practices (BMP), among many others. Current literature recognises significant cross-over regarding the definitions and terminology of measures (Fletcher et al., 2015). Therefore a broad categorisation is applied to group similar interventions in this chapter. The term ‘green infrastructure’ is applied as a generic term for drainage interventions which manage surface water by mimicking natural hydrologic processes, such as infiltration and detention (Fletcher et al., 2015).

Literature highlights the need to increase the evidence available for novel interventions through establishing new assessment frameworks which can evaluate strategy performance (Pitt, 2008). This section outlines specific interventions (as defined in Section 5.1, Figure 5.1), their effects and how they are translated into the rapid scenario screening framework. Specific interventions are simulated through high resolution representation of detailed measures and included within the model structure through spatial and temporal manipulation of cell roughness, infiltration and rainfall parameters. Parameters are determined through evaluating current literature and best practice.

A broad range of conventional and green infrastructure interventions are presented, including green roofs, rainwater capture tanks, permeable paving, infiltration techniques, sub-surface drainage measures and surface storage features. These descriptions form the basis of analysis conducted in Sections 6.2 and 6.3.

6.1.1. Green roofs

Green roofs are vegetated surfaces constructed on the roofs of buildings. There are many variations of green roof types, in line with the large variation in roof structure. In general roofs can be classified into two categories based on substrate depth: extensive and intensive roofs (Berndtsson, 2010; Woods Ballard et al., 2015). Extensive green roofs tend to have a shallow substrate depth and are suitable for planting vegetation such as grasses. Intensive green roofs are constructed using a deeper substrate and can support a wider range of planting. It should be noted that this classification is not exact; Berndtsson (2010) conducted a review of green roof literature and identified a significant overlap in substrate depths between studies (Table 6.1).

Table 6.1: Examples of soil thickness of intensive and extensive green roofs (adapted from Berndtson, 2010)

Intensive (mm)	**Extensive (mm)**	**Reference**
>500	–	Köhler et al (2002)
>300	–	Bengtsson et al (2005)
150 – 350	30 – 140	Mentens et al (2006a)
150 – 1200	50 – 150	Kosareo and Ries (2007)
>100	<100	Wong et al (2007)
>150	20 – 150	Woods Ballard et al (2015)

This thesis will adopt UK industry best practice in the form of the CIRIA (Woods Ballard et al., 2015) definition of green roofs based on a substrate depth threshold of 150 mm.

Woods Ballard et al (2015) indicate that green roofs can be installed on a variety of roofs, however installation costs will increase where building structure requires reinforcement. This is of particular note regarding intensive roofs, which tend to require significant upgrades to support increased structural load (particularly relevant when soils become saturated), access and maintenance. Structural re-design required for installation of intensive roofs adds uncertainty to the installation suitability of green roofs, particularly when considering urban retro-fit on standard properties. The high level strategic analysis undertaken through this screening method is therefore better suited for investigating the effects of extensive green roofs, which are more likely to be suitable for urban retrofit.

A number of studies have measured green roof performance across a variety of environmental and construction factors. These identify variation in the rainfall capture potential of green roofs due to substrate depth and type, planted vegetation, roof geometry (VanWoert et al., 2005; Villarreal and Bengtsson, 2005; Getter et al., 2007; Stovin et al., 2012), age and antecedent conditions such as proceeding dry periods, temperature, seasonal variation and rain event characteristics (Bengtsson et al., 2005; Villarreal and Bengtsson, 2005).

Studies indicated a range of values for the interception potential of green roofs. This thesis incorporates the variation in predicted green roof performance through adopting assumptions within the average range proposed by previous research (Table 6.2). This is equal to around 15 mm of interception for a green roof. This value will be adapted for each case study, through editing the input rainfall profiles for cells on which green roofs are situated. The process for editing rainfall values is described in Section 3.3.1.

Table 6.2: Summary of green roof performance (adapted from Woods Ballard et al., 2015)

Study	Interception provided (mm)	Substrate depth (mm)	Location
Martin (2008)	10	100	Ontario, Canada
Paudel (2009)	16.5	100	Detroit, USA
United States General Services Administration (2011)	12.5 – 19	75 - 100	USA
Stovin et al (2012)	12 – 15	80	Sheffield, UK
Fassman-Beck et al (2013)	20	100 - 150	Auckland, New Zealand

It should be noted that the variation in performance and construction suitability indicates the need to conduct detailed scoping studies in areas where preliminary screening indicates green roofs are a preferred option to manage flooding

Much of the literature indicates that green roofs are most effective for managing smaller storms (Carter and Rasmussen, 2007; Simmons et al., 2008). Limited studies have been undertaken to examine the green roof resilience to extreme events, application of green roofs within this research framework is anticipated to assist bridging this gap.

6.1.2. Rainwater capture tanks

Rainwater capture tank interventions consist of a variety of measures designed to intercept, store and release rainwater (Woods Ballard et al., 2015). Interception is typically achieved through collection from roof surfaces, although runoff can also be collected from the catchment surface. Storage is achieved using tanks across a range of scales, from small water butts through to large domestic and industrial tanks. Rainwater release can be achieved through re-use of captured grey water for uses such as gardens and toilets, infiltrated into the soil, or attenuated back into the surface water sewer system. Rainwater tanks are classified based on the combination of these three parameters (Amos et al., 2016; Melville-Shreeve et al., 2016; Campisano et al., 2017). This section will describe three common rainwater capture tanks: rainwater harvesting, attenuation tanks and water butts. Typically, rainwater capture and infiltration is considered a soakaway, which is discussed in Section 6.1.5.

Rainwater harvesting involves the capture of rainwater for re-use (Burns et al., 2015d; Amos et al., 2016; Melville-Shreeve et al., 2016; Campisano et al., 2017). Water is typically re-used at the site of capture to reduce water demand. Typically re-use is for non-potable water demand such as toilet flushing, although the addition of a treatment train means that this is not always the case.

Design of rainwater harvesting systems requires careful balance of seasonal rainfall averages, rainfall intensity, storage duration and site demand (Melville-Shreeve et al., 2014). Systems can be adapted to incorporate a range of collection and storage options, some of which are gravity fed and others which require pumping (Melville-Shreeve et al., 2016). The rate of capture can also be limited by the collection mechanisms, for example the size of the downpipe can restrict and throttle flows from the collection surface to the storage tank. As this project is primarily concerned with the potential of interventions to reduce the surface water runoff, the nuances of system design will not be examined at this point and so it is assumed that interventions operate with 100% capture efficiency until storage fills, with no throttling effects.

Attenuation tanks share very similar characteristics with rainwater re-use tanks in terms of collection and storage, however captured water is gradually attenuated back to the sewer system rather than re-using it on site. Gradual release of captured water is designed to increase the available space within surface water and combined sewer systems during an event without the cost of excavating and installing new subterranean infrastructure.

Water butts provide a cheap but low capacity rainwater capture device. However ease of installation means they are accessible for implementation across large areas, which in turn can lead to cumulative flood reduction effects.

The quantity of rainwater disconnected will be relative to the storage capacity of the system. In this study it has been assumed the only controlling factor on storage is available volume. Sufficient available storage volume within tank can be controlled using active and passive systems. Active controls combine real time forecasting with ‘smart’ tank operation to release stored water before large rainfall events, thus ensuring full tank capacity is available (Xu et al., 2018). Tank design can also include passive controls to ensure available storage capacity through compartmentalised tanks which are designed to only hold a certain volume for re-use (Figure 6.1; Gee and Hunt, 2016; Melville-Shreeve et al., 2016)

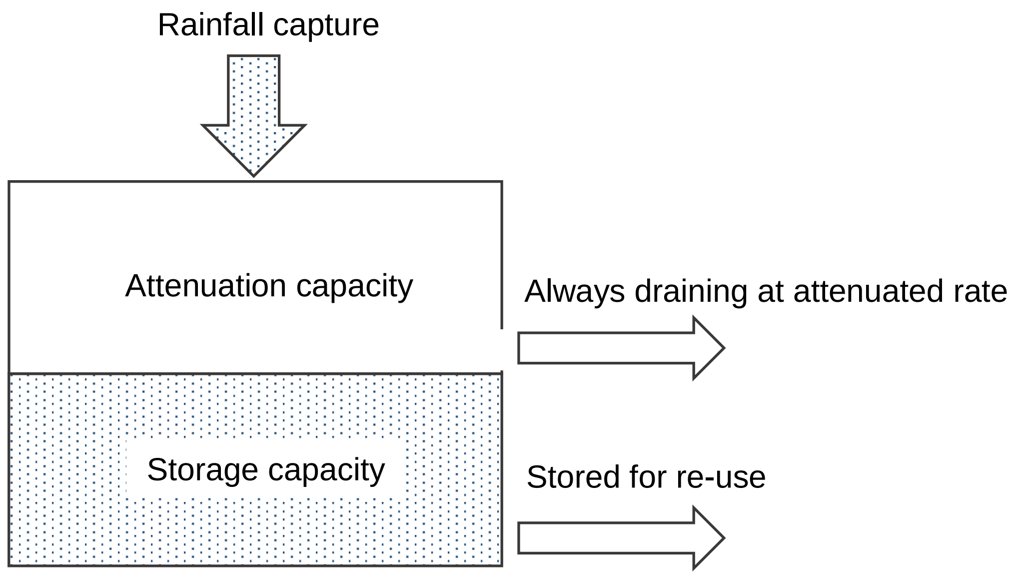

Figure 6.1: Diagram representing a compartmentalised rainwater capture tank

This thesis responds to the uncertainties regarding available storage volume and the presence of active controls through utilising assessment of many scenarios to examine a range of tank capacities for each catchment. This constitutes a sensitivity analysis, which can be used to evidence decision support using results indicative of many potential configurations. Typical sizes of domestic rainwater capture tanks range from 2500 l to 10 000 l (Rainwater Harvesting Ltd, 2018; Tanks Direct, 2018). Water butts are significantly smaller, with a typical capacity around 250 l of capacity when empty. It is assumed that water butts are unlikely to have active control mechanisms due to their relatively small capacity, therefore the study assumes a conservative available capacity of 100 l per water butt.

Rainwater capture is represented in the model using a similar approach as green roofs, where a new rainfall profile is applied to accommodate water captured by the intervention (Section 3.3.1). In the case of green roofs, the interventions rainfall capacity is controlled by the area of installation. This is not the case for rainwater capture measures, for which the capacity is specified by the tank size. Rainwater capture is instead incorporated within the modelling framework through assuming that all areas of a specified collection surface contribute to the tank equally. Therefore, the storage capacity of the tank is modelled through averaging the volume across the entire collection area through adjusting all cells rainfall inputs. For example, a 5,000 l rainwater tank draining 100 m^2 of impervious roof would be represented by capturing the first 50 mm of rainfall which fell on each cell.

This study assumes rainwater capture collection is undertaken on building roofs. Therefore roughness and infiltration will remain the same as underlying land use parameters.

6.1.3. Permeable paving

These interventions consist of paving structures which are able to permeate runoff through the catchment surface for storage or transmission (Zachary Bean et al., 2007; Collins et al., 2008a; Yong et al., 2011; Woods Ballard et al., 2015; Mohammadinia et al., 2018). Pavements are constructed using porous surface materials, which enable infiltration across the entire surface, or using impermeable materials, with infiltration only occurring at the voids between blocks. Figure 6.2 shows three types of pervious paving system: Left, a porous asphalt construction allowing infiltration across the entire surface; Middle, a series of impermeable blocks with porous jointing material; and Right, a reinforced grass and gravel structure which facilitates natural infiltration.

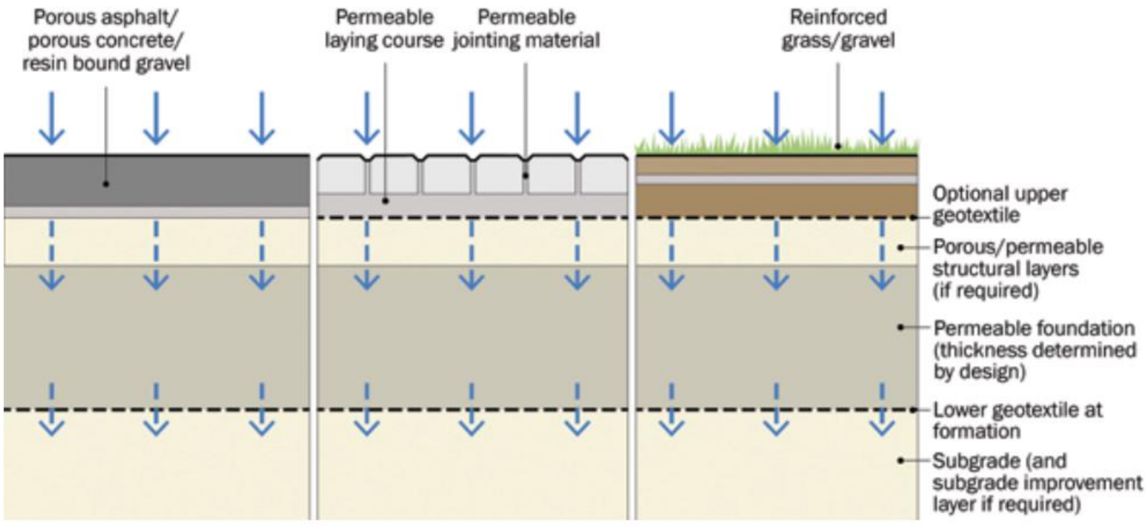

Figure 6.2: Types of pervious pavement system (Woods Ballard et al., 2015)

Once infiltrated, runoff can be stored using tanks and geo-cellular systems, infiltrated into the soil structure or collected in transmission trenches and pipes.

A large range of permeable paving systems is available, including: modular permeable paving, porous asphalt, grass reinforcement, resin bound gravel, porous concrete, macro pervious and block pervious paving (Woods Ballard et al., 2015). Paving can be installed to replace many impervious surfaces, although it is most commonly found in areas with lighter loading such as car-parks and pedestrian walkways (Scholz and Grabowiecki, 2007).

Key design requirements for effective and safe permeable pavements are presented in Table 6.3. Broad consideration of these factors is appropriate for a high level screening process, however detailed design requires a site by site investigation, with particular attention required to examine sub-surface geology, the potential for soil contamination and an appropriate maintenance regime.

Table 6.3: Practical considerations for installation of permeable paving

Consideration	**Description**
Groundwater contamination	All measures which directly infiltrate to the subsoil should be assessed for potential contamination, particularly if draining road surfaces which may build up heavy metals and motor oils. Permeable paving diverting to storage or treatment can be managed through installation of impermeable membranes (Wilson et al., 2003).
Seasonal temperature extremes	It has been documented that permeable pavements can withstand freeze-thaw conditions better than traditional pavements due to the insulating effect of air trapped within the base and the latent heat of soil moisture (Kevern et al., 2010) However, infiltration is unsuitable for managing runoff in areas with seasonally frozen ground.
Maintenance	Regular maintenance of permeable paving is required to prevent pores clogging through due to sediment build-up and shear stress (Scholz and Grabowiecki, 2007). This is of particular importance in environments where the intervention is situated in close proximity to fine particles. Full maintenance descriptions are beyond the scope of this screening project, but are available on Page 492 of the SUDS manual (Woods Ballard et al., 2015). It should be noted that other studies indicate that permeable paving operates effectively over long periods with minimal maintenance. One study indicates that over six years a permeable paved car park exhibited only minimal changes to paving structure and infiltration rates (Booth and Leavitt, 1999).

The volume reduction capability of permeable paving is controlled by the transmission speed through the medium and the availability of storage within it. However, in the case of surface water flooding caused by short duration, intense rainfall the most likely limiting factor will be the pore saturation in the upper soil or storage medium slowing transmission speeds, with the potential to limit infiltration even if storage is adequate. A range of studies have taken place which aim to quantify the volume reduction in various locations, these are presented in Table 6.4.

Table 6.4: Summary of studies measuring infiltration rates through pervious paving (adapted from Woods Ballard et al., 2015)

Study	**Infiltration rate (mm/ hour)**	**Details**
Pratt et al (2002)	2.6 – 17.2 Average 7.3	Edinburgh, UK using concrete block pervious pavement. Testing using rainfall intensity to trigger runoff.
Rankin and Ball (2004)	2.5 – 16 Average 5	Sydney, Australia using concrete block pervious pavement. Testing using rainfall intensity to trigger runoff.
Bean et al (2007)	86 median after maintenance	40 sites in North Carolina, Maryland, Virginia and Delaware, US. Testing undertake using double ring infiltrometer.
Collins et al (2008b)	> 5	Kingston, USA using concrete block pervious pavement. Testing using rainfall intensity to trigger runoff.
Collins et al (2008b)	Average 6	Kingston, USA using concrete grass grid. Testing using rainfall intensity to trigger runoff.
Drake et al (2012)	Average 7	Toronto, Canada using concrete block pervious pavement and porous concrete. Testing using rainfall intensity to trigger runoff.

A wide range of site dependant factors controls the performance of permeable paving, therefore the infiltration rate is modelled within the framework using a value of 5 mm/ hour, as indicated from the lower bounds of averages in Table 6.4. Where further information is available, it is recommended that this value is adjusted on a site by site basis using field data. This is particularly pertinent in the case of reinforced grass gravel paving structures lacking artificial storage, which are more likely to be controlled by the permeability of the underlying soil substrate. Indicative catchment specific infiltration rates for soil types are available using a variety of soil mapping products (United Nations Food and Agriculture Organisation, 2017; Cranfield Soil and Agrifood Institute, 2018).

Installing permeable paving will affect surface roughness. Concrete block based permeable paving is represented using a Manning's n coefficient of 0.015 (Arcement Jr and Schneider, 1989; XP Solutions, 2017; Butler et al., 2018). Reinforced grass gravel roughness is represented using a value of 0.030, corresponding to short grasses (Hamill, 2001; XP Solutions, 2017).

6.1.4. Infiltration techniques

Infiltration techniques consist of interventions which infiltrate runoff into the soil. A variety of techniques are commonly used, including soakaways, trenches and filter strips (Woods Ballard et al., 2015). The common feature of all infiltration approaches is the utilisation of natural soil permeability to remove runoff from the catchment surface, as such the soil structure must be permeable and unsaturated to allow percolation of water at an effective rate.

The underlying geology is considered the controlling factor in the capabilities of infiltration techniques. Full understanding of soil permeability requires detailed site specific investigations which include the influences of micro features such as rocks, preferential flow routes, soil packing and macro-pores (Ward and Robinson, 1990; Beven and Germann, 2013). This level of detail is not possible for a high level screening assessment over a broad area and so a simpler classification of infiltration capacity is achieved using broad soil type categories available from geological mapping products. This assumes relatively homogeneous conditions across the catchment based on typical rates for each soil type. Figure 6.3 presents a classification from the DEFRA and Cranfield Soil and AgriFood Institute (CSAI) Land Information System, 'LANDIS' (Cranfield Soil and Agrifood Institute, 2018).

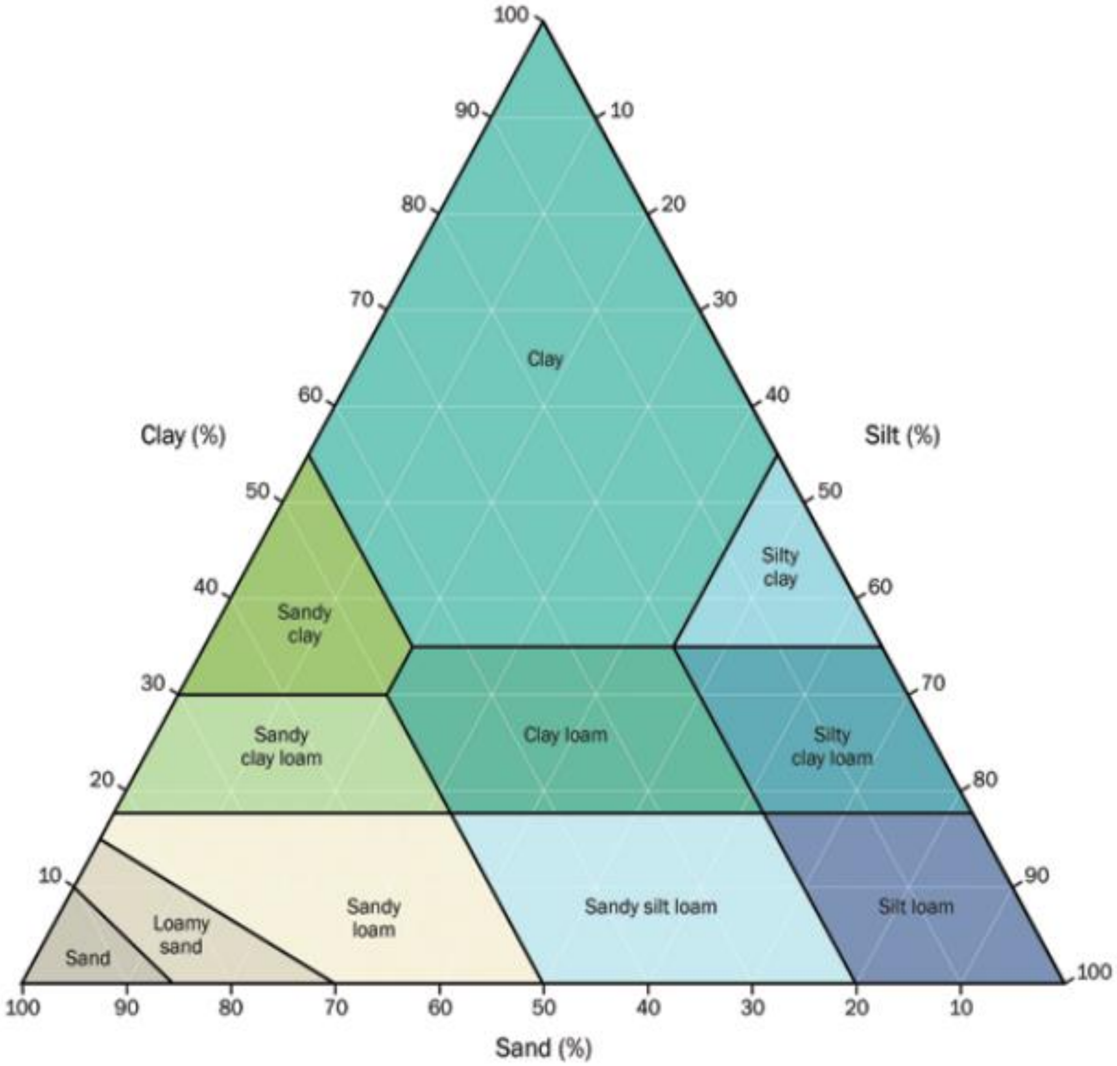

Figure 6.3: Soil texture classification (Cranfield Soil and Agrifood Institute, 2018)

LANDIS also provides catchment level mapping of soil types across the UK, which can be used in conjunction with typical permeability values to define high level infiltration rates (Table 6.5; Table 6.6).

Table 6.5: Typical infiltration rates based on soil texture (Bettess, 1996; Woods Ballard et al., 2015)

Soil type/ texture	**ISO 14688-1 description (Blake, 2010)**	**Infiltration rate (m/s)**
Gravel	Sandy GRAVEL	$3x10^{-4} – 3x10^{-2}$
Sand	Slightly silty slightly clayey SAND	$1x10^{-5} – 5x10^{-5}$
Loamy sand	Silty slightly clayey SAND	$1x10^{-4} – 3x10^{-5}$
Sandy Loam	Silty clayey SAND	$1x10^{-7} – 1x10^{-5}$
Loam	Very silty clayey SAND	$1x10^{-7} – 5x10^{-6}$
Silt loam	Very sandy clayey SILT	$1x10^{-7} – 1x10^{-5}$
Chalk (structure-less)	N/A	$3x10^{-8} – 3x10^{-6}$
Sandy clay loam	Very clayey silty SAND	$3x10^{-10} – 3x10^{-7}$
Silty clay loam	N/A	$1x10^{-8} – 1x10^{-6}$
Clay	N/A	$< 3x10^{-8}$

Table 6.6: Basic infiltration rates for soil types (United Nations Food and Agriculture Organisation, 2017)

Soil type	**Infiltration rate (mm/ hour)**
Sand	Less than 30
Sandy loam	20 – 30
Loam	10 – 20
Clay Loam	5 – 10
Clay	1 - 5

Infiltration measures are modelled based on the soil infiltration rates specified using Table 6.5 and Table 6.6. Roughness is attributed based on surface type using the typical roughness coefficients from literature (Arcement Jr and Schneider, 1989; XP Solutions, 2017; Butler et al., 2018).

Certain interventions, such as filter strips and trenches, may have additional rainfall capture capacity through void space on the top level (Melbourne Water, 2005). This additional capture capacity is included through editing the input rainfall parameter, representing an initial capture allowance for the intervention.

6.1.5. Green infrastructure rainfall detention techniques

Green detention techniques refers to a variety of methods used to temporarily capture and store surface water runoff in topographical features. Some features are used purely to store water whilst others are used to convey runoff along a channel at a predetermined rate matched to the downstream conditions. Rainfall detention measures include large scale features such as urban parks, detention basins, ponds and wetlands, as well as smaller scale features such as rain gardens and tree pits (Scholz, 2015; Woods Ballard et al., 2015).

In the case of large scale detention techniques, a storage area is created within a landscape to capture runoff. Discharge is then moderated through use of infiltration features, valves, orifices or weirs. A common detention technique involves creating a landscaped depression in a green area, referred to as a detention basin (Figure 6.4; Woods Ballard et al., 2015). Basins are typically dry except immediately following rainfall when they can offer storage to moderate runoff rates and provide a route for some runoff to infiltrate into soils. Many basins also have a secondary use as a local green space amenity.

Large scale detention features are included within the modelling approach through editing catchment elevation models to represent a depression designed to capture water. Cell output rates are specified to include the infiltration through the substrate (as discussed in Section 6.1.4) as well as the any additional attenuation rates achieved through urban drainage mechanisms. As with other interventions, roughness coefficients are specified based on surface type.

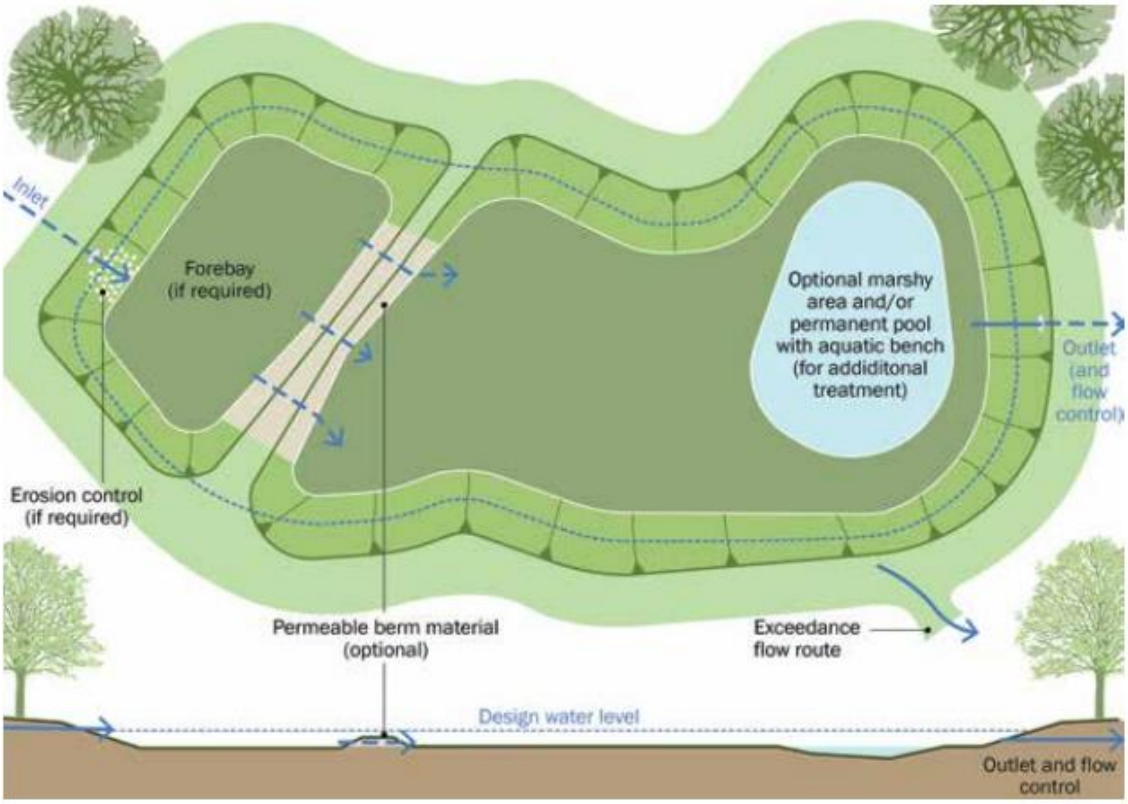

Figure 6.4: Schematic for a detention basin layout (Woods Ballard et al., 2015)

Larger detention features may also provide a basic level of treatment. Treatment can be facilitated through settling of sediments (see ‘Forebay’ in Figure 6.4) or through biological processes in wetlands and ponds. Water treatment is beyond the scope of the surface water flood research undertaken in this thesis and therefore not included within the framework. It is also assumed that wetlands are too large to be considered as a possible treatment option for urban retrofit case studies, although micro treatment facilities have been established in several urban settings.

Smaller scale rainwater detention techniques such as tree pits and rain gardens are included within the framework without editing elevation models. Instead, capture capacity is included through editing the input rain hyetograph to represent intervention capacity for storage and attenuation. Rainfall capture can be achieved through surface ponding and infiltration into a porous filter media. Surface ponding capacity is included through calculating available space on the surface. Filter media capacity is calculated through assessing the volume taking into account effective porosity of the substrate. Roughness and infiltration rates are included using the same approach as large scale detention features.

6.1.6. Upgrading catchment sewer systems

Historic management of surface water flooding has focused on construction of combined and surface water sewer systems, designed to accommodate runoff to treatment or emergency discharge (Butler et al., 2018). Upgrading sewers system capacity can comprise of several options, discussed below:

- ***Increasing pipe diameter*** involves replacing components of existing networks with larger capacity pipes, enabling systems to convey greater quantities of water. Construction requires extensive excavation of the pipe network, which brings challenges regarding disruption to the local area and designing upgrades around existing subterranean utility networks. Downstream pipes are also likely to require upgrading to manage the increased inflow.
- ***Construction of new sewers*** involves similar construction actions to increasing pipe diameters, with the addition of extra investigative and design procedures.
- ***CSO (Combined Sewer Overflow) construction*** increases the capacity of a sewer network by creating overflow compartment discharge sewerage out of the network during periods of excessive flow, typically into watercourses. CSO's release excessive water, thus preventing sewer flooding in urban areas, but at the expense of potentially significant environmental and health consequences to aquatic environments.
- ***Sewer separation*** splits the foul and surface water element of sewerage into different networks, therefore increasing the quality of surface water through removal of the foul component. It should be noted that surface water will still contain heavy metals, oils and other pollutants from surfaces, particularly as part of the 'first flush' (Sansalone and Buchberger, 1997; Lee et al., 2007). This option is unlikely to create significant additional capacity in a combined sewer system as the volume of foul flow is typically negligible relative to the large volume of surface water during extreme rainfall events. The division of different quality waste streams does however facilitate a range of discharge options.
- ***Increase capacity through monitoring, maintenance and rehabilitation.*** A further option to increase capacity, and importantly to prevent blockages, is a regular monitoring, maintenance and rehabilitation programme. Sewer collapse and blockage has the potential to lead to

surcharging networks, therefore minimising this hazard is potentially an important factor in reducing flooding (Ana and Bauwens, 2010).

CADDIES does not currently support a 1D/ 2D network, therefore it is not possible to simulate flow within a pipe network as part of the framework. Instead, surface water removal using sewer systems is included in the model through adjusting the output rate for sewer sub-catchments (Section 3.2.3). Depending on available data and designs, parametrisation can be undertaken at the sub-catchment level or through strategic analysis of increased drainage rates across the catchment. The process for modelling and parameterising sewer systems is discussed in detail in Chapter Three and validated in Chapter Four. This simplified method is found suitable for initial option screening and, as with analysis of strategic zones in Chapter Five, should be deemed indicative of the level of performance required to achieve beneficial outcomes. This understanding can then be extended into future management actions, which may involve further analysis using detailed 1D-2D models.

It should be noted that achieving drainage via the subsurface, as modelled through the cell output rate, can be achieved using a variety of measures. Therefore this intervention is referred to as ‘upgrading drainage’ rather than specifying the exact modifications to the sewer network. As this intervention is a subsurface feature it is deemed to have no effect on the roughness, rainfall or elevation model.

6.1.7. Intervention summary

Table 6.7 presents a summary of which parameters should be adjusted for representing specific interventions. Further detail and context specific adjustments are presented later in the thesis, in respect to each case study.

Table 6.7: Summary of parameter changes used to represent specific interventions

Intervention	**Elevation**	**Input**	**Output**	**Roughness**
Green roofs	✗	✓	✗	✗
Rainwater capture tanks	✗	✓	✗	✗
Permeable paving	✗	✗	✓	✓
Infiltration techniques	✗	✗	✓	✓
Green infrastructure rainfall detention techniques	✓	✗	✓	✓
Upgrading catchment sewer systems	✗	✗	✓	✗

6.2. Investigating duration effects on interventions

This section of the chapter evaluates the effects of rainfall duration on specific intervention performance in an urban catchment. A case study of a UK urban catchment is used to illustrate the advantages of the framework (Figure 6.5). Analysis is split into two stages, firstly assessing critical rainfall duration and secondly examining intervention performance across a range of rainfall intensities and durations.

6.2.1. Method

Characterising the study area

The study area examined is an urban catchment in Exeter, UK. In order to demonstrate a sequential analysis, important for establishing utility of this framework for decision support, the study area examined corresponds to the area prioritised in Chapter Five. Figure 6.5 shows the study area, highlighting the surface water catchment and building locations. The surface water catchment was identified using 1 m resolution elevation model and the ArcMap 10.3 spatial analyst function, which tracks the flow direction from each cell to define individual watersheds.

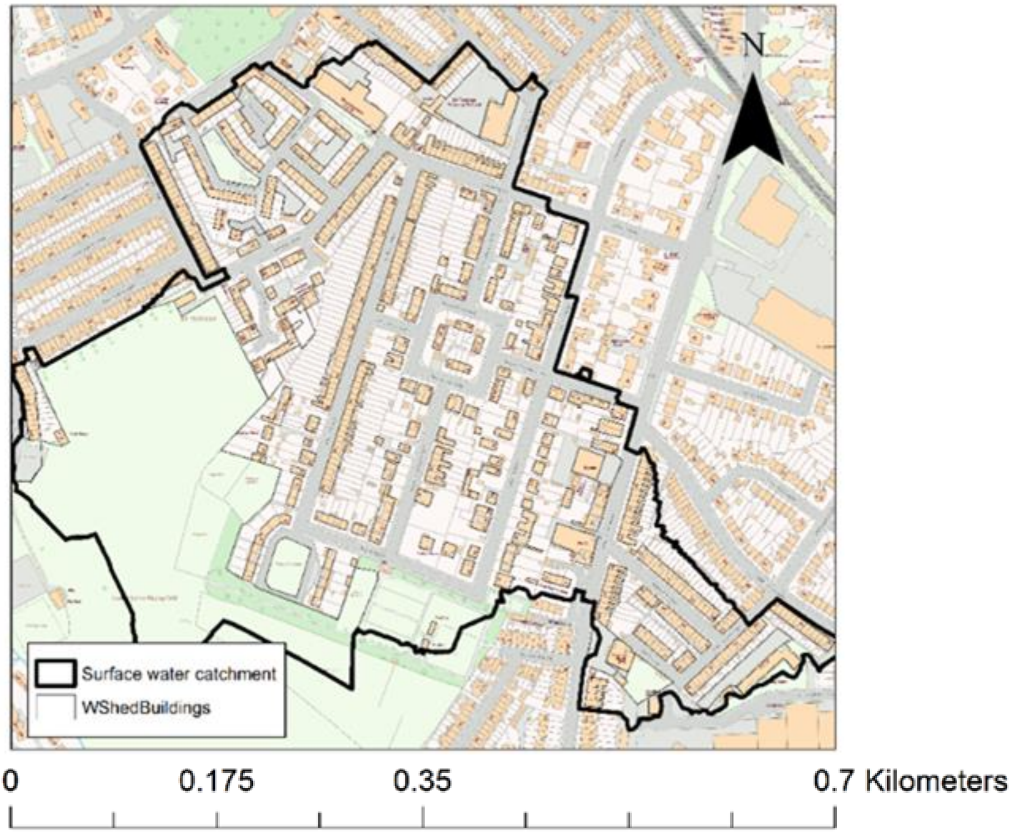

Figure 6.5: Exeter study area with surface water catchment highlighted

Characterisation of the study area was undertaken using high resolution 1 m LiDAR data to represent surface elevation. Building locations were identified using a shapefile and included in the simulation through a 0.15 m surface elevation uplift to represent a threshold level.

Land use was specified using online mapping. The effects of interventions and land use types were included through manipulation of the parameters in each cell which specified water input, output and runoff speed. An infiltration rate and roughness value was assigned to each cell based on the online mapping. Infiltration rates were specified in mm/ hour based on catchment soil types (United Nations Food and Agriculture Organisation, 2017; Cranfield Soil and Agrifood Institute, 2018). Roughness values were specified using commonly accepted Mannings 'n' coefficients found in literature (Arcement Jr and Schneider, 1989; XP Solutions, 2017; Butler et al., 2018). These values are provided in Table 5.1.

Data regarding the subterranean surface water network was unavailable, therefore the underlying drainage system was represented using a constant infiltration rate of 12 mm/hour, as specified in the Environment Agency methodology for high level surface water mapping (Environment Agency, 2013).

Catchment rainfall was simulated using 1, 2, 3, 4, 6, 12, 24 and 48 hour design rainfall events (Figure 6.6). Design rainfall events represent a constant rainfall intensity during 30, 100 and 200 year return periods (Centre for Ecology and Hydrology, 2013).

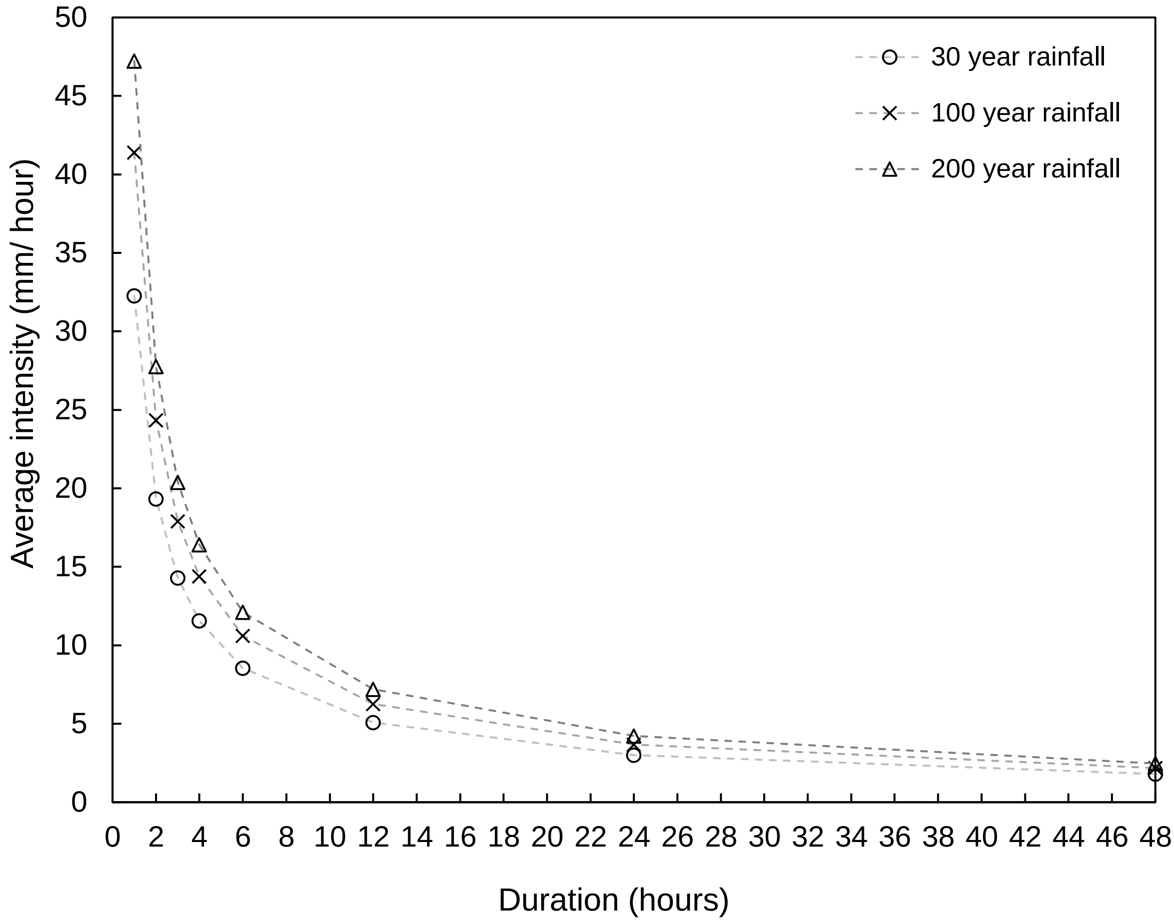

Figure 6.6: Intensity-Duration-Frequency relationship for study catchment (data from Centre for Ecology and Hydrology, 2013)

Application of rapid screening enables of a range of rainfall events to be included within intervention analysis and facilitates assessment across a range of possible future scenarios. This is of contemporary importance in the context of future uncertainties regarding climate change and is particularly relevant when assessing intervention response to changing rainfall intensities, which are predicted to increase in the future (Jones et al., 2012; IPCC, 2014). Therefore, results in this chapter are presented across a range of rainfall intensities representing many possible futures. Inclusion of many possible futures translates assessment towards decision making under deep uncertainty approaches where assessment of many simulations is undertaken instead of examining a highly optimised scenario tied to a single predicted future (Babovic et al., 2018b). This is desirable due to uncertain projections of future rainfall characteristics and a requirement for water infrastructure to retain function across extended timescales (Ana and Bauwens, 2010; Howard et al., 2010).

Representing interventions

Interventions were represented using the methodologies outlined in Section 6.1. Interventions included water butts, green roofs, rainwater capture, permeable paving, drainage upgrades and a 'Do Nothing' baseline scenario.

Conservative intervention capacity values have been applied where the literature presents a range of capacities. This may limit the observed flood reduction of strategies. Water butt and rainwater capture tank capacity was based on commercially available designs (Rainwater Harvesting Ltd, 2018; Tanks Direct, 2018) and academic literature (Section 6.2.2; Woods Ballard et al., 2015). Water butts were specified at 100 l of available storage volume and rainwater tanks at 1500 l, representing passive controls enabling half a 3000 l tank. Green roof capacity of 15 mm was based on recent published studies (Section 6.2.1; Paudel, 2009; Stovin et al., 2012). Permeable paving infiltration rates and surface roughness was based on a concrete block design rate of 5 mm and included an additional 12 mm for areas still contributing to the conventional drainage system (Section 6.2.3; Pratt et al., 2002; Bean et al., 2007; Collins et al., 2008). Drainage upgrade rates were included through doubling the Environment Agency (2013) standard rate applied for broad scale surface water modelling.

Table 6.8: Intervention strategy effects per cell for the Exeter case study

Intervention	**Rainfall capture (mm)**	**Infiltration rate (mm/hour)**	**Cell roughness (Manning n)**
Do nothing	Land use	Land use	Land use
Water butt	2.2	No effect	No effect
Green roof	15	No effect	No effect
Rainwater capture	33	No effect	No effect
Permeable paving	No effect	17	0.015
Surface drainage	No effect	24	No effect

Interventions were applied at a cell scale (1 m^2) across all suitable surfaces in the catchment. Water butts, green roofs and rainwater capture were installed on building roofs in the study area. Permeable paving and drainage upgrades were applied to all impermeable surfaces (roads, pavements and urban areas).

Simulating scenarios

Simulation was carried out across the 1 million cells which represented the 1 km by 1 km area using 0.01s time-steps. All six strategies were simulated across all eight rainfall events (1, 2, 3, 4, 6, 12, 24 and 48 hours) and all return periods (30, 100 and 200 year). Each simulation ran for the duration of rainfall, plus an additional five hours of time beyond the event to enable ponding. In total 144 scenarios were simulated.

Intervention performance assessment

Damage cost was calculated by applying a flood damage curve to peak flood depths within each building, as described in Section 3.5.4. Damage costs were industry standard figures for a three bedroom semi-detached property, typical to the study catchment, converted into an estimated cost per m^2 using average household sizes in England (Penning-Rowsell et al., 2010; DCLG, 2015). Total damage costs per scenario were calculated by adding the costs of all corresponding buildings within the watershed identified in Figure 6.5.

6.2.2. Results and discussion

Identifying the critical event duration

Figure 6.7 shows the total damage costs of design rainfall across 144 simulations, including all event return periods, durations and intervention strategies. The highest damage costs tended to occur during low probability, high magnitude rainfall, with the highest cost at each duration associated with the 200 year event. Some crossover is visible, where certain strategies lead to higher damage costs at lower probability events. A larger variation between damage costs was evident during shorter, higher intensity rainfall. This merits further analysis and is discussed later in the chapter in relation to the effectiveness of intervention strategies.

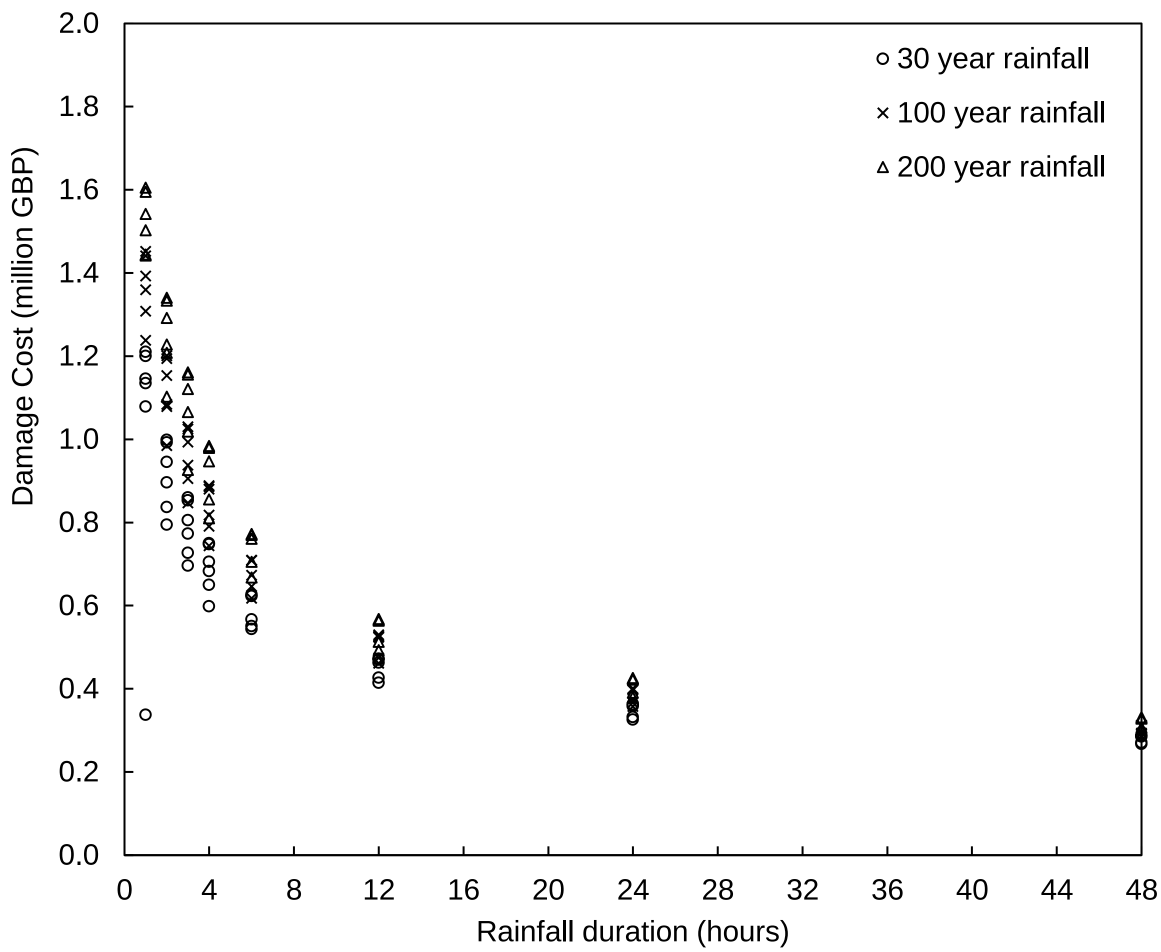

Figure 6.7: Catchment critical rainfall duration identified using a damage assessment for all interventions during 30, 100 and 200 year rainfall events (144 simulations)

The highest damage costs occurred during the one hour event. A clear trend is visible where higher damage correlates with shorter duration and more intense rainfall events. Intervention comparison for this catchment should therefore be focused on short rainfall durations. This correlates with UK government guidance indicating that short duration design events should be assessed when examining urban catchments without knowledge of critical rainfall duration (Environment Agency, 2013).

Identification of catchment flood dynamics using this approach can steer prioritisation of computationally expensive hydraulic modelling through highlighting design rainfall which is likely to lead to the peak flooding in the catchment. The advantage of this prior investigation is to streamline the modelling process whilst minimising assumptions regarding catchment flood response by evidencing selection of rainfall.

Visualising peak flooding during the critical event

Figure 6.8 shows a comparative flood depth assessment for each intervention versus the 'Do Nothing' scenario during the one hour 200 year return period rainfall event. Absolute flood depth is shown for the do nothing scenario (blue). Intervention effects on flood depth are shown on a separate scale showing improvement in a cell (green) or deeper flooding (red). This shows the largest reduction in flood extents are caused by rainwater capture tanks and upgrading sewer capacities. Reduction in flood depth across the catchment was not uniform, with interventions creating localised regions of improvement.

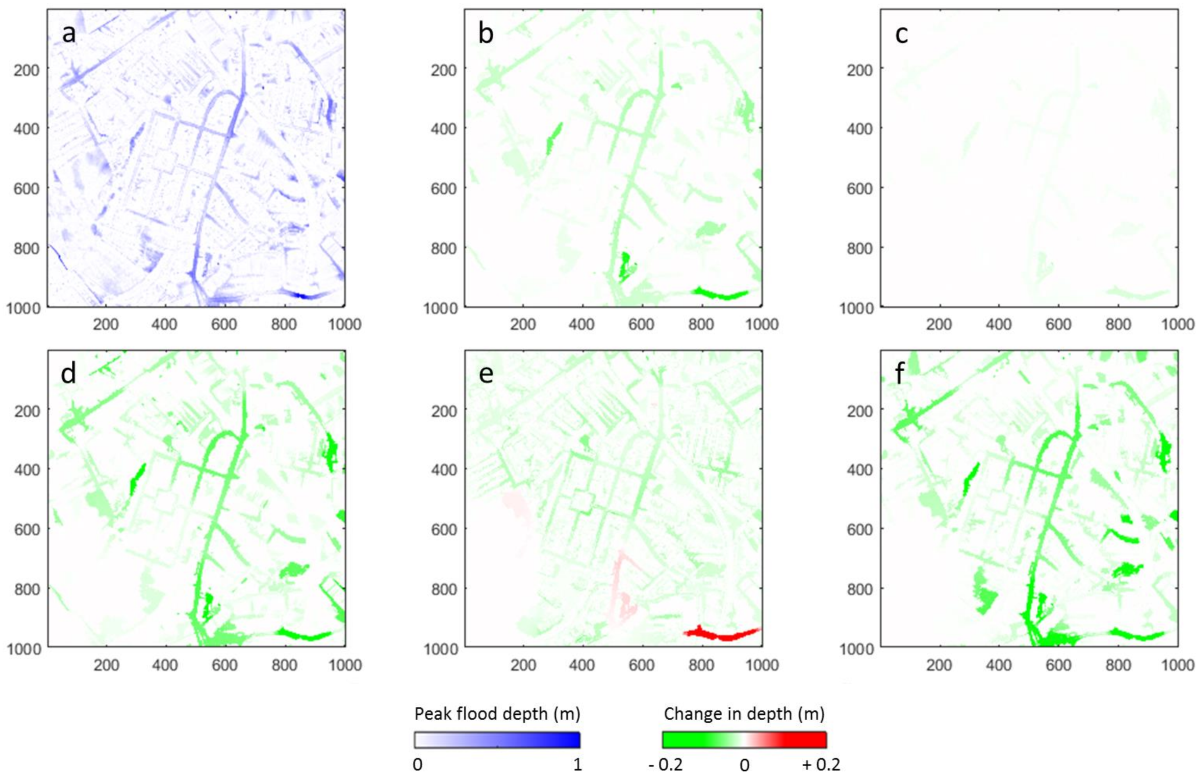

Figure 6.8: Peak flood depths during a one hour 200 year return period rainfall event: (a) do nothing, (b) green roof, (c) water butt, (d) rainwater capture, (e) permeable paving and (f) upgrade drainage

The largest flood reduction effect is visible in upgrading the drainage system and installing rainwater capture across the catchment. These strategies show around 20 cm of flood reduction across a large extent of the catchment. It should be noted that areas where no reduction is apparent can also mean that no flooding is in this region to reduce.

Examining peak depth visualises a snapshot of total flood effect, which is particularly useful for communicating hazards and an overview of strategy effects to stakeholders.

Examining the performance of intervention strategies

The previous chapter highlighted that interventions which appear to reduce flood extent most significantly do not necessarily correlate with those which show the largest damage cost reduction due to spatial variation in flood reduction effects and building locations. This distinction is important as flood management should prioritise impact reduction over hazard reduction, particularly when considering placement of surface water flood management interventions where location will effect flood extent. This highlights the advantages of a damage cost assessment (Figure 6.7) over proxy measures of impact (Figure 6.8), such as captured volume or intervention effects on a sub catchment scale, and emphasises the need to run flood simulations when comparing intervention strategies.

Figure 6.9 expands analysis to assess impact by breaking down damage costs for each intervention strategy across each event. Assessment of damage costs across each scenario indicates that short duration, high intensity rainfall generates the highest flood costs across all return periods for the Do Nothing scenario. This supports current literature emphasising prioritising investigation of short duration rainfall when assessing surface water flood management (Balmforth et al., 2006; Environment Agency, 2013; Burns et al., 2015d; Lamond et al., 2015; Schubert et al., 2017).

30 year rainfall

	Green roof	Water butt	Rainwater capture	Permeable paving	Drainage upgrade	Do nothing
1 hour	1.1	1.2	0.3	1.1	1.1	1.2
2 hour	0.9	1.0	0.8	0.9	0.8	1.0
3 hour	0.8	0.9	0.7	0.8	0.7	0.9
4 hour	0.7	0.7	0.6	0.7	0.7	0.8
6 hour	0.6	0.6	0.6	0.6	0.5	0.6
12 hour	0.5	0.5	0.5	0.4	0.4	0.5
24 hour	0.4	0.4	0.4	0.3	0.3	0.4
48 hour	0.3	0.3	0.3	0.3	0.3	0.3

100 year rainfall

	Green roof	Water butt	Rainwater capture	Permeable paving	Drainage upgrade	Do nothing
1 hour	1.4	1.4	1.2	1.4	1.3	1.5
2 hour	1.2	1.2	1.1	1.1	1.0	1.2
3 hour	1.0	1.0	0.9	0.9	0.8	1.0
4 hour	0.9	0.9	0.8	0.8	0.7	0.9
6 hour	0.7	0.7	0.7	0.6	0.6	0.7
12 hour	0.5	0.5	0.5	0.5	0.5	0.5
24 hour	0.4	0.4	0.4	0.4	0.4	0.4
48 hour	0.3	0.3	0.3	0.3	0.3	0.3

200 year rainfall

	Green roof	Water butt	Rainwater capture	Permeable paving	Drainage upgrade	Do nothing
1 hour	1.5	1.6	1.4	1.5	1.4	1.6
2 hour	1.3	1.3	1.2	1.2	1.1	1.3
3 hour	1.1	1.2	1.1	1.0	0.9	1.2
4 hour	1.0	1.0	0.9	0.9	0.8	1.0
6 hour	0.8	0.8	0.8	0.7	0.7	0.8
12 hour	0.6	0.6	0.6	0.5	0.5	0.6
24 hour	0.4	0.4	0.4	0.4	0.4	0.4
48 hour	0.3	0.3	0.3	0.3	0.3	0.3

Damage cost (million GBP): 0, 0.4, 0.8, 1.2, 1.6, 2

Figure 6.9: Flood damage costs associated with each intervention across all rainfall events for the Exeter case study

This is further supported by a trend for higher flood damage costs during shorter duration rainfall for all strategies during the 100 and 200 year return periods, and five out of six strategies for the 30 year return period. The exception to this trend is the performance of rainwater harvesting tanks in the 1 hour, 30 year rainfall event. This intervention demonstrates relatively low flood costs of £0.3 million during the one hour rainfall. This is lower than the calculated damage resulting from the 2 to 48 hour rainfall events which follow the same trend as other strategies, with shorter durations leading to increasing flood damages. This apparent anomaly can be explained by the rainwater capture capacity of 33 mm per cell, which exceeds the total rainfall volume of 32.3 mm falling during the 1 hour 30 year rainfall, and therefore captures all incoming rainfall to these cells; leading to substantially reduced flooding during this particular event. All other events exceed the capacity of the intervention, thus creating overflow; although in the case of longer flood events with lower intensities, this overflow can be accommodated by other surrounding drainage features such as the surface water system, leading to lower costs. When rainfall exceeds tank capacity, as seen during the 100 and 200 year events, the damage increases. This finding merits further analysis of tank size versus performance across a range of intensities and indicates the presence of tipping points for rainfall capture interventions, a concept which is examined in Section 6.3 in relation to intervention performance during design standard and extreme events.

The Do Nothing scenario generates the highest or equal highest flood damage costs across all return periods, demonstrating that no interventions worsen the catchment flood risk.

The largest intervention effects were observed during the shorter duration events. This is due to the existing surface water management approaches within the catchment having capacity to convey the relatively low intensities of long duration rainfall, thus minimising the observed difference between strategies.

Intervention performance ranking during longer duration events is not the same as for short duration events. The strategy with the lowest damage costs for the one hour event, particularly at lower return periods, is rainwater capture. This is despite appearing to not have as large a reduction versus drainage improvements when visually comparing peak flood depth and extent (Figure 6.8). The larger reduction in impact with a lower reduction of hazard extent is associated with the location of rainwater capture preventing surface water accumulation in and around properties, versus the drainage upgrades having a larger effect outside of these. This example highlights the importance of understanding the spatial disaggregation of hazard versus impact reduction.

Drainage upgrades tended to demonstrate the lowest damage costs during the higher intensity, longer duration events due to effective performance during prolonged rainfall. Conveyance based systems, such as drainage upgrades, can continue to function throughout the event and so lead to lower flood damages. This nuance highlights the complexity in the relative strengths of urban management strategies, indicating the benefits of rapid scenario screening able to identify characteristics of strategies designed for different rainfall durations.

Assessing resilient performance of interventions during extreme rainfall events

Resilient performance is assessed through analysis of impact in extreme events (Aldunce et al., 2015; HM Government, 2016; Butler et al., 2017). Frameworks which enable assessment of many simulations have the advantage of being able to simulate intervention response to conditions beyond design standards. Resilience is assessed relative to the magnitude and duration of failure across multiple events, in line with Butler et al (2016) which specifies that resilience minimises failure magnitude and duration. In this case failure is specified as any damage cost above zero. This research applies short duration flood costs (depth-damage) which act as a single metric that combines magnitude and duration as a monetary value.

Figure 6.10 shows the change in damage costs from each intervention strategy versus increasing rainfall intensity during the one hour rainfall event in response to design standard (30 year return period) and extreme (100 and 200 year return period) events.

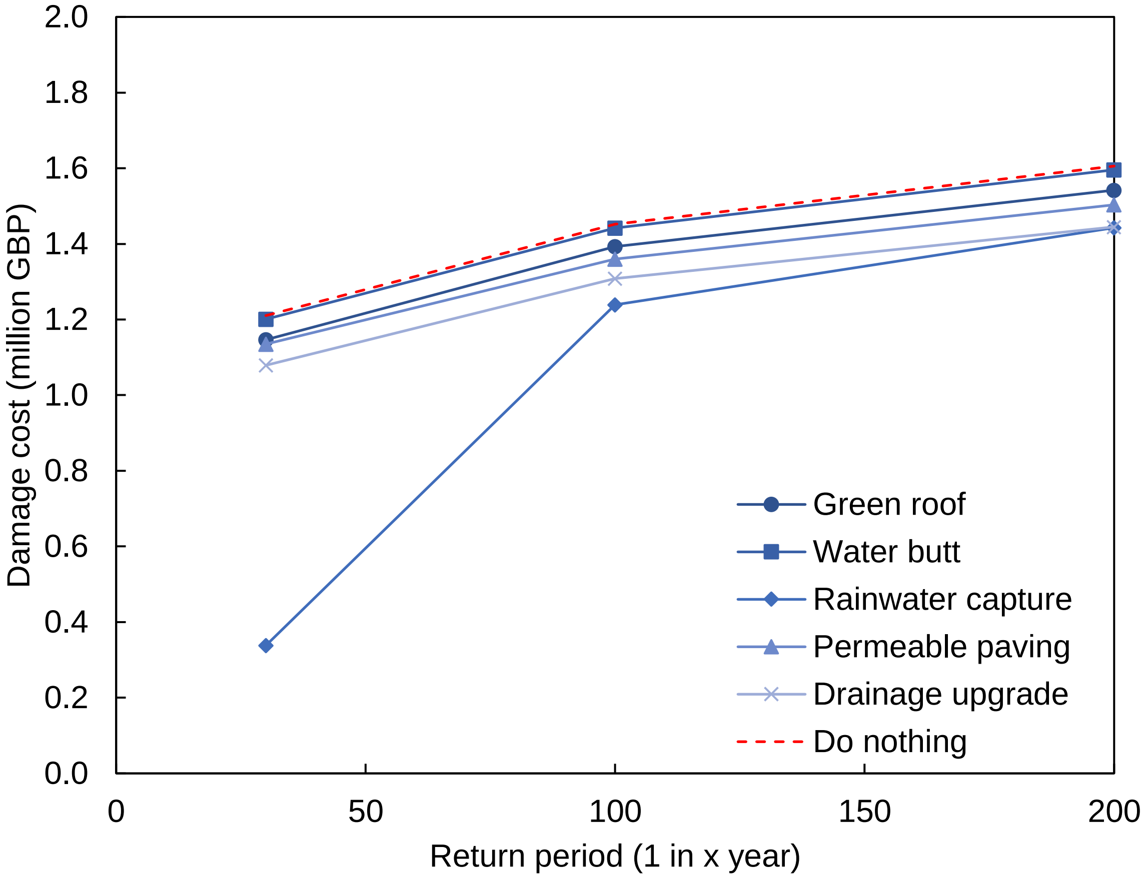

Figure 6.10: Damage cost versus increasing rainfall intensity during the one hour rainfall event for the Exeter case study

Interesting implications for resilience can be identified by the shape of the curves for each intervention in Figure 6.10. Interventions generating a shallower gradient demonstrate an ability to minimise damage beyond the standard design conditions, resulting in a more resilient performance relative to other scenarios. Some interventions exhibit a shallow curve for low return periods which steepens as higher return periods are reached, indicating failure in levels of service.

During the 30 year return period, rainwater capture results in minimum damage costs of approximately £0.3 million. All other strategies lead to at least £0.9 million more damage during this event. The same performance ranking applies to the 100 year event, however the difference between rainwater capture and the next best performing intervention, drainage upgrades, is reduced to around £0.1 million. This represents a tipping point in the performance of rainwater capture, whilst other interventions represent a more stable response to an increasing stress. During the 200 year event, the performance difference between rainwater capture and drainage upgrades becomes negligible. This change in flood damage cost rankings indicates varying levels of resilience to the increasing stress and highlights complexity in analysing intervention performance.

Figure 6.11 presents flood damage response to the two hour event. This demonstrates a similar change in ranking to Figure 6.10, where rainwater capture is initially the best performing intervention during the 30 year event, however leads to more damage than drainage upgrades over the 100 year event, and more damage than both drainage upgrades and permeable paving in the 200 year event. The role of capacity in flood resilience merits further investigation, and is explored in more detail in Section 6.3.

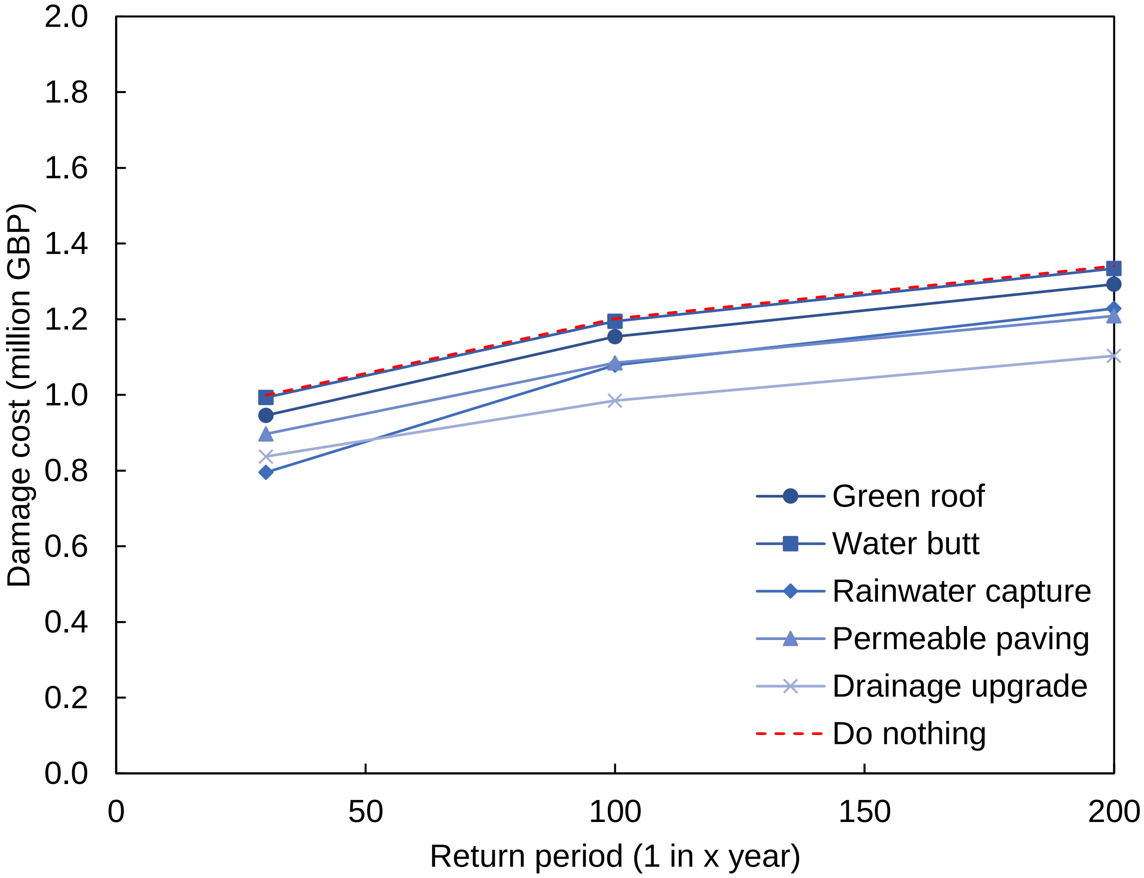

Figure 6.11: Damage cost versus increasing rainfall intensity during the two hour rainfall event

Variation in performance highlights the importance of assessing impact across multiple return periods when selecting surface water management interventions, providing evidence that the current paradigm of restricted consideration of events is not sufficient to ensure the best outcome in response to uncertainties in future climate and urban growth. This information can be presented as part of decision support to complement a standard assessment and has the potential to promote innovative interventions which meet design standards during high probability events, whilst providing additional resilience for low probability occurrences.

6.2.3. Key findings

Context specific findings relative to the intervention assessment in this catchment indicate that rainwater capture interventions exhibit the largest reduction in flood damages during short duration, high intensity events where capacity can be fully utilised; however, exceeding capture volumes leads to lower performance during longer duration or low probability extreme rainfall events. Interventions which are able to continue functioning over extended timescales, such as drainage upgrades, are more effective at managing long duration events and appear more resilient to the extreme rainfall, however damage reduction during extreme events represents a minimal saving versus the do nothing baseline scenario.

Broader findings from this section identify quantitative analysis of flood depths and damage costs provide a simple metric to evidence decision support, offering an advantage versus fast but qualitative screening tools such as stakeholder ranking (Ellis et al., 2004; Martin et al., 2007; Makropoulos et al., 2008; Young et al., 2010) and GIS analysis (Weng, 2001; Makropoulos et al., 2007; Viavattene and Ellis, 2013). Variation in intervention performance ranking relative to the complex relationships between event intensity, duration and frequency highlight the advantages applied through simulation of many events. Therefore, the key recommendation from this section is to screen the flood dynamics of many strategies prior to detailed design. Expanding analysis will also benefit and inform the resilience assessment principles presented here by expanding the response to a larger range of return periods.

Performance sensitivity to changes in rainfall duration, in particular the high costs associated with short term burst of high intensity rainfall, even when averaged across an event, indicate that assessment may benefit from a more detailed representation of events. It is therefore recommended that future analysis applies input hyetographs at a finer temporal resolution to simulate the effects of peak intensities within rainfall events.

Further capabilities could be added to the assessment framework through examination of intervention cost effectiveness and whether variation of intervention location will lead to preferential cost benefit ratios through taking into consideration the spatial differences between flood reduction effects and building locations.

6.3. Applying a cost effectiveness metric to assess the effect of intervention placement on performance

This section of the chapter responds to recommendations made in Section 6.2.3. Namely, the utility of including a cost effectiveness measure within option screening, investigating intervention performance across different locations and examining intervention response to design and extreme rainfall events. These recommendations correspond to objectives six, four and seven, respectively.

The intention of this section is to advance new methods which can be applied to complement established detailed modelling techniques through initial prioritisation of intervention cost effectiveness, suitable for evidencing and directing further detailed analysis using techniques which can be applied quickly and with limited data. Interventions include both green infrastructure and conventional solutions modelled at the property scale. Cost effectiveness is assessed by comparing an estimated cost of constructing and operating interventions versus an expected annual damage reduction cost.

6.3.1. Method

The study area is the same surface water catchment of a residential suburb in a UK city as applied in Section 6.2 (Figure 6.5). The catchment is approximately 700 m x 700 m and was identified using a GIS watershed analysis with 1 m resolution LiDAR. Predominant land use is residential, comprised of minor roads and semi-detached and terraced housing. A main road connects the north and south of the catchment. A large area of open recreational green space is located in the south west.

Characterising the study area

The study area elevation and land use was represented using the same process described in Section 6.2.1.

The investigation presented in Section 6.2 was used as a basis for evaluating a catchment critical rainfall duration through calculating flood damages using FEH design rainfall events at 1, 2, 3, 4, 6, 12, 24 and 48 hour durations across 30, 100 and 200 year return periods (Centre for Ecology and Hydrology, 2013). Peak flooding in all return periods was observed during one hour rainfall, therefore analysis of intervention performance was made relative to this event (Figure 6.7; Figure 6.9).

The number of return periods was expanded for intervention analysis to include 2, 5, 10, 20, 30, 50, 100, 200 and 1000 year rainfall events, all provided by the FEH database (Centre for Ecology and Hydrology, 2013). Rainfall was represented using hyetographs at a one minute resolution, this better represented the peak rainfall intensities during events, which earlier analysis found to correlate with the highest flood damage outcomes (Figure 6.12). Summer design rainfall profiles were selected due to characteristic higher peak intensities which are more likely to exceed drainage capacity and result in surface water flooding (Jones et al., 2012; Butler et al., 2018).

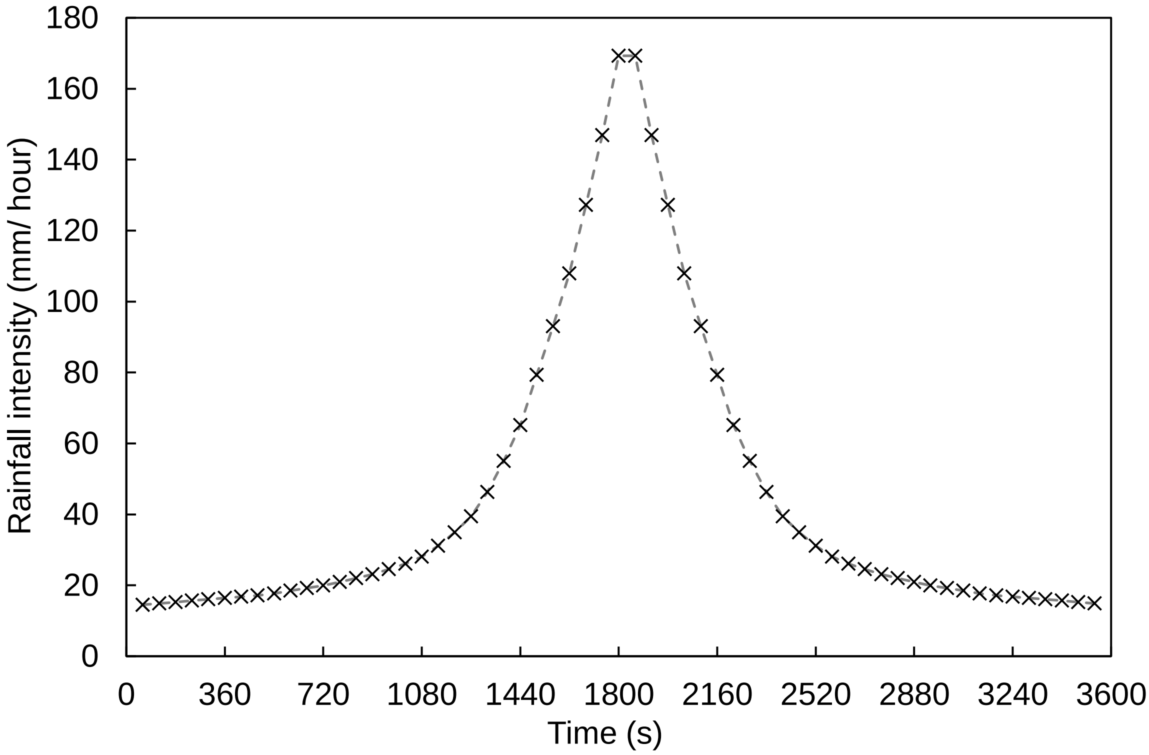

Figure 6.12: Input hyetograph for one hour rainfall at a 100 year return period for the Exeter case study catchment

Representing interventions

This study applied a more diverse range of interventions than in Section 6.2. In particular, this analysis was designed to examine the effects of different scale rainwater capture measures, in response to the indication of a rainfall capture tipping point as discussed in Section 6.2.2. The study investigates green roofs, water butts, rainwater capture tanks, permeable paving and drainage upgrades. This section outlines the modelling approach applied for each intervention, which is presented in Table 6.4 and summarised below. All strategies are modelled using the same process outlined in Section 6.1.

Table 6.9: Summary of intervention effects per 1 m^2 cell

Type	Rainfall capture (mm)	Output rate (mm/ hour)	Roughness (Manning's n)
Green roof	15	Land use	Land use
Water butt (100 l)	2	Land use	Land use
RW capture (1 500 l)	33	Land use	Land use
RW capture (3 000 l)	66	Land use	Land use
RW capture (5 000 l)	110	Land use	Land use
RW capture (10 000 l)	220	Land use	Land use
Permeable paving	-	17	0.015
Drainage upgrade (+12 mm/hr)	-	24	Land use
Drainage upgrade (+24 mm/hr)	-	36	Land use

Green roofs are represented by capturing 15 mm of rainfall prior to generating runoff. It is assumed that the substrate can capture rainfall with 100% efficiency until saturation occurs. As this intervention consists of water capture above the model domain surface, it will have no effect on surface roughness or infiltration rate.

Rainwater capture tanks are modelled based on the assumption the only controlling factor on storage is available volume, not the throttling effects of down pipes. Sensitivity to intervention capacity and tipping points is modelled through inclusion of four capture volumes: 1500 l, 3000 l, 5000 l and 10 000 l. Water butts were modelled using the same approach but with a conservative available capacity of 100 l per water butt. This low capture volume also functions as part of a sensitivity analysis to set a benchmark for very low capacity rainwater capture function.

Volume reduction properties of permeable pavements are controlled by the infiltration rate through the surface and available storage. Several studies have taken place to identify infiltration rates into commonly used surface materials (Pratt et al., 2002; Zachary Bean et al., 2007; Collins et al., 2008a). These studies found infiltration rates for concrete block pervious paving have been recorded from 2.6 up to 17.2 mm/hour, with average rates around 5 to 7 mm/hour, therefore a conservative estimate of 5 mm is applied for this measure. Roughness values are taken from Manning's n coefficients for concrete (Section 6.2.1).

No data for the underlying surface water sewers are available, therefore drainage upgrades have been included by increasing water output rates linked to the drainage system (12 mm/ hour) by an additional 12 mm/ hour and 24 mm/hour, representing a doubled and tripled rate from Environment Agency (2013). This also functions as a sensitivity analysis for the effects of draiange capacity assumptions.

Intervention placement scenarios

Examining the performance of a baseline scenario and nine interventions (Table 6.9) across a combination of locations in the catchment generated 88 scenarios for simulation. Each scenario represented placing one intervention type across a location (or locations) in the catchment. Eight locations were selected using the street layout of the study area as shown in OS Mastermap (Figure 6.13) and the areas of flooding identified during the preliminary analysis of critical rainfall duration (Figure 6.8). Green roofs, water butts and rainwater capture tanks were applied to building roofs. Drainage upgrades were applied to the catchment surface. Permeable paving was applied across carparks in the residential zone. The 88 scenarios consisted of interventions applied across: the entire catchment (16); locations 1 to 8 individually (64); locations where flooding was identified including 1, 2, 3 and 4 combined (8) and 1, 2, 3, 4 and 6 combined (8). A scenario also represented applying permeable paving to car parks (1) and another represented a catchment baseline where no interventions were applied (1).

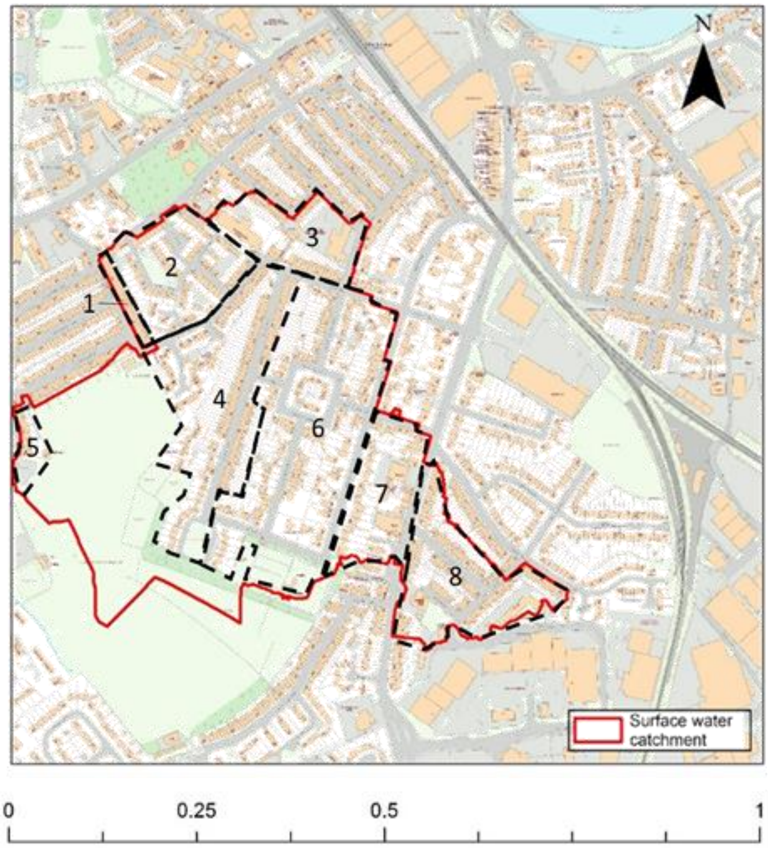

Figure 6.13: Defining eight regions for basing intervention placement locations across the Exeter study catchment

The intention of examining multiple locations was to demonstrate the utility of the framework for screening multiple scenarios, responding to a need for tools to simulate flood dynamics of surface water management across many possible locations and to prioritise future modelling using preliminary analysis. This is particularly important given the spatial variation of intervention effects, which is not typically included in intervention screening (Section 2.2.2).

Simulating scenarios

Each intervention scenario was simulated across nine return periods, resulting in a total of 792 simulations. The simulation was run using an ‘Nvidia Tesla K20c’ (2496 CUDA cores) at a grid resolution of 1 m^2 and a minimum simulation time-step of 0.01 s.

Simulation speed for the most intensive simulation, one hour duration 1 in 1000 year summer design rainfall, was six minutes. This simulation was extended by four hours of model time to ensure sufficient time for all runoff processes. This took an additional 21 minutes to run.

6.3.2. Intervention performance assessment using a cost effectiveness measure

Decision support should be enhanced through a transparent evidence base, using clear performance metrics (House of Commons, 2016). Flood management actions are expected to demonstrate positive benefit to cost ratios (Environment Agency, 2014). Therefore, it is important that a framework intended to enhance catchment screening and prioritise further actions is capable of indicative cost estimates for schemes.

It should be noted that certain artificial economic incentives, such as financial penalties for DG5 (property sewer) flooding, and water company performance outcomes may skew the economic analysis and in certain cases prioritise non cost beneficial projects to ensure legislative compliance (Ofwat, 2012, 2017, 2018). However, screening project costs and identifying multiple routes to the reach performance measures is still of benefit to practitioners.

This section outlines development of a cost effectiveness metric, intended to assist steering strategic design. The scope of analysis is high level estimation using approximated average costs to identify trends. Analysis also refers to 'cost-effectiveness' rather than 'cost-benefit' in recognition of the extended multiple benefits, particularly those associated with green infrastructure, which are not included in this analysis but are the subject of a wide body of current research (MWH, 2014; CIRIA, 2015; Fletcher et al., 2015; Hammond et al., 2015; Jose et al., 2015; Norton et al., 2015; Mijic et al., 2016; Bowen and Lynch, 2017; Kunapo et al., 2018). The metrics applied to this case study use non contextual average estimates from literature, it is anticipated that practical application of the approach would apply site specific cost data available from catchment partner organisations.

Intervention capital costs

Capital costs of interventions are presented in Table 6.9. These have been calculated based on academic and government studies which provide a range of average costs, discussed in detail below. Where multiple cost estimates are available the higher cost was used to develop a safety margin.

Capital costs have been converted to present day (2018) values using UK inflation rates (Office for National Statistics, 2018). Operational costs are calculated using discounting at a rate of 3.5% over a 30 year period (Environment Agency, 2010; HM Treasury, 2013). Costs are translated to a value per 1 m^2 cell through dividing the intervention total cost by the area for which the intervention is situated, typically across a roof (45.5 m^2 in DCLG, 2015) or per m^2 for surface based interventions. A similar method was applied in Environment Agency (2007).

It should be noted that in practice the costs of interventions are heavily influenced by locational and project context, therefore these values should be considered indicative for the purposes of demonstrating the methodology. Where this method is applied practically it is recommended that contextual cost models are applied.

Table 6.10: Cost estimates for intervention construction, operation and routine maintenance per 1 m^2 cell, adjusted to 2018 values

Intervention	**Capital cost per measure (£)**	**Capital cost per cell (£/m^2)**	**30 year operational cost per cell (£/m^2)**	**30 year total cost per cell (£/m^2)**
Green roof	131.40 per m^2	131.40	469.87	601.27
Water butt (100 l)	335.34 per butt	7.37	4.25	11.62
RW capture (1 500 l)	3050.00 per system	67.03	10.12	77.15
RW capture (3 000 l)	4270.00 per system	93.85	10.12	103.96
RW capture (5 000 l)	4880.00 per system	107.25	10.12	117.37
RW capture (10 000 l)	5856.00 per system	128.70	10.12	138.82
Permeable paving	74.52 per m^2	74.52	10.12	84.64
Drainage upgrade (+12 mm/hr)	648.42 per 1 m pipe	3.10	0.13	3.23
Drainage upgrade (+24 mm/hr)	648.42 per 1 m pipe	3.72	0.17	3.89

Literature states rainwater capture tanks (adjusted for 2018 values) cost £3050 for 1500 l, £4270 for 3000 l, £4880 for 5000 l and £5856 for 10 000 l (Roebuck et al., 2011). Other studies corroborate this range of values (Environment Agency, 2007a). Green roofs are estimated to cost £131.40 per m^2 in 2018 prices (Bamfield, 2005). Water butts were estimated to cost £335.34 per unit in 2018 prices (Stovin et al., 2007). It has been assumed that water butts will be replaced after 15 years at a discounted rate of £193.39, this represents the more conservative assumption from available literature (Environment Agency, 2007a; Ossa-Moreno et al., 2017). Permeable paving costs are based on present day concrete block pervious paving in literature of approximately £74.52 per m^2 (Environment Agency, 2007a; Stovin et al., 2007; Woods Ballard et al., 2007).

Cost of sewers are provided as a conservative upper estimate of £648.82 per m of 450 mm diameter pipe laid under an urban highway (Environment Agency, 2015). A cost per m^2 has been estimated by calculating the area which a single pipe could drain at full flow during the time of concentration. Flow rates were estimated using the Colebrook-White equation with dimensions typical of an urban surface water drainage system designed to reach a self-cleaning velocity, as described in Chapter Three and Four (Butler et al., 2018). Application of this method included the standard assumptions of a pipe roughness of 0.6×10^{-3} m and a kinematic viscosity of 1.14 x 10-6 m^2/s.

Flow rate was calculated using a shallow gradient of 1:200, indicative of a safety margin representing slow flowing sewers. The pipe full flow rate was linked to the increase in cell output rate by attributing pipe flow capacity to a sub-catchment where each cell drained at the rate of +12 mm/hour or +24 mm/hour. The sub-catchment was assumed to be rectangular where the pipe was laid in a straight line through the middle of the area. This calculation estimates a 450 mm diameter pipe can drain at 12 mm/hour across a 280 m x 280 m region, at 24 mm/hour across a 200 m x 200 m region and at 36 mm/hour across a 160 m x 160 m region. The cost of the pipe length was divided between each cell within these regions to calculate an approximate cost per m^2 drained. This method assumes connection to an existing sewer system without additional resizing of downstream pipes or treatment. This cost is an indicative figure, designed to test model application.

Intervention operation and maintenance costs

Maintenance costs are shown in Table 6.9, these are indicative estimates of routine maintenance which do not consider decommissioning costs or out of the ordinary maintenance issues. All costs are converted to 2018 values (Office for National Statistics, 2018).

Literature indicates green roofs require £3650 per year for the initial two years and £876 a year maintenance afterwards (Bamfield, 2005). Rainwater capture maintenance is estimated to cost £0.55 per m^2 / year (Environment Agency, 2007a). Water butts are assumed to have a negligible annual maintenance cost, but are replaced after a 15 year design life (Environment Agency, 2007a). Average costs for operation and maintenance in sewers are specified in industry estimation advice (Hunter Water Corporation, 2013). A 450 mm gravity fed sewer is estimated to cost £1512 per km/ year. This cost was translated into a cost per m^2 scaled by the catchment size of each pipe network to calculate an indicative cost per cell.

Measuring intervention performance using cost effectiveness

Costs of property damage have been calculated using the process described in Section 3.5.4. Damage costs have been taken from industry standard depth damage curves for an average residential property (Penning-Rowsell et al., 2010). This relates the direct and tangible costs of short duration inundation (less than 12 hours), typical of surface water flooding, to the building fabric and household inventory. Damage is only related to depth, without consideration of velocity or other damaging factors such as contamination. Intangible and indirect costs have not been included within this assessment (Hammond et al., 2015).

Costs and qualitative assessment of multiple benefits have been omitted from this research due to data and modelling requirements being beyond the scope of an initial project screening, analysis of these can be found in other studies (Ashley et al., 2002; CIRIA, 2015; Woods Ballard et al., 2015; Mijic et al., 2016; Ossa-Moreno et al., 2017; Kunapo et al., 2018).

Projecting costs across a 30 year design life

The annual effect of damages was calculated through projecting EAD over a thirty year design life (Environment Agency, 2010). EAD for each strategy is calculated through sampling cost damage across a range of different probability events to generate a curve representing damage versus annual exceedance probability. This curve represents damage costs in low probability high magnitude events as well as high probability, low magnitude events. This analysis has included a wide range of probabilities by sampling 2, 5, 10, 20, 30, 50, 100, 200 and 1000 year events. A full description of the calculation technique is provided in Section 3.5.4.

As intense local precipitation is the controlling factor in creating surface flooding it is reasonable to assume the return period of the rainfall can be applied as the return period for the flood (University of Exeter, 2014).

EAD for each intervention was used to quantify benefit through avoided flood damage relative to a baseline. Future costs were calculated over a thirty year period using a discount rate of 3.5% per year, as specified by the UK Government (Environment Agency, 2010; HM Treasury, 2013). It should be noted that discounting adjusts net present value for future economic costs, and does not adjust costs in relation to potential future changes to probabilities of events. The design life of all interventions, bar the water butts, was assumed to be the same.

Intervention performance was assessed using a simple cost effectiveness metric which compared the cost of the intervention over thirty years with the benefit of damage avoided over the same period.

Assessing intervention resilience

A long-standing critique of resilience science has been a lack of operational and quantitative application of theories (Aldunce et al., 2015). This is of particular note in complex systems, such as surface water management in cities.

The Safe & SuRe project at Exeter University (Butler et al., 2014, 2017) has proposed a definition for resilience which allows a quantification of resilience in a practical setting by measuring the failure magnitude and duration during extreme events. This model has been applied to a range of challenges, including: wastewater treatment (Sweetapple et al., 2014, 2017), water distribution (Diao et al., 2016), urban drainage (Mugume et al., 2015) and urban catchment management (Casal-Campos et al., 2015). As of yet this research has not been applied to assessing surface water flooding interventions.

The Safe and SuRe project specifies resilience as ‘general’, the ability of a system to as limit failure duration to all threats, and ‘specified’, limiting failure to a particular threat based on an operational goal. Specified resilience is applied in this research to identify the resilience of cities to surface water flooding.

Resilience is measured as a function of the magnitude and duration of failure. This research applies short duration flood costs (depth-damage) which act as a single metric that combines magnitude depth and duration.

6.3.3. Results and discussion

Comparison of interventions when applied across all available surfaces

Figure 6.14 shows the damage cost versus mean rainfall intensity for interventions applied across all suitable areas within the catchment. Results were used to develop a performance curve representing the damage cost response of intervention strategies to a range of rainfall intensities, including design standard and high magnitude events. Figure 6.14 maps the intensity of each one hour rainfall event (primary x axis) to a return period, expressed in terms of a '1 in X year' event (secondary x axis).

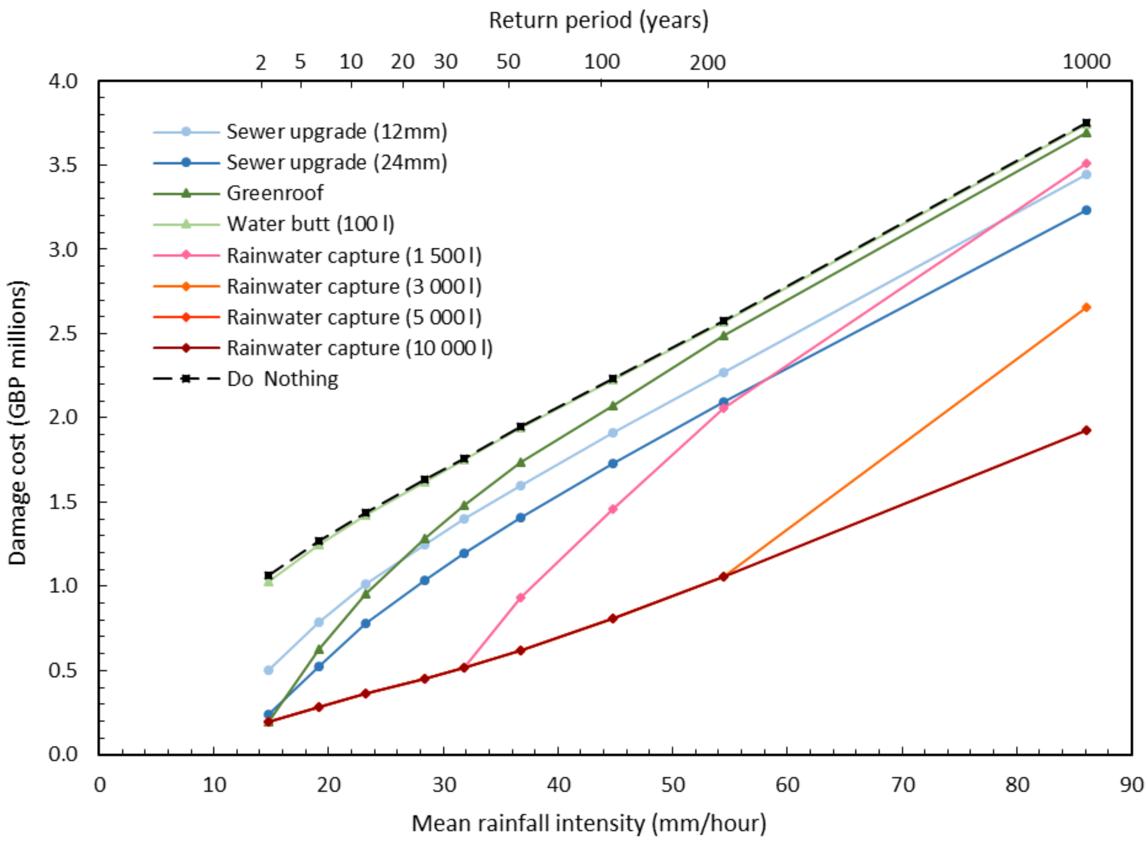

Figure 6.14: Damage cost versus mean rainfall intensity for interventions applied across all suitable locations for the Exeter case study

The damage cost resulting from each intervention strategy rises as the rainfall intensity increases. The highest flood damage costs during each event are consistently observed in the ‘Do Nothing’ baseline, where no interventions are applied.

Large rainwater capture tanks (> 5 000 l) generate the lowest flood damage costs at all rainfall intensities. Smaller tanks perform well at low return periods, but lead to very large damage costs at higher return periods as rainfall exceeds storage capacity. This generates a spike in the damage curve for these interventions, indicating low resilience to events above design conditions. Drainage upgrades do not provide as great a damage reduction as rainwater capture interventions, however exhibit a relatively gradual and consistent increase in damage as a response to higher magnitude events. This implies a higher resilience to larger magnitude events, as indicated in Section 6.2. During the 1 in 1000 year event drainage upgrades perform better than rainfall capture at 1 500 l and below.

Permeable paving shows only a slight improvement over the ‘Do Nothing’ scenario, this is attributed to a very small area within the catchment being suitable for construction relative to the large areas suitable for other interventions. As such this line was not discernible and has been omitted from the Figures 6.14 and 6.15. Conclusions regarding the effect of permeable paving should be considered in the mitigating context that the very small area of permeable paving applied is unfavourable relative to the much larger footprints of other interventions in this study.

Figure 6.15 illustrates the percentage of total damage avoided by each intervention, highlighting the drop in damage avoided as rainwater capture interventions exceed storage capacity. This occurs at around 31 mm/hour for 1500 l tanks and 54 mm/hour for 3000 l tanks. During high rainfall intensities these interventions approach zero damage reduction due to storage filling too early and shifting the time of flood concentration rather than reducing magnitude. The ability of surface drainage to reduce damage by a more consistent value is attributed to continually removing runoff across the event, rather than having a finite storage volume filled.

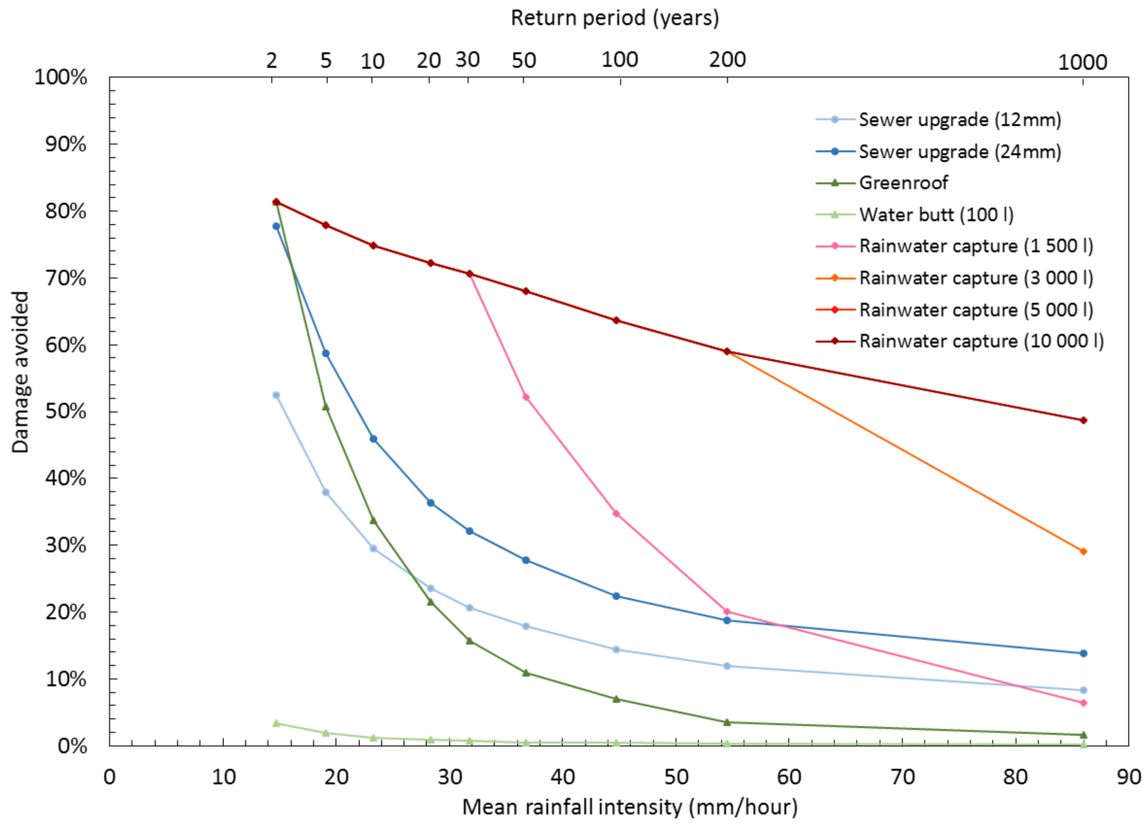

Figure 6.15: Percentage of damage avoided versus mean rainfall intensity for interventions applied across all suitable locations in the Exeter case study

Spatial variation of intervention EAD

Interventions were also examined when placed on suitable surfaces in the regions indicated in Figure 6.5. Figure 6.16 presents the EAD and cost effectiveness for each intervention when applied across a different region of the catchment.

Intervention strategies demonstrate a wide range of EAD outcomes. The interventions which demonstrated the largest reduction in EAD were catchment wide installation of large capacity (> 5000 l) rainwater capture tanks, which reduced damage costs as low as £196 000, representing a 76% saving versus the baseline scenario (Figure 6.15). These interventions demonstrate the lowest EAD for every placement option (Figure 6.16). The lowest EAD is generated when the intervention is applied across all available areas, however application of these measures in Areas One, Four and Six can achieve damage costs around £400 000, representing a 51% saving versus the Do Nothing scenario and a similar result to applying other interventions across the entire catchment.

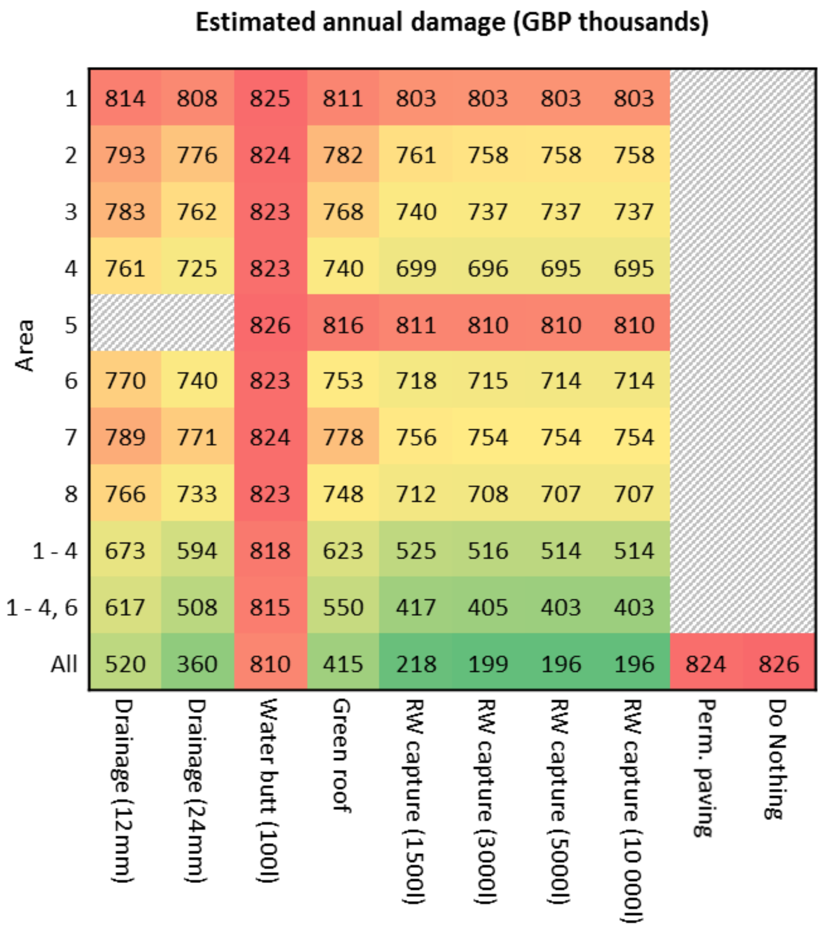

Figure 6.16: Comparison of EAD for interventions across all placement scenarios in the Exeter case study

The worst performing scenario is the Do Nothing baseline, which equates to an EAD of £826 000. The worst performing intervention is permeable paving, however it should be noted that this intervention was only investigated across a very small scale application due to uncertainties regarding suitability of application across the catchment. Water butts generate the largest EAD when compared with other interventions applied across the same areas, and only represents a marginal improvement of up to £16 000 less than the Do Nothing scenario. Poor performance is attributed to the low capacity of water butts not providing sufficient storage to prevent peak flooding during the event, particularly when applied across small areas. Better performance is seen at catchment wide application as the cumulative effects of capture reduce runoff volume, however the saving is still marginal relative to the high intensity rainfall experienced during the one hour rainfall event. This corroborates existing literature calling for catchment scale approaches to realise flood reduction benefits (Wong and Brown, 2009; Burns et al., 2012, 2015a; Palla and Gnecco, 2015).

Catchment wide application of interventions led to lowest EAD for every intervention studied, however selection of different intervention placement locations leads to variation in resulting EAD. Application of interventions in Area Four demonstrates the lowest EAD for all strategies, relative to other single locations. The worst locations for placement are Areas Five and One. These regions demonstrate EAD's up to £115 000 higher than the same strategies applied elsewhere in the catchment. Both of these locations are relatively small and isolated regions on the periphery of the catchment.

Variation in performance highlights the importance of investigating multiple intervention locations when designing strategies. A trend of lower EAD associated with catchment wide solutions versus individual locations supports current literature emphasising a need for broad scale implementation of strategies to manage catchment scale flood hazards (Wong, 2006; Wong and Brown, 2009). This is of particular significance when considering the better performance of dispersed and lower capacity measures such as green roofs and 1500 l tanks relative to intensive application of high capacity measures in a single location.

Spatial variation of intervention cost effectiveness

It is crucial to consider a range of metrics when evaluating intervention performance. Figure 6.17 presents the cost effectiveness of interventions applied across the locations identified in Figure 6.5. This figure highlights that strategies which generate the lowest EAD (Figure 6.16) do not correlate with damage avoided per GBP spent (Figure 6.17). Cost effectiveness is an important consideration in flood management as long term investment decisions must represent value relative to other public spending requirements (Environment Agency, 2007a, 2010; DEFRA, 2010; Jayasooriya and Ng, 2014).

Estimated avoided damage over 30 years per GBP spent

Area	Sewer (12mm)	Sewer (24mm)	Water butt (100l)	Green roof	RW capture (1500l)	RW capture (3000l)	RW capture (5000l)	RW capture (10 000l)	Perm. paving	Do nothing
1	20.7	26.0	0.7	0.3	4.0	3.0	2.7	2.3		
2	10.6	13.3	0.6	0.3	3.3	2.5	2.2	1.9		
3	16.3	20.1	0.9	0.4	4.1	3.2	2.8	2.4		
4	9.6	12.3	0.5	0.3	3.3	2.5	2.3	1.9		
5			0.3	0.3	3.0	2.3	2.1	1.8		
6	13.5	17.3	0.6	0.3	3.4	2.6	2.3	1.9		
7	13.8	17.1	0.8	0.3	3.8	2.9	2.6	2.2		
8	6.6	8.5	0.4	0.3	2.9	2.2	2.0	1.7		
1 - 4	11.6	14.6	0.7	0.3	3.6	2.7	2.4	2.0		
1 -4, 6	12.1	15.3	0.6	0.3	3.5	2.7	2.4	2.0		
All	10.6	13.4	0.6	0.3	3.4	2.6	2.3	2.0	0.1	0.0

Figure 6.17: Comparison of intervention cost effectiveness over a 30 year period for all placement scenarios in the Exeter case study

Surface drainage upgrades demonstrate the most cost beneficial application within this catchment. When applied across the entire catchment, upgrading by 24 mm/hour reduced EAD to £360 000, representing a 56% saving on the do nothing scenario and achieving a damage cost reduction of £13.4 per £1 spent. Catchment wide high volume rainwater capture, which demonstrated the lowest EAD, generates a damage cost reduction of £2 per £1 spent. This result highlights a variation in ranking when assessing intervention performance using different metrics.

Green roofs and water butts demonstrate the lowest cost effectiveness in all location permutations. This is likely to be due to the high costs of green roof maintenance within the catchment and the very conservative capacity assumptions (100 l) made for water butts. These are the only interventions which demonstrate greater costs of damage than intervention installation, operation and maintenance. However, it should be noted that green infrastructure is likely to offer other additional benefits to catchments which are not costed within this assessment. This includes multiple benefits such as reduction in ambient heat, opportunities to re-use water and increasing biodiversity (Oberndorfer et al., 2007; CIRIA, 2015; Norton et al., 2015; Woods Ballard et al., 2015; Bowen and Lynch, 2017; Kunapo et al., 2018).

Comparatively high cost effectiveness of surface drainage may be a result of low costs not accounting for the total drainage build and maintenance costs. It should be noted that the strong performance of sewer based interventions could also be achieved using extensive infiltration based measures, which may also convey additional benefits to the catchment.

The same intervention applied across a range of locations demonstrated a range of cost effectiveness outcomes. Different types of technique (i.e. rainwater capture and water removal from the surface) resulted in different optimum locations. Surface drainage measures applied in Area One demonstrated the highest damage reduction ratio of up to 26 times the intervention cost. This area is a small region in the north west of the catchment which experiences deep flooding (Figure 6.5). The strong performance of catchment surface drainage indicates the advantages of small scale application of high volume measures in areas with significant flood hazards. Rainwater capture measures were most effective when applied across Area Three, a larger and more dispersed region. Here, interventions demonstrate up to a 4.1 ratio of benefit to cost. These measures perform similarly well in Area One, where they demonstrate a ratio of up to 4.0.

Although less cost effective, rainwater capture interventions demonstrated more consistent ratio for the same intervention applied across each location than surface drainage counterparts. The variation in maximum and minimum ratios for rainwater capture interventions is a difference of 0.6 through to a difference of 1.2. Surface drainage interventions indicate a much wider variation, with ranges of 14.1 and 17.5. Consistency of rainwater capture methods despite different placement strategies supports more reliable performance when applied in new locations. This is attributed to these measures removing runoff from the site of damage (buildings), which is less reliant on the spatial difference in runoff patterns present on the surface.

The range of cost effectiveness across all scenarios varies from 0.3 (Area Five green roofs) through to 26.0 (Area One drainage). Wide range of performance values indicates the importance of examining multiple placement strategies for intervention options, and supports application of rapid scenario screening techniques to screen urban flood management actions.

Resilience of interventions to extreme rainfall

Findings specific to the case study interventions indicate that that relative performance of interventions is inconsistent as the intensity of the rainfall increases from design standard to extreme rainfall. Interventions which minimise damage during higher probability, low intensity rainfall are not always observed to be the most effective interventions in lower probability, higher intensity rainfall events. This is expressed most clearly when considering how the ranking of interventions changes during each return period (Figure 6.14).

All rainwater capture tanks are the equal best ranking intervention during two to thirty year return period rainfall. Large (> 5 000 l) tanks remain the best ranking intervention for all events; however smaller tanks demonstrate inconsistent performance, with a large increase in flood damage costs and a consequent reduction in ranking during higher return periods. The increase in flood damage costs is observed with 1 500 l tank scenario from the fifty year return period onward. By the 200 year return period this intervention is only marginally better performing than the 24 mm/ hour sewer upgrade, although it is still the second ranked strategy. By the 1000 year return period the intervention is now ranked fifth and demonstrates considerably worse performance than interventions it outperformed during the lower intensity events.

Sewer upgrades perform well during lower intensity rainfall, but demonstrate higher damage costs as intensity increases. Damage costs rise in a relatively consistent and incremental values in response to the increasing stress. This stable rise in cost results in the interventions improving their ranking during the higher return periods, despite initially ranking poorly. The strategies still demonstrate increasingly high costs relative to the best performing interventions in each event.

Green roofs rank equal first place during the two year return period but fall to sixth following the twenty year event. The cost increase steps are particularly large relative to other interventions between these two return periods.

Both small capacity water butts and the do nothing base case demonstrate the worst performance and are respectively ranked eighth and ninth at each return period. Damage cost increases in stable steps relative to increasing rainfall intensity.

In this example, the proposed mechanisms controlling damage increase are the storage capacity and rate of runoff removal from each cell, parameterised for each intervention. Storing rainfall is an effective damage reduction technique when storage is able to contain all rainfall, however as rainfall exceeds intervention capacity the damage increases significantly at each additional increase in intensity. Removing rainfall at a set rate from a cell via infiltration or increasing the drainage capacity did not perform as effectively as capturing all of it, however a more consistent response to the increasing rainfall events was observed due to a continuing reduction effect across the event. The mechanisms presented here are simplified: In practice the output rate is controlled by a variety of physical processes including hydraulic limitations in the piped system and saturation in soils, therefore these findings can only be considered indicative of high level strategic implications of the actual strategies.

Generally, these findings indicate that intervention performance during a high probability event is not an indicator of performance during low probability events. This is of major significance when considering a planning environment focused on meeting specified design standards versus environmental hazards which are increasing in severity as a response to climate change, urbanization and aging infrastructure systems (Chocat et al., 2007; Wheater and Evans, 2009; Howard et al., 2010; IPCC, 2014). Planning based solely on design standard events is not guaranteed to develop systems which are able to cope with extreme events (Butler et al., 2017). Future developments to planning methods should include analysis of a range of events and conditions so decision makers are able to better manage system shocks.

It should also be recognised that this analysis is focused on homogenous intervention strategies whereas in practice catchment management is likely to be developed through multiple integrated intervention types designed to accommodate a range of rainfall intensities. Hoang and Fenner (2015) describe how an integrated portfolio of interventions interacts to effectively manage day to day rainfall and design standard rainfall alongside extreme events. The research indicates that green infrastructure can be applied to accommodate every day and design standard hydrological function whilst achieving enhanced outcomes for urban ecology and societal benefits (Woods Ballard et al., 2015; Fenner, 2017). Complementary large capacity surface water management infrastructure can then be installed to manage extreme rainfall with dispersed green interventions attenuating the peak volume. The analysis presented in this chapter supports Hoang and Fenner's conclusions regarding intervention effectiveness across the spectrum of flood events through the observed variation in performance between intervention types in response to increasing rainfall intensities. Therefore, although day to day hydrological function and enhanced multiple benefits are outside the scope of this thesis, it is significant that conclusions regarding the performance of green infrastructure during extreme rainfall should be considered within the context of developing integrated catchment management strategies which interact to manage every day, design standard and extreme events.

Application of simplified simulation approaches is one way of including extreme events within design. These approaches have the advantage of assessing many scenarios and expanding understanding of catchments, but encounter several drawbacks regarding the simplification of underlying physical processes. These approaches require understanding of hazard characteristics (i.e. rainfall IDF) in order to simulate surroundings. As such they are best applied at an initial strategic level of design, with findings advanced and corroborated by further more detailed analysis.

Alternative approaches of including extreme events within planning include application of 'middle state' failure analysis or emergency planning (Mugume et al., 2015; Butler et al., 2017; Sweetapple et al., 2017). Middle state analysis removes the need for understanding of hazards by systematically assessing how a system operates as more components fail. This has been applied with success to pipe networks (water distribution systems and sewers) where components can clearly be identified and changed. So far this approach has not been applied to surface water management for which the spatial complexity of hazards makes defining the middle state failure theoretically and computationally challenging.

On the other hand, emergency planning approaches encourage planners to develop contingency plans for failure as part of an understanding that unprecedented and unknown events may take place, so advance communication on strategies for managing failures gracefully becomes necessary (Cutter et al., 2010; Alexander, 2013; Scolobig et al., 2015).

On balance, a combination of these approaches is likely the best outcome for managing resilience, however the advantages of applying visualisation of resilience to decision support should not be understated. Particularly in light of potential for quick wins, where a selection of similarly costed strategies may all meet design standards but certain interventions may provide additional benefits beyond this.

Framework utility

The speed of simulation using the framework enables analysis of intervention performance across many return periods. This facilitates analysis of intervention resilience to extreme events alongside evaluating design standard performance. The observed variation in intervention performance across events highlights the importance of evaluating a range of conditions when designing strategic infrastructure. Interventions which perform well within standard conditions may fail to provide protection to high magnitude events.

Cost assumptions were focused on developing a fast but high level assessment and do not take into account site specific costs. Uncertainty has been managed through cost valuation at the high end of estimated ranges which may lead to overestimation of intervention costs. Estimation of sewer costs using a value per area drained is only suitable as an initial estimate due to the complexities and costs associated with installing pipes and connecting (or resizing) to existing networks and treatment facilities. It is recommended that this approach is only used for screening, and is validated on a catchment basis by comparison with costs of similar schemes.

The cost effectiveness metric applied during this study is a simplified metric focused on avoided direct flood damage to buildings. Future development of this work could enhance this metric through inclusion of additional benefits certain interventions may provide. In particular, studies indicate that green infrastructure may provide significant and tangible benefits including a reduction in the urban heat island effect, improvements in air quality and use of captured rainfall. Intangible benefits such as a reduction in risks to life, prevention of psychological impacts, amenity value and mitigation of climate change are also relevant when comparing infrastructure options (CIRIA, 2015). These benefits are difficult to monetise without detailed investigations using specific models, however studies have begun to develop mechanisms for estimating these (Ashley et al., 2002; Ossa-Moreno et al., 2017). Studies indicate inclusion of multiple benefits within option screening is likely to increase the cost effectiveness of interventions, particularly green infrastructure (Woods Ballard et al., 2015).

6.3.4. Key findings

This section demonstrated a resource efficient analysis of intervention cost effectiveness in a UK catchment through applying a rapid screening framework requiring minimal setup time, readily available data and simulation speeds of less than six minutes per scenario. Resource efficient analysis enabled screening of many intervention types, placement locations and rainfall scenarios, including extreme events not normally modelled within surface water management. The main utility of the approach is early catchment screening to develop evidence to inform and steer future detailed design.

Catchment scale application of large rainwater capture interventions achieved the largest reduction in flood damage costs across the case study in all scenarios. The most cost effective intervention was found to be localised surface drainage upgrades; however, discussion indicates that cost estimates for these upgrades are high level and in practice they may be more expensive due to the costs required in connecting to existing drainage networks.

This work identifies that performance of strategies during low magnitude events is not reflective of a strategies response to extreme events. A paradigm based on design standard planning therefore misses assessing resilient performance. A range of approaches can be used to assess resilience and it is important that these feature in future urban design in order to ensure preparedness for unexpected, unprecedented and extreme events. Visualisation of resilience curves using rapid simulation of many scenarios is one way of achieving this.

6.4. Chapter conclusions

This chapter responded to the need to include novel flood management strategies within decision support frameworks through developing representations of specific interventions and then evaluating performance across a wide range of rainfall scenarios, including design standard and resilience focused events.

Section 6.1 reviewed current literature to develop representations of specific interventions within the structure of the rapid screening framework presented in Chapter Three. These interventions can be investigated following initial catchment screening of strategic intervention zones, such as that demonstrated in Chapter Five. This responded to Objective Four by enabling a methodology to investigate intervention flood reduction performance.

Section 6.2 evaluated 144 intervention performance scenarios representing rainfall ranging from 1 to 48 hour duration. Key findings indicated:

- Short duration rainfall led to the highest flood damages, corroborating guidance indicating the importance of assessing these events when planning surface water management strategies.
- Performance of rainwater capture interventions is limited by total volume, whereas the performance of surface drainage interventions is limited by rainfall intensity. Rainwater capture interventions outperform surface drainage interventions in high intensity, short duration events, however limited capacities mean that surface drainage provides a more consistent performance to manage lower intensity, longer duration rainfall.
- Interventions which are able to continue functioning over extended timescales, such as drainage upgrades, are more effective at managing long duration events and appear more resilient to the extreme rainfall.

Section 6.3 developed a cost effectiveness metric, which was then applied as part of intervention performance analysis including design standard and extreme rainfall. The analysis evaluated 792 scenarios and identified a wide range of intervention performances, dependent on location and event characteristics. Key findings included:

- Rapid scenario screening can advance current best practice through including analysis of many scenarios within high level screening. This responds to limitations in current approaches such as narrow analysis of future uncertainties, for example evaluating strategies using a design storm for a fixed return period, and restricting permutations of novel surface water management interventions.

- Although centralised interventions provide benefit at smaller scales, catchment based strategies are required to substantially reduce flood extent and estimated annual damage costs across urban areas. Catchment wide high capacity rainwater capture measures (> 5000 l) generated the lowest EAD, indicating a saving of 76% versus a baseline scenario.
- Dispersed lower volume catchment wide interventions performed better than concentrated higher volume measures. Decentralised 1500 l rainwater capture tanks demonstrated a lower EAD (£0.2 million) than centralised, high volume tanks (£0.7 to £0.8 million). This supports future development of catchment wide surface water management.
- Intervention type, location and scale have significant impacts on cost effectiveness. Analysis of hundreds of scenarios indicates a wide range of cost effectiveness ratios for interventions, ranging from a £0.10 to £26.0 damage reduction per £1 spent, with the most cost effective interventions identified as high volume localised drainage interventions targeted in areas of intense flooding.
- Rainwater capture demonstrates lower but more consistent cost effectiveness across multiple scales and locations than surface drainage. Rainwater capture effectiveness ranges from £1.7 to £4.1, whereas surface drainage ranges from £6.6 to £26.0. This is attributed to surface based interventions demonstrating sensitivity to spatial variation of surface runoff. When considered in combination with the large reduction in EAD attributed to rainwater capture, this supports catchment scale decentralised application of rainwater capture, but highlights the advantages of strategically targeting complementary drainage interventions.
- Interventions which generate the lowest EAD do not necessarily correlate with the most cost effective application. The most cost effective intervention, strategically targeted surface drainage, has a cost effectiveness of £26.0 per £1 spent but only reduces EAD by £18 000 per

year. The intervention predicted to have the lowest EAD, catchment wide rainwater capture, reduces EAD by £607 000 but only demonstrates a cost effectiveness of £2.3 per £1 spent.

- Intervention performance during design standard rainfall is not indicative of resilience to extreme events. Interventions demonstrated performance tipping points where damage costs increased over a threshold. Rainwater capture based interventions were most susceptible to this due to storage capacities being exceeded during high magnitude events.

Overall, the wide performance variation highlights the advantages of evaluating the complex permutations of intervention type, scale and distribution through applying a rapid scenario screening framework to generate evidence and understanding prior to detailed design. The next chapter of this thesis will apply the framework in conjunction with catchment stakeholders to assess the utility of the approach in practice.

7. APPLICATION OF THE FRAMEWORK TO AN INTERNATIONAL CASE STUDY

This chapter responds to Objectives Four, ‘investigate the flood reduction performance of strategic and specific interventions’, Six, ‘verify application of the framework through practical application with catchment stakeholders’ and Seven ‘investigate the relationship between resilience and reliability of interventions’. Objective Four is met through evaluating the performance of surface water management interventions across a case study in Melbourne, Australia. Objective Six is met through collaboratively designing interventions and implementing the framework for decision support alongside partnership with catchment stakeholders. Objective Seven is met through examining intervention performance from design standard through to extreme rainfall events.

This chapter is structured through introducing a case study catchment and describing how this is represented using the modelling framework. The chapter then details the collaborative process applied to design a range of intervention strategies. Intervention performance is assessed through analysis of flood depth and velocity at key locations in the catchment. The discussion evaluates the performance of interventions and the lessons learnt applying this approach in practice.

This work presented here is drawn from the paper, ‘Is green infrastructure a viable strategy for managing urban flooding’, which is currently in review in the ‘Urban Water Journal’. Elements of the chapter have also been published in the proceedings of the 11th International Conference on Urban Drainage Modelling, which took place in Palermo, Italy, in September 2018.

7.1. Surface water flood management case study in Melbourne, Australia

7.1.1. Study catchment

The study area is a surface water catchment in the City Centre of Melbourne, Australia (Figure 7.1). The catchment is intensely urbanized and constitutes a major hub of commerce, entertainment and governmental function in Melbourne. The majority of buildings in the study area are high rise commercial and residential structures, several of which are recognised with national heritage status. Significant infrastructure sites are located in the catchment, including the City’s major railway station, local government offices and transport connections across the city. The study catchment is at the lowest point in the Melbourne CBD and is built on top of a natural creek which flows into the City's focal river (River Yarra). The north of the catchment includes urban parkland, national heritage sites, hospitals and the campus' of several large universities.

Surface water flooding in the catchment is of concern due to large historic floods in 1972 and 2010. During these events, intense rainfall across the catchment generated surface water runoff, in some cases deeper than 1 m, which flowed down the catchment's main street before ponding in front of the central railway station. These flows carried cars and manhole covers down the street and resulted in risk to life alongside damage to transport infrastructure, shops and properties.

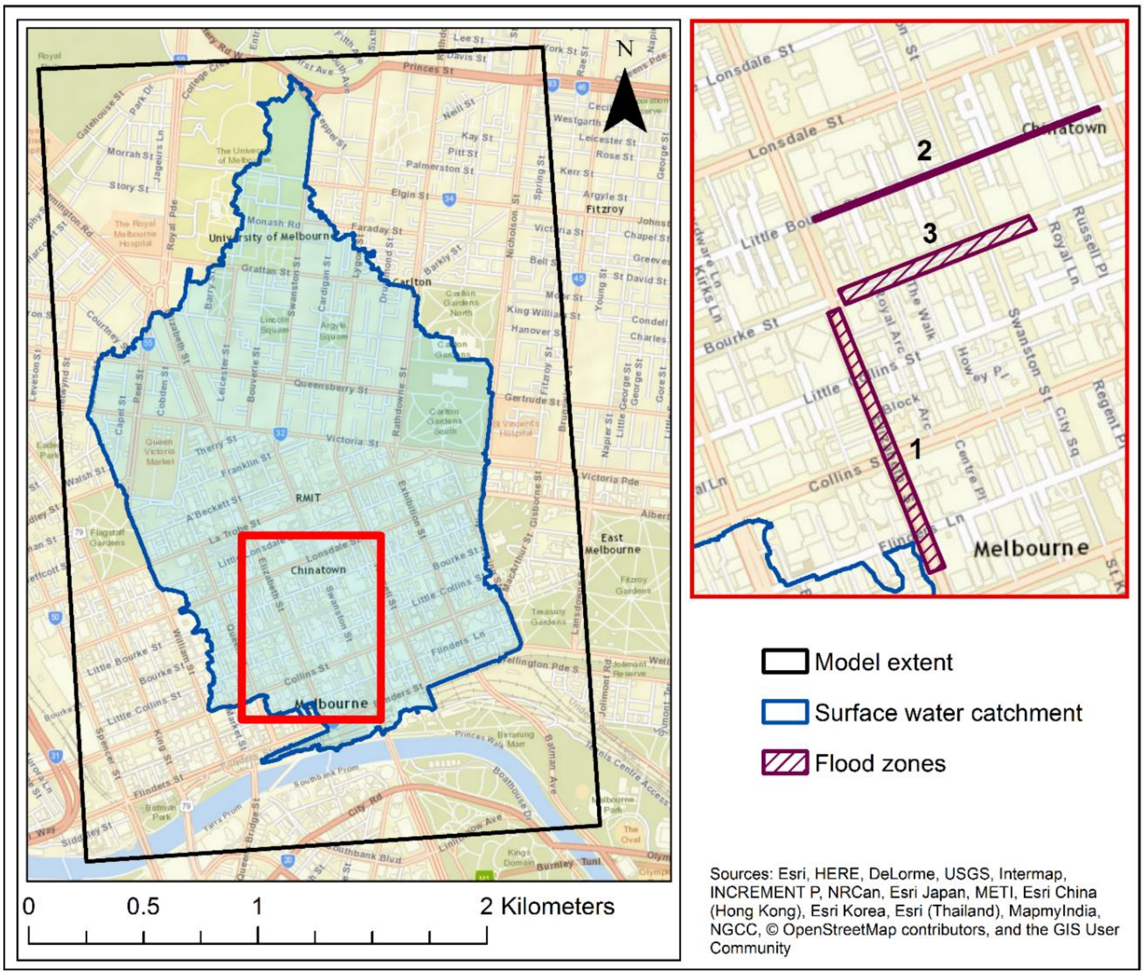

Figure 7.1: Identifying the modelled study catchment in Melbourne, Australia. Inset shows the extent the three flood zones evaluated during analysis

7.1.2. Catchment stakeholders

The research benefitted from engaged and motivated catchment stakeholders who represented decision makers in the study catchment. Partnership with experienced decision makers enabled research to explore the practical application and utility of developing and screening interventions using the framework. Stakeholders included within the project are presented in Table 7.1.

Table 7.1: Catchment stakeholders who helped design the interventions

Organisation	**Responsibility**
City of Melbourne	Local government with oversight of Melbourne CBD, including urban planning and flood management responsibilities. Their role in flood management is limited to catchments less than 60 ha, with larger catchments managed by the regional water authority.
Melbourne Water	Regional water authority responsible for water supply, sewage, and drainage.
Water Technology	Consultants responsible for identifying green infrastructure strategies for flood management in the CBD. Water Technology provided access to data and expertise from previous flood modelling studies in the catchment.
University of Melbourne	Researchers and major land owners in the catchment, responsible for joint R&D partnership to identify green infrastructure opportunities in the city centre.

Access to these stakeholders with their wide range of case study specific and general surface water management expertise provided the study with support to validate the flood modelling, identify opportunities for siting green infrastructure and review intervention performance modelling to steer application of the framework.

7.2. Methods

The viability of green infrastructure to manage urban flooding was tested using the rapid scenario screening framework presented in Chapter Three. A range of intervention strategies was devised in collaboration with the catchment's stakeholders. The performance of these interventions was measured through evaluating the resulting flood depth and velocity benefits of each strategy versus a baseline scenario. Details on each of these steps is described below.

7.2.1. Characterising study area

Catchment land use and elevation

Catchment elevation was represented using a 1 m resolution DEM provided by the City of Melbourne and Water Technology. The DEM comprised the entire Elizabeth Street surface water catchment as validated by Melbourne Water and the City of Melbourne.

Land use was characterised into 8 specifications, representing urban spaces, buildings, vegetation and transport infrastructure. Parameters associated with each of these classifications are outlined in Table 7.2. Some of these values differ from the ranges applied in other chapters of this thesis due to aligning modelling with previous studies undertaken by catchment stakeholders within this study area. This classification was made using digital land use mapping provided by Water Technology. The distribution of this classification is presented in Figure 7.2.

Table 7.2: Land use parameterisation in the study catchment

Land use type	**Roughness (Manning's n)**	**Infiltration rate (mm/ hour)***
Residential high density space	0.350	15
Buildings	0.400	15
Cemetery	0.100	1
Minimal vegetation	0.040	1
Moderate vegetation	0.060	1
Heavy vegetation	0.090	1
Roads and pavements	0.020	15
Railway	0.125	15

*Higher infiltration rates associated with impervious surfaces represent losses due to the underground surface drainage system.

Surface roughness was attributed based on specifications for Manning's n coefficient from available literature (Arcement Jr and Schneider, 1989; Hamill, 2001; Syme, 2008; XP Solutions, 2017; Butler et al., 2018). Buildings were represented using a high Manning's n coefficient to represent water being held up within structures (Syme, 2008). Infiltration was specified based on typical rates of clay soils from the region (City of Melbourne, 2018).

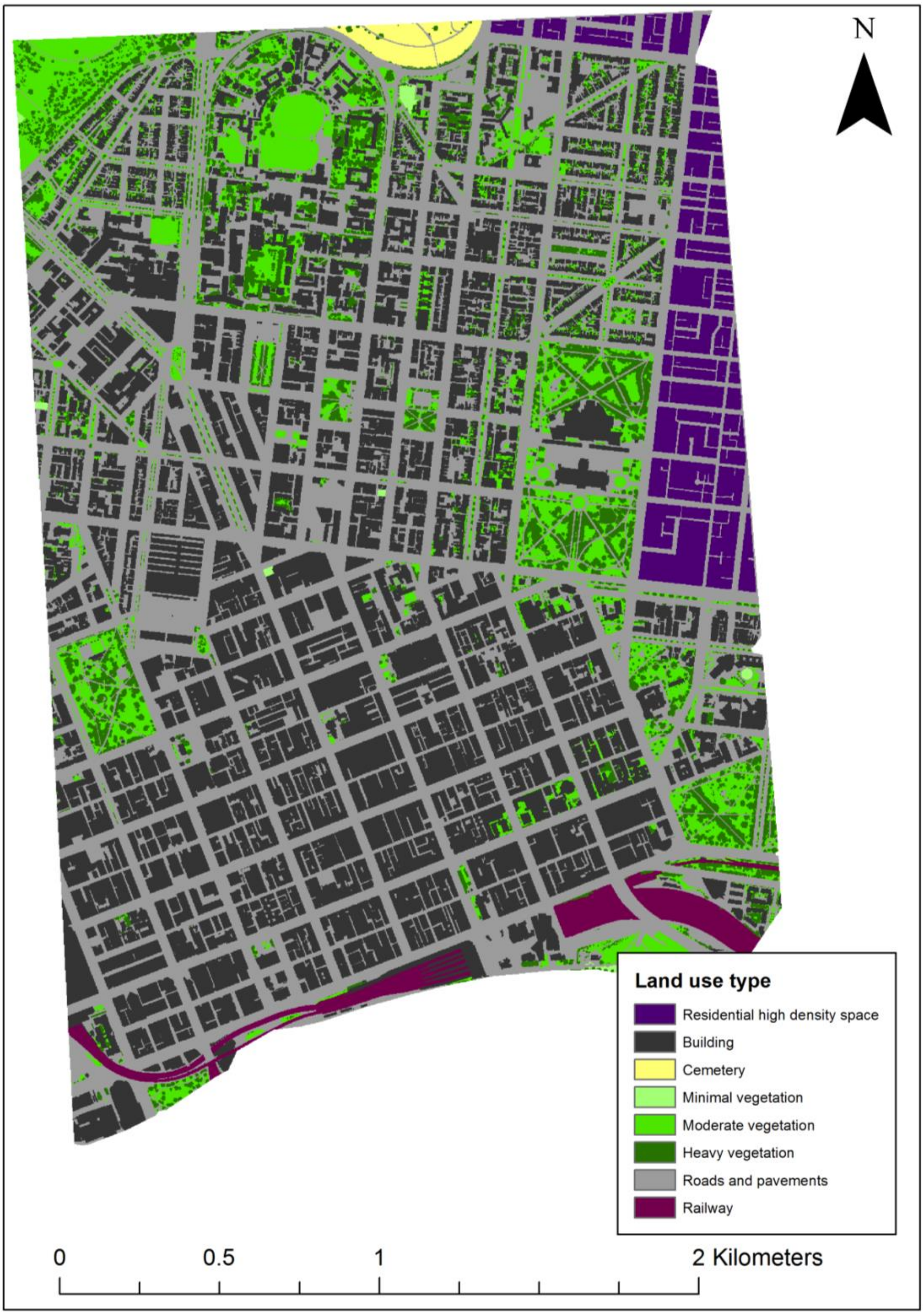

Figure 7.2: Classifying Melbourne City Centre land use into eight categories

Representing surface water drainage systems

The underground surface drainage system was represented through application of a constant cell output rate to areas assumed to be drained by the underground piped surface water system. This included residential high density space, buildings, railway, roads and pavements. A rate of 15 mm/ hour was applied to represent a system designed to convey the average intensity of an 18% AEP, 2 hour event, which was specified by stakeholders as a conservative estimate of the system's flow capacity.

Consideration of finer resolution representation of drainage through variation depending on the trunk capacity in sub-catchments was not possible for this assessment due to incomplete data regarding the pipe network in the area. Catchment stakeholders have previously undertaken extensive monitoring studies of the network, however significant data gaps still remain, making 1D modelling challenging and subject to many assumptions. As such, screening using an output rate provides an opportunity to examine the catchment without expensive data collection, and prioritise areas for future monitoring work.

This simplified representation of drainage systems facilitated fast screening of interventions and was able to simulate overland flooding as rain volume exceeds pipe capacity. As discussed in Chapter Four, it should be noted that this trade-off between model complexity and speed is only suitable for initial option comparison and not for detailed design of options.

Validation using records from the 1972 flood event

To validate the simple representation of the catchment's underground surface drainage system, predicted model outputs from a large rainfall event were compared against available observational evidence including photographs, flood histories and anecdotal information. The flood model was driven using the hyetograph of the 1972 event, one of the most intense on record, where rainfall intensities exceeded 100 mm/hour (Figure 7.3).

It should be noted that land cover parameters applied to represent the 1972 flood were not available, therefore current catchment land use was applied to model this event. The implication of this is that historic land use may influence the catchment runoff relative to current day values, particularly if the historic catchment was less urbanised that observed in current conditions. However, this is mitigated through the broad categorisation and parametrisation of urban space applied within this analysis not representing specific building types and the fact that analysis is focused on the historic CBD, which has been highly urbanised with the same historic park space for many years. This assumption would not be valid for the suburbs of Melbourne, within which urbanisation is rapidly expanding.

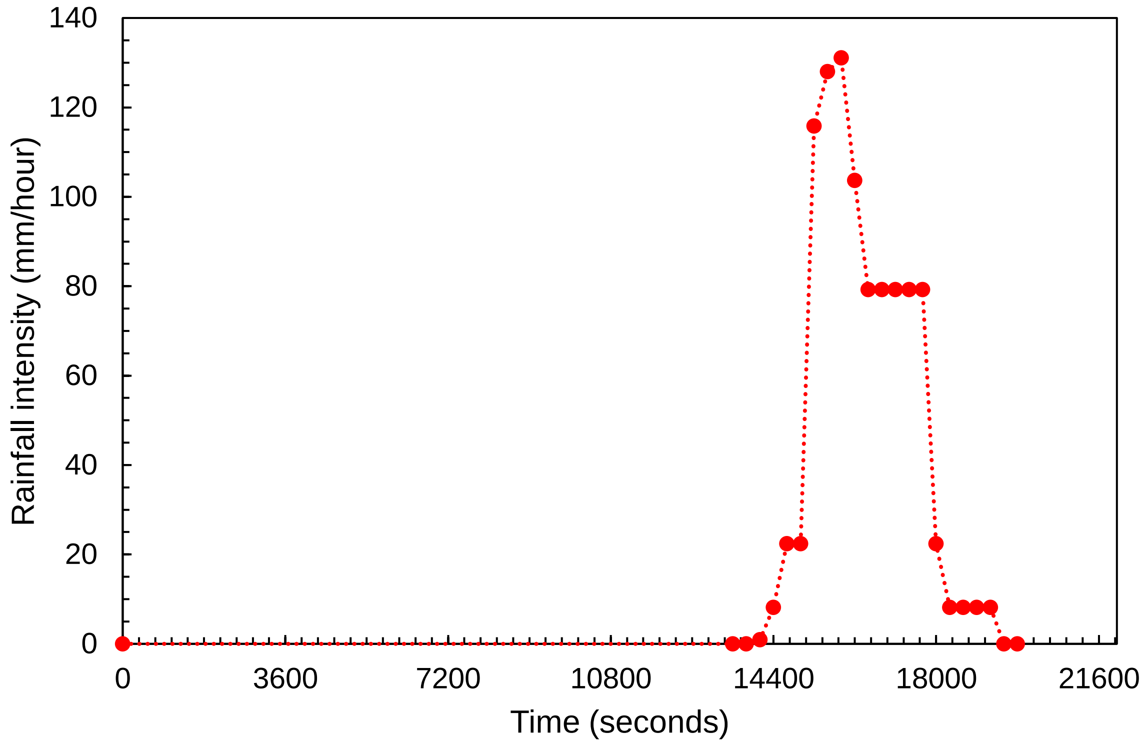

Figure 7.3: Hyetograph from the 1972 Melbourne flood event showing very high rainfall intensities over a one hour period

Flood maps indicating the peak depth and velocity of flooding during the event were evaluated with stakeholders during workshopping using comparison with photographs and reports from the 1972 event, as well as in house flood mapping derived from standard 1D-2D coupled models.

Figure 7.4 presents peak flood depth during the event. This predicts maximum flood depths of 1 to 1.5 m. Figure 7.5 supports this through presenting peak velocities during the flood, which was predicted to reach up to 2.5 m/s. These results compare well with photographs depicting flood waters exceeding the height of cars and the extent of flooding across the event. It is noted that full data regarding the depths of the actual flood event are not available due to the difficulty of practically monitoring high spatial and temporal resolution of surface water flows across urban streets (Neal et al., 2009), particularly in 1972; However, this high level validation builds confidence that the flow routes and approximate depths are acceptable to use for an initial and relative assessment, aimed at high level option comparison.

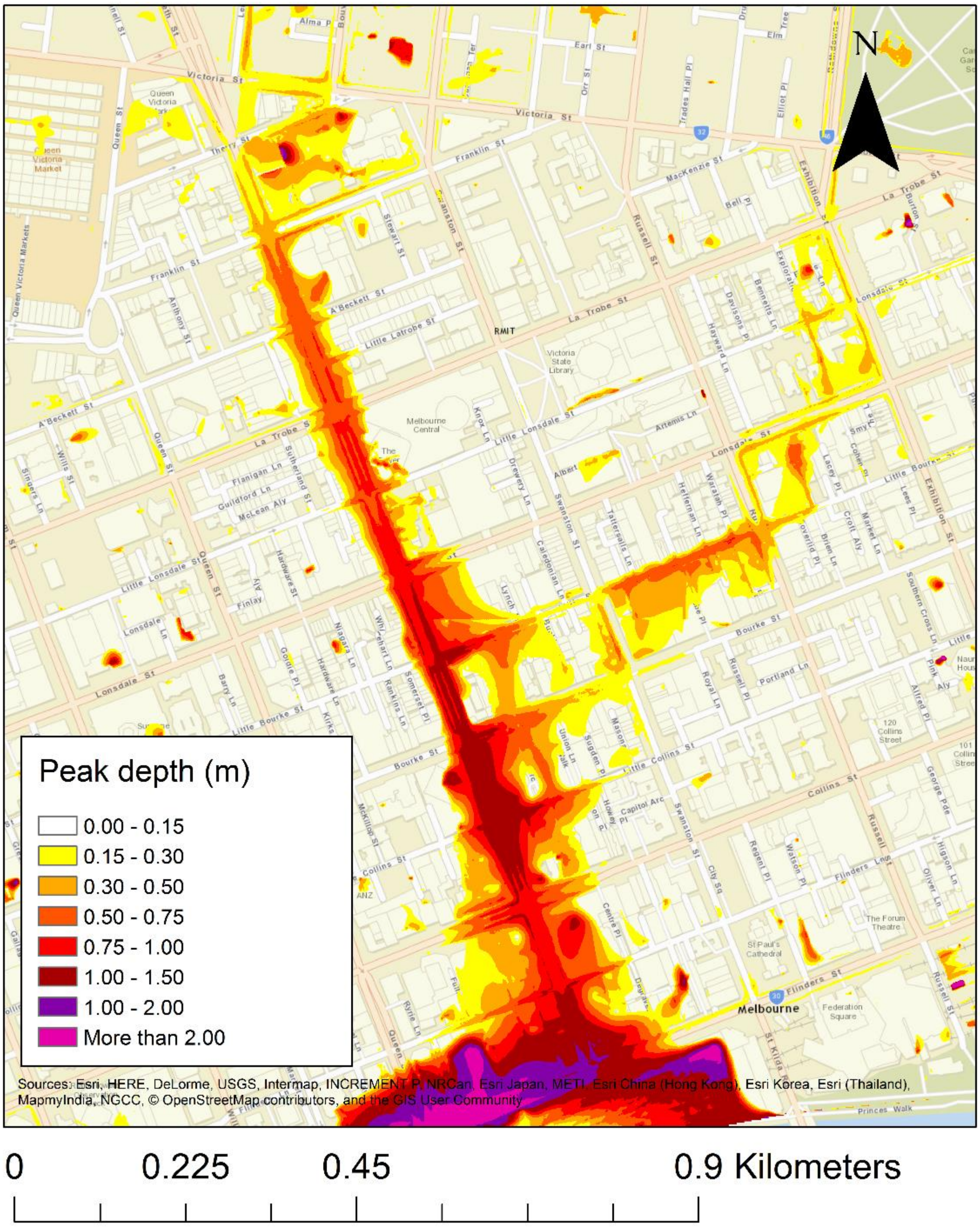

Figure 7.4: Flood map showing peak flooding on Elizabeth Street during the 1972 flood event

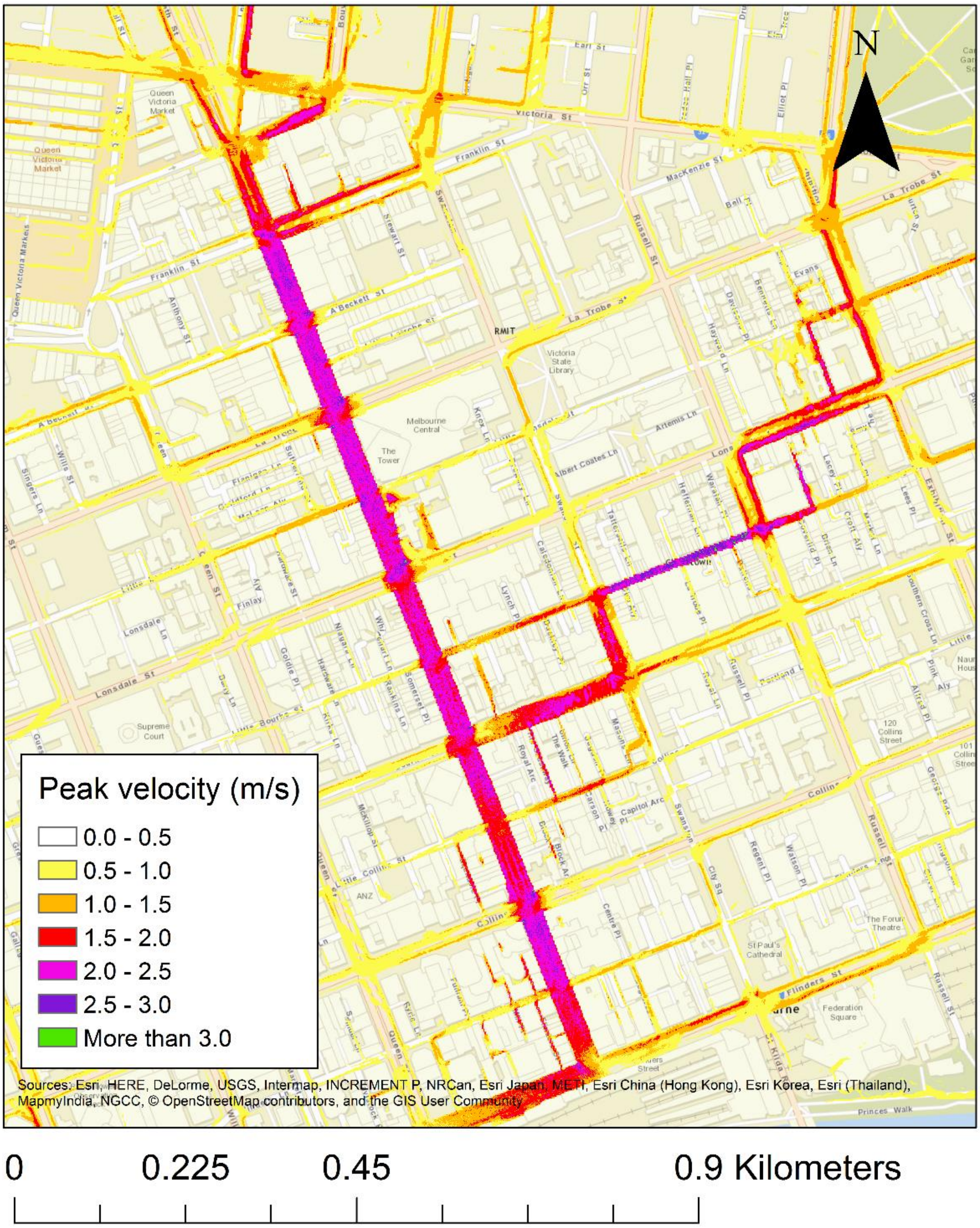

Figure 7.5: Map showing peak water velocity on Elizabeth Street during the 1972 flood event

Initial assessment of the 1972 flood event, combined with stakeholder expertise, identified three flood zones where flood risk was highest. These zones were used as the basis for examining intervention performance and are indicated in Figure 7.1. The intervention performance assessment is discussed in more detail in Section 7.2.4, later in this chapter.

Design rainfall generation and identification of a 'catchment critical event'

Engineers typically base designs for surface water management systems on a critical duration event where all upstream areas are contributing rainfall to a specific location. This approach is not possible when considering flood hazard across an entire catchment due to the spatial complexities and differing intensities generated using different rainfall profiles. The study overcame this restriction by taking advantage of fast simulation speeds to analyse a range of rainfall profiles and compare maximum flood depths to identify the catchment's critical event. A total of thirty rainfall events were assessed, including five different frequencies (18%, 10%, 5%, 2% and 1% AEP) across six different durations (30, 60, 120, 180, 270 and 480 minutes). Design rainfall was derived using methods outlined in the Australian Rainfall and Runoff guidance (Ball et al., 2016), which involve disaggregating burst rainfall depths (e.g. 50 mm in 30 min) into synthetic events. This analysis identified peak flooding during the one hour rainfall profile for all AEPs, therefore this was used for analysing intervention effectiveness.

Throughout the study the likelihood of each design rainfall is described in terms of AEP. This terminology was selected to address potential ambiguity for stakeholders unfamiliar with hydrology, due to misconceptions regarding implied periods of 'safety' between event recurrences when presenting findings using return periods, for example '1 in X year'. This is important for a study partially responding to the need for decision-support suited to stakeholders of varied training and professional backgrounds.

7.2.2. Representing interventions

Stakeholder engagement to design strategies

The research team collaborated with the catchment stakeholders to devise potential surface water management strategies which could be applied within the area. This involved a series of workshops with key organizational staff from a range of departments including engineering, environment, and planning. Workshop participants identified a range of strategies, which included specific sites for green infrastructure retrofit along with the possibility of broad implementation of interventions across the entire catchment.

Translation of the strategies into the rapid scenario screening framework was undertaken using the same approach as described in Sections 3.3 and 6.1. Parameterisation of specific interventions were adjusted in line with Australian literature and stakeholder expertise regarding the possibilities for application across the catchment. This section outlines each of the strategies examined by describing the adjustments to cell parameters and extent of the interventions effect relative to the total area of the catchment. Intervention map handouts, created as part of workshopping interventions with catchment stakeholders, are presented to indicate the scale and distribution of each strategy. A summary of all strategies is presented in Table 7.3.

Base case

The base case represented a business-as-usual scenario where the catchment was simulated as described in Section 7.2.1, with no interventions applied. This was used as a comparative baseline to measure the performance of each intervention strategy against. The effects of additional intervention scenarios are overlaid on top of this base case setup.

Green roofs applied across the entire catchment

This strategy represented retrofitting green roofs on all buildings within the catchment. Application across all roofs constituted 39% of the total catchment area (Figure 7.6). This was deemed to be an aspirational strategy, achievable in the medium to long term through changes in city level planning. Green roofs were modelled through editing input hyetographs to represent capturing rainfall within a cell. Each m^2 of a green roof captured 10 mm of rainfall, based on a conservative value from a review of the green roof literature (Mentens et al., 2006; Martin, 2008; Paudel, 2009; Stovin et al., 2012; Woods Ballard et al., 2015). Such levels of rainfall retention represented a conservative estimate, taking into account a range of typical values associated with varying roof slope, substrate storage capacities and climates.

Figure 7.6: Map showing the distribution of green roofs installed across all buildings in the model extent

Green roofs applied across the upper catchment

This strategy represented a smaller scale application of green roofs devised by the local government to investigate the effect of scale and capacity assumptions on green roof performance. The scenario only added green roofs to specific buildings in the upper catchment, representing 8% of the total catchment area (Figure 7.7). Rainfall capture with this intervention was limited to a more conservative 7.5 mm per cell.

Figure 7.7: Map showing the distribution of green roofs installed across the upper catchment

Rainwater capture tanks applied across the entire catchment

This strategy included installation of rainwater capture tanks across all buildings in the catchment. It was assumed that rainfall would be captured on roof surfaces and transmitted to storage tanks within each building. Application to all buildings in the catchment constituted 39% of the total catchment area (Figure 7.8).

A storage capacity of 2500 l per 100 m^2 of roof space was applied across all buildings. This value represents an estimate for rainwater capture supported by literature and common practice (Hamel and Fletcher, 2014; Burns et al., 2015b, 2015c; Schubert et al., 2017; Xu et al., 2018). It was assumed that the entire capacity was available for storage, attributed to real time control operation draining the tank in preparation of a predicted rainfall event (Campisano et al., 2017; Xu et al., 2018), and that the downpipe would not throttle water into the tank.

Figure 7.8: Map showing the distribution of rainwater capture tanks installed across all buildings in the model extent

Rain gardens distributed across the entire catchment

Rain gardens were applied across impermeable areas within the catchment. A 2 m^2 garden was specified to drain 100 m^2 of impervious area. The rainfall capture effect was represented through a uniform application of this capture capacity across all contributing cells, representing 43% of the total catchment area (Figure 7.9).

Rainfall is captured in rain-gardens through surface ponding and infiltration into porous filter media. Surface ponding was specified to a depth of 200 mm of water across the 2 m^2 surface (equating to 400 l of storage). The filter media was assumed to be 500 mm deep with a porosity of 0.4, but an effective porosity of 0.3 to account for likely antecedent soil moisture. Thus each rain-garden had a total storage capacity of 700 l (400 l at the surface and 300 l within the filter media). The filter media was lined and assumed to flow into the surface water sewer system, so no allowance for infiltration was included within the intervention. The value of 700 l was applied uniformly across all cells in the 100 m^2 catchment to generate a representative average capture effect of 7 mm of rainfall per m^2. It was assumed that this capacity was not limited by an infiltration rate into the substrate.

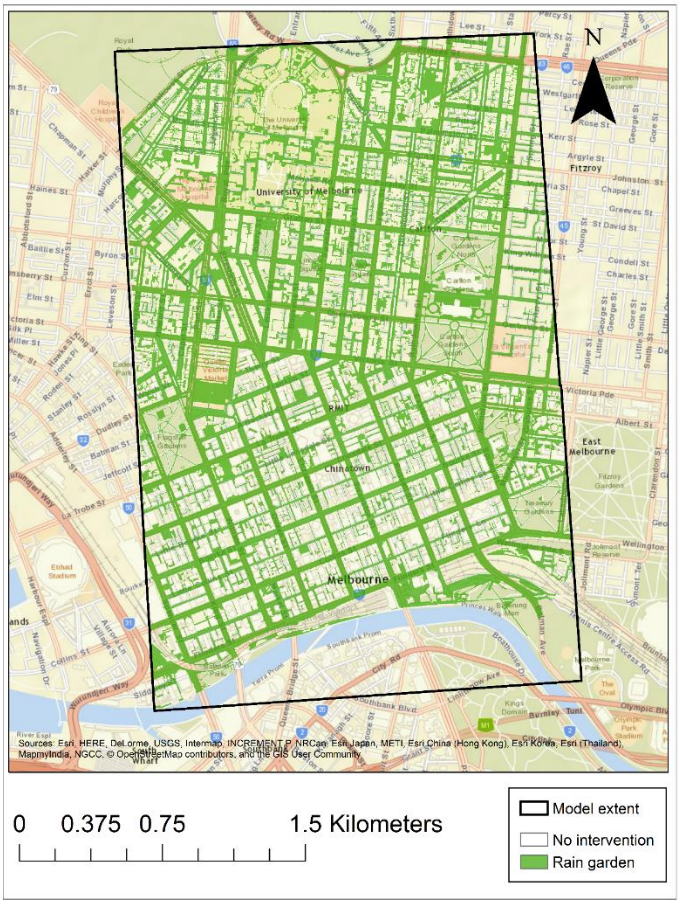

Figure 7.9: Map showing the distribution of rain gardens installed across the entire study area

Tree pits distributed across the upper catchment

The effect of locating 1000 tree pits across the upper catchment was modelled through assuming the storage capacity of a 1 m^2 tree pit to be 350 l, using the same assumptions as for rain gardens (above). This capacity was multiplied by 1000 and then applied as a uniform capture rate of 0.12 l.m^2 across the entire upper catchment, which constituted 43% of the total study area (Figure 7.10).

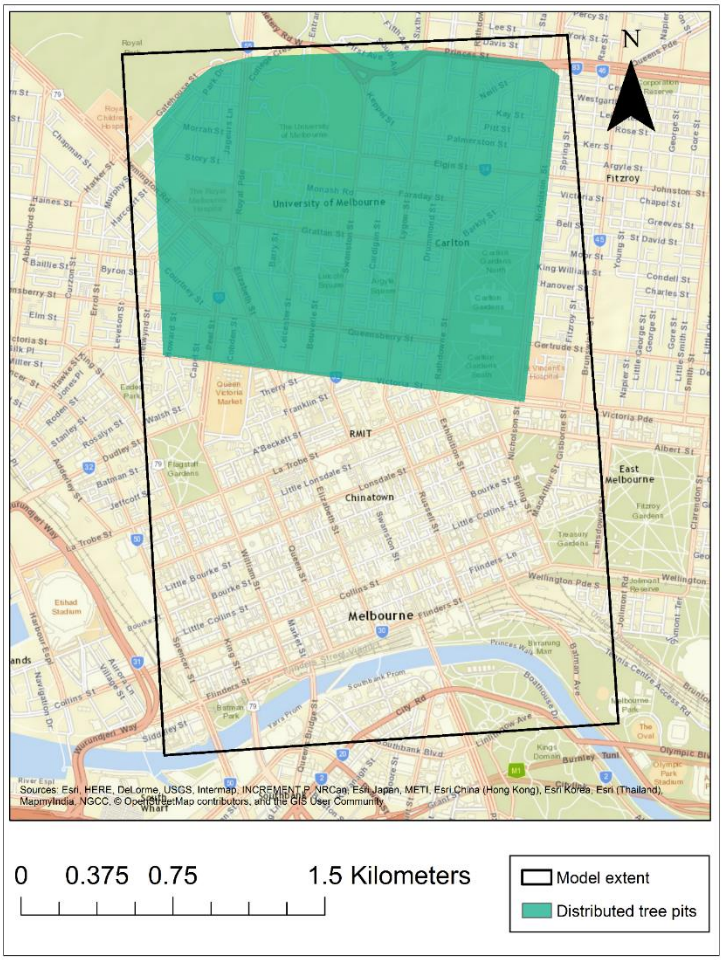

Figure 7.10: Map showing the distribution of tree pit effects across the upper catchment

Permeable paving distributed across the entire catchment

Permeable paving was modelled through assuming all impermeable areas within the catchment, constituting 44% of the study area, could runoff to a permeable paving unit (Figure 7.11). Typical paving structure comprises of 200 mm depth gravel with a porosity of 0.5 (Melbourne Water, 2005; Yong et al., 2011; Mohammadinia et al., 2018). This equates to 100 mm of interception across each 1 m^2 paving unit. It was assumed that 1 permeable paving unit served 10 m^2 of contributing area, therefore this effect was averaged and distributed evenly, represented through 10 mm captured from each contributing cell. An ongoing infiltration rate of 1 mm/ hour was based on typical permeability of the underlying clay soils in the catchment.

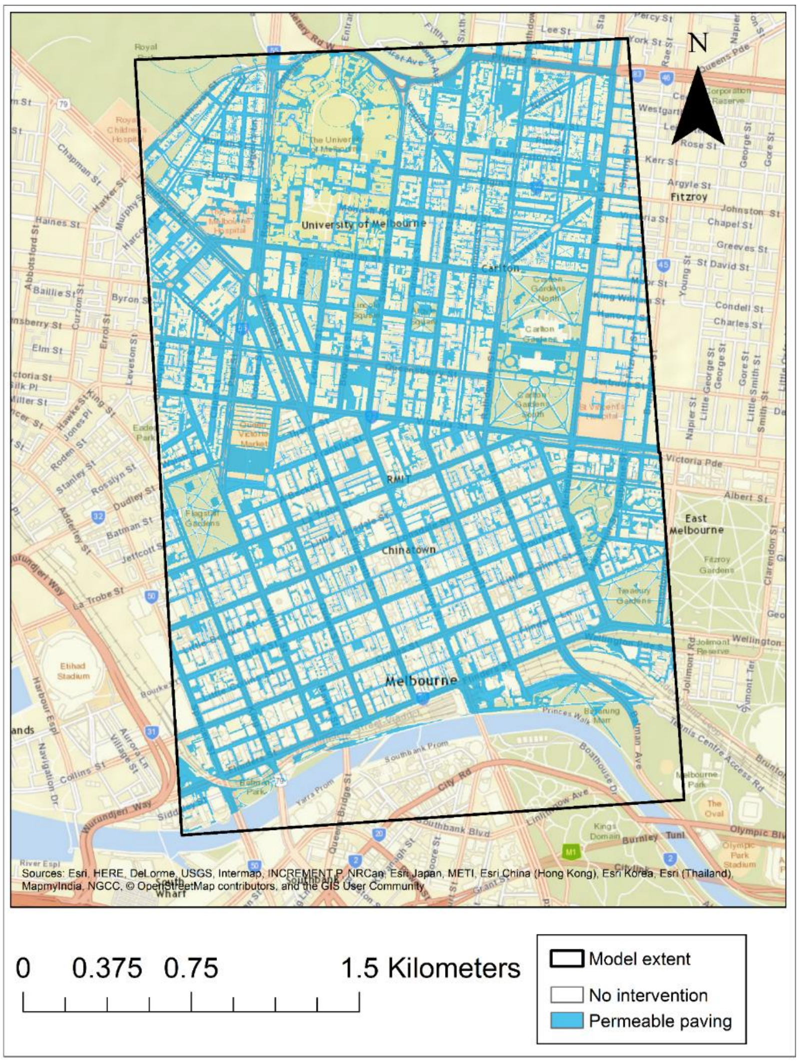

Figure 7.11: Map showing the distribution of permeable paving across the entire model extent

Enhanced storage across the upper catchment

The local government was interested to test the potential combined effect of large-scale distributed storage applied across the entire catchment. This scenario represented the possible effects of a collaboration between all property planners and owners (both public and private) in the catchment. It was assumed storage would be implemented through a wide application of sustainable drainage features, which may also offer additional benefits to the city.

Previous investigations by the local government found that a value of 4.5 l/m^2 could be achieved across the catchment and an enhanced storage capacity of 8.1 l/m^2 would be possible in strategically targeted areas of the upper catchment. This is a strategic development zone within the city where extensive works are currently being planned in collaboration with major landowners. No detail could be provided regarding locating sites at this early stage of option analysis, therefore this intervention was modelled through capturing rainfall landing within each cell of the catchment. Standard storage capacity was applied across 52% of the total catchment area and the enhanced capacity was applied across a further 16% of the catchment area (Figure 7.12).

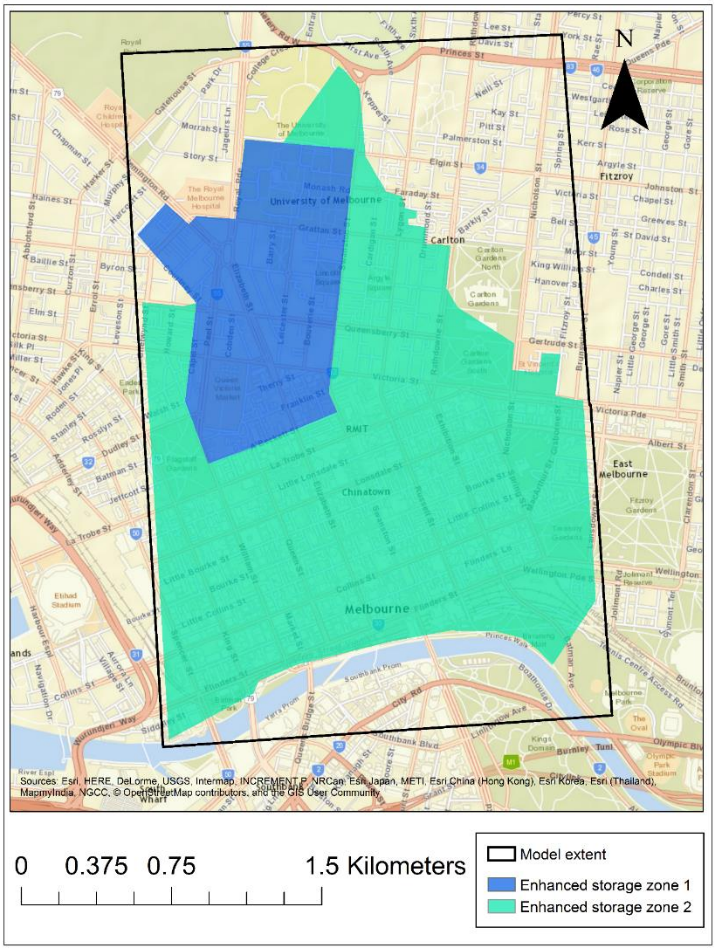

Figure 7.12: Map showing the distribution of enhanced catchment storage zones specified by catchment stakeholders

Storage at a major university campus

Further storage was considered across the City's major university campus in the north of the CBD. A total of 1.5 Ml of storage was proposed, achievable through intensive application of surface water control measures such as permeable pavement, rain-gardens and rainwater capture across the campus. Storage was implemented in the modelling framework using an assumption of uniform capacity across the entire campus, which constituted 6.6% of the total catchment area (Figure 7.13). The effect over this area was estimated to be 3.3 l/m^2.

Figure 7.13: Map showing the distribution of storage across upper catchment university campus

Storage at university buildings integrated within the city centre

Similar storage was proposed across the other university in the catchment. These buildings are located across multiple sites clustered in the north of the catchment. It was proposed that 1 Ml could be captured on roofs of campus buildings the northern subset and 0.5 Ml could be captured on roofs in the southern distribution. This was modelled through capture volumes of 46 l/m^2 in the north (0.3% of the total catchment area) and 10 l/m^2 in the south (0.7%) (Figure 7.14).

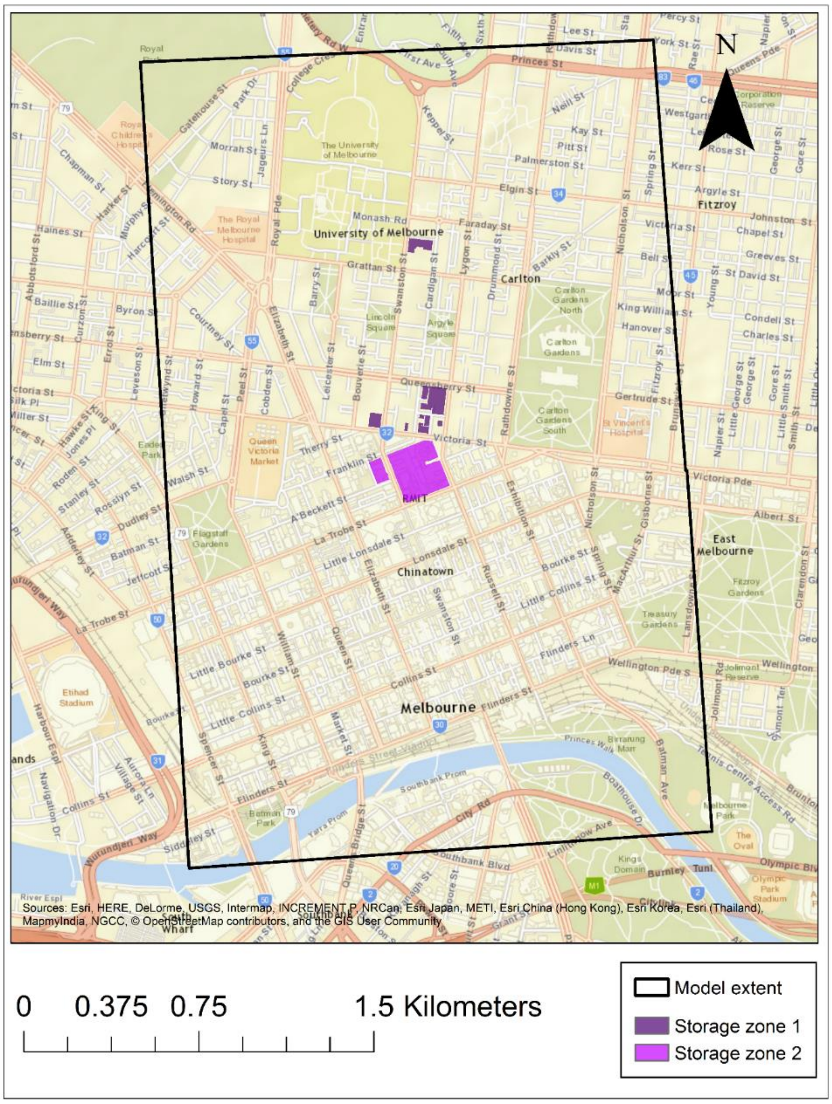

Figure 7.14: Map showing the distribution of storage installed at urban university campus buildings

Park expansion at city squares

The local government proposed expanding the pervious area of three major parks in the catchment. The parks were expanded across the roads to increase the park space up to 0.9% of the total catchment area (Figure 7.15). This strategy was modelled with the assumption that the rate of capture across the space would be equivalent to permeable paving applied uniformly across the area. Each 1 m^2 section of permeable paving was calculated to capture 100 l of rainfall, with a continuing rate of 1 mm/ hour infiltrating into the underlying clay soil. Roughness was attributed a uniform Manning's n coefficient of 0.040 to represent minimal vegetation coverage across the square.

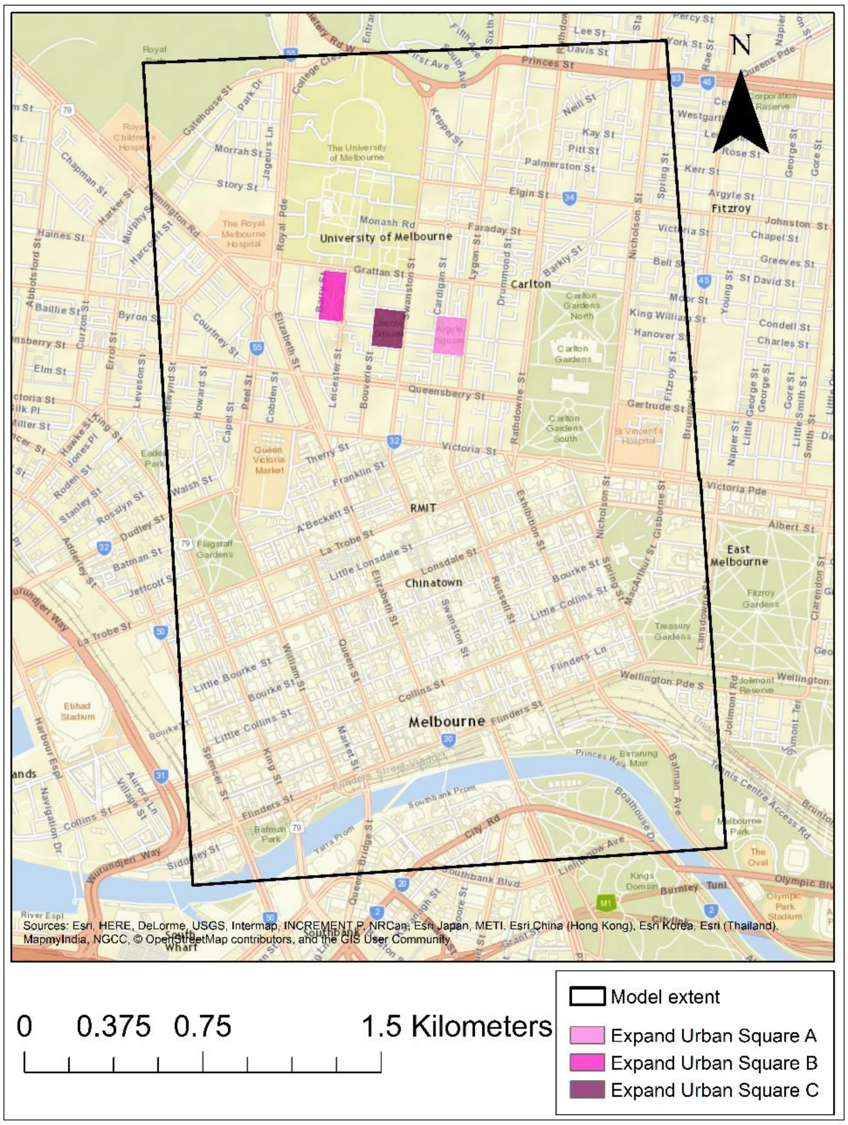

Figure 7.15: Map showing the distribution of park expansions at city squares

Increasing drainage capacity in the strategic sub-catchments

A grey intervention was proposed to increase the drainage capacity in two key areas through duplicating current pipes in two surface water drainage sub-catchments, representing 17% of the total catchment area (Figure 7.16). Limited data exists regarding the pipe capacities (Section 7.2.1), so a high level assumption was used to represent the scenario where the drainage rate used in the analysis were doubled to 30 mm/hour output per cell across both areas.

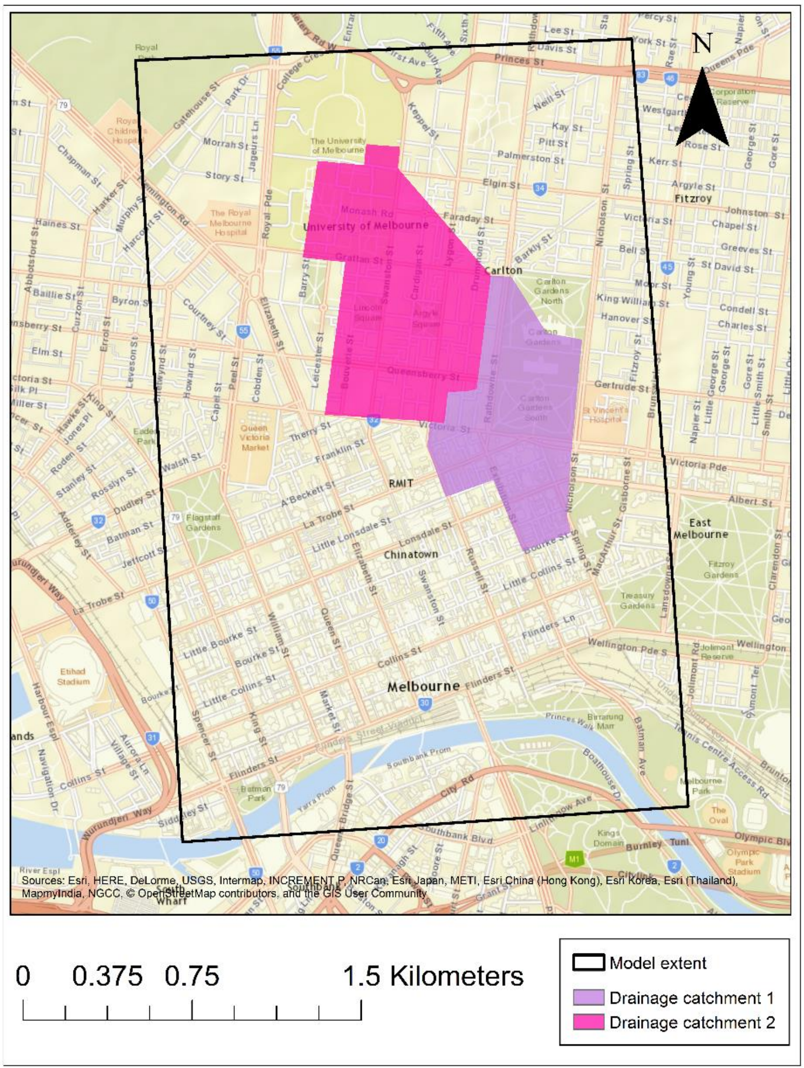

Figure 7.16: Map showing the distribution of increasing drainage capacity in the strategic sub catchments within the upper catchment

Summary of intervention effects

Table 7.3 presents a summary of all twelve intervention strategies and outlines cell parameter values applied to represent each intervention. The relative scale of strategies can be ascertained through the proportion of the study area which each approach is applied across.

Table 7.3: Intervention scenarios applied to the catchment

Intervention	**Distribution (% of study area)**	**Roughness (Manning's n)**	**Infiltration (mm/ hour)**	**Rainfall capture (l per cell)**
Base case model	n/a	Land use	Land use	n/a
Green roofs on all buildings	39.4	Land use	Land use	10.0
Green roofs in the upper catchment	7.8	Land use	Land use	7.5
Rainwater capture tanks on all buildings	39.4	Land use	Land use	25.0
Rain gardens across all impermeable spaces*	43.5	Land use	Land use	7.0
Tree pits in the upper catchment*	43.4	Land use	Land use	0.1
Permeable paving (catchment wide)*	43.5	Land use	Land use (plus 1.0)	10.0
Rainfall storage in the university campus	6.6	Land use	Land use	3.3
Rainfall storage in university buildings	0.3 (zone 1) 0.7 (zone 2)	Land use	Land use	46.0 (zone 1) 10.0 (zone 2)
Enhanced catchment storage	15.6 (zone 1) 52.2 (zone 2)	Land use	Land use	8.1 (zone 1) 4.5 (zone 2)
Increase park space	0.9	0.040	1.0	100.0
Pipe duplication	17.1	Land use	30.0	n/a

* Intervention capture rates averaged over all cells within the specified area.

7.2.3. Simulating scenarios

In total, 60 scenarios were examined, which consisted of the twelve intervention strategies (Section 7.2.2) applied across the five rainfall magnitudes (Section 7.2.1). Simulation was undertaken using CADDIES run on an ‘Nvidia Tesla K20c’. Average simulation time for each scenario was 2.12 hours at a minimum model time step of 0.01 s.

7.2.4. Analysing intervention performance

Areas of investigation

Intervention performance was assessed in three zones across the catchment (Figure 7.1). These were selected through correlating flood ponding and conveyance routes during the base case scenario with expertise and observations from catchment stakeholders. Each zone corresponded to a major road within the catchment and were corroborated as PFS through historical flood observations (Section 7.2.1).

Using peak flood depth and velocity as flood hazard metrics

Performance of interventions was assessed using peak flood depths and velocities in cells within each flood zone identified in Figure 7.1. Peak values were chosen for analysis as these represent the worst case flooding and allow one image to effectively communicate overall flood hazard to stakeholders. This approach is advantageous for quick comparison of interventions effects on flooding to all buildings, infrastructure and features. Storing one output per cell per simulation also greatly reduces the memory requirements for large numbers of simulations.

Data limitations and commercial sensitivities meant that detailed flood damage cost curves were unavailable for this case study. However, the three flood zones evaluated consist of high density urban commercial buildings of relatively consistent value and type, resulting in a relatively homogenous level of vulnerability across each zone. Therefore, spatial variation in flooding within these regions is unlikely to have a significant impact on flood damage costs, therefore it is deemed acceptable for performance assessment to be based on analysis of peak depth and velocity in this instance.

7.3. Results

This section evaluates the effects of each intervention strategy across the 18%, 10%, 5%, 2% and 1% AEP events simulated during the analysis. Performance of each intervention across each of the three flood zones was undertaken through comparing the distribution of peak flood and peak velocity values from each cell in the zones (Figure 7.1). The distributions are presented as a box plot which shows the mean, 25% and 75% percentiles as a box, bounded by the full range of data.

Surface water flooding was observed during the one hour rainfall event in all scenarios and at all AEPs. Analysis of the distribution of peak flood depths per simulation indicates that the deepest mean and maximum peak depth and velocity are observed in the base case scenario for all zones during all AEPs, demonstrating that no intervention had a negative effect on flooding within the study area.

7.3.1. Intervention performance during the 18% AEP event

Performance during the 18% AEP event is shown in Figure 7.17 (peak depth) and Figure 7.18 (peak velocity). Flood depths and velocities within this event are the lowest across all the events studied. All scenarios, including the baseline demonstrate mean peak flood depths of less than 20 cm across all three flood zones. All scenarios in all zones also demonstrate cells with no flooding. The base scenario demonstrates the deepest peak flooding, at up to 40 cm in Flood Zone One.

Intervention performance shows the largest reduction in flood depth in Flood Zone One. Differences in other flood zones are less noticeable, with only minor changes (less than 5 cm) between the base case mean peak flood depth and any other intervention.

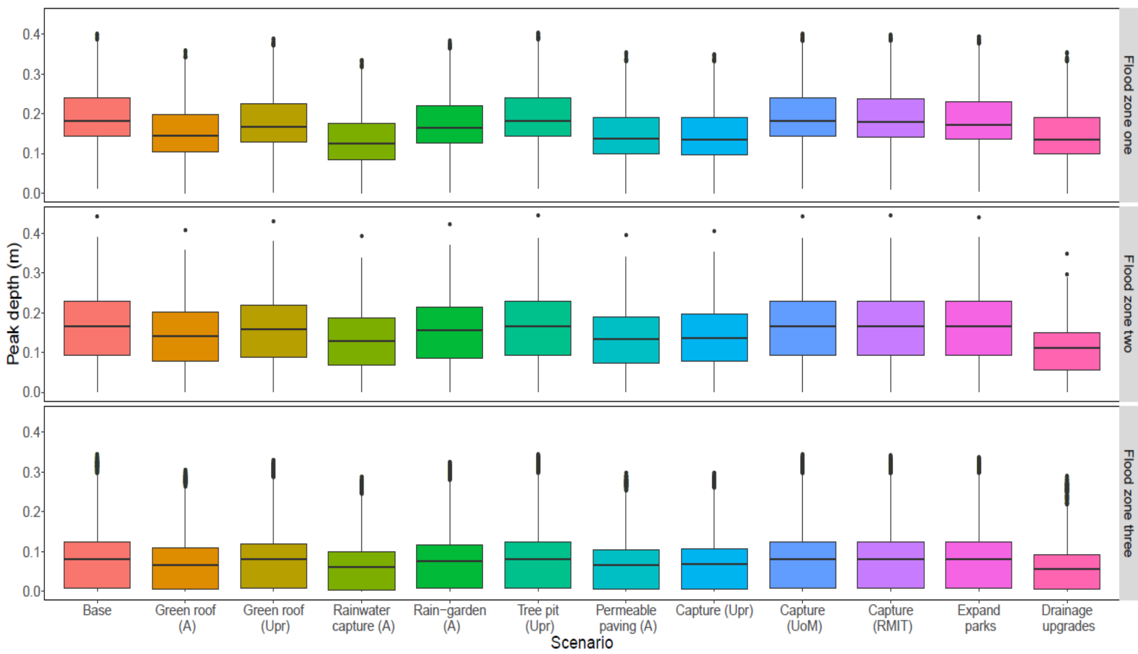

Figure 7.17: Comparison of peak flood depth distribution for all flood zones in the 18% AEP event

The distribution of peak velocity in each cell across each flood zone demonstrates a similar pattern to peak depths, with the base case demonstrating the fastest velocities. As with peak depth, the largest effect of interventions is demonstrated in Flood Zone One, with only minor variation apparent in Flood Zone Two and Three.

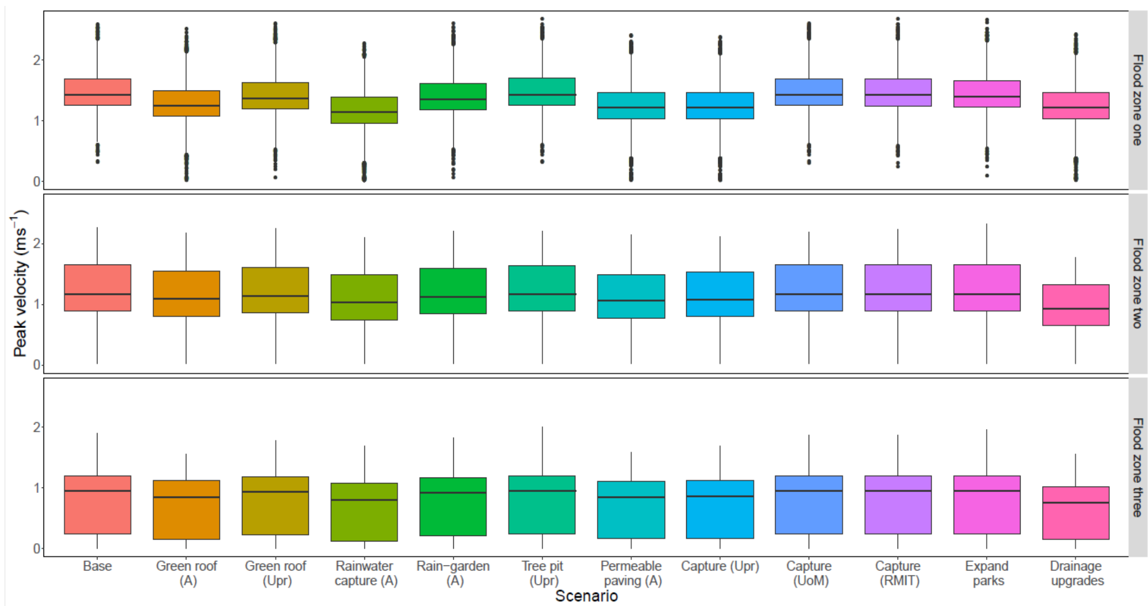

Figure 7.18: Comparison of peak runoff velocity distribution for all flood zones in the 18% AEP event

7.3.2. *Intervention performance during the 10% AEP event*

Peak flood depth (Figure 7.19) and peak flood velocity (Figure 7.20) during the 10% AEP event demonstrate a relatively similar pattern to results from the 18% AEP events. The most notable performance variation is observed in Flood Zone One with much smaller variation in performance across Flood Zones Two and Three. The largest difference is that the base scenario in Flood Zone One has no cells with zero flood depth, however four strategies (rainwater harvesting, permeable paving, the strategic capture zone and drainage upgrades) all generate areas with zero flood depth.

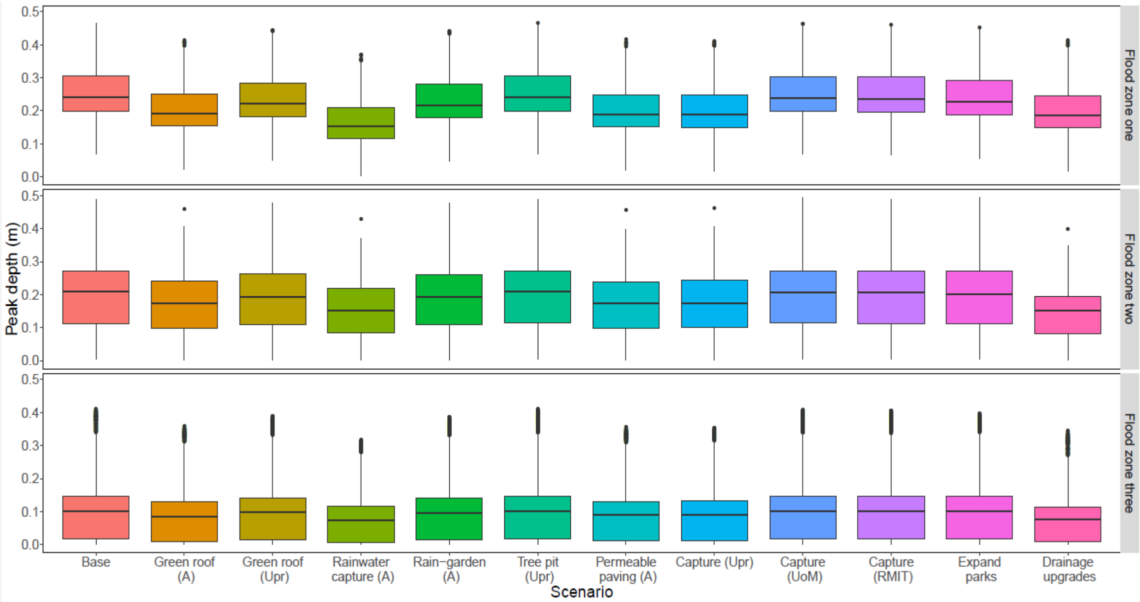

Figure 7.19: Comparison of peak flood depth distribution for all flood zones in the 10% AEP event

Differences in the distribution of values attributable to intervention performance are more pronounced for this event. In particular for Flood Zone One, where rainwater capture is the most effective intervention for reducing flood depths and velocities. Drainage upgrades demonstrate the largest difference in Flood Zone Two, attributable to the upgrades removing runoff from the catchments which directly flow into this zone.

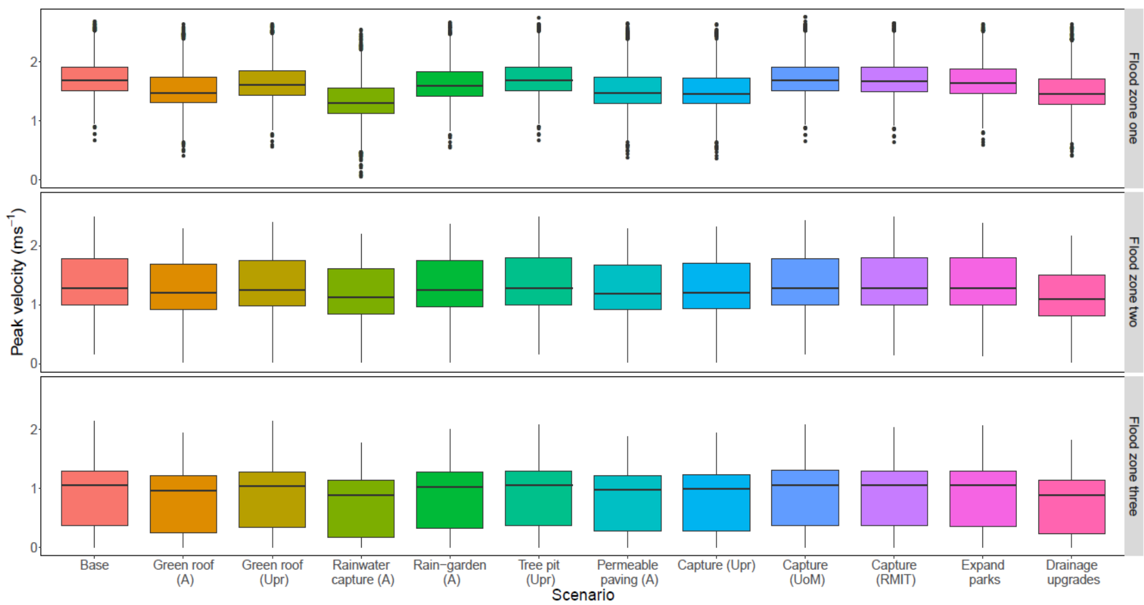

Figure 7.20: Comparison of peak runoff velocity distribution for all flood zones in the 10% AEP event

7.3.3. Intervention performance during the 5% AEP event

The majority of interventions reduced flood depth across the three zones during the 5% AEP rainfall event. Figure 7.21 shows the distribution of peak flood depth across all cells for all strategies in each of the three zones. The maximum mean peak depth in the base case scenario, approximately 0.38 m, was observed in Flood Zone One. The deepest peak flood depth, over 0.61 m, was also identified in this zone. Zone One is located at the furthest downstream point of the catchment and will have the largest contributing area. Flooding across the other zones was on average shallower, due to smaller contributing areas.

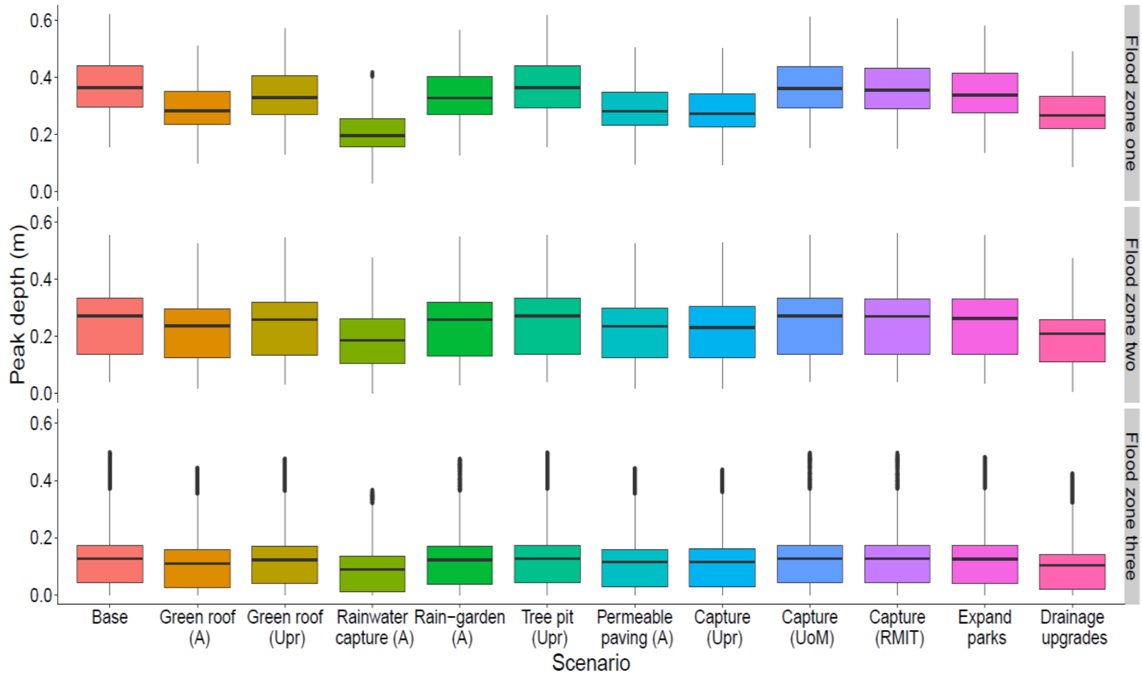

Figure 7.21: Comparison of peak flood depth distribution for all flood zones in the 5% AEP event

The largest reduction in peak depth was observed in Zone One, where several strategies reduced the mean peak flood depth by 25 to 50%. The most effective interventions were those applied across large areas of the catchment, including rainwater capture, green roofs, permeable paving and the introduction of enhanced storage in the upper catchment. Rainwater capture was consistently the most effective intervention, reducing the mean peak flood depth to less than 0.2 m in Zone One and Two, and to less than 0.1 m in Zone Three. The strategy of increasing drainage capacities also demonstrates flood reduction. Tree pits and capturing runoff at the city's universities demonstrated a negligible reduction in flood depth versus the base scenario. It is suggested that, in this instance, these interventions only capture enough rainfall to delay the timing of the flood peak, rather than reduce its magnitude.

Interventions demonstrate similar performance rankings in each zone. The most effective performances were observed in Zone One with similar, albeit a smaller range of, values exhibited in the other study areas. No interventions completely eliminated flooding, however rainwater capture was predicted to remove all flooding from certain cells in Flood Zone Two, a benefit which is not present in the base scenario. All interventions demonstrated cells with no flooding in Flood Zone Three.

Figure 7.22 shows the intervention effects on peak flood velocity in each of the three flood zones. The distribution of values has a similar, albeit less pronounced, pattern to the change in peak flood depths. The most effective intervention was catchment scale rainwater capture, however even in Flood Zone One, where the effect is most noticeable, this only reduced mean peak velocities from 2 m/s to 1.6 m/s.

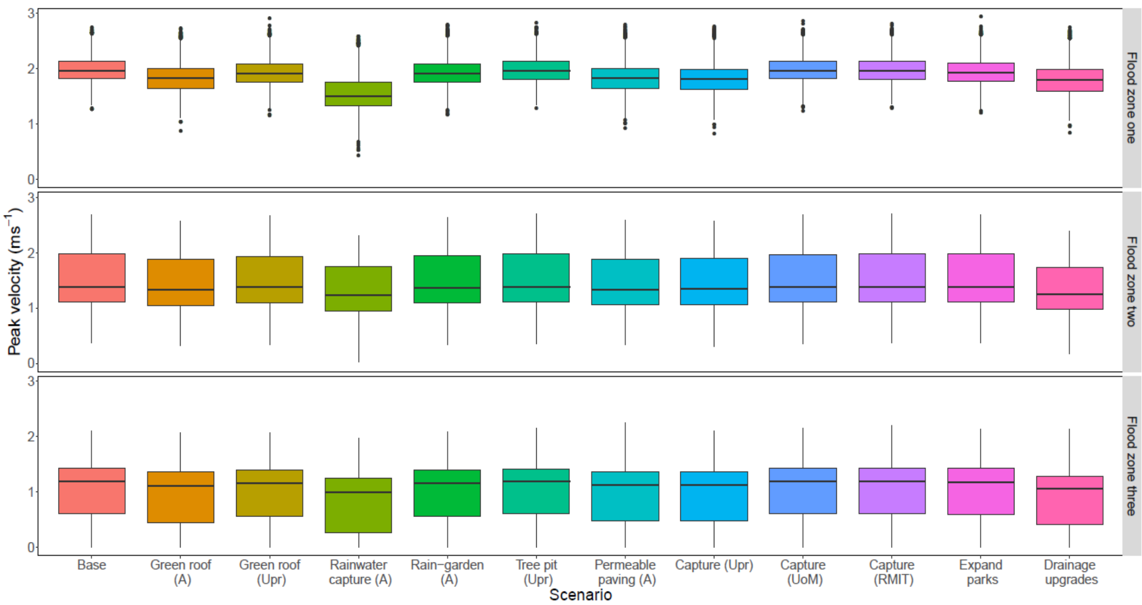

Figure 7.22: Comparison of peak runoff velocity distribution for all flood zones in the 5% AEP event

7.3.4. Intervention performance during the 2% AEP event

Figure 7.23 presents peak flood depths during the 2% AEP event. The distribution of peak depths is deeper than the more frequent AEP events, but still follows the same pattern; with the most noticeable intervention performances observed in Flood Zone One. The most effective interventions in this area are the catchment wide strategies, with rainwater capture reducing flooding by the largest value in all flood zones.

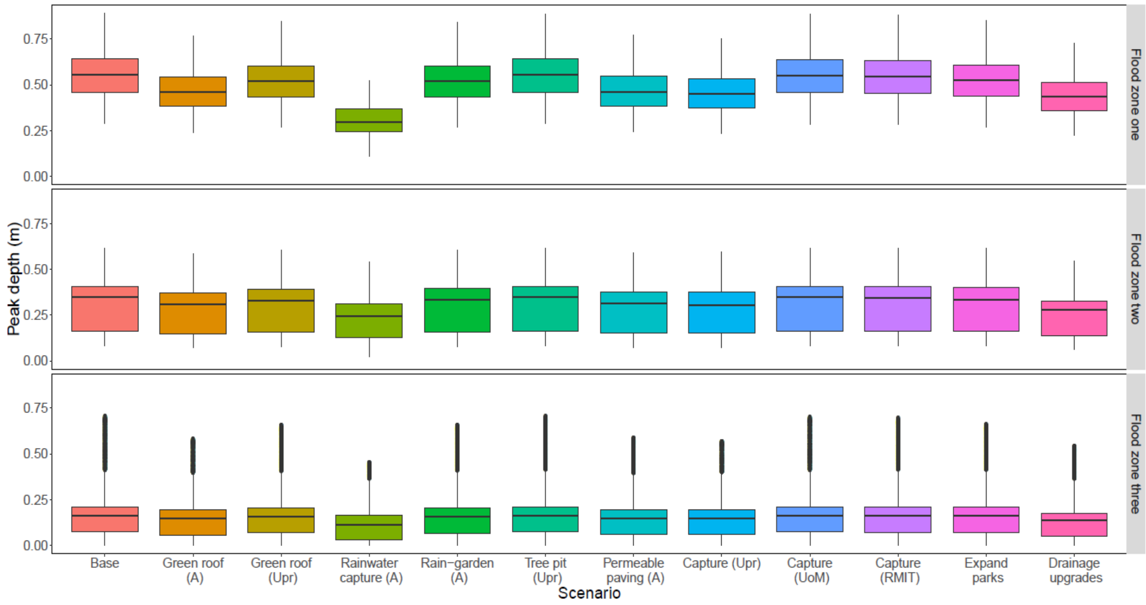

Figure 7.23: Comparison of peak flood depth distribution for all flood zones in the 2% AEP event

Interventions have a less pronounced effect on the peak flood velocity, which remains relatively consistent across all intervention strategies. The best performing intervention, rainwater capture, reduced the mean by approximately 0.2 m/s across all scenarios. Other interventions demonstrated negligible effects on peak velocities.

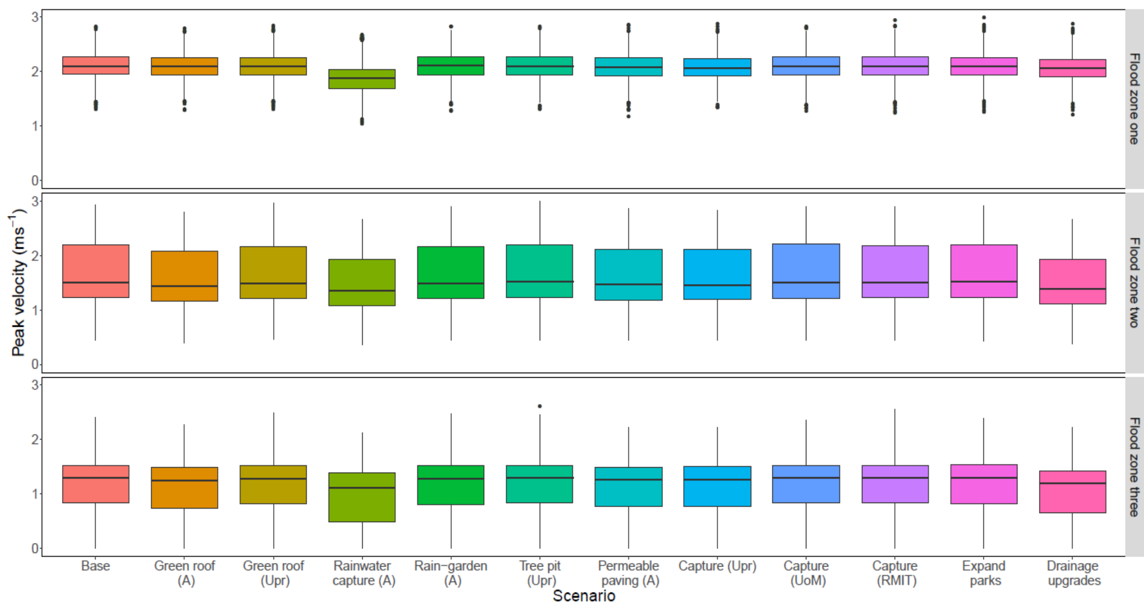

Figure 7.24: Comparison of peak runoff velocity distribution for all flood zones in the 2% AEP event

7.3.5. Intervention performance during the 1% AEP event

The deepest flooding across all scenarios was predicted to occur during the 1% AEP event (Figure 7.25). Intervention performance generally approached the base case scenario, with less variation in performance relative to lower return periods. Ranking of interventions remained consistent, but the degree of variation between strategies was less noticeable.

No interventions worsen flood depths, however tree pits and capturing 1.5 Ml at the University of Melbourne and RMIT campuses show negligible differences to the base case across all zones. Other strategies based on a defined capture volume, such as green roofs in the upper catchment, rain gardens and park expansion also have little impact on flood depths during the most intense rainfall event. Limited performance is attributable to rainfall exceeding capture capacities during the event and therefore interventions leading to a delay, rather than reduction, in peak runoff rates. This is similar to findings regarding intervention performance tipping points discussed in Chapter Six. This effect is partly mitigated by strategies applied across the whole catchment or large areas, such as rainwater capture, green roofs, enhanced catchment storage and permeable paving which capture sufficient volume to reduce peak depths. Strategically targeted and intensive options such as increasing drainage capacities also demonstrate an improvement versus the base case.

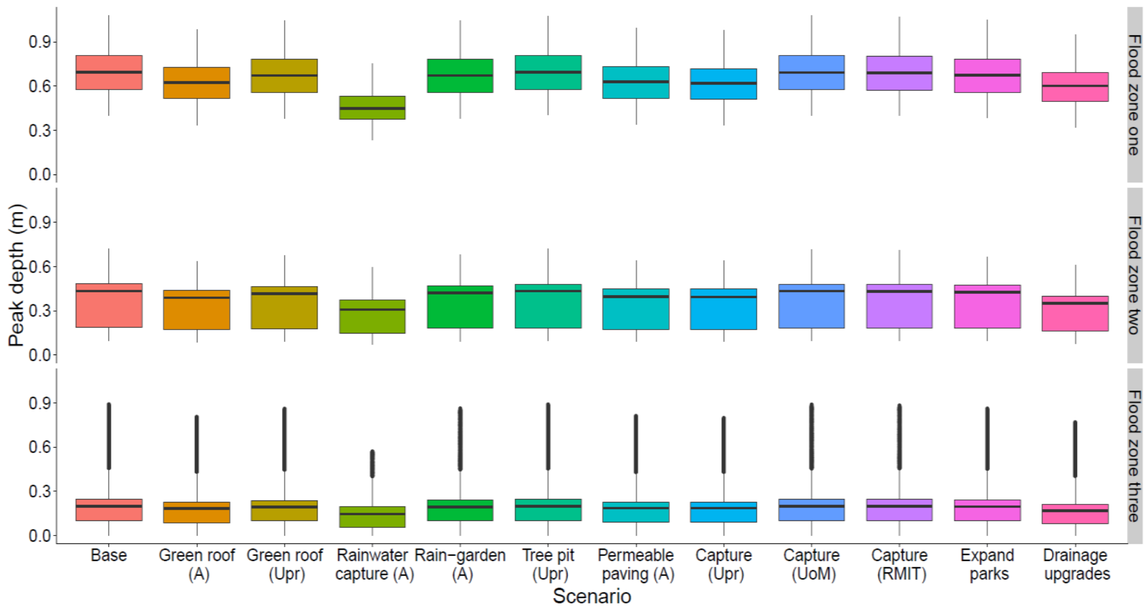

Figure 7.25: Comparison of peak flood depth distribution for all flood zones in the 1% AEP event

Interventions demonstrate a limited effect on flooding in Zones Two and Three. It is suggested that this is due to these zones receiving runoff from smaller areas of the catchment, and consequently reach a time of concentration faster after rainfall volume exceeds capture capacities.

Analysis of depth distribution is a useful tool for identifying strategic performance trends during decision support. However, it is also important for decision makers to consider the location of flooding, in order to conceptualise and manage risk. Figure 7.26 presents a visualisation of peak depths in the base case scenario during the 1% AEP event (panel A) with a comparison of the reduction in flooding created by application of the most effective intervention, rainwater capture (panel B). The analysis created maps like this for each of the scenarios and rainfall events investigated, however these have been omitted from this results section to facilitate a concise analysis. It is important to note that these maps are available, as they formed a useful tool for visualising and exploring the effectiveness of interventions during workshops with the city council and are discussed later in the chapter.

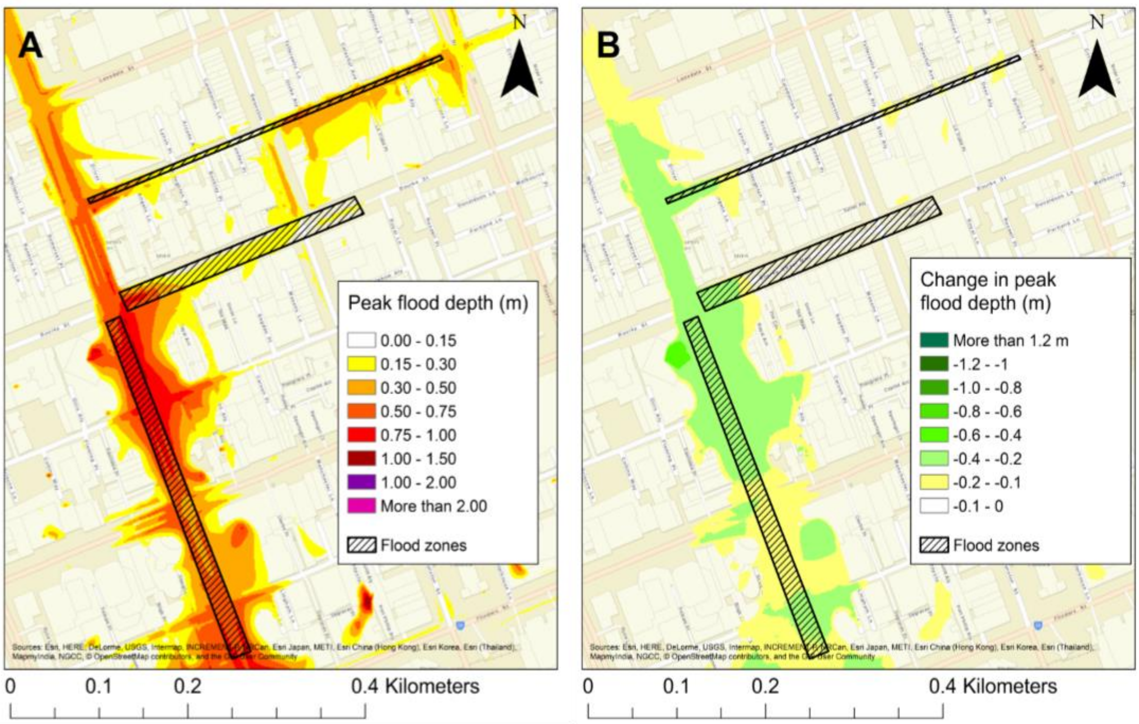

Figure 7.26: Effect of catchment scale rainwater capture on peak flooding during the 1% AEP event. (A) Base case and (B) Difference in peak floods using rainwater capture

This figure shows that rainwater capture reduces peak flood depths by 0.2 to 0.6 m across the entire width of Zone One along approximately 300 m of the transect. This demonstrates a substantial public safety and damage reduction improvement versus the base case scenario.

Figure 7.27 indicates that interventions only have a marginal impact on reducing peak flood velocities in the study area. This is attributed to the interventions delaying the timing, rather than reducing the magnitude of peak runoff during the event, and is similar to the observations regarding the minor changes in peak flood depths predicted during this scenario. The velocity is also controlled by the topography and roughness of the road surface, which has not been altered using these interventions.

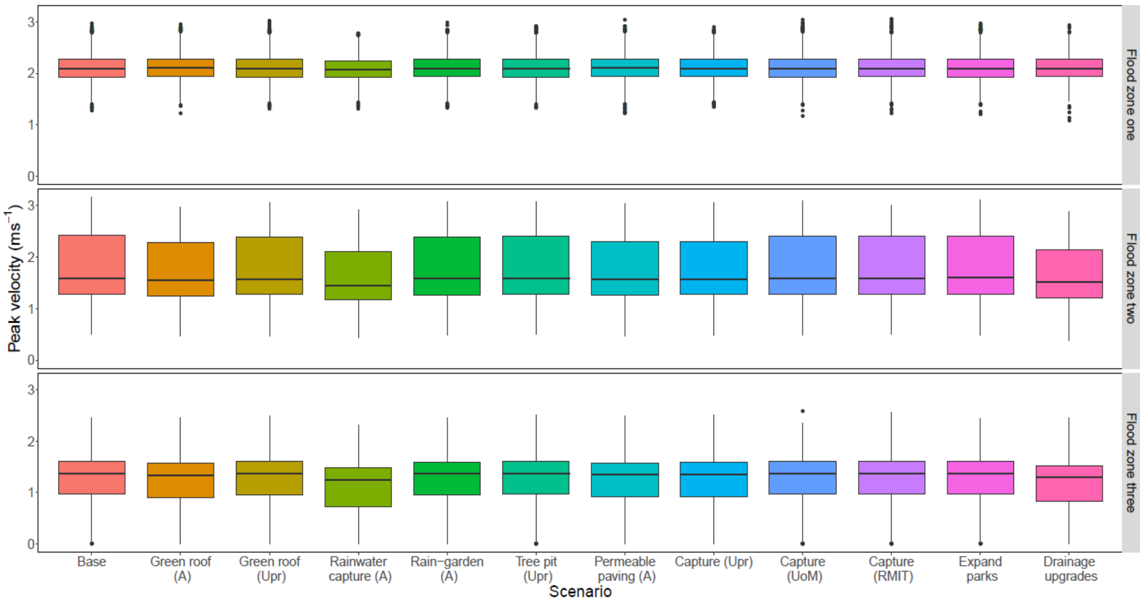

Figure 7.27: Comparison of peak runoff velocity distribution for all flood zones in the 1% AEP event

7.4. Discussion

7.4.1. Green infrastructure to manage urban surface water flooding

Many interventions reduced peak flood depths and velocities, and no strategy performed more poorly than the base case approach in any scenario. Of particular note were interventions applied as part of a catchment-wide strategy, which were predicted to achieve the largest reductions in flood depth and velocity in all scenarios.

The apparent limited performance of certain localised or smaller scale interventions, even when capturing a high volume) should be considered in the context of an incremental development step towards a larger catchment management approach. In addition to this, although local strategies may not reduce the peak depth in the downstream catchment, their local effect and delaying flood peaks may facilitate more effective movement around areas of severe flooding which are consistent with large rainfall events. Additionally, such interventions are likely to be effective for reducing localised nuisance flooding in strategically targeted regions.

A reduction in peak flood depth will likely correlate with a decrease in flood damage costs (Penning-Rowsell et al., 2005; Hammond et al., 2015). Reduction in flood volume will also reduce the duration of flooding, which in turn will reduce the disruption and hazard exposure to the general public. Delaying the occurrence of peak flood depths will also provide an opportunity for additional warning time, enabling more effective application of flood resilience measures and early warning systems, potentially providing safer emergency evacuations of the at risk areas and allowing diversions to limit disruption to key economic, cultural and social activity in the city (Parker et al., 2011).

No interventions completely prevented surface water flooding in the zones studied during any rainfall event. This is likely due to the very high capacity of storage and conveyance systems required to capture all runoff across the large contributing area when subject to the highly intense rainfall predicted during short duration events. Despite this, many strategies demonstrated a reduction in flood depth within the flood zones studied. Many interventions also created safe areas, where water levels were reduced to either zero or very low values. Safe areas prevent damage in the locality, but also have a far wider reach in minimising disruption and consequences through establishing evacuation routes which can provide the public with an opportunity to minimise hazard exposure.

Although the zones investigated still flood, interventions were predicted to reduce flood depths on streets in the upper catchments which act as tributaries to the main conveyance route. Accumulation of volume downstream masks this reduction in the regions studied, however flood management upstream will both reduce risk and make movement more manageable around the areas of higher flood risks.

Application of frameworks which facilitate analysis of many simulations may have a role in iteratively combining smaller local strategies to project the impact of combined future projects and develop towards greener urban catchments. This mode of analysis could also provide utility in identifying tipping points, where the combined effect of interventions will reduce rather than delay the peak flood and to limit the requirements for financially and environmentally expensive sub surface drainage upgrades.

7.4.2. Effect of AEP on flood intervention effectiveness

Fast analysis of strategies enabled evaluation of performance across multiple rainfall AEP scenarios. This analysis has identified a clear trend where, as events become more intense (i.e. AEP decreases), intervention performance to reduce peak flood depth and velocity become less effective. It should be noted that this finding is made relative to the peak values and does not include the interventions effect on hazard duration. However, in the case of surface water flooding, the peak depth rather than flood duration is likely to be the controlling factor in flood damage (Penning-Rowsell et al., 2010).

It should be noted that this effect is partially obscured during more common AEP events (18% and 10%) as the large capacity, catchment scale interventions do not appear to make such a difference, this is attributed to the fact that the full potential of these large scale interventions is not utilised during lower intensity rainfall events.

The reduction in intervention performance during more intense rainfall is attributed to strategies reaching capacity and then ceasing to reduce the flow rate versus the base case scenario. This is particularly relevant to peak flood velocity, which will be controlled by the peak flow rate of incoming runoff via topography. This pattern of interventions becoming less effective at reducing peak values during high intensity events is observed during the 1% AEP event (Figure 7.27), where differences between interventions were negligible. This finding correlates with evaluation of rainfall capture strategies from Chapter Six, where performance tipping points were predicted as rainfall volume exceeded storage capacities.

This effect is less apparent when comparing peak flood depths, which reflects a distinction in the controlling factor of depth being the total flood volume leading to ponding, rather than peak flows which control peak velocities. This argument is supported by observations of a larger variation in intervention peak flood depths at higher intensities (lower AEP's) than is present when examining the changes in peak velocities.

Assessing the response of green infrastructure to changing rainfall intensities is significant as it informs understanding the effective flood management beyond design standards. Green infrastructure is frequently cited as a desirable method with which to manage surface water and build resilience in urban environments (Balmforth et al., 2006; Environment Agency, 2007b; Duffy et al., 2008; Wong and Brown, 2009; Woods Ballard et al., 2015; Bowen and Lynch, 2017). Finding that performance of certain interventions reduces in response to high intensity rainfall indicates that resilience of interventions needs to be assessed in relation to a range of events when building urban resilience, particularly in light of increasing intensity and frequency of future hazards (Goonetilleke et al., 2005; Ana and Bauwens, 2010; Howard et al., 2010; Barbosa et al., 2012; IPCC, 2014). This supports conclusions regarding to the resilience of intervention strategies in Chapter Six, where certain storage based interventions demonstrate a sudden decrease in performance as a capacity threshold is exceeded.

7.4.3. Supporting practical application of green infrastructure through collaborative strategy screening

It is imperative that catchment stakeholders understand the performance of flood management techniques in order for benefits to be applied to cities (Pitt, 2008; MWH, 2014; Burns et al., 2015c; Woods Ballard et al., 2015; Schubert et al., 2017). Historic approaches have been limited by restrictions on time, budget and data which can lead to decision makers considering only tried and tested interventions, resulting in institutional inertia and stifling innovation (Cettner, 2012; O'Donnell et al., 2017). This research sought to develop application of new methods to address this institutional barrier through collaborating with key personnel from local government to devise intervention strategies. There is thought that such civic experimentation can change standard practice (Karvonen, 2011), and in this case, could increase the capacity of the local government to implement green infrastructure. This is particularly important because case study results suggest that substantial reductions in flood risk are only possible when green infrastructure is applied across large areas of a catchment, requiring buy in and communication between many stakeholders. Fast analysis using the framework enabled a series of workshops in quick succession, in which stakeholders could communicate and test strategies with fast feedback upholding a collaborative momentum.

Achieving high levels of green infrastructure implementation will likely take time and trusting partnerships between those involved (Burns et al., 2015a). Planners therefore need to develop and articulate aspirational strategies which gradually implement actions towards catchment wide surface water flood management. It is important to note that although the more localised interventions appear less effective, these will play a vital role in achieving larger scale ambitions. In fact, Burns et al. (2015d) found that localised projects increase the confidence of using new surface water management interventions. Communicating that substantial outcomes could take time will also be an important part of stakeholder consultation efforts. This observation is influenced by the decision to focus assessment, in line with local government priorities, on reducing the flooding in three flood zones which act as the principal areas of flood hazard in the catchment. Localised application of flood interventions will have an impact in smaller areas of the upper catchment, which may not be captured in this analysis. These may form an important role managing disruption of floods in the lower catchment through providing increased opportunity to navigate and divert around areas of risk. Understanding and utilising these opportunities is an important consideration for maintaining functionality of the urban environment during flooding.

7.5. Chapter conclusions

This chapter demonstrates a practical and applied implementation of the framework described in Chapter Three. The chapter finds several key conclusions regarding application of the framework:

- Implementation of this framework in collaboration with catchment stakeholders provided a clear and concise strategic intervention development mechanism which generated evidence towards the utility of the framework in screening urban flood management strategies.
- Efficient option analysis achieved using the framework enabled a collaborative screening process which could be undertaken iteratively over

the course of several workshops. Holding workshops in quick succession (over several weeks) upheld the collaborative momentum of the project.

- The simplified development of intervention strategies provided a clear communication tool which supported the multi-disciplinary investigations required for urban planning in a complex environment. This was evidenced through engaging multiple catchment stakeholders and departments ranging from engineering, environment, and urban landscape planning.

The chapter also finds several conclusions regarding application of green infrastructure in managing urban surface water flooding:

- Analysis of interventions indicated a range of strategies which were effective at reducing flooding when built up across the catchment, and that multiple smaller intervention strategies accumulate towards catchment scale benefits.
- The most effective strategy was found to be high volume rainwater capture tanks applied across the catchment. Strategically targeted drainage upgrades also demonstrated significant reductions in flooding.
- Green infrastructure flood reduction performance declines when managing high intensity rainfall events. This is of particular significance given a trend in literature to present green infrastructure as an intrinsically resilient solution to extreme rainfall and demonstrates the need for future evaluation of interventions to apply context specific analysis across many rainfall scenarios to build resilient surface water management.

8. CONCLUSIONS AND RECCOMENDATIONS

This chapter details conclusions and recommendations emerging from the research presented in this thesis. This is structured through outlining a summary of the work undertaken, presenting conclusions pertinent to each objective, synthesising conclusions towards general guidance for evaluating intervention performance using rapid scenario screening and identifying emerging opportunities for future research.

8.1. Thesis summary

Surface water flooding is the predominant cause of flood risk in the UK and contributes significantly to global flood impact (Pitt, 2008; Committee on Climate Change, 2017; Löwe et al., 2017; DEFRA, 2018b; Guerreiro et al., 2018; Wing et al., 2018). Many studies predict these impacts to increase in response to climate change, urbanisation and a reliance on aging urban drainage infrastructure (Wheater and Evans, 2009; Howard et al., 2010; Djordjević et al., 2011). Despite significant flood impacts and a growing realisation of future hazards, management of surface water flooding has historically been overshadowed by prioritisation of fluvial and coastal flooding counterparts (Douglas et al., 2010; DEFRA, 2018b). In response, contemporary research has developed a technical understanding of many management interventions, however inclusion of measures within strategic planning methodologies is still limited.

The aim of this thesis was to develop rapid scenario screening to investigate the performance of surface water management strategies in urban catchments across design standard and extreme events.

To achieve this aim it was first necessary to identify current tools applicable for measuring the performance of interventions. A review of current literature identified a general trade off regarding the ability of available tools to quantitatively measure intervention performance versus the capability to practically screen the multitude of possible permutations of strategies and scenarios. Consequently, this has historically limited the scope of intervention comparisons, typically at the expense of examining novel interventions and strategy responses to extreme rainfall events.

The research responded to this gap through developing a rapid scenario screening framework, capable of high level quantitative screening of many scenarios through application of easily accessible input data, computationally efficient cellular automata flood models and a simplified representation of interventions.

Application to evaluate intervention performance required validation of this simplified approach and an understanding of its potential advantages and limitations versus conventional assessment methodologies. Framework validation was achieved through testing the approach versus industry standard modelling, undertaken and published as part of the Cambridgeshire SWMP (Arcadis, 2012). Analysis indicated that the rapid scenario screening provided comparable results to published model outputs, whilst retaining advantages regarding simulation speed and set-up time.

Validation of the framework enabled the method to be applied for screening surface water flood risk and potential interventions. This was first applied through assessing the effect of strategic intervention zones, intended to identify the scope and effects of various intervention strategies required for effective flood management in a case study catchment. Rapid scenario screening was used to identify priority flood spots and flood damage estimates for a case study in Exeter, UK.

The framework was advanced to incorporate representation of specific interventions. The performance of a range of intervention types and placement strategies was assessed across 144 rainfall scenarios and 792 placement and AEP permutations. This provided analysis of intervention response to design standard and extreme rainfall. A cost effectiveness metric, based on EAD reduction versus capital, operation and maintenance costs over a thirty year planning horizon was developed and used to enhance analysis towards developing evidence to screen intervention strategies in urban catchments. This stage of research contributed a progression in methodological approaches and understanding of intervention reliability and resilience.

The final stage of research was to verify the theoretical opportunities provided by using the rapid scenario screening framework through application to a practical case study, alongside catchment stakeholders. This was achieved through framework application to assess interventions in a case study of Melbourne City Centre, Australia. The framework was demonstrated as an appropriate tool for collaborative catchment screening, whilst providing insight into the scale versus effectiveness of green infrastructure strategies for surface water flood management in a highly developed urban environment.

It is recommended that the outcomes from rapid scenario screening are applied to support initial and high level strategic decisions, such as: influencing the direction of further detailed modelling, highlighting additional data requirements; stakeholder engagement; scenario exploration; and, including novel intervention strategies within the initial stages of the decision support process. This screening approach can also be used to explore many permutations of strategies and their responses to future uncertainties, such as the increases in precipitation intensity and changes in rainfall characteristics evaluated within this thesis. This supports other research indicating the advantages of enhancing decision support through evaluating many possible futures.

8.2. Conclusions

This section presents the main conclusions regarding each objective of the thesis.

8.2.1. Review literature regarding screening intervention performance under design standard and extreme rainfall events

Chapter Two reviewed literature regarding available surface water management interventions and current methodologies for evaluating intervention performance during design standard and extreme rainfall. Key conclusions drawn from the review are:

- Screening and comparing intervention performance is currently achieved using a wide range of qualitative and quantitative approaches. However, a trade-off exists between fast but low resolution methods, capable of qualitatively screening many interventions, and high resolution but computationally intensive flood simulation, which can only evaluate a limited number of scenarios. New rapid approaches, such as cellular automata flood modelling, provide an opportunity to overcome this trade-off and increase enhance consideration of intervention type, scale and distribution when evaluating performance of strategies. Despite documented speed and accuracy of these techniques, their application to surface water management is currently limited.

- Current performance evaluation is also focused on implementation of design standards, which neglects the importance of building resilience to extreme events and represents a gap in current surface water flood management.

8.2.2. Develop a screening framework to enable assessment of many intervention scenarios at the urban catchment scale

A rapid scenario screening framework was developed to address the gaps identified during the literature review. The scope of the framework is aimed at generating evidence for decision support using fast preliminary option assessment, and therefore is designed to use data requirements and assumptions commensurate with this utility. Chapter Three details development of this framework, with the following key messages:

- Research has contributed a novel rapid scenario screening framework which delivers insight into how intervention performance can deliver maximum benefits given the many permutations of intervention type, scale and distribution possible within urban catchments. The framework applies easy to access data, a simplified representation of interventions and a computationally efficient cellular automata flood model to quantitatively screen scenarios at an urban catchment scale.
- Utility of the framework is designed to screen many strategies at a high level to enhance understanding of performance and develop evidence towards surface water management actions. Application towards this goal is achieved through assessing performance using readily comparable quantitative metrics, including flood depth, extent and damage costs.

8.2.3. Validate the framework against industry best practice

The rapid scenario screening framework was validated through comparison with industry standard modelling techniques in Chapter Four. The overall conclusion was that rapid scenario screening is a promising tool for evaluating flood dynamics and intervention performance as part of scenario exploration to aid decision support. Specifically, research found:

- Rapid scenario screening demonstrates close correlation with outputs from current industry standard modelling when evaluating surface water flood hazards within priority flood spots across a UK case study. This finding applies to models constructed to multiple levels of detail, including worst case overland flow (97.4%), inclusion of the sub-surface drainage system (98.5%) and addition of interventions to the catchment surface (98.5%). This supports application of the framework for screening priority surface water flood spots and high level flood dynamics at the urban catchment scale.
- The new framework can advance current best practice through including analysis of many scenarios within high level screening. This responds to limitations in current approaches such as narrow analysis of future uncertainties, for example evaluating strategies using a design storm for a fixed return period, and restricting permutations of novel surface water management interventions.

8.2.4. Investigate the flood reduction performance of strategic and specific interventions

Performance of interventions was evaluated through Chapters Five, Six and Seven. Chapter Five examined effects of strategic intervention zones, Chapter Six evaluated specific interventions in Exeter, UK, and Chapter Seven measured performance of green infrastructure applied to Melbourne, Australia. Several general conclusions regarding intervention performance can be drawn from these three chapters:

- Intervention performance varies significantly in response to the duration and intensity of rainfall. Short duration, high intensity rainfall was predicted to cause the deepest flooding and highest flood damage costs. This finding corroborates existing guidance indicating the importance of managing high intensity and extreme events in urban environments (Meehl et al., 2000; Environment Agency, 2013).
- Although centralised interventions provide benefit at smaller scales, catchment based strategies are required to substantially reduce flood extent and estimated annual damage costs across urban areas. The most effective intervention was consistently found to be extensive application of

decentralised rainfall capture, which reduced estimated annual damage in a UK case study by 76% versus a business as usual baseline.

- Multiple smaller intervention strategies accumulate towards catchment scale benefits. Dispersed lower volume catchment wide interventions performed better than concentrated higher volume measures. For example, in Exeter, decentralised 1500 l rainwater capture tanks demonstrated a lower EAD (£0.2 million) than centralised, high volume (up to 10 000 l) tanks (£0.7 to £0.8 million). This finding is supported by analysis in the Melbourne case study, which indicates catchment wide approaches are more effective at reducing flood depths than high volume centralised interventions. This supports future development of catchment wide surface water management.

8.2.5. Evaluate intervention cost effectiveness across many rainfall scenarios

Chapter Six responded to this objective by expanding analysis of intervention performance through developing a cost effectiveness metric. Key conclusions pertinent to this objective are:

- Intervention type, location and scale have significant impacts on cost effectiveness. Analysis of hundreds of scenarios indicates a wide range of cost effectiveness ratios for interventions, ranging from £0.10 to £26.0 damage reduction per £1 spent, with the most cost effective interventions identified as high volume localised drainage interventions targeted in areas of intense flooding. The implications of spatially varying cost effectiveness are two-fold: Firstly, future intervention performance analysis should include spatial simulation of flood dynamics; and secondly, development of decentralised catchment scale strategies should be complemented by application of targeted and cost effective high volume interventions in areas of high risk.
- Objective Four indicates that catchment wide surface water management should be pursued as an aspirational goal, however extensive implementation will take time, resources and co-operation between multiple stakeholders. The implication from this objective is that progress towards this can be implemented incrementally and cost effectively using tools such as rapid scenario screening. Analysis supports this through

demonstrating all interventions generated their strongest cost effectiveness ratios when strategically targeted.

8.2.6. Verify application of the framework through practical application with catchment stakeholders

Chapter Seven responded to verification of the framework through designing and evaluating intervention performance across a case study in Melbourne City Centre in collaboration with catchment stakeholders. Key conclusions were:

- Development and analysis of many intervention strategies was enabled through rapid setup and simulation using easy to access data. The simplified development of intervention strategies provided a clear and concise communication tool leading to a collaborative and efficient option screening process which supported the multi-disciplinary investigations required for urban planning in a complex environment.
- Catchment screening identified a clear hierarchy of interventions, highlighting the effective flood reduction of catchment wide surface water management strategies, which could be achieved through iteratively developing smaller local strategies to project the impact of combined future interventions towards greener urban catchments.
- Application of the framework is also supported by Chapter Four, which indicates close correlation between industry standard screening methodologies versus the rapid scenario screening approach (as discussed in Objective Three).

8.2.7. Investigate the relationship between resilience and reliability of interventions

Chapters Six and Seven respond to Objective Seven through exploring the performance of interventions across design standard and extreme events. Typically studies can only assess a limited range of events due to resource costs of surface water modelling, the rapid scenario screening framework addresses this challenge. Analysis of intervention performance across events identifies several key conclusions:

- Performance of strategies during low magnitude events is not reflective of a strategies response to extreme events. This is evidenced through rainwater capture interventions demonstrating low flood damages where capacity can be fully utilised, but reaching a tipping point where exceeding capture volumes leads to a substantial performance decrease. Interventions which are able to continue functioning over extended timescales, such as drainage upgrades, are more effective at managing long duration events and appear more resilient to the extreme rainfall beyond design standards. This is of major significance when considering a planning environment focused on meeting specified design standards versus environmental hazards which are increasing in severity as a response to climate change, urbanization and aging infrastructure systems. Planning based solely on design standard events is not guaranteed to develop systems which are able to cope with extreme events.
- Chapter Seven indicates that green infrastructure effectiveness declines when managing high magnitude rainfall events. This is of particular significance given a trend in literature to present green infrastructure as an intrinsically more resilient solution to extreme rainfall (Balmforth et al., 2006; Environment Agency, 2007; Duffy et al., 2008; Wong and Brown, 2009; Woods Ballard et al., 2015; Bowen and Lynch, 2017). The implication of this finding is that developing strategies with resilient performance requires evaluation of many rainfall scenarios and that interventions with resilient properties, such as green infrastructure, do not necessarily achieve resilient performance.

8.2.8. Develop recommendations for practical application of the methodology

This thesis has the intention of developing reliable and resilient surface water management through contributing a rapid scenario screening framework applicable to complement and direct the existing detailed urban drainage tools currently available. A crucial component of contributing an actionable framework is developing a set of recommendations for future application. The following section outlines recommendations pertaining to practical considerations, structuring the application of the tool and framework utility.

Development, validation and application of rapid scenario screening has identified several key considerations when implementing the methodology:

- Studies have identified the importance of high resolution elevation models which incorporate macro and micro topographical features to accurately route runoff across urban surfaces (Schubert et al., 2008; Fewtrell et al., 2011; Dottori and Todini, 2013); therefore, wherever possible, catchments should be represented using high resolution data.
- Chapter Six identified that intervention performance during design standard events is not reflective of resilience to extreme rainfall. Interventions with similar cost effectiveness and performances during low magnitude events were found to exhibit substantial differences during high magnitude events. Therefore, application of the framework should examine intervention performance across a range of rainfall events, particularly given likely increases to future rainfall intensities (Westra et al., 2014).
- Analysis using constant intensity design storms (Section 6.2) and variable intensity design storms (Section 6.3) in the same catchment demonstrates increased flood damage when high resolution peak intensities are represented. The influence of peaks on flood damage indicates future modelling should represent these when calculating damages.
- Selection of flood metrics should be considered carefully depending on each context. Performance analysis based on peak flood depth, flood damage costs and intervention cost effectiveness did not always return the same intervention performance rankings. Therefore, evaluation of intervention performance should considered relative to the spatial disaggregation and context of metrics assessed. This is particularly relevant in catchments with a high degree of spatial variation in land use, where the location of flooding may be significant in determining damage. It should be noted that this isn’t always the case, for example high density and broadly homogenous land uses (i.e. main streets in urban city centres) may not demonstrate a large difference between intervention rankings based on depth or damage costs.

- Parameterisation of cell output rates to represent the sub surface drainage system demonstrated high correlation with 2D-1D modelling (Chapter Four), however data confidentiality, record uncertainties and legacy assets mean that detailed schematics of surface sewers may not always be available, particularly at the initial stages of intervention screening. Low sensitivity of cell output parameters during high intensity rainfall indicates that broad scale parameterisation, such as that undertaken as part of Environment Agency (2013) surface water flood mapping, is suitable for preliminary screening where this is the case. In this case, preliminary modelling using the framework can also be utilised towards identifying where additional data is required to develop opportunities for future detailed modelling.
- Rapid scenario screening should be applied subject to a preliminary analysis of catchment flood mechanisms. The strengths of the methodology lie in computationally efficient setup and analysis of surface water runoff, and the approach is not intended to examine other causes of flooding. It is recommended that catchment flood mechanisms should be assessed to scope framework suitability prior to investigations taking place. This can be achieved through investigating published flood reports, incident logs, flood histories and discussion with catchment stakeholders.

The following conclusions relate to a implementing a staged process when applying the framework. It is recommended that rapid scenario screening is applied iteratively to evidence, direct and explore the complex permutations of intervention type, scale and distribution:

- The first stage of analysis should identify and prioritise flood hazards across catchment(s) through assessment of a baseline scenario(s) (Chapters Four and Five). Identification of priority flood spots and general trends in flood dynamics can then be applied to focus subsequent analysis on specific sub-catchments, both enhancing the direction of intervention design and achieving computational efficiency through refining the area where further modelling is required.
- Intervention screening should start with analysis of strategic intervention zones (Chapter Five). This broad scale of analysis will direct requirements

for future modelling by establishing the scope, scale and effects of interventions required to best achieve surface water flood management outcomes within the catchment.

- Assessment of strategic intervention zones can then inform analysis of specific intervention type and distribution (Chapter Six). This may also be undertaken iteratively with catchment stakeholders (Chapter Seven), to enable development and exploration of flood management scenarios.
- Iterative analysis following this procedure will form a structured and well evidenced direction of investigation which can be used to justify requirements for subsequent flood management actions within the catchment, as recommended by UK Government guidance (House of Commons, 2016).

Rapid scenario screening using the framework has the following key utilities for surface water flood management:

- Scoping requirements for surface water flood management projects through preliminary investigation of catchment flood dynamics and identification of priority flood locations across city scale catchments.
- Enhancing analysis through evaluating complex permutations of intervention type, scale and distribution in urban catchments.
- Generating evidence for decision support and directing future actions through the screening the performance of strategic and specific flood management interventions across multiple scenarios. Future actions may consist of capturing additional data, further detailed modelling, implementing interventions or developing strategic catchment management plans.
- Evaluating resilience of catchments to extreme rainfall using performance analysis across a wide range of rainfall IDF characteristics.
- Exploration of management scenarios with catchment stakeholders, structured through iterative stages of analysis and corresponding workshops to utilise fast and simplified assessment as a basis for evaluating options.

8.3. Recommendations for future research

A number of future research topics were identified.

8.3.1. Develop intervention 'cost effectiveness' towards 'cost benefit'

The cost effectiveness metric applied during this study is a simplified metric focused on avoided direct flood damage to buildings, suitable for high level screening. Future development of this work could enhance the metric towards cost 'benefit' through a more detailed consideration of damages and benefits.

The damage element of the cost effectiveness metric could be developed through finer spatial analysis of building classifications alongside inclusion of indirect damages and intangible impacts such as flooding effects on human health (Ahern et al., 2005; Bowen and Lynch, 2017). Current studies support this direction of research through highlighting the range of impacts resulting from indirect damages (Messner et al., 2007; Merz et al., 2010; Penning-Rowsell et al., 2010; Hammond et al., 2015; Chen et al., 2016) and the potential for damages to cascade through inter-connected urban environments (Cavallo and Ireland, 2014).

The benefit element of the metric could be enhanced through inclusion of the wide range of tangible benefits attributable to interventions. Many studies are currently evaluating these multiple benefits, with specific focus on the potential of urban green infrastructure (MWH, 2014; CIRIA, 2015; Jose et al., 2015; Norton et al., 2015; Mijic et al., 2016; Bowen and Lynch, 2017; Fenner, 2017; Kunapo et al., 2018). In particular studies indicate benefits such as a reduction in the urban heat island effect, improvements in air quality and use of captured rainfall. Intangible and difficult to quantify benefits such as a reduction in risks to life, prevention of psychological impacts, amenity value and mitigation of climate change are also relevant when comparing infrastructure options (CIRIA, 2015). These benefits are difficult to monetise without detailed investigations using specific models, however studies have begun to develop mechanisms for strategic level analysis (Ashley et al., 2002; Ossa-Moreno et al., 2017). Inclusion of multiple benefits within quantitative rapid scenario screening methodologies underpinned by a high resolution simulation is likely to provide additional evidence to support installation of multi-functional infrastructure.

8.3.2. Investigate timing and duration of surface water flooding

Analysis of resilience within this thesis is made relative to minimising the duration and magnitude of failure (Butler et al., 2014). These elements, along with recovery costs, are captured through application of flood damage curves inclusive of short duration flood damage (Penning-Rowsell et al., 2010). Resilience literature also encapsulates additional criteria such as the speed, timing and recovery duration of failures (Hashimoto et al., 1982; Linkov et al., 2014). This is identified within this research through assessment of intervention tipping points (Chapter Six) where intervention performance becomes less effective across an event intensity threshold. This is attributed to interventions exceeding storage capacity and shifting the timing, rather than reducing the magnitude, of the downstream time of concentration. This concept merits further investigation through analysis of intervention effects on failure timing across catchments, and evaluation of how timing and duration will affect the consequences of failure.

8.3.3. Develop detailed scenarios including urban and population change

Scenario screening within this thesis is focused on intervention response to design standard and extreme rainfall. Literature also emphasises the need to manage future urban and population growth (Marlow et al., 2013; Mikovits et al., 2015; Lu et al., 2018). Future research has the potential to apply rapid scenario screening to a greater range of scenarios to evaluate the effects and interactions of changing landscapes, rainfall characteristics and intervention strategies.

8.3.4. Enhance parameterisation within the CADDIES model

The CADDIES model applied for simulation delivers a computationally efficient simulation of spatial flood dynamics, suitable for catchment screening. However, the approach applies several simplifications of physical processes which could be adapted to include more functionality within the approach. The main recommendation for this is to enhance the cell output parameter to incorporate temporal variability. This could be achieved using a similar mechanism as applied through the cell input hyetograph. Temporal variation in output rates would enable finer resolution representation of the physical processes controlling urban drainage system function and infiltration to soils. Application of surface storage volumes would also enhance possibilities for representing surface drainage features. However, it is noted that computational efficiency is a pre-requisite for assessing many scenarios, therefore any increase in model complexity should be evaluated relative to the trade off in simulation time versus improvements to utility.

8.3.5. Develop and test model application towards continuous simulation of rainfall events

This research has focused on intervention performance in response to short duration, high intensity rainfall; a typical cause of high magnitude surface water flooding. Many interventions evaluated within this framework are also applicable to manage everyday rainfall events over extended periods. Application towards this analysis could be enhanced through development of the modelling approach using continuous simulation to verify parameterisation of interventions. This could be further enhanced through implementation alongside the recommendations outlined in Section 8.3.4.

8.3.6. Align work within context of 'Decision Making under Deep Uncertainty'

Application of rapid scenario screening to explore many scenarios and generate extensive and robust decision support tools using simplified modelling techniques draws significant parallels with current research in the field of 'Decision Making under Deep Uncertainty' (DMDU) (RAND, 2013; Babovic et al., 2018b). A link between rapid scenario screening and DMDU could be developed through implementing this and similar simplified modelling approaches to address the wider urban system when evaluating options, for example through evaluating resilience in the context of a broader societal-infrastructure relationship, advancing on the intervention perspective developed in this work.

8.3.7. Implement machine learning to optimise surface water management

Machine learning is frequently applied to optimise multi-dimensional water engineering problems for which systematic evaluation of every alternative is not possible. Methods include application of techniques such as genetic algorithms and neural networks (Ostfeld et al., 2013). These techniques have been available for decades, however require carefully formulated problems, treatable using solvers which have traditionally only been applicable to simplified representations of systems such as pipe networks, reservoirs and water treatment (Savic and Walters, 1997; Sweetapple et al., 2014). The fast simulation speeds and simplification of model parameters applied within this research provide the potential for future work to apply simplifications as the basis for developing machine learning approaches which include the spatially complex datasets required for surface water flood management. Automatic adjustment of parameters, fast simulation and programming multi-objective goal seeking algorithms could enable advanced exploration of multi-objective decisions and develop optimisation within surface water flood management.

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“For a moment, nothing happened. Then, after a second or so, nothing continued to happen.”

Douglas Adams, The Hitchhiker’s Guide to the Galaxy