Received: 25 October 2017 | Revised: 14 November 2018 | Accepted: 19 December 2018

DOI: 10.1111/jfr3.12523

CIWEM Chartered Institution of Water and Environmental Management WILEY Journal of Flood Risk Management

ORIGINAL ARTICLE

Comparing cost-effectiveness of surface water flood management interventions in a UK catchment

James L. Webber | Guangtao Fu | David Butler

Centre for Water Systems, College of Engineering, Mathematics and Physical Sciences, University of Exeter, Exeter, UK

Correspondence
James Webber, Centre for Water Systems, College of Engineering, Mathematics and Physical Sciences, University of Exeter, North Park Road, Exeter EX4 4QF, UK.
Email: jw616@exeter.ac.uk

Funding information
Engineering and Physical Sciences Research Council, Grant/Award Number: EP/K006924/1EP/L016214/1; Natural Environment Research Council, Grant/Award Number: NE/P011217/1

Despite significant consequences caused by recent events, surface water flooding has historically been of lower priority relative to fluvial and coastal risks in UK flood management. Legislation and research proposes a variety of innovative interventions to address this; however, widespread application of these remains a challenge due to a number of institutional, economic, and technical barriers. This research applies a framework capable of fast and high-resolution assessment of intervention cost-effectiveness as an opportunity to improve available evidence and encourage uptake of interventions through analysing permutations of type, scale, and distribution in urban catchments. Fast assessment of many scenarios is achieved using a cellular automata flood model and a simplified representation of interventions. Conventional and green strategies are examined across a range of design standard and high-magnitude rainfall events in an urban catchment. Results indicate high-volume rainwater capture interventions demonstrate a significant reduction in estimated annual damage costs, and localised surface water drainage interventions exhibit high cost-effectiveness for damage reduction. Analysis of performance across a wide range of return periods enhances available evidence for option comparison decision support and provides a basis for future resilience assessment of interventions.

KEYWORDS
decision support, resilience, surface water, sustainable drainage systems

1 | INTRODUCTION

Surface water flooding is a significant hazard, which regularly impacts communities in the United Kingdom. The Summer 2007 floods highlighted the significant hazard associated with surface water flooding (Pitt, 2008) when intense precipitation overwhelmed drainage systems and created surface flows which flooded over 35,000 properties in England and Wales (Parker, Priest, & McCarthy, 2011; Priest, Parker, Hurford, Walker, & Evans, 2011). The direct and tangible damage of this event was estimated to cost £4 billion, with further significant intangible impacts (Environment Agency, 2010a). More recently, Winter 2015–2016 broke rainfall records as Storm Desmond caused more than £5 billion in damage (House of Commons, 2016). Current estimates place 4 million homes at risk from surface water, resulting in this mode of flooding recognised as the leading cause of flood risk in the United Kingdom (DEFRA, 2012).

There is an emerging realisation regarding the importance of surface water management in the United Kingdom. Recent studies and government reports have identified a historic paradigm focused on fluvial and coastal flooding and highlight the need to respond by investing in new approaches to minimise risk (Douglas et al., 2010; Ellis & Revitt, 2010; Pitt, 2008). In particular, studies and legislation emphasise application of innovative management interventions, for example, green infrastructure, to manage consequences of events (House of Commons, 2016). Addressing flooding using effective management interventions is of contemporary importance given changes to climate, land use, and demographics that are likely to exacerbate future flood hazards (Barbosa, Fernandes, & David, 2012; Chocat et al., 2007; Goonetilleke, Thomas, Ginn, & Gilbert, 2005; Howard et al., 2010; IPCC, 2014).

Technical understanding and availability of a range of both tested and novel surface water flood management interventions already exists. Interventions such as upgrading urban piped drainage and novel interventions such as sustainable drainage systems (SuDS), green infrastructure, property-level resilience measures, nature-based solutions, and catchment management are frequently discussed in academic literature and government reports (Butler & Davies, 2011; Fletcher et al., 2015; Schanze, 2017; Woods Ballard et al., 2015). Application of these interventions is supported by current legislation such as the UK Flood and Water Management Act 2010, which specifies for local flood risk strategies to be developed and implemented (HM Government, 2010). However, despite clear and established legislation, recent government reviews indicate application of new intervention strategies still faces multiple challenges (Committee on Climate Change, 2015). 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 of novel interventions (O'Donnell, Lamond, & Thorne, 2017; Ossa-Moreno, Smith, & Mijic, 2017).

This study addresses the complexities of evaluating the large range of potential interventions in an urban catchment, given the many possible permutations of type, spatial distribution, and scale. Selecting and applying appropriate interventions requires a performance comparison of the many available options, which is typically assessed in relation to the resultant flood depth and extent associated with each intervention strategy across a range of scenarios. Current standard techniques to quantify flood depth and extent typically apply computationally demanding 2D flood modelling based on hydrodynamic equations (Elliott & Trowsdale, 2007; Hunter et al., 2008). Studying intervention performance using these techniques provides detailed analysis but carries a high resource cost to set up and run, therefore restricting the number of scenarios which can be practically assessed during option development (Emanuelsson et al., 2014). Where these models include novel interventions, they are often constrained to analysis of specific measures, which may result in the need to apply several models to assess a range of options (Zhou, 2014). Alternative option comparison approaches can increase the number of scenarios assessed by sacrificing simulation of flood dynamics and instead apply a qualitative comparison of options. This includes approaches such as expert-led multicriteria decision-making, geographic information system (GIS)-based techniques, and expert judgement (Birgani, 2013; Ellis, Deutsch, Mouchel, Scholes, & Revitt, 2004; Makropoulos, Liu, Natsis, Butler, & Memon, 2007; Makropoulos, Natsis, Liu, Mittas, & Butler, 2008; Young, Younos, Dymond, Kibler, & Lee, 2010). Qualitative analysis greatly increases the speed and scope of analysis but at a tradeoff against the simulation of flood dynamics required to provide detailed evidence for decision support.

Recent advances in research address this tradeoff through developing frameworks which quickly screen catchment flood dynamics and generate evidence to support and steer the application of computationally expensive detailed modelling (Webber, Gibson, Chen, Fu, & Butler, 2018). Catchment screening is achieved using application of accessible input data (representative of high-level screening data with low processing requirements available to a UK practitioner at the start of a project), a simplified representation of interventions and the fast processing speeds of a cellular automata-based flood routing model, "CADDIES" (Cellular Automata Dual DraInagE Simulation) (Ghimire et al., 2013; Gibson et al., 2016; Guidolin et al., 2016). Data available to a UK practitioner in the preliminary stages of a design project are likely to include access to elevation models from clients, coarse land use data (through online imagery), catchment rainfall through national datasets (Centre for Ecology and Hydrology, 1999, 2013), potential intervention strategies, and high-level cost models. Practitioners are unlikely to have commissioned surveys to capture catchment specific processes such as traffic counts or undertaken detailed intervention specific modelling at this stage. Availability, resolution, and quality of data are likely to vary on a country-by-country basis. Previous application of this framework has been limited to large-scale analysis of strategic interventions, applied in blocks across a catchment (Webber, Fu, & Butler, 2018).

The aim of this study is to develop and apply a rapid intervention assessment framework for application as an initial option screening tool, applicable using data likely to be available in the initial scoping stages of a strategic design project. The intention of the research is to advance new methods that can be applied to complement established detailed modelling techniques through initial prioritisation of intervention cost-effectiveness. Initial prioritisation will provide utility towards evidencing and directing further detailed analysis using techniques that can be applied quickly and with limited data. Interventions include both green infrastructure and conventional solutions modelled at the property scale. This paper applies the term "green infrastructure" to refer to interventions that achieve surface water management through decentralised rainfall capture and infiltration techniques that replicate natural catchment processes. Within this paper, this "green infrastructure" can be considered synonymous with other terms such as SuDS, best management practices, and low impact development (Fletcher et al., 2015). Cost-effectiveness is assessed by comparing an estimated cost of constructing and operating interventions versus an expected annual damage reduction cost over a 30-year planning period. The paper describes model setup, presents an example of application in a UK catchment, and discusses the advantages and limitations of fast screening techniques.

2 | MATERIAL AND METHODS

The surface water intervention assessment framework presented in Webber et al. (2018, b, c) is a four-stage process that includes: characterising the study area, representing interventions, simulating scenarios, and analysing intervention performance. Flood simulation is undertaken using the CADDIES model. CADDIES is a cellular automata flood model that applies Manning's equation to simulate rainfall runoff between cells across a regular grid modelling domain (Figure 1). Water movement is controlled by elevation, input, output, and roughness parameters that are changed on a cell-by-cell basis to represent the effects of land use and interventions. A full description of this modelling process is available in Ghimire et al. (2013), Guidolin et al. (2016), and the CADDIES website - (Centre for Water Systems, 2017). Accuracy of the approach is validated in Gibson et al. (2016) and Webber, Booth, et al. (2018).

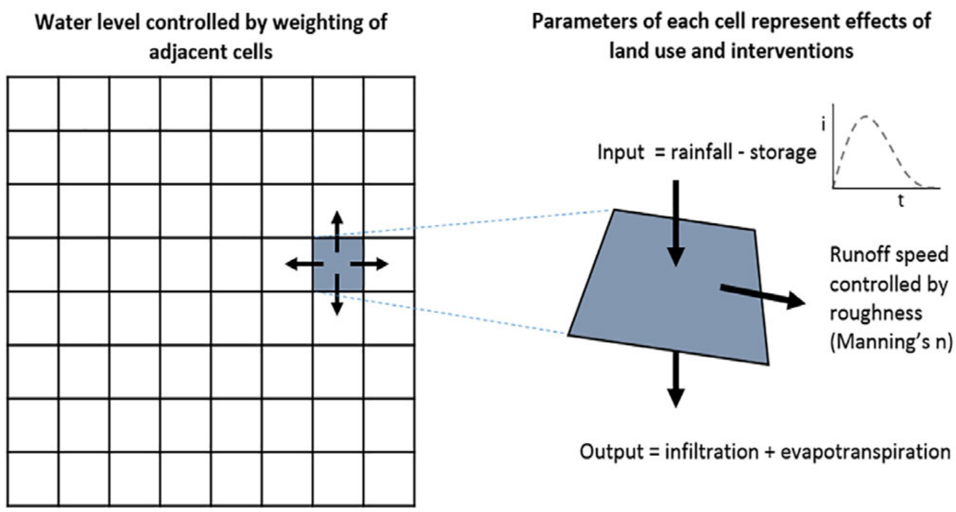

FIGURE 1 Cellular Automata Dual DraInagE Simulation modelling across a regular grid with parameters describing water input, output, and runoff speed at a cell-by-cell resolution (Webber, Fu, & Butler, 2018)

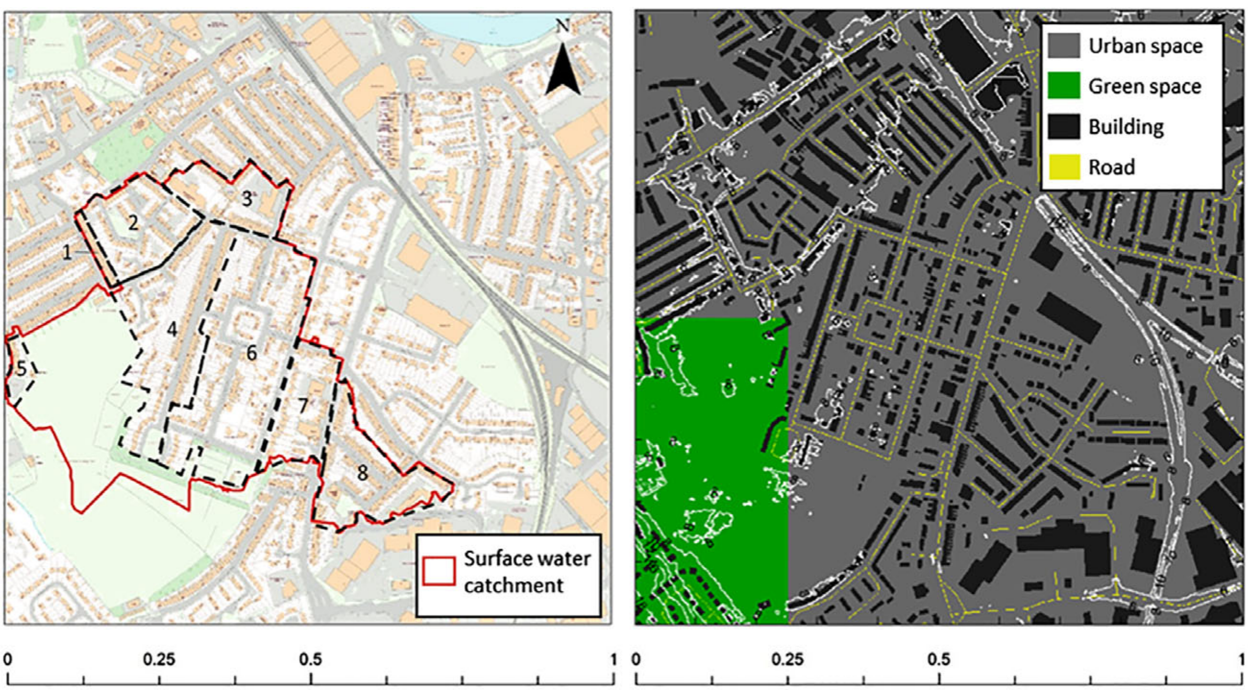

FIGURE 2 (Left) Map of study area. (Right) Land use classification

2.1 | Characterising study area and rainfall

The study area is a surface water catchment of a residential suburb in a UK city (Figure 2). The catchment is approximately 700 m × 700 m and was identified using a GIS-watershed analysis with 1-m resolution light detection and ranging (LiDAR). Predominant land use is residential, comprised of minor roads and semidetached 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.

The study area was processed to generate a land use classification and an elevation map. Land uses were classified into four categories using ordinance survey mapping: urban space, green space, buildings, and roads (Figure 2). Classification is expressed through changes to roughness, infiltration, and rainfall capture parameters associated with each grid cell in the catchment. Roughness values are taken from Manning's n coefficient for concrete (Arcement & Schneider, 1989; Butler & Davies, 2011; XP Solutions, 2017) and short grass (Hamill, 2001). Buildings have an artificially high-roughness coefficient in order to represent water being temporarily held within a structure (Syme, 2008). Infiltration rates are based on the sandy loam soil type identified in the region (Cranfield Soil and Agrifood Institute, 2017; UNFAO, 2017). Elevation information was provided using 1-m resolution LiDAR, processed to include the threshold levels of buildings within the catchment. The combined sewer system is represented using the Environment Agency surface water mapping method of increasing infiltration rates to 12 mm/hr (Environment Agency, 2013). This increase in output rate was applied to all 1 m^2 cells classified as "urban space" or "roads" in the catchment (Figure 1, Figure 2).

Rainfall was selected through preliminary analysis of critical storm durations using flood estimation handbook design rainfall events at 1-, 2-, 3-, 4-, 6-, 12-, 24-, and 48-hr durations across 30, 100, and 200 year return periods (Centre for Ecology and Hydrology, 1999, 2013). Peak flooding in all return periods was observed during 1-hr rainfall; therefore, analysis of intervention performance was made relative to this event. The number of return periods was expanded for intervention analysis to include 2, 5, 10, 20, 30, 50, 100, 200, and 1,000 year rainfall events, all provided by the flood estimation handbook database (Centre for Ecology and Hydrology, 1999, 2013). Summer design storms were selected due to characteristic higher peak intensities, which are more likely to exceed drainage capacity and result in surface water flooding (Butler & Davies, 2011; Jones, Fowler, Kilsby, & Blenkinsop, 2013).

2.2 | Representation of specific interventions

This section describes the intervention types and modelling approaches used in the study.

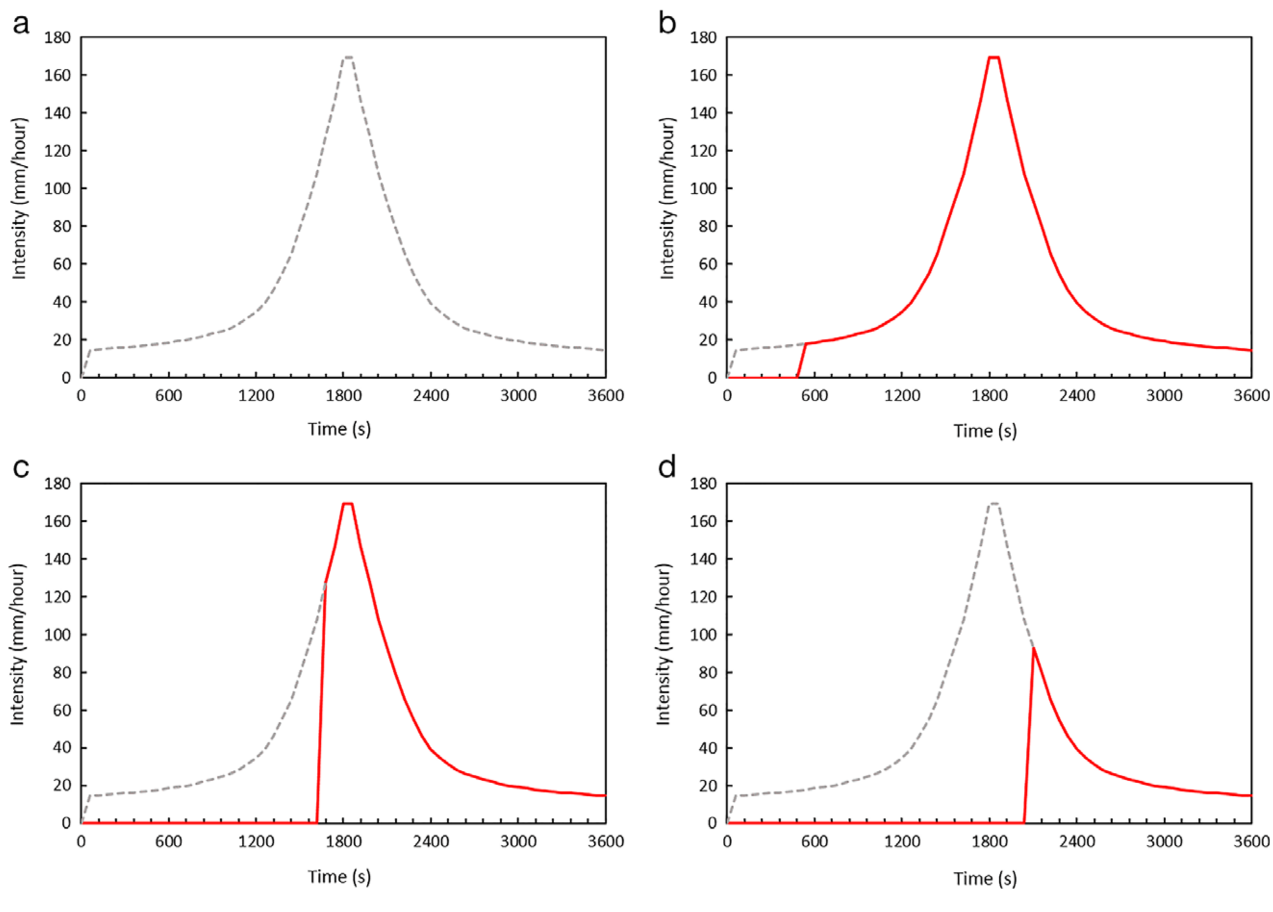

FIGURE 3 Representation of rainfall capture in a 1 m^2 cell through hyetograph manipulation during 100 year 1-hr rainfall: (a) no storage, (b) 2.2 L water butt, (c) 15 L green roof, (d) 33 L rainwater capture

2.2.1 | Modelling approach using CADDIES

Interventions were represented through spatial and temporal manipulation of cell roughness, input, and output parameters (Figure 1, Webber et al. 2018, b, c). Interventions which alter surface roughness are represented in CADDIES via Manning's "n" coefficient to control flow speed across a cell. Interventions which increase infiltration and subsurface drainage are included in the model through adjusting a water output rate per cell (Figure 1). Rainfall capture interventions are represented by adjusting the rainfall input rate to cells (Figure 3). 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. An average roof size in the United Kingdom is 45.5 m^2 (DCLG, 2015). Therefore, a 100-L water butt collecting from this surface would capture 2.2 L of rainfall per 1 m^2 cell. Figure 4 shows how the hyetograph is manipulated to achieve this. Figure 3a shows an unedited rainfall profile for a 1 in 100-year event. Figure 3b–d shows edited profiles representing capturing rainfall. This approach assumes 100% efficiency of rainfall capture interventions.

2.2.2 | Green roofs

Green roofs are vegetated surfaces constructed on top of structures with the intention of capturing rainwater within a soil substrate. Potential rainfall capture of a green roof varies depending on a number of factors including the depth of substrate, evapotranspiration potential, and roof geometry (Mentens, Raes, & Hermy, 2006). Various studies have measured the interception provided by green roofs around the globe with results typically indicating between 10 and 20 mm of rainfall storage using substrate depths between 75 and 150 mm (Martin, 2008; Paudel, 2009; Stovin, Vesuviano, & Kasmin, 2012; Woods Ballard et al., 2015).

Green roofs are represented by capturing 15 mm of rainfall prior to generating runoff (Figure 3). 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.

2.2.3 | Rainwater capture

Rainwater capture refers to interventions designed to intercept and store incoming rainfall. This is typically achieved through collection from a roof surface into tanks for water reuse or attenuation to sewers and soakaways (Woods Ballard et al., 2015). Volume reduction is controlled by the storage size and conveyance capacity of an installed system. In this study, it has been assumed that the only controlling factor on storage is available volume, not the throttling effects of down pipes. This intervention has been included in the model using the same approach as green roofs, where a new rainfall profile is applied to accommodate water captured by the intervention (Figure 3). Sensitivity to intervention capacity is modelled through inclusion of four capture volumes: 1,500, 3,000, 5,000, and 10,000 L.

2.2.4 | Water butts

Water butts provide a rainwater capture device when attached to the downpipes from roofs. Water butts are typically much smaller and cheaper than other rainwater capture methods, with approximately 250 L of capacity when empty. This study assumes a conservative available capacity of 100 L per water butt.

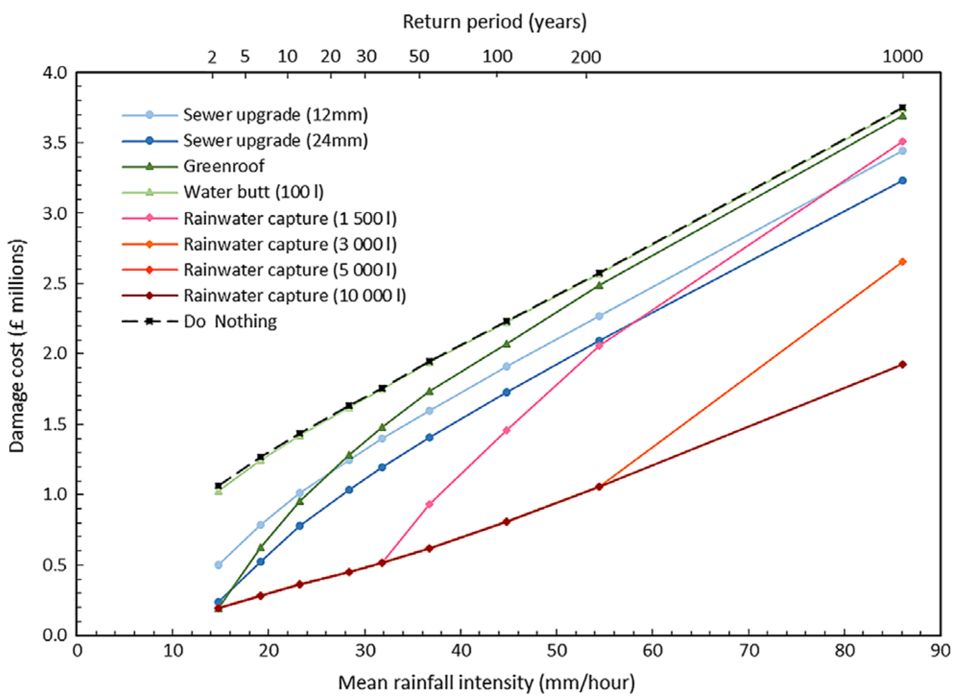

FIGURE 4 Damage cost versus mean rainfall intensity for interventions applied across all suitable locations

2.2.5 | Permeable paving

Permeable pavements replace traditional nonporous materials with permeable surfaces able to infiltrate surface water through the surface and into underlying storage. Water can be stored through geocellular systems, infiltrated into the soil structure, or collected in transmission trenches or pipes (Woods Ballard et al., 2015). Infiltration can occur directly through the surface material or through voids between tiles. A range of materials can be used depending on loading requirements. 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 (Collins, Hunt, & Hathaway, 2008; Pratt, Wilson, & Cooper, 2002; Zachary Bean, Frederick Hunt, & Alan Bidelspach, 2007). These studies found that infiltration rates for concrete block pervious paving have been recorded from 2.6 up to 17.2 mm/hr, with average rates around 5 to 7 mm/hr. Roughness values are taken from Manning's n coefficients for concrete (as discussed in the land use classification).

2.2.6 | Drainage upgrades

Upgrading storm sewers represents a common conventional approach to managing surface water within urban catchments. Upgrades primarily consist of increasing diameters, constructing combined sewer overflows, sewer separation, and construction of new sewer branches (Butler & Davies, 2011). These methods are designed to increase capacity in the piped system through increasing transmission volume or allowing controlled discharge. Capacity of systems can be preserved through regular maintenance and cleaning to prevent build-up or blockage.

CADDIES is not linked to a 1D representation of the piped system. This results in faster simulation times but means representation of sewer networks in urban catchments must be simplified in order to include this as an intervention within the model. For this study, upgrades have been included by increasing water output rates linked to the drainage system (12 mm/hr) by an additional 12 and 24 mm/hr, representing a doubled and tripled rate from Environment Agency (2013).

2.3 | Intervention placement scenarios

Examining the performance of a baseline scenario and nine interventions (identified in Table 1) across a combination of locations in the catchment (Figure 2) 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 2), and the areas of flooding identified during the preliminary analysis of critical storms (see catchment setup). 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, 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).

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.

2.4 | Flood simulation

Each intervention scenario was simulated across nine return periods, resulting in a total of 792 simulations. Fast simulation speeds were achieved using CADDIES, which minimises the computational requirements usually associated with 2D modelling (Ghimire et al., 2013; Guidolin et al., 2016). The simulation was run using an "Nvidia Tesla

TABLE 1 Costs of intervention construction, operation, and routine maintenance per 1 m^2 cell

Type	Capital cost (£ in 2018 value)	Capital cost per cell (£/m^2)	Operational cost over 30 years per cell (£/m^2)	Total cost over 30 years 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)	3,050.00 per system	67.03	10.12	77.15
RW capture (3,000 L)	4,270.00 per system	93.85	10.12	103.96
RW capture (5,000 L)	4,880.00 per system	107.25	10.12	117.37
RW capture (10,000 L)	5,856.00 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

Note. RW: rain water.

K20c" (2,496 compute unified device architecture cores) at a grid resolution of 1 m^2 and a minimum simulation time step of 0.01 s. CADDIES outputs water depth and velocities at user-defined output time steps (5 min). The model also outputs the peak water depths across the simulation period. Simulation speed for the most intensive simulation, 1-hr duration 1 in 1,000-year summer design rainfall, was 6 min. This simulation was extended by 4 hr of model time to ensure sufficient time for all runoff processes. This took an additional 21 min to run.

2.5 | Cost of interventions and flood damages

2.5.1 | Intervention capital costs

Capital costs of interventions are presented in Table 1. These have been calculated based on academic and government studies that 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 (ONS, 2018). Operational costs are calculated using discounting at a rate of 3.5% over a 30-year period (Environment Agency, 2010b; 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: DCLG, 2015) or per metre square 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.

Literature states rainwater capture tanks (adjusted for 2018 values) are £3,050 for 1,500 L, £4,270 for 3,000 L, £4,880 for 5,000 L, and £5,856 for 10,000 L (Roebuck, Oltean-Dumbrava, & Tait, 2011). Other studies corroborate this range of values (Environment Agency, 2007). Green roofs are estimated to cost £131.40/m^2 in 2018 prices (Bamfield, 2005). Water butts were estimated to cost £335.34/unit in 2018 prices (Stovin, Swan, Stovin, & Swan, 2007). It has been assumed that water butts will be replaced after 15 years at a discounted rate of £193.39, which represents the more conservative assumption from the available literature (Environment Agency, 2007; Ossa-Moreno et al., 2017). Permeable paving costs are based on present day values in the literature of £74.52/m^2 (Environment Agency, 2007; Stovin et al., 2007; Woods Ballard et al., 2007).

Cost of sewers is provided as a conservative upper estimate of £648.82/m of 450-mm-diameter pipe laid under an urban highway (Environment Agency, 2015). A cost per metre square has been estimated by calculating the area in 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 stormwater drainage system designed to reach a self-cleaning velocity (Butler & Davies, 2011). 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 × 10–6 m^2/s. Flow rate was calculated using a shallow gradient of 1:200, indicative of a safety margin for shallow gradient sewers.

The pipe full flow rate was linked to the increase in cell output rate by attributing pipe flow capacity to a subcatchment where each cell drained at the rate of +12 or +24 mm/hr. The subcatchment 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 that can drain at 12 mm/hr across a 280 m × 280 m region, at 24 mm/hr across a 200 m × 200 m region, and at 36 mm/hr across a 160 m × 160 m region. The cost of the pipe length was divided between each cell within these regions to calculate an approximate cost per metre square 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.

2.5.2 | Intervention operation and maintenance costs

Maintenance costs are shown in Table 1, 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 (ONS, 2018).

Literature indicates that green roofs require £3,650 per year for the initial 2 years and £876 a year maintenance afterwards (Bamfield, 2005). Rainwater capture maintenance is estimated to cost £0.55 per m^2/year (Environment Agency, 2007). Water butts are assumed to have a negligible annual maintenance cost (Environment Agency, 2007). 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 £1,512 per km/year. This cost was translated into a cost per metre square scaled by the catchment size of each pipe network to calculate an indicative cost per cell.

2.5.3 | Measuring intervention performance using cost-effectiveness

Costs of property damage have been calculated using a GIS-based flood damage tool (Chen, Hammond, & Djordjevic, 2016). The tool operates through applying a function that attributes a damage cost for each cell based on the peak flood depth within it (Webber, Fu, & Butler, 2018). CADDIES is used to calculate peak flood depth maps, which alongside flood depth-damage curves provide the basis for the calculation. Damage costs have been taken from industry standard depth-damage curves for an average residential property (Penning-Rowsell, Viavattene, & Parode, 2010). This relates the direct and tangible costs of short-duration inundation (<12 hr), 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 (Merz, Kreibich, Schwarze, & Thieken, 2010). Intangible and indirect costs have not been included within this assessment (Hammond et al., 2016). 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). It should be noted that these approaches require significant user input of costs and benefits through environmental and economic assessments, and as such are better suited to analysis of schemes at a detailed design stage rather than high-level analysis of large numbers of options.

Estimated annual damage (EAD) represents the expected damage per year when averaged over a long-time period and represents a useful metric to describe the damage avoidance of intervention strategies (Equation 1). 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 1,000 year events. EAD is determined by calculating the area under this curve (University of Exeter, 2014).

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

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

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 the benefit of each intervention through damage avoided. Future costs were calculated over a 30-year period using a discount rate of 3.5% per year, as specified by the UK Government (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 30 years with the benefit of damage avoided over the same period.

3 | RESULTS

3.1 | Comparison of interventions when applied across all available surfaces

Figure 4 shows the damage cost versus mean rainfall intensity for interventions applied across all suitable areas within the catchment. Consideration of intervention effects across the range of return periods generates performance curves, which describes effects in design standard and high-magnitude flooding. All interventions demonstrate a reduction in flood damage relative to the "do nothing" scenario.

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

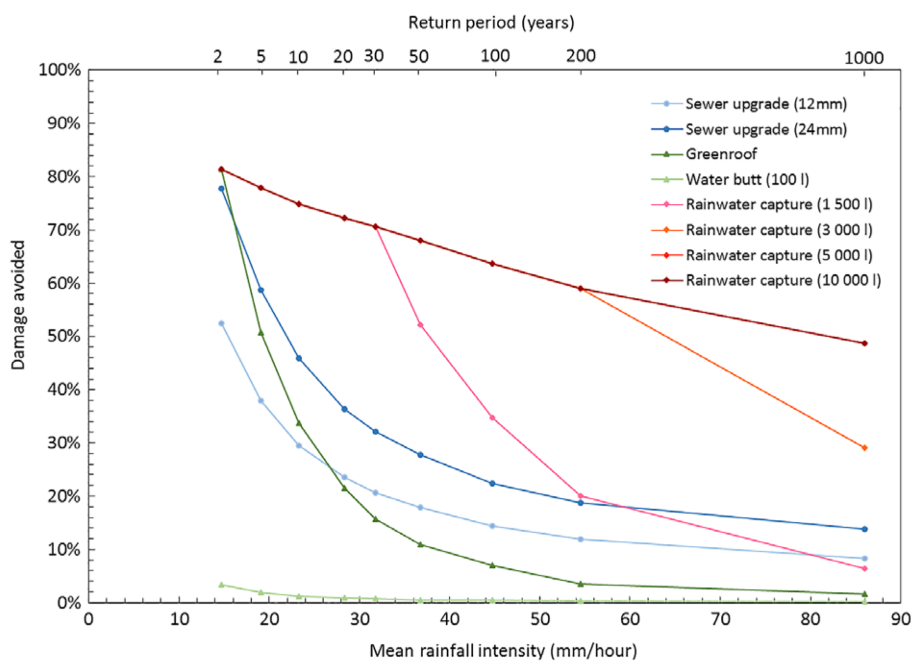

FIGURE 5 Percentage of damage avoided versus mean rainfall intensity for interventions applied across all suitable locations

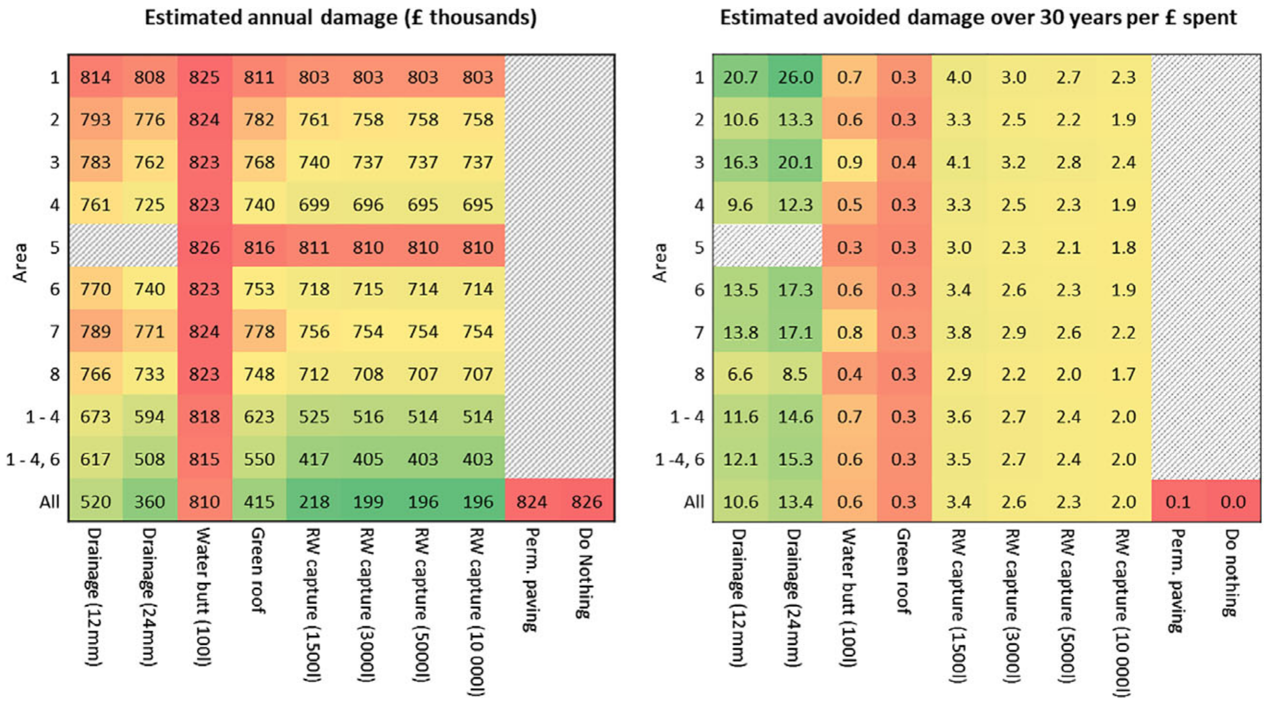

FIGURE 6 Comparison of estimated annual damage and cost-effectiveness for interventions across all placement scenarios. GBP, great British pound(s)

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, they 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. During the 1 in 1,000 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.

Figure 5 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/hr for 1,500 L tanks and 54 mm/hr for 3,000 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 an ability to continue removing runoff across the event, rather than having a finite volume filled.

3.2 | Comparison of interventions when applied across different locations

Interventions were also examined when placed on suitable surfaces in the regions indicated in Figure 2. Direct damage costs are typically lowest, when interventions are applied across all available areas (Figure 6). 5,000 and 10,000 L rainwater capture interventions generate the lowest damage costs across the whole catchment. Application of these measures in areas 1, 4 and 6 can achieve similar benefits to applying other interventions across the entire catchment. Water butts generate the largest damage costs when compared with other interventions in the same area. Permeable paving is the least effective intervention; however, this is attributed to the very limited area of application. Areas 4 and 6 show the largest reduction in catchment damage costs. Areas 1 and 5 show the least effective application.

Damage avoided per great British pound(s) spent does not correlate with the direct flood damage costs (Figure 6). The most cost effective intervention placement for surface-based interventions (drainage) is attributed to Area 1. The most effective area to implement roof-based interventions is Area 3. Areas 3, 6, and 7 also demonstrate better cost-effectiveness relative to other locations. Drainage upgrades are the most cost effective intervention, demonstrating above a 20-fold return on investment in Area 1. Green roofs perform poorly due to expensive operational costs.

4 | DISCUSSION

The study identified that rainwater capture interventions demonstrated the largest reduction in EAD (up to 76%), with the 5,000 and 10,000 L tanks generating the lowest damage costs in all rainfall events (Figure 4). Large reductions in EAD can also be achieved through placement of interventions in areas 1, 4, and 6 (Figure 6). The ability to screen interventions to higher magnitude events provides information that can be used to complement design standard assessments and highlights the importance of examining performance curves (Figure 4) alongside single measurements such as EAD and cost benefit ratios.

Surface drainage upgrades appeared to be the most cost beneficial intervention within this catchment. When applied across the entire catchment, upgrading by 24 mm/hr reduced EAD to £360,000, representing a 56% saving on the do nothing scenario. This achieved a damage cost reduction of £13.4 per £1 spent. When this intervention was applied in individual areas, it realised an estimated cost reduction factor of up to 26 (Figure 6). This demonstrates the importance of examining multiple placement strategies for intervention options. 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 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 (Butler et al., 2014). Managing extreme events is of particular relevance in the field of surface water management due to the future probability of high-magnitude events increasing in response to growing cities and a changing climate (Chocat et al., 2007; Howard et al., 2010; IPCC, 2014).

A tradeoff between simulation speed and model complexity has resulted in the simplification of several physical processes within the model. The primary limitation is the lack of a 1D pipe modelling scheme. Representation of sewers using a constant infiltration rate simplifies the complex flow dynamics, in particular occurrences of pipe surcharge. A constant rate can be calibrated to meet design standards within sewer subcatchments and represents the dynamics of above ground flow when runoff volume exceeds pipe capacities (Webber, Booth, et al., 2018). Treating soils infiltration rate as a constant value also simplifies the physical limitations associated with soil saturation, which has been managed by application of a conservative rate.

Cost assumptions were focused on developing a fast but high-level assessment and do not take into account site specific costs. Costing household-scale interventions at a metre square scale can result in under or over estimation of costs where property sizes differ from the UK average. Costs are likely to be more accurate over larger schemes where these variations may average out. 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 an estimated cost 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; Woods Ballard et al., 2015). These benefits are difficult to monetise without detailed investigations using specific models; however, recent studies have begun to develop mechanisms for estimating these (Ashley et al., 2002; Ossa-Moreno et al., 2017). Inclusion of multiple benefits within option screening is likely to increase the cost-effectiveness of interventions, particularly green infrastructure (Ciria, 2015).

5 | CONCLUSIONS

This research demonstrated a resource efficient analysis of intervention cost-effectiveness in a UK catchment through applying a fast assessment framework requiring minimal setup time, readily available data and simulation speeds of less than 6 min per scenario. Resource efficient analysis enabled screening of many intervention types, placement locations, and rainfall scenarios, including extreme events not normally included 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 expense of connecting to existing drainage networks. Future developments to the research should evaluate how multiple benefits associated with green infrastructure may improve the cost benefit ratio for green interventions, in particular due to positive outcomes to urban heat islands, public health and air quality.

ACKNOWLEDGEMENTS

The authors would like to thank Mike Gibson and Albert Chen (University of Exeter) for their help applying the CADDIES model and damage cost tool. This research was supported by the UK Engineering and Physical Sciences Research Council through the Water Informatics Science and Engineering Centre for Doctoral Training (EP/L016214/1) and the Safe and SuRe research fellowship (EP/K006924/1) and the UK Natural Environment Research Council "South West Partnership for Environmental and Economic Prosperity, 'SWEEP'" (NE/P011217/1).

ORCID

James L. Webber https://orcid.org/0000-0002-1158-4895

REFERENCES

Arcement, G. J., & Schneider, V. R. (1989). *Guide for selecting Manning's roughness coefficients for natural channels and Flood Plains* (Technical Report, Geological Survey Water-Supply) (p. 38). Washington, DC: United States Government Printing office.

Ashley, R., Blackwood, D., Butler, D., Jowitt, P., Oltean-Dumbrava, C., Davies, J., … Bouchart, F. (2002). Making more sustainable decisions for asset investment in the water industry - sustainable water industry asset resource decisions - The SWARD Project. In E. W. Strecker & W. C. Huber (Eds.), *Global Solutions for Urban Drainage* (pp. 1–16). Reston, VA: American Society of Civil Engineers.

Bamfield, B. (2005). *Whole Life Costs & Living Roofs* (pp. 1–12). Bridgewater: The Springboard Centre, Forum American Bar Association.

Barbosa, A. E., Fernandes, J. N., & David, L. M. (2012). Key issues for sustainable urban stormwater management. *Water Research*, *46*(20), 6787–6798.

Birgani, Y. T. (2013). Role of resilience in sustainable urban stormwater management. *Scientific Professional Quarterly*, *1*(1), 42–50.

Butler, D., & Davies, J. (2011). *Urban Drainage* (3rd ed.). Abingdon: SPON Press.

Butler, D., Farmani, R., Fu, G., Ward, S., Diao, K., & Astaraie-Imani, M. (2014). A new approach to urban water management: Safe and Sure. *Procedia Engineering*, *89*, 347–354.

Centre for Ecology and Hydrology. (1999). *Flood Estimation Handbook*. Wallingford, England: Centre for Ecology and Hydrology (formally the Institute of Hydrology).

Centre for Ecology and Hydrology. (2013). *Flood estimation handbook web service [Online].* Retrieved from https://fehweb.ceh.ac.uk/

Centre for Water Systems. 2017. *CADDIES framework.* Retrieved from http://emps.exeter.ac.uk/engineering/research/cws/resources/caddies-framework/caddies-2d/

Chen, A., Hammond, M., & Djordjevic, S. (2016). From hazard to impact: Flood damage assessment tools for mega cities. *Natural Hazards*, *82*(2), 857–890.

Chocat, B., Ashley, R., Marsalek, J., Matos, M. R., Rauch, W., Schilling, W., & Urbonas, B. (2007). Toward the sustainable management of urban stormwater. *Indoor and Built Environment*, *16*(3), 273–285.

Ciria. (2015). *BeST (Benefits of SuDS Tool) W045d BeST – User Manual Release Version 1* (pp. 1–22). London: CIRIA.

Collins, K. A., Hunt, W. F., & Hathaway, J. M. (2008). Evaluation of various types of permeable pavement with respect to water quality improvement and flood control. In N. She (Ed.), *Low Impact Development* (pp. 275–289). Reston, VA: American Society of Civil Engineers.

Committee on Climate Change. (2015). *Committee on climate change—Progress in preparing for climate change.* 2015 Report to Parliament.

Cranfield Soil and Agrifood Institute. (2017). *Landis soilscapes [Online].* Retrieved from www.landis.org.uk/soilscapes

DCLG. (2015). *English Housing Survey*. London: Department for Communities and Local Government.

DEFRA. (2012). *Climate Change Risk Assessment for the Floods and Coastal Erosion Sector*. London: Department for Environment, Food and Rural Affairs.

Douglas, I., Garvin, S., Lawson, N., Richards, J., Tippett, J., & White, I. (2010). Urban pluvial flooding: A qualitative case study of cause, effect and nonstructural mitigation. *Journal of Flood Risk Management*, *3*(2), 112–125.

Elliott, A. H., & Trowsdale, S. A. (2007). A review of models for low impact urban stormwater drainage. *Environmental Modelling and Software*, *22*(3), 394–405.

Ellis, J. B., Deutsch, J. C., Mouchel, J. M., Scholes, L., & Revitt, M. D. (2004). Multicriteria decision approaches to support sustainable drainage options for the treatment of highway and urban runoff. *Science of the Total Environment*, *334–335*, 251–260.

Ellis, J. B., & Revitt, D. M. (2010). The management of urban surface water drainage in England and Wales. *Water and Environment Journal*, *24*(1), 1–8.

Emanuelsson, M. A. E., Mcintyre, N., Hunt, C. F., Mawle, R., Kitson, J., & Voulvoulis, N. (2014). Flood risk assessment for infrastructure networks. *Journal of Flood Risk Management*, *7*(1), 31–41.

Environment Agency. (2007). *Cost-Benefit of SUDS Retrofit in Urban Areas*. Bristol: Environment Agency.

Environment Agency. (2010a). *The Costs of the Summer 2007 Floods in England*. Bristol: Environment Agency.

Environment Agency. (2010b). *Flood and Coastal Erosion Risk Management Appraisal Guidance*. Bristol: Environment Agency.

Environment Agency. (2013). *What Is the Updated Flood Map for Surface Water?* (pp. 1–51). Bristol: Environment Agency.

Environment Agency. (2015). *Cost Estimation for SUDS - Summary of Evidence*. Bristol: Environment Agency.

Fletcher, T. D., Shuster, W., Hunt, W. F., Ashley, R., Butler, D., Arthur, S., … Viklander, M. (2015). SUDS, LID, BMPs, WSUD and more – The evolution and application of terminology surrounding urban drainage. *Urban Water Journal*, *12*(7), 525–542.

Ghimire, B., Chen, A. S., Guidolin, M., Keedwell, E. C., Djordjević, S., & Savić, D. a. (2013). Formulation of a fast 2D urban pluvial flood model using a cellular automata approach. *Journal of Hydroinformatics*, *15*(3), 676–686.

Gibson, M. J., Savic, D. A., Djordjevic, S., Chen, A. S., Fraser, S., & Watson, T. (2016). Accuracy and computational efficiency of 2D Urban surface flood modelling based on cellular automata. *Procedia Engineering*, *154*, 801–810.

Goonetilleke, A., Thomas, E., Ginn, S., & Gilbert, D. (2005). Understanding the role of land use in urban stormwater quality management. *Journal of Environmental Management*, *74*(1), 31–42.

Guidolin, M., Chen, A. S., Ghimire, B., Keedwell, E. C., Djordjević, S., & Savić, D. A. (2016). A weighted cellular automata 2D inundation model for rapid flood analysis. *Environmental Modelling & Software*, *84*, 378–394.

Hamill, L. (2001). *Understanding Hydraulics* (2nd ed.). Basingstoke, England: Palgrave Macmillan.

Hammond, M. J., Chen, A. S., Djordjević, S., Butler, D., Mark, O., Chen, A. S., … Urban, O. M. (2016). Urban flood impact assessment: A state-of-the-art review. *Urban Water Journal*, *12*(1), 14–29.

HM Government. (2010). *Flood and Water Management Act 2010*. London: The Stationary Office.

HM Treasury. (2013). *Green Book Supplementary Guidance: Discounting*. London: HM Treasury.

House of Commons (2016). In The Environment, Food and Rural Affairs Committee (Ed.), *Future Flood Prevention Second Report of Session 2016–17*. London: Department of Environment, Food and Rural Affairs.

Howard, G., Charles, K., Pond, K., Brookshaw, A., Hossain, R., & Bartram, J. (2010). Securing 2020 vision for 2030: Climate change and ensuring resilience in water and sanitation services. *Journal of Water and Climate Change*, *1*(1), 2.

Hunter, N. M., Bates, P. D., Neelz, S., Pender, G., Villanueva, I., Wright, N. G., … Mason, D. C. (2008). Benchmarking 2D hydraulic models for urban flooding. *Proceedings of the Institution of Civil Engineers - Water Management*, *161*(1), 13–30.

Hunter Water Corporation. (2013). *Operating and Maintenance Costs*. New South Wales: Hunter Water Corporation.

IPCC. (2014). *Climate Change 2014: Synthesis Report.* Contribution of Working Groups I, II and III to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change. Geneva, Switzerland: Intergovernmental Panel on Climate Change.

Jones, M. R., Fowler, H. J., Kilsby, C. G., & Blenkinsop, S. (2013). An assessment of changes in seasonal and annual extreme rainfall in the UKbetween 1961 and 2009. *International Journal of Climatology*, *33*(5), 1178–1194.

Makropoulos, C., Liu, S., Natsis, K., Butler, D., & Memon, F. A. (2007). Supporting the choice, siting and evaluation of sustainable drainage systems in new urban developments. *Water Practice and Technology*, *2*(2).

Makropoulos, C. K., Natsis, K., Liu, S., Mittas, K., & Butler, D. (2008). Decision support for sustainable option selection in integrated urban water management. *Environmental Modelling and Software*, *23*(12), 1448–1460.

Martin, B. (2008). *The Dynamic Stormwater Response of a Green Roof.* Canada: Doctoral dissertation, University of Guelph.

Mentens, J., Raes, D., & Hermy, M. (2006). Green roofs as a tool for solving the rainwater runoff problem in the urbanized 21st century? *Landscape and Urban Planning*, *77*(3), 217–226.

Merz, B., Kreibich, H., Schwarze, R., & Thieken, A. (2010). Review article 'assessment of economic flood damage'. *Natural Hazards and Earth System Science*, *10*(8), 1697–1724.

O'Donnell, E. C., Lamond, J. E., & Thorne, C. R. (2017). Recognising barriers to implementation of blue-green infrastructure: A Newcastle case study. *Urban Water Journal*, *14*(9), 964–971.

Office for National Statistics (ONS). (2018). *RPI all items: Percentage change over 12 months.* Retrieved from https://www.ons.gov.uk/economy/inflationandpriceindices/timeseries/czbh/mm23

Ossa-Moreno, J., Smith, K. M., & Mijic, A. (2017). Economic analysis of wider benefits to facilitate SuDS uptake in London, UK. *Sustainable Cities and Society*, *28*, 411–419.

Parker, D. J., Priest, S. J., & McCarthy, S. S. (2011). Surface water flood warnings requirements and potential in England and Wales. *Applied Geography*, *31*(3), 891–900.

Paudel, B. (2009). *Calibration and testing of simple mass balance model for quantifying Stormwater management benefit of an extensive green roof* (Electronic Theses and Dissertations). Canada: Doctoral dissertation, University of Windsor.

Penning-Rowsell, E., Viavattene, C., & Parode, J. (2010). The benefits of flood and coastal risk management: A handbook of assessment techniques-2010 (multi-coloured manual). *Flood Hazard Research*. London: Middlesex Univeristy Press.

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

Pratt, C., Wilson, S., & Cooper, P.; Construction Industry Research and Information Association. (2002). *Source Control Using Constructed Pervious Surfaces: Hydraulic, Structural and Water Quality Performance Issues*. London: CIRIA.

Priest, S. J., Parker, D. J., Hurford, a. P., Walker, J., & Evans, K. (2011). Assessing options for the development of surface water flood warning in England and Wales. *Journal of Environmental Management*, *92*(12), 3038–3048.

Roebuck, R. M., Oltean-Dumbrava, C., & Tait, S. (2011). Whole life cost performance of domestic rainwater harvesting systems in the United Kingdom. *Water and Environment Journal*, *25*(3), 355–365.

Schanze, J. (2017). Nature-based solutions in flood risk management – Buzzword or innovation? *Journal of Flood Risk Management*, *10*(3), 281–282.

Stovin, V., Vesuviano, G., & Kasmin, H. (2012). The hydrological performance of a green roof test bed under UK climatic conditions. *Journal of Hydrology*, *414*, 148–161.

Stovin, V. R., Swan, A. D., Stovin, V. R., & Swan, A. D. (2007). *Retrofit SuDS—Cost Estimates and Decision-Support Tools* (pp. 207–214). London: Thomas Telford Ltd.

Syme, W. J. (2008). *Flooding in Urban Areas - 2D Modelling Approaches for Buildings and Fences [online].* In 9th National Conference on Hydraulics in Water Engineering: Hydraulics 2008. (pp. 25–32). Barton: A.C.T.: Engineers Australia.

United Nations Food and Agriculture Organisation. (2017). *Irrigation water management. Annex 2 Infiltration rate and infiltration rate. [Online].* Retrieved from http://www.fao.org/docrep/s8684E/s8684e0a.htm

University of Exeter. (2014). *Flood Damage Model Guidelines. CORFU Project (WP3 Output)*. Exeter: University of Exeter.

Webber, J. L., Booth, G., Gunasekara, R., Fu, G., & Butler, D. (2018). Validating a rapid assessment framework for screening surface water flood risk. *Water and Environment Journal*. Retrieved from https://onlinelibrary.wiley.com/doi/full/10.1111/wej.12415.

Webber, J. L., Fu, G., & Butler, D. (2018). Rapid assessment of surface water flood intervention performance. *Water Science & Technology*, *77*(8), 2084–2092.

Webber, J. L., Gibson, M. J., Chen, A. S., Fu, G., & Butler, D. (2018). Rapid assessment of surface water flood management options in urban catchments. *Urban Water Journal*, *15*(3), 210–217.

Woods Ballard, B., Kellagher, R., Martin, P., Jefferies, C., Bray, R., & Shaffer, P. (2007). *The SuDS Manual C697*. London, England: CIRIA.

Woods Ballard, B., Wilson, S., Udale-Clarke, H., Illman, S., Scott, T., Ashley, R., & Kellagher, R. (2015). *The SuDS Manual (C753)*. London, England: CIRIA.

XP Solutions. (2017). *Manning's roughness tables [Online].* Retrieved from http://xpsolutions.com/webhelp/SECTION_11__GLOBALS/11_4_Infiltration/Manning_s_Roughness.htm

Young, K. D., Younos, T., Dymond, R. L., Kibler, D. F., & Lee, D. H. (2010). Application of the analytic hierarchy process for selecting and modelling Stormwater Best management practices. *Journal of Contemporary Water Research & Education*, *146*(1), 50–63.

Zachary Bean, E., Frederick Hunt, W., & Alan Bidelspach, D. (2007). Evaluation of four permeable pavement sites in eastern North Carolina for runoff reduction and Water quality impacts. *Journal of Irrigation and Drainage Engineering*, *133*(6), 583–592.

Zhou, Q. (2014). A review of sustainable Urban drainage systems considering the climate change and urbanization impacts. *Water*, *6*(4), 976–992.

How to cite this article: Webber JL, Fu G, Butler D. Comparing cost-effectiveness of surface water flood management interventions in a UK catchment. *J Flood Risk Management*. 2019;e12523. https://doi.org/10.1111/jfr3.12523