Title: Using green infrastructure as a social equity approach to reduce flood risks and address climate change impacts: a comparison of performance between cities and towns

1. Introduction

Flood risk management is confronted with significant challenges related to population growth, accelerated urbanisation and expansion of impervious surfaces. These challenges are compounded as a result of climate change (IPCC, 2014). The combination of urbanisation pressures and climate change projections suggests a significant increase in future flood risk (Alves, Gomez, Vojinovic, Sanchez, & Weesakul, 2018). These risks tend to be even higher for underprivileged people and communities (Bird et al., 2013). As a response to the uncertainty of climate change and increased flood risks, the use of non-traditional drainage measures, such as green infrastructure[1] (GI), has been increasingly promoted (Fletcher et al., 2015; Mell, 2017). In particular, it is argued that using GI options in combination with conventional drainage systems could improve flood risk management outcomes (Mguni, Herslund, & Jensen, 2015), while also delivering a range of wider social, environmental and health benefits (Benedict & McMahon, 2006).

Many scholars (e.g., Carter, Handley, Butlin, & Gill, 2018; Ercolani, Chiaradia, Gandolfi, Castelli, & Masseroni, 2018; Goncalves et al., 2018) have linked GI to flood risk. For example, Jia et al. (2015) stated that GI is efficient at minimising flood risks as it can lower loads on stormwater infrastructure, increase rainfall infiltration and retain

[1] GI can be described as an instrument for improving human well-being via its environmental, social and economic values, based on the multifunctional use of ecosystems (Vallecillo et al., 2018) GI comprises both natural, semi-natural, and artificial networks of ecological systems at all spatial scales with significant multifunctional interconnection roles, contributing to the protection of biological diversity and preserving habitat integrity (Alves et al., 2018). In terms of flood risk management, GI is related to other concepts used in different regions of the world, such as nature-based solutions, water sensitive urban design, blue-green infrastructure, low impact development – LID, and sponge cities (Fletcher et al., 2015).

and store stormwater runoff. Goncalves et al. (2018) similarly assessed how different GI combinations can lower flood risks in areas of high-intensity rainfall. Their results demonstrate that GI can, on average, reduce total flood volume between 30% and 75%. Bai, Zhao, Zhang, and Zeng (2018) evaluated scenarios with four different types of GI and concluded that the combination of alternatives with good infiltration and high storage capacity had the best performance in minimising runoff. The authors also stated that, if properly planned, GI is effective in reducing the risk of urban flooding due to extreme rainfall events.

Despite growing evidence of performance, researchers also emphasise the difficulty in incorporating GI within decision making systems. For example, the assessment of GI performance in minimising flood risk can be a major challenge due to time and cost involved in addressing multiple potential scenarios (e.g., different combinations of types and quantities of GI, suitable locations, different temporal scales) (Webber et al., 2019). The use of computational models can help assess potential challenges to GI implementation by analysing trajectories, emerging risks and informing possible strategic decisions by recognising and mitigating unforeseen or unintended outcomes (Brewington, Keener, & Mair, 2019; Toimil, Losada, Nicholls, Dalrymple, & Stive, 2020). These models can be more efficient if they connect socio-political issues and development demands. For instance, the introduction of social-technical dimensions can inform land-use planning decisions for low socio-economic urban areas, which may be in greater need of new public green spaces and more likely to be severely affected by flood events (Toxopeus et al., 2020), whilst also considering land-use planning constraints and climate change projections (Reu Junqueira, Serrao-Neumann, & White, 2021b).

Carefully identifying areas for GI implementation is important as cities need to avoid locating those in areas where they can exacerbate patterns of social exclusion and gentrification (Van der Jagt, Bernadett, Shunsuke, & Wakana, 2021), especially because GI can be employed to revitalise degraded neighbourhoods. While this usually attracts new investors, it may ignore underlying spatial concerns (e.g., processes of social exclusion and displacement) and fail to address wider societal challenges (e.g., the need to protect and restore ecosystems which can greatly contribute to public health and well-being) (Sekulova et al., 2021). As such, the incorporation of social-equity considerations can help cities improve inequalities in the provision of environmental amenities or prevent green gentrification (Anguelovski, Connolly, Masip, & Pearsall, 2018). For this, GI implementation requires careful consideration of context specific distributive, operational and environmental dimensions of equity so that it is neither excessive nor insufficient (Pallathadka et al. 2022, La Rosa & Pappalardo 2020). Adding to this, for GI implementation to be considered successful careful consideration should also be given to how it is framed, designed and implemented to ensure it effectively contributes to achieving sustainability and social justice goals (Shi, 2020; Van der Jagt et al., 2021). It is challenging, however, to develop spatially-explicit GI scenario analysis able to link urban, climate and social aspects so as to provide more certainty for planners and decision-makers (Brewington et al., 2019).

From an urban retrofitting perspective, a key benefit linked to GI is their ability to have good performance in terms of infrastructure function (such as storage and runoff peak rate reduction) (Zhu & Chen, 2017), while potentially minimising effects on private land or property (Dover, 2015). Green roofs, planter boxes, bio-retention cells (also known as bioswales), permeable pavements, rain gardens and urban forestry are viable retrofit options for enhancing urban resilience to flooding (Reu Junqueira, Serrao-Neumann, & White, 2021a). However, many of these solutions require uptake by private landowners and consequently are more complex to initiate at scale. In general, bio-retention cells, rain gardens and permeable pavements are solutions that can be conveniently retrofitted in the public realm as they demand small spaces to be implemented and may not have to be negotiated with private property owners (e.g., as seen in the case of green roofs installed in central business district areas) (Sun, Li, Liu, Xu, & Liu, 2014). For example, bio-retention cells can be incorporated along sidewalks, rain gardens can be introduced in small areas such as parks and public gardens, and permeable pavements can replace the cover in existing parking lots and driveways.

Flood risk management, land-use planning and decision-making processes also need to consider a greater range of temporal and spatial scales to deal with future climate change impacts (Ran & Nedovic-Budic, 2016). Flood risks are inherently uncertain but are likely to increase in the future (Struck & Lichten, 2010), challenging urban planning processes and political structures that usually have a presentist tendency (White & Haughton, 2017). To address this impasse, temporal scales should consider both current conditions and climate change projections. Additionally, the consideration of a variety of spatial scales in flood risk management, from the street to the city, can lead to improved land-use planning decisions (Jayawardena & van Roon, 2017), as water pathways are interconnected and part of a wider urban system.

While the hydrological performance of GI has received scholarly attention (Ercolani et al., 2018; C. Li et al., 2018; Wang, Zhang, Cheng, & Tan, 2019), there is still lack of knowledge regarding the different efficiency of these options within differing contexts. This gap in knowledge also extends to the incorporation of both future climate risks and equity dimensions to better inform land-use planning decisions (Reu Junqueira et al., 2021b). Aotearoa – New Zealand is no exception. While the country has been using GI alternatives for over a decade, they have not been effectively used in spatial planning nor for urban stormwater management (Ira & Simcock, 2019). Additionally, many areas in the country have a legacy of past unequal fund allocation both across regions and within jurisdictions, with some localities (e.g., Auckland north) receiving twice as much investment as others to create new or enhance existing public open spaces (Biddle, 2020; Latif, 2021). Hence, it is important to carefully consider where GI alternatives may be implemented so as to avoid exacerbating such resource disparity and further contribute to green gentrification. This paper contributes to filling this gap in knowledge by developing a method to assess the performance of different combinations and quantities of GI options (i.e., bio-retention cells, permeable pavement and rain gardens) in reducing both pluvial flood risks under different extreme rainfall events (various intensities and durations) and link this to social equity issues. This study comprises two case studies in Aotearoa – New Zealand which are both exposed to urban flooding, a highly urbanised city (especially pluvial flash flooding) and a small regional town (both pluvial and fluvial flooding). The findings, however, may be applicable to other similar urban contexts, and towns and cities beyond Aotearoa – New Zealand.

2. Research approach and design

This study adopted a scenario analysis approach (Amer, Daim, & Jetter, 2013) to evaluate the performance of different GI alternatives under various rainfall event intensities, and applied this to two case study areas in Aotearoa-New Zealand. Scenario analysis is a method used to identify trends and challenges, and help decision-makers to understand and evaluate the uncertainties affecting decisions (Varum & Melo, 2010). Scenario analysis is used in different fields, including land-use planning (Hashimoto et al., 2018; He, Li, Zhang, Liu, & Zhang, 2017), climate change adaptation (Bannayan & Eyshi Rezaei, 2012; Vermaat et al., 2017), water resource management (Graveline, Aunay, Fusillier, & Rinaudo, 2014; Matias, Matias, Johnes, & Johnes, 2012) and GI performance (C. Li et al., 2018; Wang, Zhang, Su, Dong, & Tan, 2018), and thus fits with existing decision-making practices. It typically enables the assessment of multiple strategies, their comparison against both business-as-usual scenarios and each other to determine their potential and limitations (Amer et al., 2013). In our study, GI scenarios were developed based on three aspects: social-equity dimensions; projections of rainfall data under climate change conditions and GI types and spatial configurations (see Figure 1). A five-stage approach was used to carry out the study, each of which comprising of a range of steps. An overview is shown in Figure 2.

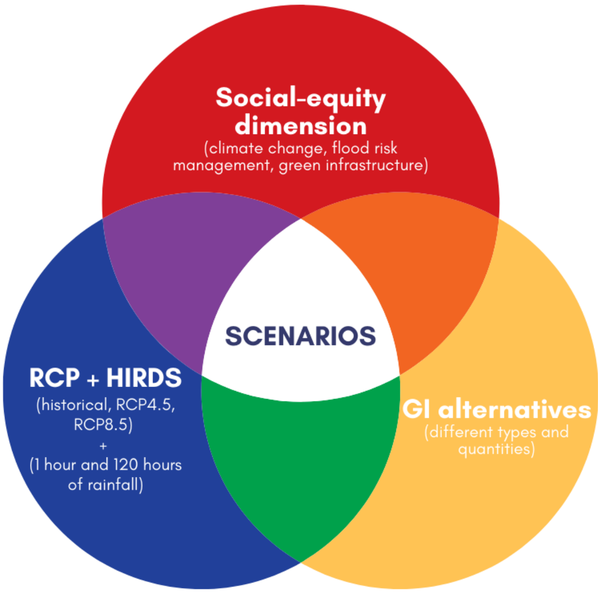

Figure 1 – Key scenario development aspects

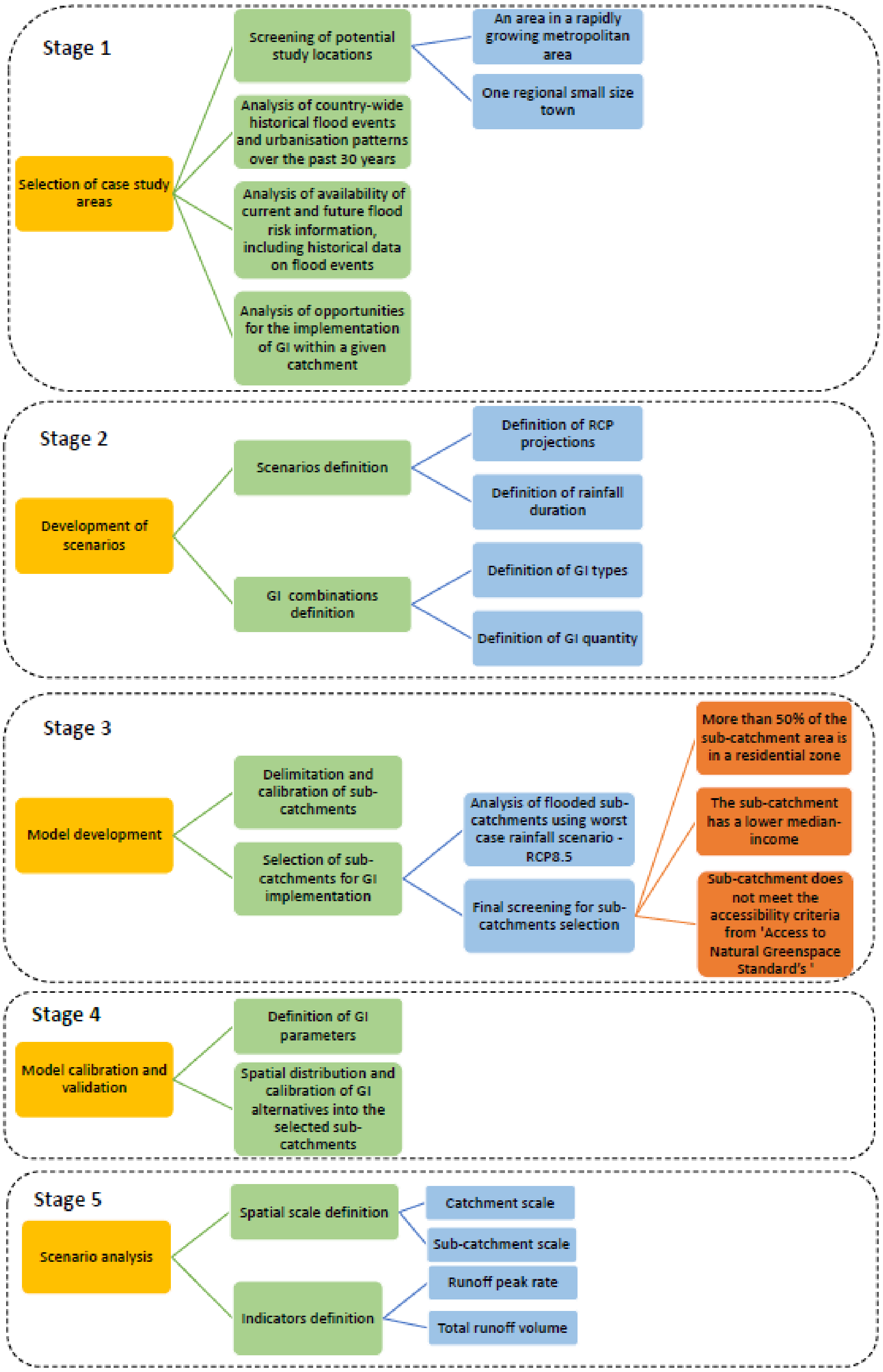

Figure 2 - Overview of research design, including stages and associated steps involved in the scenario development and analysis process

2.1 Stage 1: Selection of case study areas

Similar to other regions worldwide, climate change is likely to increase the frequency and intensity of rainfall events affecting Aotearoa-New Zealand and increase the risk of urban flooding (Ministry for the Environment, 2020). The two case study locations were selected on the following criteria:

(i) Screening of urban areas at risk of flooding to identify:

a. an area of urban infill development in a rapidly growing metropolitan area;

b. one regional small size town with a population between 10,000 and 15,000 inhabitants;

(ii) Country-wide historical analysis of flood events and urbanisation patterns over the past 30 years;

(iii) Availability of current and future flood risk information, including historical data on flood events (e.g., significant flood damages reported within the catchment, quality of data sets); and,

(iv) Opportunities for the implementation of GI within a given catchment.

We determined the selection of a large metropolitan city and a smaller town as case study locations would enable a more holistic analysis of GI performance when applied to different urbanisation patterns. To this end, Auckland, in the North Island, and Gore, in the South Island, were selected (see Figure 3). Both locations have been identified as being at high risk from climate change impacts (NIWA, 2018). Auckland’s selected catchment area is mostly residential with some local schools and industrial sites. Gore’s selected catchment area is primarily residential, with some significant industrial and commercial zones, and has a lower impervious area compared to Auckland’s (see Table 1 for more details).

Table 1 - Main characteristics of the two case study areas

Attribute	Auckland	Gore
Total area (km²)	~22	~14
Population (inhabitants)*	79,356	12,396
Population density (inhabitants/km²)	3,595	900
Private dwellings*	26,169	5,577
Flood vulnerability area (km²)	~4	5
Impervious area (%)**	~34	~23

*Stats NZ 2018 Census
** Considering building footprints and roads

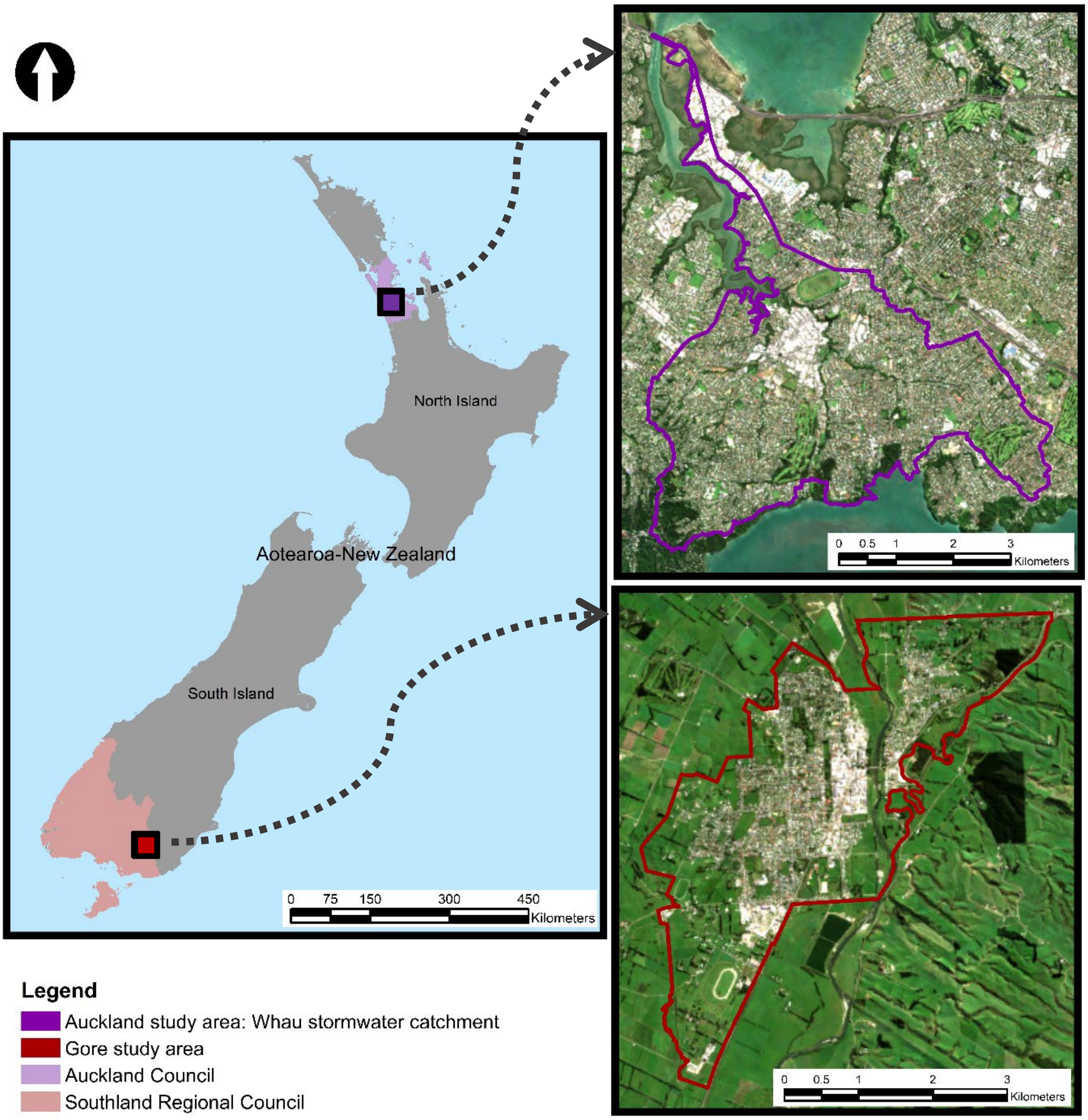

Figure 3- Location of case study areas

2.2 Stage 2: Development of scenarios

Step 1: Definition of scenarios

To account for climate change impacts and associated flood risk, we used the Representative Concentration Pathways (RCPs) described in the Assessment Report (AR-5) prepared by the International Panel on Climate Change (IPCC, 2014) and High Intensity Rainfall Design System V4 (HIRDS) projections prepared by New Zealand's National Institute of Water and Atmospheric Research (NIWA), which estimates how the intensity of rainfall events differs under conditions of climate change (NIWA, 2018 p.8). The RCPs include a low greenhouse gases (GHG) emissions scenario (RCP2.6), two mid emissions scenarios (RCP4.5 and RCP6.0) and a high emissions scenario (RCP8.5) (IPCC, 2014). HIRDS provides a variety of potential rainfall depths and intensities for 1-hour and 120-hour events under various climate change projections (NIWA, 2018). Our scenarios used RCP4.5 and RCP8.5 data for the period 2031-2050. Both RCPs are familiar to decision-makers as they were considered in the National Climate Change Risk Assessment for New Zealand (Ministry for the Environment, 2020), which seeks to assist decision-makers to properly evaluate climate change risks.

Six possible scenarios (S1-6) were then developed to evaluate the risks and opportunities for effectively implementing GI alternatives in the selected case study areas. From a GHG emissions perspective, scenarios considered: (i) a minimal or null approach (historical); (ii) a moderate approach (RCP4.5); and, (iii) a worst-case approach (RCP8.5). As GI alternatives can perform differently during short and long rainfall events (Qin, Li, & Fu, 2013; Wang et al., 2019), scenarios considered events of 1 and 120 hours of duration according to HIRDS projections (NIWA, 2018). This resulted in six different temporal scales; two for each RCP for each case study area assuming no changes to current land-use (see Table 2).

Table 2- High-intensity rainfall events and duration used to develop contrasting scenarios

Scenario	Duration of Rainfall	High-intensity rainfall event
S1	1 hour	Historical*
S2		RCP 4.5 (range year 2031-2050)
S3		RCP 8.5 (range year 2031-2050)
S4	120 hours	Historical*
S5		RCP 4.5 (range year 2031-2050)
S6		RCP 8.5 (range year 2031-2050)

*The historical rainfall scenario is based on currently available data to enable the evaluation of the current condition and was used in the HIRDS analysis to derive future extreme rainfall intensities.

Step 2: GI combination definition

Different types and quantities of GI were added to the current urban fabric of selected catchments. This aimed to evaluate the best options for reducing stormwater runoff and/or stormwater retention (or flood mitigation capacity) based on GI types and their spatial configuration/distribution. A total of 14 GI possibilities were defined on the basis of the potential combination of selected GI alternatives (see Table 3) and were tested in all six scenarios. Selected GI types included bio-retention cells (BR), permeable pavement (PP) and rain gardens (RG) as these are known to be suitable urban retrofit alternatives (Dover, 2015; Sun et al., 2014). That is, these alternatives require minimum space to be implemented (e.g., BR and RG) and have a relatively low cost for implementation (e.g., BR and PP) (Dover, 2015; Zhu & Chen, 2017) while also being able to absorb or reduce stormwater runoff (e.g., BR), slow down stormwater runoff (e.g., PP) (Wang et al., 2019) and provide temporary storage (e.g., RG) (Zhang, Oyake, Morimoto, Niwa, & Shibata, 2019). Different quantities of GI were initially investigated (with 1, 2, 5, 10 and 20% of each sub-catchment area converted into GI alternatives), but some were considered unsuitable for different reasons (such as minimal variation from the business-as-usual situation or availability of areas for implementation). For example, anything beyond 10% conversion would require significant intervention on private properties and therefore would not be easily implemented within a given sub-catchment without substantial costs to local authorities. In particular, prohibitive costs associated with land resumption would apply to most cases and in cities like Auckland, where both past development did not consider the local hydrology (Silva, 2018) and land has reached exorbitant prices (Fernandez & Martin, 2020). In the end, a total GI implementation area of 5% and 10% within each sub-catchment was selected for analysis because both percentages would require minimal space and costs and consequently be a politically viable urban retrofit option, whilst showing to be effective in reducing localised flood risk (Goncalves et al., 2018).

Table 3 – Different combinations of GI types and quantity applied to each of the six scenarios

GI Combination	GI alternative	Quantity of GI alternative	Total of GI in the sub-catchment
BR	Bio-retention cells	5%	5 %
PP	Permeable pavement	5%	
RG	Rain garden	5%	
BR+PP	Bio-retention cells	2.5%	
	Permeable pavement	2.5%	
RG+PP	Rain garden	2.5%	
	Permeable pavement	2.5%	
RG+BR	Rain garden	2.5%	
	Bio-retention cells	2.5%	
RG+BR+PP	Rain garden	1.5%	
	Bio-retention cells	1.5%	
	Permeable pavement*	2%	
BR	Bio-retention cells	10%	10 %
PP	Permeable pavement	10%	
RG	Rain garden	10%	
BR+PP	Bio-retention cells	5%	
	Permeable pavement	5%	
RG+PP	Rain garden	5%	
	Permeable pavement	5%	
RG+BR	Rain garden	5%	
	Bio-retention cells	5%	
RG+BR+PP	Rain garden	3%	
	Bio-retention cells	3%	
	Permeable pavement*	4%	

BR - Bio-retention cells; PP - Permeable pavement; RG - Rain garden

* GI alternatives were not equally distributed when using the combination of RG+BR+PP, due to urban design patterns.

2.3 Stage 3: Model development

The study used the following software to process the data and simulate GI performance under different scenarios: Personal Computer Stormwater Management Model (PCSWMM Professional 2D) and ESRI's ArcGIS 10.8. Stormwater Management Model (SWMM) software is frequently used for hydrological modelling and analysis of GI (Goncalves et al., 2018). Added to that, both software were chosen as they allow the integration between hydrological and spatial analysis as well as the implementation of multiple temporal scales and GI designs (Reu Junqueira et al., 2021b). The model development comprised two main steps: delineation and calibration of the sub-catchments, and selection of the sub-catchments for GI implementation.

Step 1: Delineation and calibration of sub-catchments

The case study areas were divided into sub-catchments[2], and the outlet point for each sub-catchment was identified. The sub-catchments were defined in PCSWMM using GIS data (DEM and river lines). This process generated 184 sub-catchments for Auckland and 135 sub-catchments for Gore[3].

Sub-catchments were then manually calibrated in PCSWMM and main parameters were defined. Impervious areas such as roads, sidewalks, parking lots and building footprints were calculated using ArcGIS. Data were collected from local and regional territorial authorities, Land Information New Zealand (LINZ), Land Resource Information Systems and Manaaki Whenua - Landcare Research and Stats NZ (for more information about data used and the main parameters refer to Supplementary Material Tables SM1 and SM2).

Step 2: Equity and the Selection of sub-catchments for GI implementation

This step involved the selection of sub-catchments for the implementation of GI alternatives. Firstly, a simulation was run considering the highest intensity GHG emissions (RCP8.5) without the implementation of GI alternatives to identify which sub-catchments were likely to flood. Then, a social equity dimension was applied to these sub-catchments to determine preferred locations for GI alternatives. The stage is important because: (i) socio-economically disadvantaged residents and communities tend to be more impacted by the effects of extreme weather events (Bird et al., 2013); (ii) new GI in higher socio-economic areas may lead to increases in housing costs and property values, enhancing green gentrification (Anguelovski et al., 2018), (iii) the implementation of GI as a retrofit option of existing built-up areas may be costly, hence local authorities need to be selective in justifying their implementation (Webber et al.,

[2] According to the User's Guide to SWMM5, sub-catchments are hydrological land units whose topographic and drainage system elements direct surface runoff to a single discharge point. (James, Rossman, & James, 2010)
[3] Mataura river catchment was used to generate and calibrate the model, however, the analysis focused on Gore's urban area.

2019); and, (iv) there is a need to redirect public investment towards more socio-economically deprived communities, which both struggle to win arguments for scarce public funds and have less access to green space (Hoffimann, Barros, & Ribeiro, 2017). To narrow down the number of sub-catchments and define which ones would be suitable for implementing GI options, the following screening criteria were applied:

(i) Sub-catchment has more than 50% of its total area occupied by residential areas;

(ii) Residents in the sub-catchment have lower median annual income (0-$30,000) compared with the average annual income in Aotearoa-New Zealand; and,

(iii) The current sub-catchment does not meet the accessibility criteria established by the Access to Natural Greenspace Standard (Natural England, 2010) – that is, green areas of at least 2 hectares in size and no more than 300 metres from any given residence.

Out of the 184 Auckland sub-catchments and 135 Gore sub-catchments, 12 and 4, respectively met all three criteria and were selected for analysis (see Figure 4).

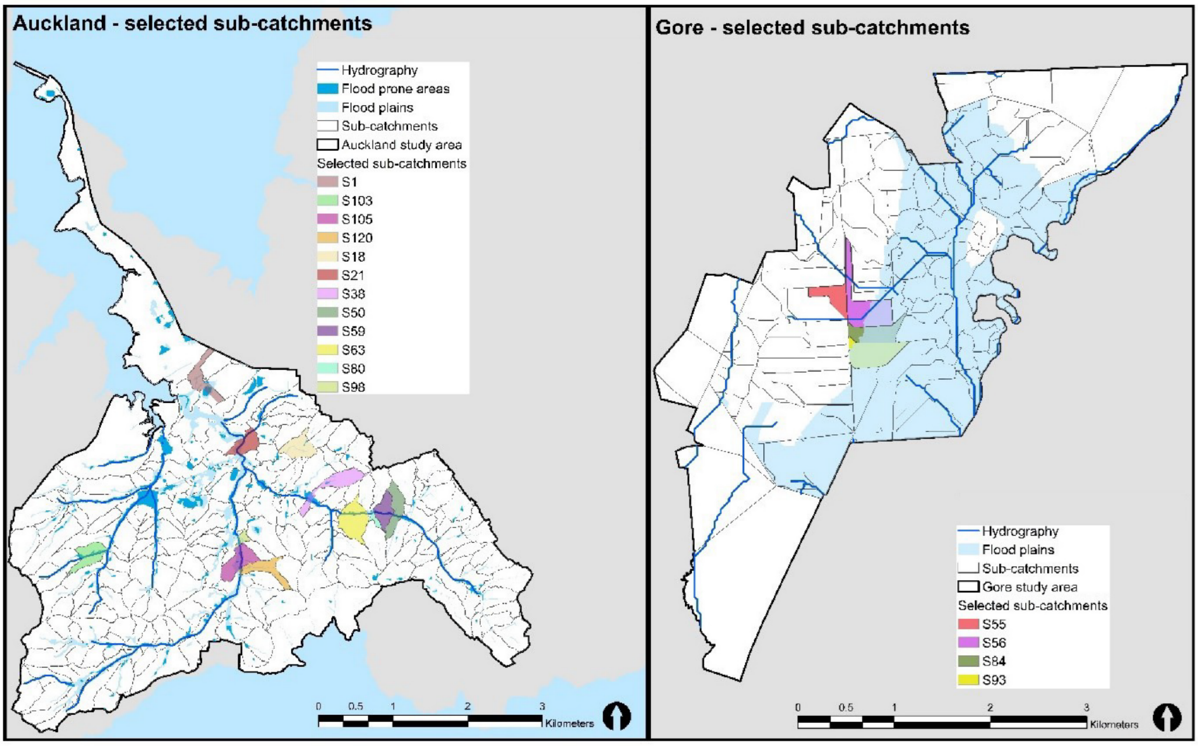

Figure 4 – Case study areas and selected sub-catchments (sub-catchment numbers is automatically generated by PCSWMM)

2.4 Stage 4: Model calibration and validation

The implementation of GI alternatives in PCSWMM was performed in two steps. First, several parameters (such as layer thickness, berm heights, porosity, hydraulic conductivity) were selected and their technical properties defined for the GI alternative (see Table 4). The second phase included the incorporation and spatial distribution of GI in the sub-catchments of the study areas. This was done following the 'User's Guide to SWMM5' (James et al., 2010) range suggestions for the parameters, which are based on many GI design manuals.

Table 4- Main parameters applied to GI alternatives

Parameters		Bio-retention cells	Permeable pavement	Rain garden
Surface	Berm height (mm)	150	0.0	450
	Vegetation volume (fraction)	0.5	0.0	0.5
	Surface roughness (Manning's n)	0.1	0.012	0.24
	Surface slope	0.5	0.5	0.5
Pavement	Thickness (mm)	-	100	-
	Void ratio (voids/ratio)	-	0.15	-
	Impervious surface (fraction)	-	0	-
	Permeability (mm/hr)	-	300	-
	Clogging factor	-	0	-

Parameters		Bio-retention cells	Permeable pavement	Rain garden
	Regeneration interval (days)	-	0	-
	Regeneration fraction	-	0	-
Soil	Thickness (mm)	600	-	600
	Porosity (volume fracion)	0.5	-	0.5
	Field capacity (volume fraction)	0.2	-	0.2
	Wilting point (volume fraction)	0.1	-	0.1
	Conductivity (mm/hr)	76.2	-	76.2
	Conductivity slope	10	-	10
	Suction head (mm)	60.9	-	60.9
Storage	Thickness (mm)	600	300	-
	Void ratio (voids/solids)	0.75	0.4	-
	Seepage rate (mm/hr)	750	750	750
	Clogging factor	0	0	-
Underdrain	Drain coefficient (mm/hr)	0.5	0	-
	Drain exponent	0.5	0.5	-
	Drain offset height (mm)	150	100	-
	Open level (mm)	0	0	-
	Closed level (mm)	0	0	-

2.5 Stage 5: Scenario analysis

GI performance was evaluated using two different spatial scales, catchment and sub-catchment. The catchment-scale allows an evaluation of the whole system affected, while the sub-catchment addresses the impact of GI intervention at the surrounding area. Two indicators were used to assess the efficiency of GI implementation for each of the six scenarios: (i) runoff peak rate (catchment and sub-catchment); and, (ii) total runoff volume (catchment and sub-catchment). Both indicators are among the most important parameters for evaluating flood mitigation performance (Goncalves et al., 2018). A reduction on runoff peak rate indicates lower risk of sewer overflow and less flooding and water quality problems, while reduction on total runoff volume means a decrease in the total quantity of precipitation that fails to infiltrate, evaporate or being stored during a rainfall event (Fahy & Chang, 2019). PCSWMM provides information regarding maximum runoff and total runoff volume for both catchment and sub-catchment levels. A reduction in these values would mean that GI alternatives were effective in reducing the risk of flooding and the damage if it occurred. For each scenario, the indicators were compared to a business-as-usual approach[4]. The effectiveness of implementing GI alternatives was assessed based on how they affected peak runoff rate and total runoff volume.

The modelling configuration applied to the scenarios is a simplification of infrastructure in real life (e.g., the drainage network and ideal pipelines). It is assumed, however, that these issues can be resolved through infrastructure design.

3. Results

3.1 Runoff peak rate

When GI is implemented in 10% of the sub-catchment area, the reduction in the catchment runoff peak rate is best accomplished by BR, PP and BR+PP; the worst performance was shown by RG for both case study areas for all six scenarios (see Figures 5 and 6). Interestingly, at the catchment scale, RG+BR and RG+PP had a similar performance to RG in reducing runoff peak rate for high emission and long-duration rainfall scenario (S6) with 10% of GI in the Auckland case.

Scenarios that considered 5% of GI implementation in the selected sub-catchments yielded similar results (see Figures 7 and 8). In Auckland, the highest reduction in runoff peak rate was obtained by BR, PP and BR+PP for all six scenarios; whereas in the less urbanised Gore, RG+BR were the best alternative to reduce runoff peak rate at the catchment scale (see Supplementary Material SM3 and SM4 for further information). RG showed the lowest efficiency for reducing runoff peak rate.

The reduction in runoff peak rate observed at the sub-catchment scale differed greatly when compared to the whole catchment. For example, when 10% of GI implementation was applied to selected sub-catchments, the reduction in runoff peak rate ranged between 49%-72% (Auckland) and between 44%-61% (Gore) (see Supplementary Material Tables SM5 and SM6, respectively). When 5% was applied,

[4] Business-as-usual was used to simulate what would happen if no GI alternative was implemented in each of the six scenarios.

the reduction in runoff peak rate ranged between 25%-41 in Auckland, and between 25%-42% in Gore (Tables SM5 and SM6, respectively).

There was a variation from sub-catchment to sub-catchment regarding the best GI alternatives for lowering runoff peak rate. In Auckland, sub-catchment 01 obtained the best results when PP was tested for S1, S2 and S3 (5% and 10%); and BR, PP and BR+PP S4, S5 and S6 (Tables SM5 and SM7). Nine out of 12 sub-catchments had better performance deploying BR+PP on short rainfall scenarios (S1, S2 and S3) with 10% of GI. BR, PP and BR+PP performed the best on prolonged rainfall events (S4, S5 and S6) with 5% and 10% of GI, and short rainfall (S1, S2 and S3) with 5% of GI. In both study areas and all six scenarios, the sub-catchments showed RG as the alternative with the lowest reduction in runoff peak rate. Interestingly, there seems to be no difference in which GI combination performs better or worse when different GHG emissions projections are tested.

In Gore, there was significant variation between scenarios in the sub-catchments (Tables SM6 and SM8). For example, when 10% of GI were tested for sub-catchment 93, the best alternatives for S1 were PP and RG+BR+PP; while BR+PP was the best for S2; and BR for S3. Sub-catchment 55 appeared to achieve better results when BR+PP was tested for all six scenarios. Repeatedly, for all combinations and all six scenarios, RG reported the lowest reduction in the runoff peak rate.

There was no substantial reduction in GI efficiency among different scenarios. Even with more extreme rainfall, GI alternatives are still capable of reducing the runoff peak rate. Notably, in the Auckland case, both at catchment and sub-catchment scales, GI tend to perform better for scenarios involving extremes (i.e., RCP8.5 and 120-hour extreme rainfall).

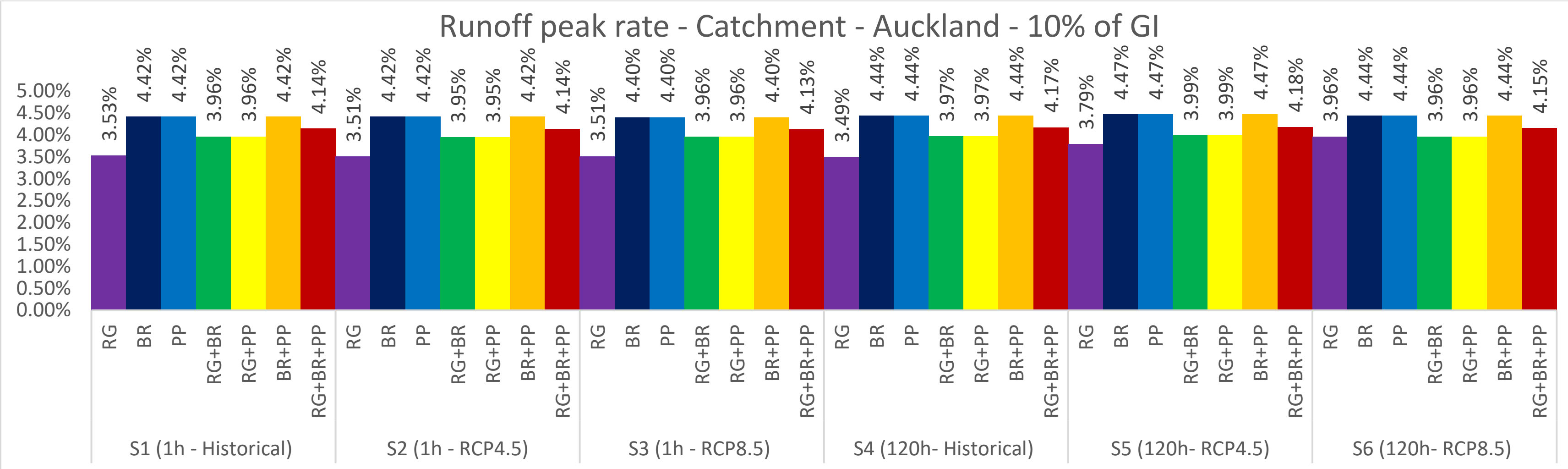

Figure 5 - Reduction in runoff peak rate compared to a business-as-usual scenario - Auckland catchment - converting 10% of selected sub-catchments into GI

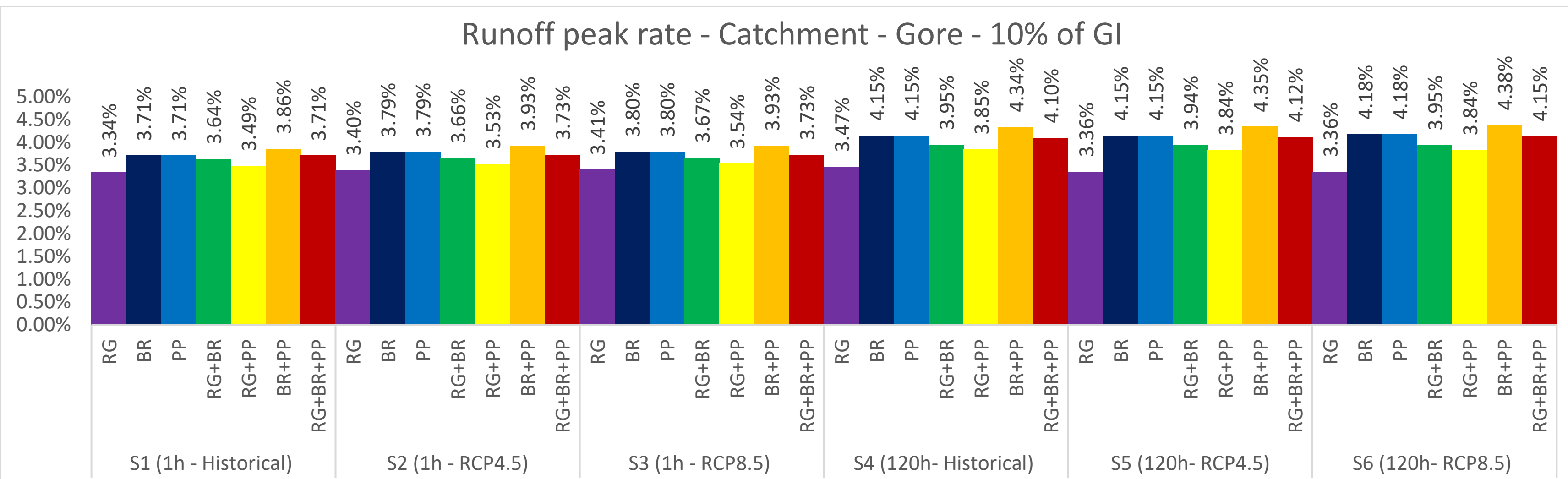

Figure 6 - Reduction in runoff peak rate compared to a business-as-usual scenario - Gore catchment - converting 10% of selected sub-catchments into GI

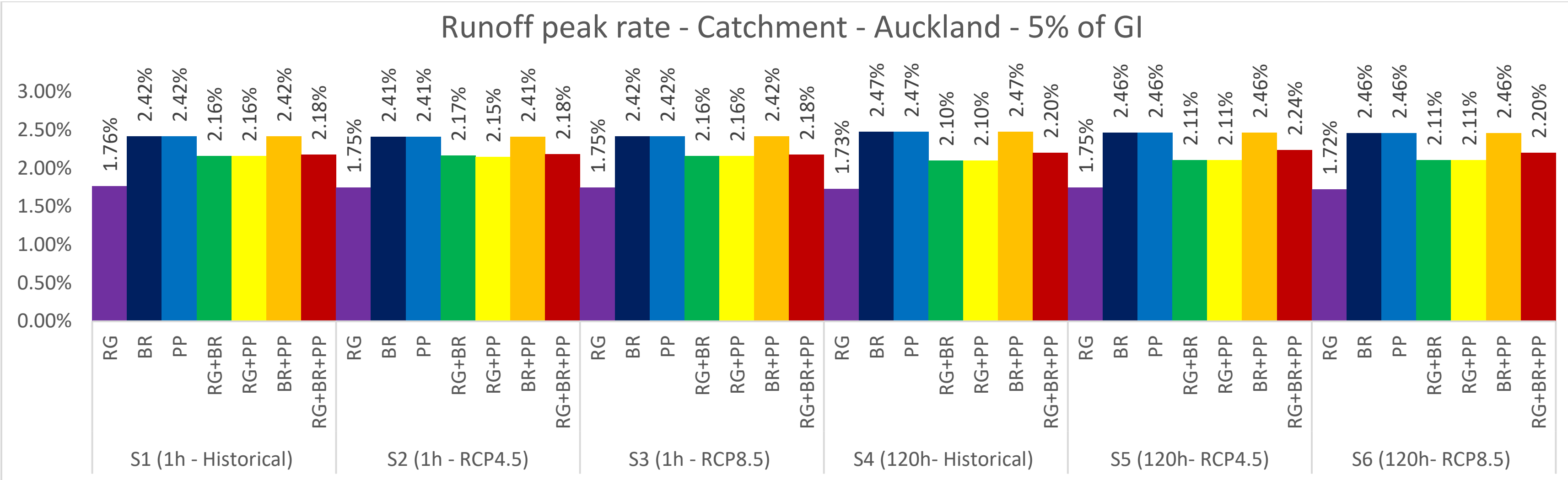

Figure 7 - Reduction in runoff peak rate compared to a business-as-usual scenario - Auckland catchment - converting 5% of selected sub-catchments into GI

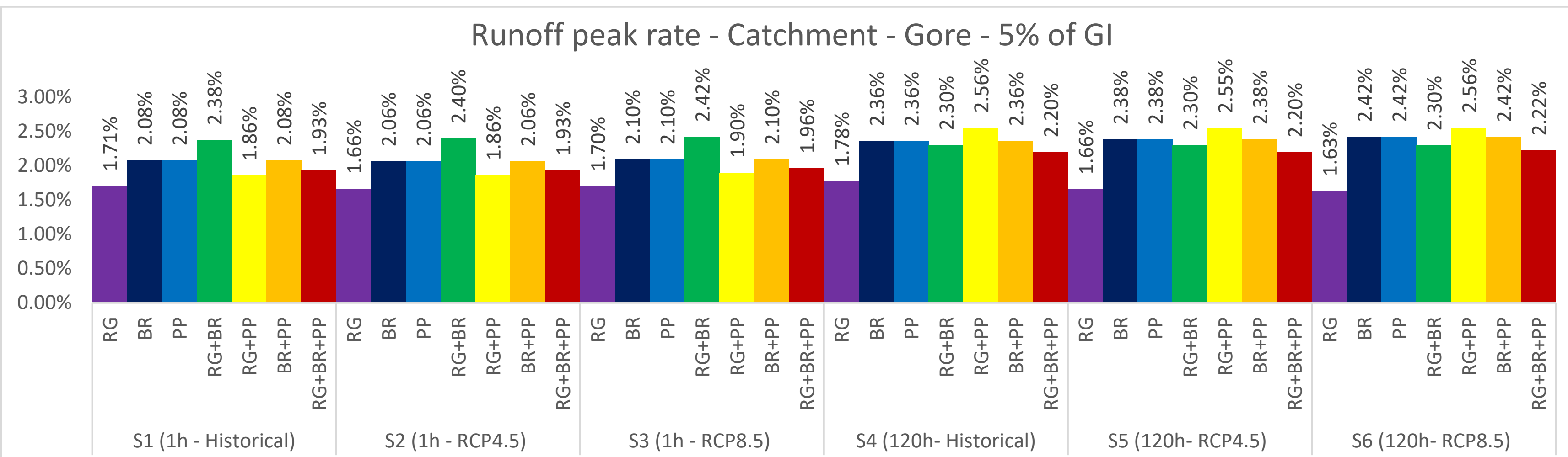

Figure 8 - Reduction in runoff peak rate compared to a business-as-usual scenario - Gore catchment - converting 5% of selected sub-catchments into GI

3.2 Total runoff volume

When testing 10% of GI implementation for short rainfall events (S1, S2 and S3), total runoff volume in Auckland (Figure 9) and Gore (Figure 10) showed greater reduction when PP and BR+PP were applied. For both case study areas, BR+PP showed the highest performance results for long rainfall (S4, S5 and S6). BR and PP as single options also showed the same performance as BR+PP for Auckland.

Similarly, the best solutions for all six scenarios were BR+PP and PP, deploying 5% of GI on the selected sub-catchments in Auckland (Figure 11). For prolonged rainfall (S4, S5 and S6), BR also showed a strong capacity to minimising total runoff volume. On the other hand, for Gore, the best results were obtained by RG+BR followed by PP for short rainfall (S1, S2 and S3) and by RG+PP for long rainfall (S4, S5 and S6) (Figure 12).

There was a significant reduction in the total runoff volume at the sub-catchment scale compared to what is shown in Figures 9-11[5]. The studied sub-catchments presented a very similar performance compared to the results for lowering runoff peak rate. In Auckland, in S1, S2 and S3, the decrease ranged between 50-74% (10% of GI) and between 25-43% (5% of GI); and between 50-75% (10% of GI) and 25-42% (5% of GI) in S4, S5 and S6. Interestingly, in all sub-catchments, PP appeared to be the best solution for reducing total runoff volume (Tables SM7 and SM9), regardless of the scenario.

Similar to runoff peak rate results, in Gore, the best alternative to lowering total runoff volume varied for each sub-catchment and depended on rainfall intensity and duration (see

[5] This reduction is due to a change in the spatial scale analysed – that is, the whole catchment area or each sub-catchment. For example, considering S3, sub-catchment 01 in Auckland showed a reduction range between 49.78%-65.89% in its runoff peak rate. By comparison, for the whole catchment where this sub-catchment is located, the reduction ranged between 3.41%-3.8% (see Supplementary Material Tables SM9, SM10, SM11 and SM12 for more detailed information regarding total runoff volume for each sub-catchment).

Tables SM10 and SM12). Deploying 5% of GI, S56 showed better results with RG+BR for S1, S2 and S3; BR, PP and BR+PP for S4 and S5; and BR+PP showed the highest results for S6.

Comparable to the other indicators, all GI combinations have proved to be efficient in reducing total runoff volume at both catchment and sub-catchment scales. Even when extremes were considered (RCP8.5 and 120-hour rainfall), the results have shown GI's ability to minimise total runoff volume.

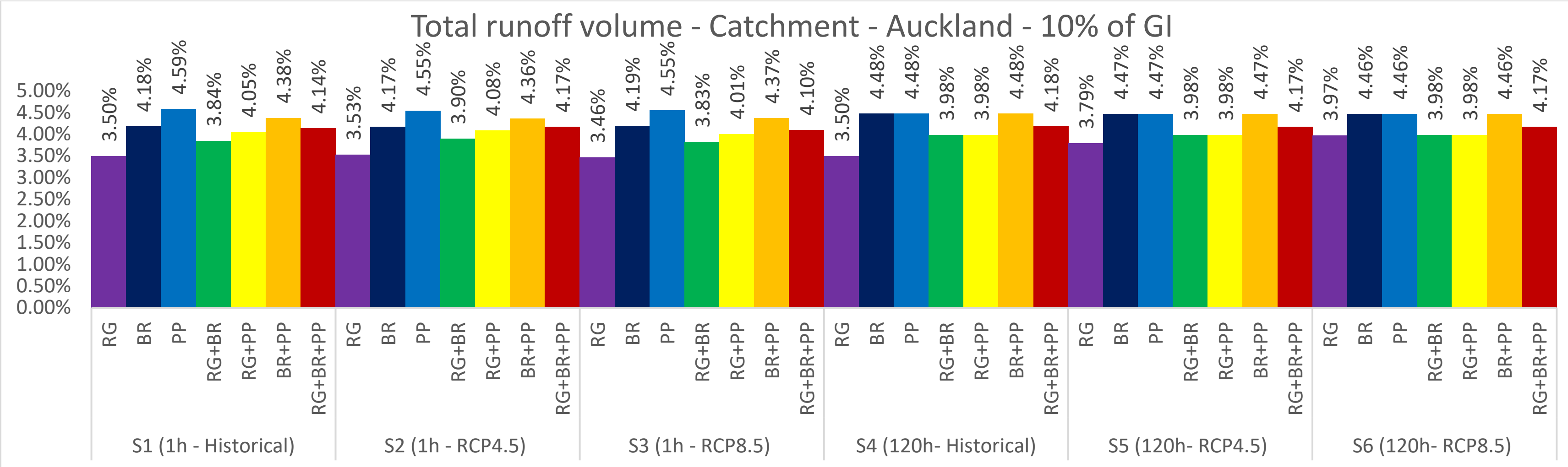

Figure 9 -Reduction in total runoff volume compared to a business-as-usual scenario - Auckland catchment - converting 10% of selected sub-catchments into GI

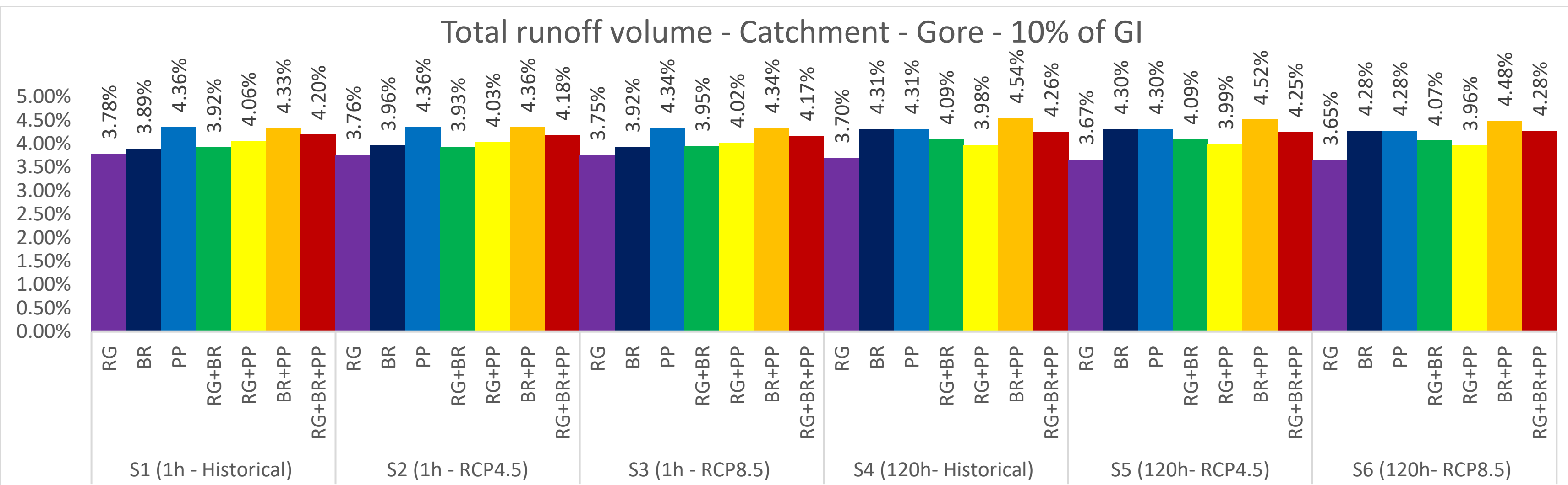

Figure 10 - Reduction in total runoff volume compared to a business-as-usual scenario - Gore catchment - converting 10% of selected sub-catchments into GI

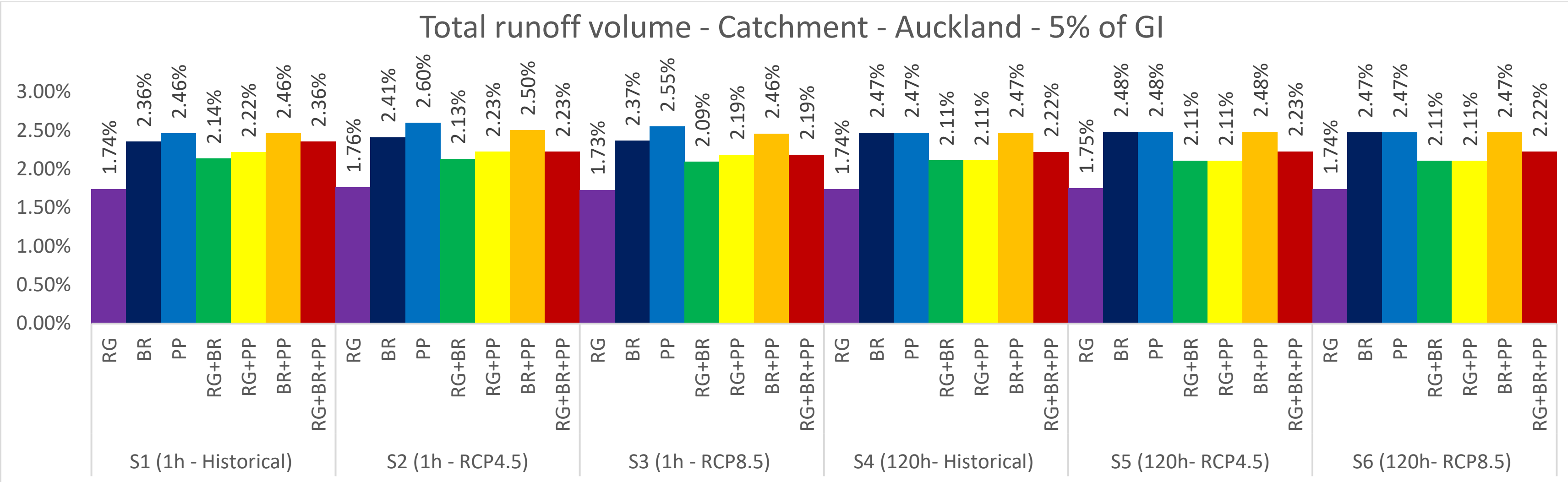

Figure 11 - Reduction in total runoff volume compared to a business-as-usual scenario - Auckland catchment - converting 5% of selected sub-catchments into GI

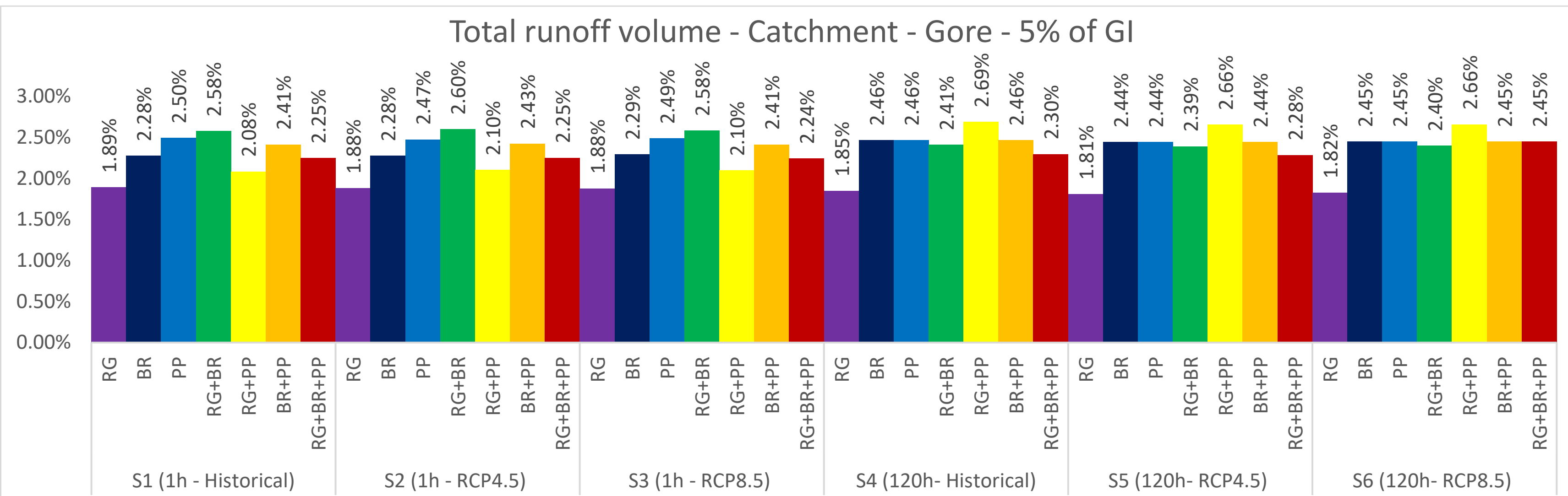

Figure 12 - Reduction in total runoff volume compared to a business-as-usual scenario - Gore catchment - converting 5% of selected sub-catchments into GI

4. Discussion

Our overall findings concur with other studies (Goncalves et al., 2018; Mei et al., 2018) and confirm that GI alternatives can reduce flood risk and potentially minimise the effects of climate change in urban areas. In particular, our findings confirmed that GI efficiency in dealing with different rainfall intensities (in our case 1 and 120-hour duration, RCPs 4.5 and 8.5) is directly related to their targeted spatial distribution and quantity within catchments and sub-catchments. Importantly, we identify three key points for informing decisions regarding land-use planning and flood risk management, with the aim to adapt to climate change impacts whilst improving social outcomes more broadly.

4.1 Small catchment areas converted into GI can have significant impact in reducing flood risk

Scholars (Bai et al., 2018; Qin et al., 2013; Wang et al., 2019) have previously demonstrated that GI alternatives are a feasible approach to flood risk management. In particular, the implementation of GI in small areas within a catchment can affect catchment scale performance and, consequently, reduce flood risk (Golden & Hoghooghi, 2018). Our results confirm this potential as all GI combinations in all six scenarios showed some capacity to reduce runoff peak rate and total runoff volume. Notably, even when a very small area of the whole catchment was converted into GI alternatives (less than 1% in both case studies and all six scenarios), there was reduction in the potential impacts of extreme precipitation events. Of particular interest from our results is the efficiency of relatively cheaper GI alternatives such as PP and BR (Chui, Liu, & Zhan, 2016). These alternatives can be easily implemented in existing built-up areas (Hu, Zhang, Li, Yang, & Tanaka, 2019) and, when combined with other GI options (e.g., rain gardens, urban forestry), they can also help deliver water quality improvements, heat island minimisation, and greater accessibility to urban greenspaces.

Urbanisation pressures and competition for space in urban areas prevents a wide implementation (and acceptance) of GI alternatives (Reimer & Rusche, 2019), but our study shows that the conversion of small areas within the catchment may be a valuable and effective approach. Hence, rather than investing in large scale GI projects, small areas such as driveways, sidewalks and small gardens could be converted into GI options to maximise their potential. From a spatial planning perspective, this means that when GI implementation considers multiple spatial scales, from the localised intervention area to a whole suburb or city, and retrofitted into existing urban form, their impact is maximised (Lafortezza, Davies, Sanesi, & Konijnendijk, 2013; Li, Uyttenhove, & Van Eetvelde, 2020). For example, our findings demonstrate that the flood risk reduction benefits provided by GI did not increase linearly - that is, by doubling the amount of GI implemented, flood risks did not reduce two-fold. Thus, our findings confirm that the strategically conversion and distribution of small GI areas along the catchment appears to be the most effective way of maximising their benefits, especially when different alternatives are combined (Fahy & Chang, 2019).

4.2 GI alternatives can play a significant role in addressing climate change impacts but this differs between highly urbanised cities and small regional towns

It is argued that uncertainty related to climate change and future risks are expected to escalate in the future, including changes in rainfall patterns and flood risks (Ran & Nedovic-Budic, 2016). More extreme and prolonged rainfall events, higher peak flows and greater flood risks are expected to affect the performance of conventional (and aged) stormwater systems with direct implications for infrastructure planning and corresponding funding requirements (Zevenbergen, Veerbeek, Gersonius, & van Herk, 2008; Zhang, Manuelpillai, Raut, Deletic, & Bach, 2019). Our findings concur with previous studies which have demonstrated the potential of GI alternatives in addressing flood risks set to be exacerbated by climate change (Carter et al., 2018; Chen, Tung, & Li, 2017; Derkzen, van Teeffelen, & Verburg, 2017).

However, we develop this agenda further by providing data comparing settlement size and urbanisation patterns. Our scenarios demonstrate that highly urbanised cities can achieve better results if GI alternatives such as BR and PP (or their combination) are incorporated throughout their catchments. These alternatives substitute impermeable surfaces and increase the catchment's capacity of retaining or slowing down runoff (Chui et al., 2016). Small towns, however, can also benefit from the introduction of RG combined with other(s) alternative(s) such as BR or PP. This is an important finding as rain gardens can be paired with storage tanks to provide temporary storage (Jia, Tang, Luo, Li, & Zhou, 2016) while adding biodiversity and aesthetic value and enhancing urban landscape (Yuan, Dunnett, & Stovin, 2017). Rain gardens can also function as green spaces, providing leisure areas for surrounding communities. Moreover, if properly framed, designed and implemented (Shi, 2020), these alternatives have the potential of also reducing social inequity and related distribution of environmental and social benefits. A summary of differences between cities and towns, and the most viable options for decision-makers in each area is provided in Figure 13.

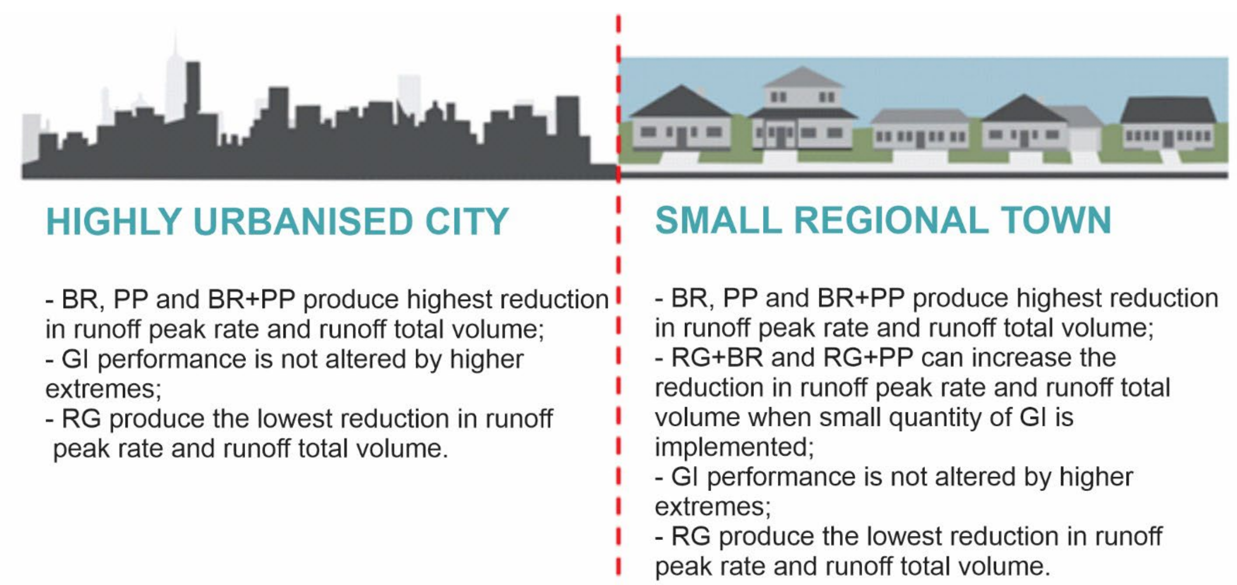

Figure 13 - Summary of GI measures most appropriate to larger cities and small regional towns.

4.3 Prioritising GI implementation to maximise social outcomes

While the implementation of large extensions of GI is considered to be the ideal situation (Webber et al., 2019), this is not possible in many existing urban areas where the built form is already defined, or financial resources are limited. Additionally, GI implementation has been mainly driven by traditional engineering design (Kuller, Bach, Ramirez-Lovering, & Deletic, 2018), often leaving socio-economic factors overlooked (Iojă, Badiu, Haase, Hossu, & Niță, 2021; Li et al., 2020). It should be noted that from a purely hydrological perspective there may be similar effects in peak rates or volume in different sub-catchments within the two case studies, but the selection of sub-catchments we modelled, and the additional inclusion of criteria outlined in step 2 of section 2.3, was designed to provide specific data on spatial equity to inform decision-making that has tended towards technical considerations. We argue that by including data on income and current access to green space, scarce public funds have greater potential to be invested in areas that are more deprived, typically have less political power, and which also tend to have less access to environmental goods. As outlined elsewhere (e.g., Shi 2020, Sekulova et al. 2021, Pallathadka et al. 2022), the focus would then need to turn to ensuring that framing, design and implementation of GI strategies can adequately capture those wider underlying drivers of spatial inequality to avoid green gentrification. Acknowledging inequality within the initial modelling process is an important step in this regard, although in practice the situation is complex and context-dependent. According to Shokry, Connolly, and Anguelovski (2020), investment can also lead to displacement and green gentrification can be intensified by ‘omission’ or ‘commission’. They argue that acts of omission occur from unequal and inequitable climate protection as underprivileged people or communities are excluded from interventions. Acts of commission are those that may aggravate social vulnerabilities over time, most of which are caused by gentrification or relocation of low-income people.

To add to the literature on GI and equity, our study tested the implementation of GI alternatives in areas where many residents have relatively lower income compared to the national average (e.g., up to $30,000/year while the national average was $45,744/year) and confirmed the efficiency of potentially low-cost GI alternatives (PP and BR) in reducing flood risk in those areas. Whilst we acknowledge issues concerning inequality are complex, we argue that the explicit use of these layers of data holds potential in both preventing the initial erasure of spatial equity considerations in public flood risk investment and elevating these issues within implementation discussions. Overall, we argue that there is clear potential for cities to apply a social-equity dimension when investing in, or for prioritising, GI alternatives in land-use planning (Nordman, Isely, Isely, & Denning, 2018) – particularly if included in the modelling process. While we do focus on specific sub-catchments, as a planning strategy there is a degree of transferability regarding the use of a social-equity dimension to help justify spending and encourage decision-makers to prioritise GI implementation in areas which are both at greater risk of flooding and inhabited by more vulnerable populations. In this way, the implementation of GI using a clear social-equity dimension can be a helpful approach to minimise green gentrification and reduce social inequalities in the distribution of environmental amenities (Anguelovski et al., 2018; Rigolon & Németh, 2018). Using this approach, to maximise GI benefits, spatial and strategic plans may set priority areas for GI implementation, especially areas which have greater proportion of socio-economic disadvantaged residents, less access to greenspaces, and greater flood risks.

Importantly, the methodology applied on this study has potential to e an efficient decision-making tool to prioritise GI implementation. By applying a social-equity, temporal and spatial dimensions our findings indicate that not only is GI efficient in addressing climate change impacts, but it can also help cities overcome social inequalities in the provision of environmental amenities whilst preventing green gentrification.

5. Conclusion

This paper used scenario analysis to investigate the performance of implementing various combinations of GI alternatives in two distinct locations in Aotearoa-New Zealand. Six scenarios were created using historical and future rainfall data, including 1 and 120 hours of rainfall duration under RCPs 4.5 and 8.5 (2038-2050), and the implementation of 5 and 10% of GI alternatives along selected sub-catchments in Auckland and Gore. GI performance was evaluated based on their capacity to reduce relative peak runoff rate and total runoff volume compared with a business-as-usual approach.

Our findings confirmed that GI alternatives are effective in reducing flood risks even when a very small area within the whole catchment is converted into GI options (less than 1% in both case studies and all six scenarios). Notably, these included relatively low-cost alternatives such as PP and BR. PP and BR are politically appealing in terms of retrofitting options because they do not require significant space and may be installed along sidewalks, cycleways and driveways (as opposed to rain gardens that cannot be easily implemented in those areas). Additionally, PP and BR maintenance may be included into typical urban maintenance routine, resulting in no additional costs to local authorities. GI alternatives, therefore, are proved to be a feasible strategy in minimising the impacts of extreme precipitation events as a result of climate change. By developing and testing a methodology that investigates GI performance in areas of socio-economic disadvantage, our study also demonstrated how a social-equity dimension could form a key criterion in spatial and strategic planning for investing and prioritising the implementation of GI alternatives, and help overcome financial constraints whilst delivering maximum social outcomes and benefits. We also anticipate that the information would be of interest to policy makers in both highly urbanised cities and smaller regional towns, despite each having its own constraints and opportunities.

Nevertheless, we acknowledge that the incorporation of GI alternatives is still affected by the willingness of the public to participate, political decision-making and challenges in the project execution (such as budget and competition for space). Lastly, although the findings indicated the potential of GI in reducing flood risks at both catchment and sub-catchment scales, it should be noted that their performance is closely linked to the attributes of the drainage system.

By testing multiple GI combinations under different rainfall patterns, this paper helps strengthen the connection, and interdependence, between flood risk management and land-use planning. It also helps lessen the gap between the potential of GI and its application in practice. While the knowledge gained by this study may be applied to similar planning contexts, it is important to stress that variables such as soil type, permeability rate and urban density can have an impact on the performance of GI alternatives. Hence, caution should be taken when transferring the outputs of the models used in this study to on-the-ground GI implementation in other locations. This also needs to be informed by specific engineering inputs relative to designated locations. Additionally, we also suggest that future studies could focus on economic assessments of GI implementation (e.g., construction and maintenance costs) to better inform land-use planning decisions.

References

Alves, A., Gomez, J. P., Vojinovic, Z., Sanchez, A., & Weesakul, S. (2018). Combining co-benefits and stakeholders perceptions into green infrastructure selection for flood risk reduction. *Environments, 5*, 23.

Amer, M., Daim, T. U., & Jetter, A. (2013). A review of scenario planning. *Futures, 46*, 23-40.

Anguelovski, I., Connolly, J. J. T., Masip, L., & Pearsall, H. (2018). Assessing green gentrification in historically disenfranchised neighborhoods: a longitudinal and spatial analysis of Barcelona. *Urban Geography, 39*, 458-491.

Bai, Y., Zhao, N., Zhang, R., & Zeng, X. (2018). Storm water management of low impact development in urban areas based on SWMM. *Water, 11*.

Bannayan, M., & Eyshi Rezaei, E. (2012). Future production of rainfed wheat in Iran (Khorasan province): climate change scenario analysis. *Mitigation and Adaptation Strategies for Global Change, 19*, 211-227.

Benedict, M. A., & McMahon, E. T. (2006). *Green infrastructure : Linking landscapes and communities*. Washington: Washington: Island Press.

Biddle, D.-L. (2020). South Auckland parent's five-year battle to get a playground. In *Stuff*. New Zealand.

Bird, D., King, D., Haynes, K., Box, P., Okada, T., & Nairn, K. (2013). Impact of the 2010-11 floods and the factors that inhibit and enable household adaptation strategies. In *Synthesis and Integrative Research - Final Report*. Gold Coast: National Climate Change Adaptation Research Facility.

Brewington, L., Keener, V., & Mair, A. (2019). Simulating land cover change impacts on groundwater recharge under selected climate projections, Maui, Hawai. *Remote Sensing, 11*.

Carter, J. G., Handley, J., Butlin, T., & Gill, S. (2018). Adapting cities to climate change-exploring the flood risk management role of green infrastructure landscapes. *Journal of Environmental Planning and Management, 61*, 1535-1552.

Chen, P. Y., Tung, C. P., & Li, Y. H. (2017). Low impact development planning and adaptation decision-making under climate change for a community against pluvial flooding. *Water, 9*.

Chui, T. F. M., Liu, X., & Zhan, W. (2016). Assessing cost-effectiveness of specific LID practice designs in response to large storm events. *Journal of Hydrology (Amsterdam), 533*, 353-364.

Derkzen, M. L., van Teeffelen, A. J. A., & Verburg, P. H. (2017). Green infrastructure for urban climate adaptation: How do residents' views on climate impacts and green infrastructure shape adaptation preferences? *Landscape and Urban Planning, 157*, 106-130.

Dover, J. W. (2015). *Green infrastructure: incorporating plants and enhancing biodiversity in buildings and urban environments*: New York : Routledge, Taylor & Francis Group, Earthscan from Routledge.

Ercolani, G., Chiaradia, E. A., Gandolfi, C., Castelli, F., & Masseroni, D. (2018). Evaluating performances of green roofs for stormwater runoff mitigation in a high flood risk urban catchment. *Journal of Hydrology, 566*, 830-845.

Fahy, B., & Chang, H. (2019). Effects of Stormwater Green Infrastructure on Watershed Outflow: Does Spatial Distribution Matter? *International Journal of Geospatial and Environmental Research*, *6*(1), 5.

Fernandez, M. A., & Martin, S. L. (2020). Staged implementation of inclusionary zoning as a mechanism to improve housing affordability in Auckland, New Zealand. *International Journal of Housing Markets and Analysis, 13*, 617-633.

Fletcher, T. D., Shuster, W., Hunt, W. F., Ashley, R., Butler, D., Arthur, S., Trowsdale, S., Barraud, S., Semadeni-Davies, A., Bertrand-Krajewski, J.-L., Mikkelsen, P. S., Rivard, G., Uhl, M., Dagenais, D., & Viklander, M. (2015). SUDS, LID, BMPs, WSUD and more – The evolution and application of terminology surrounding urban drainage. *Urban Water Journal, 12*, 525-542.

Golden, H. E., & Hoghooghi, N. (2018). Green infrastructure and its catchment-scale effects: an emerging science. *WIREs Water, 5*, e1254.

Goncalves, M. L. R., Zischg, J., Rau, S., Sitzmann, M., Rauch, W., & Kleidorfer, M. (2018). Modeling the effects of introducing low impact development in a tropical city: A case study from Joinville, Brazil. *Sustainability, 10*.

Graveline, N., Aunay, B., Fusillier, J. L., & Rinaudo, J. D. (2014). Coping with urban & agriculture water demand uncertainty in water management plan design: The interest of participatory scenario analysis. *Water Resources Management, 28*, 3075-3093.

Hashimoto, S., DasGupta, R., Kabaya, K., Matsui, T., Haga, C., Saito, O., & Takeuchi, K. (2018). Scenario analysis of land-use and ecosystem services of social-ecological landscapes: implications of alternative development pathways under declining population in the Noto Peninsula, Japan. *Sustainability Science, 14*, 53-75.

He, C., Li, J., Zhang, X., Liu, Z., & Zhang, D. (2017). Will rapid urban expansion in the drylands of northern China continue: A scenario analysis based on the Land Use Scenario Dynamics-urban model and the Shared Socioeconomic Pathways. *Journal of Cleaner Production, 165*, 57-69.

Hoffimann, E., Barros, H., & Ribeiro, A. I. (2017). Socioeconomic inequalities in green space quality and accessibility—Evidence from a Southern European city. *International Journal of Environmental Research and Public Health, 14*, 916.

Hu, M. C., Zhang, X. Q., Li, Y., Yang, H., & Tanaka, K. (2019). Flood mitigation performance of low impact development technologies under different storms for retrofitting an urbanized area. *Journal of Cleaner Production, 222*, 373-380.

Iojă, C. I., Badiu, D. L., Haase, D., Hossu, A. C., & Niță, M. R. (2021). How about water? Urban blue infrastructure management in Romania. *Cities, 110*, 103084.

IPCC. (2014). Climate change 2014: Synthesis report. In (The Core Writing Team, Rajendra K. Pachauri, Leo Meyer ed.). Geneva, Switzerland: Intergovernmental Panel On Climate Change.

Ira, S., & Simcock, R. (2019). *Understanding costs and maintenance of WSUD in New Zealand: Activating WSUD for healthy resilient communities.* Wellington: Building Better Homes Towns and Cities National Science Challenge.

James, W., Rossman, L. A., & James, W. R. C. (2010). User's guide to SWMM 5. In. Guelph, Ontario, Canada: CHI.

Jayawardena, H. M. I. D. P., & van Roon, M. (2017). Water sensitive planning and design as an ecologically inspired approach to delivering flood resilient urban environment in Sri Lanka. *Water Practice and Technology, 12*, 964-977.

Jia, H., Yao, H., Tang, Y., Yu, S. L., Field, R., & Tafuri, A. N. (2015). LID-BMPs planning for urban runoff control and the case study in China. *Journal of Environmental Management, 149*, 65-76.

Jia, Z., Tang, S., Luo, W., Li, S., & Zhou, M. (2016). Small scale green infrastructure design to meet different urban hydrological criteria. *Journal of Environmental Management, 171*, 92-100.

Kuller, M., Bach, P. M., Ramirez-Lovering, D., & Deletic, A. (2018). What drives the location choice for water sensitive infrastructure in Melbourne, Australia? *Landscape and Urban Planning, 175*, 92-101.

La Rosa, D., & Pappalardo, V. (2020). Planning for spatial equity-A performance based approach for sustainable urban drainage systems. *Sustainable Cities and Society*, 53, 101885.

Lafortezza, R., Davies, C., Sanesi, G., & Konijnendijk, C. C. (2013). Green Infrastructure as a tool to support spatial planning in European urban regions. *IForest-Biogeosciences and Forestry, 6*, 102-108.

Latif, J. (2021). Auckland Council fight looms as under-funded areas demand more cash for playgrounds and community facilities. In *Stuff*. New Zealand.

Li, C., Liu, M., Hu, Y., Han, R., Shi, T., Qu, X., & Wu, Y. (2018). Evaluating the hydrologic performance of low impact development scenarios in a micro urban catchment. *International Journal of Environmental Research and Public Health, 15*.

Li, L., Uyttenhove, P., & Van Eetvelde, V. (2020). Planning green infrastructure to mitigate urban surface water flooding risk – A methodology to identify priority areas applied in the city of Ghent. *Landscape and Urban Planning, 194*, 103703.

Matias, N.-G., Matias, N.-G., Johnes, P. J., & Johnes, P. J. (2012). Catchment phosphorous losses: An export coefficient modelling approach with scenario analysis for water management. *Water Resources Management, 26*, 1041-1064.

Mei, C., Liu, J., Wang, H., Yang, Z., Ding, X., & Shao, W. (2018). Integrated assessments of green infrastructure for flood mitigation to support robust decision-making for sponge city construction in an urbanized watershed. *Science of the Total Environment, 639*, 1394-1407.

Mell, I. C. (2017). Green infrastructure: reflections on past, present and future praxis. *Landscape Research: Green Infrastructure, 42*, 135-145.

Mguni, P., Herslund, L., & Jensen, M. B. (2015). Green infrastructure for flood-risk management in Dar es Salaam and Copenhagen: exploring the potential for transitions towards sustainable urban water management. *Water Policy, 17*, 126-142.

Ministry for the Environment. (2020). *National climate change risk assessment for New Zealand: Main report – Arotakenga Tūraru mō te Huringa Āhuarangi o Āotearoa: Pūrongo whakatōpū.* Ministry for the Environment (Wellington).

Natural England. (2010). *Nature nearby: Accessible natural greenspace guidance.* Natural England (Peterborough).

NIWA. (2018). *High Intensity Rainfall Design System*. National Institute of Water & Atmospheric Research Ltd - NIWA (Wellington: Envirolink).

Nordman, E. E., Isely, E., Isely, P., & Denning, R. (2018). Benefit-cost analysis of stormwater green infrastructure practices for Grand Rapids, Michigan, USA. *Journal of Cleaner Production, 200*, 501-510.

Pallathadka, A., Sauer, J., Chang, H., & Grimm, N. B. (2022). Urban flood risk and green infrastructure: Who is exposed to risk and who benefits from investment? A case study of three US Cities. *Landscape and Urban Planning*, 223, 104417.

Paris Agreement. (2015). The Paris Agreement. Retrieved from https://unfccc.int/resource/docs/2015/cop21/eng/l09r01.pdf

Qin, H. P., Li, Z. X., & Fu, G. (2013). The effects of low impact development on urban flooding under different rainfall characteristics. *Journal of Environmental Management, 129*, 577-585.

Ran, J., & Nedovic-Budic, Z. (2016). Integrating spatial planning and flood risk management: A new conceptual framework for the spatially integrated policy infrastructure. *Computers, Environment and Urban Systems, 57*, 68-79.

Reimer, M., & Rusche, K. (2019). Green infrastructure under pressure. A global narrative between regional vision and local implementation. *European Planning Studies, 27*, 1542-1563.

Reu Junqueira, J., Serrao-Neumann, S., & White, I. (2021a). Managing urban climate change risks: Prospects for using green infrastructure to increase urban resilience to floods. In T. M. Letcher (Ed.), *The Impacts of Climate Change* (pp. 379-396): Elsevier.

Reu Junqueira, J., Serrao-Neumann, S., & White, I. (2021b). A systematic review of approaches for modelling current and future impacts of extreme rainfall events using green infrastructure. *Journal of Cleaner Production, 290*, 125173.

Rigolon, A., & Németh, J. (2018). “We're not in the business of housing:” Environmental gentrification and the nonprofitization of green infrastructure projects. *Cities, 81*, 71-80.

Sekulova, F., Anguelovski, I., Kiss, B., Kotsila, P., Baró, F., Palgan, Y. V., & Connolly, J. (2021). The governance of nature-based solutions in the city at the intersection of justice and equity. *Cities, 112*, 103136.

Shi, L. (2020). Beyond flood risk reduction: How can green infrastructure advance both social justice and regional impact? *Socio-Ecological Practice Research*.

Shokry, G., Connolly, J. J. T., & Anguelovski, I. (2020). Understanding climate gentrification and shifting landscapes of protection and vulnerability in green resilient Philadelphia. *Urban Climate, 31*, 100539.

Silva, C. (2018). Auckland's urban sprawl, policy ambiguities and the peri-urbanisation to Pukekohe. *Urban Science, 3*, 1.

Struck, S., & Lichten, K. H. (2010). *Low impact development 2010 : Redefining water in the city*. Reston, United States: American Society of Civil Engineers.

Sun, Y. W., Li, Q. Y., Liu, L., Xu, C. D., & Liu, Z. P. (2014). Hydrological simulation approaches for BMPs and LID practices in highly urbanized area and development of hydrological performance indicator system. *Water Science and Engineering, 7*, 143-154.

Toimil, A., Losada, I. J., Nicholls, R. J., Dalrymple, R. A., & Stive, M. J. F. (2020). Addressing the challenges of climate change risks and adaptation in coastal areas: A review. *Coastal Engineering, 156*.

Toxopeus, H., Kotsila, P., Conde, M., Katona, A., van der Jagt, A. P. N., & Polzin, F. (2020). How 'just' is hybrid governance of urban nature-based solutions? *Cities, 105*, 102839.

Vallecillo, S., Polce, C., Barbosa, A., Perpiña Castillo, C., Vandecasteele, I., Rusch, G. M., & Maes, J. (2018). Spatial alternatives for Green Infrastructure planning across the EU: An ecosystem service perspective. *Landscape and Urban Planning, 174*, 41-54.

Van der Jagt, A. P. N., Bernadett, K., Shunsuke, H., & Wakana, T. (2021). Nature-based solutions or debacles? The politics of reflexive governance for sustainable and just cities. *Frontiers in Sustainable Cities, 2*.

Varum, C. A., & Melo, C. (2010). Directions in scenario planning literature – A review of the past decades. *Futures, 42*, 355-369.

Vermaat, J. E., Hellmann, F. A., van Teeffelen, A., van Minnen, J., Alkemade, R., Billeter, R., Beierkuhnlein, C., Boitani, L., Cabeza, M., Feld, C. K., Huntley, B., Paterson, J., & WallisDeVries, M. F. (2017). Differentiating the effects of climate and land use change on European biodiversity: A scenario analysis. *Ambio, 46*, 277-290.

Wang, M., Zhang, D., Cheng, Y., & Tan, S. K. (2019). Assessing performance of porous pavements and bioretention cells for stormwater management in response to probable climatic changes. *Journal of Environmental Management, 243*, 157-167.

Wang, M., Zhang, D. Q., Su, J., Dong, J. W., & Tan, S. K. (2018). Assessing hydrological effects and performance of low impact development practices based on future scenarios modeling. *Journal of Cleaner Production, 179*, 12-23.

Webber, J. L., Fletcher, T. D., Cunningham, L., Fu, G., Butler, D., & Burns, M. J. (2019). Is green infrastructure a viable strategy for managing urban surface water flooding? *Urban Water Journal*, 1-11.

White, I., & Haughton, G. (2017). Risky times: Hazard management and the tyranny of the present. *International Journal of Disaster Risk Reduction, 22*, 412-419.

Yuan, J., Dunnett, N., & Stovin, V. (2017). The influence of vegetation on rain garden hydrological performance. *Urban Water Journal, 14*, 1083-1089.

Zevenbergen, C., Veerbeek, W., Gersonius, B., & van Herk, S. (2008). Challenges in urban flood management: travelling across spatial and temporal scales. *Journal of Flood Risk Management, 1*, 81-88.

Zhang, K. F., Manuelpillai, D., Raut, B., Deletic, A., & Bach, P. M. (2019). Evaluating the reliability of stormwater treatment systems under various future climate conditions. *Journal of Hydrology, 568*, 57-66.

Zhang, L., Oyake, Y., Morimoto, Y., Niwa, H., & Shibata, S. (2019). Rainwater storage/infiltration function of rain gardens for management of urban storm runoff in Japan. *Landscape and Ecological Engineering, 15*, 421-435.

Zhu, Z. H., & Chen, X. H. (2017). Evaluating the effects of low impact development practices on urban flooding under different rainfall intensities. *Water, 9*.

Fahy, B., & Chang, H. (2019). Effects of Stormwater Green Infrastructure on Watershed Outflow: Does Spatial Distribution Matter? *International Journal of Geospatial and Environmental Research*, *6*(1), 5.