Closing the climate adaptation gap for water supply utilities

A thesis presented for the degree of Doctor of Philosophy

School of Geography and the Environment



Olivia Becher

St Anne's College

University of Oxford

Supervisors: Dr Raghav Pant and Prof Jim Hall

October 2023

“When water trow weh, it cyaan pick up back.”

Translation: When water is thrown away, it cannot be picked up back.

Jamaican proverb

Abstract

Resilient water supply infrastructure is fundamental for human life and wellbeing. In addition to the number of people who are not yet connected to a water utility network, millions suffer from frequent and prolonged climate-induced water supply disruptions. This problem is particularly acute in the developing world, where countries lack sufficient financing for water infrastructure maintenance and development, and where extreme climatic and hydrological variability make the challenge of designing resilient water systems even greater. The goal of this DPhil is to develop and apply a set of complementary, applied, frameworks for analysing climate risks to water supply systems to guide decision making and enable the quantification of adaptation benefits. This overarching aim is divided into three research objectives. 1) To develop a framework for analysing widespread, multi-hazard climate risks to water supply utilities. 2) To develop an approach for quantifying the costs and benefits of adaptation for both water supply utilities and their customers, including those in poverty. 3) To develop a global framework for quantifying drought risk at the utility scale, as a platform for comparing benefit-cost ratios (BCR) across multiple types of adaptation actions.

The first two modelling frameworks were developed around, and applied to, a case study of the National Water Commission (NWC) in Jamaica, where water supplies are vulnerable to multiple climatic hazards and subject to frequent disruptions. As presented in chapter 4, the total annual expected disruption incurred by fluvial and pluvial flooding, extreme cyclonic wind and drought is approximated to be 5 days per year per utility customer under present conditions, with significant spatial variation in risk across the utility's systems. This risk is projected to increase by a factor of between 2 and 2.5 when end-of-century climate scenarios are propagated through the model. Based on the flood and drought risk estimates, the costs of flood and drought risk disruptions are estimated in chapter 5 to prioritise across a portfolio of adaptation options. Of the 240 sub-systems that make up the NWC's water supply network, about 34% would benefit from flood defences and 53% would benefit from leakage reduction to adaptation to droughts. The cumulative annual benefit that could be achieved by implementing all system optimised adaptation options is estimated to be 720 million Jamaican dollars [4.65 million USD] per year, representing a substantial saving for the utility and its customers, including low-income customers. The third framework, presented in chapter 6, is applied at the global scale to find hotspots of present-day and future drought risk under future scenarios of climate change and demand growth (namely, SSP1-2.6, SSP3-7.5, SSP5-8.0). We estimate the mean rate of unsustainable or disrupted utility supply at 15% (interquartile range,

0-26%) and project a global increase in risk of between 30-45% under future scenarios. Implementing the most cost-effective adaptation action identified per utility would mitigate additional future risk by 75-80%. However, implementing the subset of cost-effective options that generate sufficient tariff revenue to provide a benefit-cost ratio that is greater than 1 would only achieve 5-20% of this benefit. The results underline the challenge of attracting the financing required to close the climate adaptation gap for water supply utilities.

Although water utilities are uniquely positioned to deliver climate adaptation services for society, offering a direct channel to protect vital services from climate extremes, efforts to close the climate adaptation gap are fraught with challenges. One challenge faced by utilities and investors lies in the coherent assessment of risk and adaptation needs in the context of multiple hazards, distributional impacts triggered by a range of disruption channels and wide-ranging possible adaptation pathways. This DPhil thesis introduces a set of interdisciplinary frameworks that can be applied at both national and global scales to support investment decision making in the water supply sector. The frameworks also serve to demonstrate the impact of investments towards climate adaptation and wider development targets, supporting the business case for adaptation investments that also respond to national development priorities. With the climate agenda rapidly gaining momentum, there is an opportunity to strengthen recognition that the utilities receive in high level dialogue on climate finance and channel greater financial flows towards closing the climate adaptation gap for water supply utilities.

Acknowledgements

This work has benefitted from a wide range of collaborations. The chapters (4 and 5) that were developed around a case study of the National Water Commission in Jamaica were made possible by the team involved in realising the 'Jamaica - Systemic Risk Assessment Tool'. This includes, notably, Anaitee Mills, Edson Williams and collaborators from the National Water Commission and the University of the West Indies. Chapter 6 was developed during the Young Scientist's Summer Programme at the International Institute of Applied Systems Analysis where I was supervised by Dr Mikhail Smilovic and Dr Sylvia Tramberend. For chapters 4 and 5, data collaborations were established with JBA for flood inundation modelling and the National Water Commission in Jamaica for datasets detailing their potable supply network, capital improvement plans and data on past disruptions. For chapter 6, data collaborations were established with the International Institute for Applied Systems Analysis for hydrological outputs of Community Water Model (CWatM) and the Global Water Intelligence for the global utility database. The work was funded by the Engineering and Physical Sciences Research Council (EPSRC) under grant number EP/T517811/1.

I would firstly like to thank my supervisor Prof Jim Hall for everything I have gained and learnt from this DPhil. It has been a very difficult and challenging experience, but I have benefitted immensely both personally and professionally from his example, mentorship and the overall process. I would like to thank my co-supervisor Dr Raghav Pant for his keen eye for detail and wise inputs throughout the DPhil journey. I would like to thank Dr Jasper Verschuur for always being there to help and offer guidance, and also for setting an inspiring example on combining creativity with competence and discipline. I hope I can bring all my learnings forward with me into the future and sorry for being a (maybe not so) occasional source of stress. I would also like David Ford for helping me with the School of Geography Linux cluster, which was essential for the work, despite being incredibly busy. Lastly, I would like to thank my boyfriend, Konstantinos Tsochas, family and friends, in particular Rio Ruskin-Tompkins, Andrew McMahon, Jasper Verschuur, Tina Christmann, Sophie Erfurth, Sarah Gall, Maryam Malik and Ana Lemyre for their support and filling the past three years with happy memories.

Publications

The work presented in chapters 4 to 6 has either been published or is under review with the following peer-reviewed journals.

1. Becher, O., Pant, R., Verschuur, J., Mandal, A., Paltan, H., Lawless, M., Raven, E. and Hall, J., 2023. A Multi-Hazard Risk Framework to Stress-Test Water Supply Systems to Climate-Related Disruptions. *Earth's Future*, 11(1), p.e2022EF002946. **(Chapter 4)**

2. Becher, Verschuur, Pant and Hall (in review with *Environmental Research: Infrastructure and Sustainability* [manuscript status is 'Accept in Principle', conditional on meeting editorial requests]), Prioritising climate adaptation options to minimise financial and distributional impacts of water supply disruptions. **(Chapter 5)**

3. Becher, Verschuur, Smilovic, Pant, Tramberend and Hall (in review with Nature Communications Earth and Environment [manuscript status is 'Accept in Principle', conditional on meeting editorial requests]), *Closing the climate adaptation gap for water supply utilities.* **(Chapter 6)**

Contents

1. Introduction 1

1.1. Research Gaps 5

1.1.1. Lack of multi-hazard risk assessment framework for water supply systems 5

1.1.2. No climate adaptation prioritisation framework that captures both utility and household cost of water supply disruptions 7

1.1.3. Limited quantification of global climate drought and adaptation benefits 9

1.2. Aims and objectives 11

1.3. Structure 12

1.4. Key terms 14

2. Literature review 15

2.1. Climate hazards impacting potable water supply services 18

2.1.1. Drought hazard modelling 18

2.1.2. Flood and extreme wind hazard modelling 19

2.1.3. Hazard event sets 20

2.1.4. Climate change impact on hazards 23

2.1.5. Summary 25

2.2. Climate risk to water supply utilities and their customers 27

2.2.1. Infrastructure as a network to understand climate risk propagation 27

2.2.2. Potable water supply climate-related disruption pathways 31

2.2.3. Impacts to utilities and customers 34

2.2.4. Summary 36

2.3. Adaptation strategies for water utilities 37

2.3.1. Adaptation optioneering approaches 38

2.3.2. Just adaptation frameworks 46

2.3.3. Summary 48

3. The case of the Jamaican National Water Commission....50

4. A multi-hazard risk framework to stress-test water supply systems to climate-related disruptions....53

4.1. Introduction....55

4.2 Materials and Methods....59

4.2.1 Overview of generally applicable framework....59

4.2.2. Application of framework to Jamaica case study....60

4.3. Results....71

4.3.1. Utility-scale risk in terms of CDD....71

4.3.2. Spatial correlation of CDD....73

4.3.3. System level annual expected CDD and event duration....75

4.4. Conclusions and Discussion....77

5. Prioritising climate adaptation options to minimise financial and distributional impacts of water supply disruptions....81

5.1. Introduction....83

5.2. Materials and Method....88

5.2.1. Overview....88

5.2.2. Application of multi-hazard risk approach for optioneering....91

5.3. Results and Discussion....100

5.3.1. Benefit cost ratio of all options....100

5.3.2. System level optioneering....101

5.4. Assumptions, limitations, uncertainty and merits....108

5.5. Concluding remarks....111

6. Closing the climate adaptation gap for water supply utilities....113

6.1. Introduction....115

6.2. Results....118

6.2.1. Present-day and future utility drought risk....118

6.2.2. The business case for adaptation....120

6.2.3. Optimal adaptation investment 124
6.3. Discussion 126
6.4. Methods 131
6.4.1. Overview 131
6.4.2. Water supply utility database 131
6.4.3. Drought risk method 134
6.4.4. Climate adaptation optioneering 137
7. Conclusions 142
7.1. Summary of findings 143
7.1.1. Magnitude of multi-hazard climate risk faced by the NWC 143
7.1.2. Priority climate adaptation options for the NWC and its customers 144
7.1.3. Global distribution of drought adaptation needs and priority actions 144
7.2. Discussion & Limitations 146
7.2.1. Risk metrics for understanding multi-hazard water supply disruptions 146
7.2.2. The importance of climate resilient water supply utilities for society and development 147
7.2.3. Generalising the quantification of adaptation benefits 148
7.2.4. Adaptation finance to reverse the vicious cycle? 149
7.2.5. Summary of peer review 150
7.3. Recommendations for future research 153
7.3.1. Use of alternative datasets for climate risk analysis 153
7.3.2. Water-energy nexus 153
7.3.3. Inclusion of a wider range of hazards for multi-hazard framework 154
7.3.4. Global multi-hazard risk analysis for water supply utilities 154
7.3.5. Accounting for water quality deterioration during extremes 154
7.3.6. Development of decision-relevant metrics 155
7.3.7. Co-benefits of leakage reduction 155

7.3.8. Alternative adaptation solutions 156
8. Closing remarks 157
9. References 159
10. Appendices 199
Appendix A: Co-author paper contribution statements 200
Appendix B: Supplement to chapter 4 209
Supplementary Text B-1: Spatial synthetic flood events 209
Supplementary Text B2: Infrastructure criticality 215
Supplementary Text B3: Uncertainty analysis 218
Appendix C: Supplement to chapter 5 224
Supplementary Text C1: Overview of multi-hazard risk analysis (Becher *et al.*, 2023). 224
Supplementary Text C2: Water balance (Becher *et al.*, 2023). 229
Appendix D: Supplement to chapter 6 232
Supplementary Text D1. Community Water Model. 237

List of Figures

Figure 1 Schematic overview of a water supply system supplying a city. Interactions between climate hazards and system components annotated in red, with droughts impacts water resources and floods and/or extreme wind impacts assets. Water supply system schematic was redrawn and adapted based on (Summaryplanet, 2023). 2

Figure 2 Conceptual overview of how risk is a function of climate hazards, social exposure and vulnerability. Figure taken from (Zscheischler et al., 2018).....15

Figure 3 Shows the extremeness of river flows (relative to historic records) across river gauges in the UK during three widespread flood events, where the larger and darker point represent a more extreme flow. Figure taken and adapted from (Towe, Tawn and Lamb, 2018). 22

Figure 4 Conceptual representation of how floods and drought hazards interact with system-based vulnerability characteristics to result in asset damage and water shortage risk. Water supply system schematic was redrawn based on (Summaryplanet, 2023). 30

Figure 5 Interaction between climate-related disruptions, climate adaptation and the water utility vicious cycle. 38

Figure 6 Graphical representation of the investment costs, in million British pounds, (x-axis) of a portfolio of 30 adaptation plans against the residual drought risk, in million British pounds, that would remain after each option's implementation (y-axis). Figure taken from (Borgomeo et al., 2018, p. 480). 40

Figure 7 Conceptual representation of the interaction between utility vicious cycle and poverty trap for low-income customers. 46

Figure 8 Illustration of decision-making framework tailored to analyse trade-offs between maximizing access to low welfare population and maximizing overall access. Figure taken from (Garcia et al., 2021, p. 11).....48

Figure 9 Potable asset data comprising 129 plants, 409 pumping stations, 169 wells and 2 dams that are owned and operated by the NWC Utility in Jamaica, the NWC (National Spatial Management Branch, 2010). The locations of major city locations are also provided (OpenStreetMap, 2018).....51

Figure 10 Methodological framework 60

Figure 11 Potable asset data comprising 129 plants, 409 pumping stations, 169 wells and 2 dams that are owned and operated by the NWC Utility in Jamaica, the NWC (National Spatial Management Branch, 2010). The locations of major city locations are also provided (OpenStreetMap, 2018).....67

Figure 12 Customer Disruption Days (CDD) in millions and as a % of the maximum potential annual NWC customer-base disruption for water shortage (top, a) and asset damage (bottom, b) risks in Jamaica under baseline climate conditions (light grey) and end of century RCP2.6 (orange), 4.5 (pink) and 8.5 (dark red) up to a 1 in 1000-year event.....72

Figure 13 Annual expected CDD per climate change scenario and per hazard driver in CDD and as a % of the maximum potential annual NWC customer-base disruption: cyclone wind (dark blue), non-cyclone related flood (light blue) and water shortages (dark red). 73

Figure 14 Likelihood of a water supply disruption in affecting two parishes simultaneously, (a) outlines the parish locations and distribution of households (OpenStreetMap, 2018). The correlation between parish-level CDD calculated as the correlation coefficient between the fraction of total parish customers disrupted multiplied by the days of disruption for water shortage events (red, b) and asset damage events (blue, c). 74

Figure 15 Annual expected CDD per system induced by water shortages (left) and asset damages (right). The baseline climate scenario output is shown as the spatial pattern is similar across climate scenarios. 76

Figure 16 Annual expected drought-induced water shortage duration in months under baseline climate scenario (left) and asset damage disruption duration in days under baseline climate scenario (right). 76

Figure 17 Methodological framework for multi-hazard optioneering with respect to economic and equity (poverty benefit) incentives.....90

Figure 18 Cost (DCs) and benefit (DBs) per leakage reduction option (top) and flood defence option (bottom), where points are coloured by the tankering costs saved by customers in poverty. All costs are provided in million Jamaican dollars. Vertical lines associated with each scatter point represent the uncertainty associated with the benefit of each option. The black line

illustrated the 1-1 relationship between benefit and cost; therefore, options are associated with BCRs that are >1. 101

Figure 19 BCRs across all considered protection levels (20-, 50-, 100-, 200- and 500-year return period) $BCRspd$ for leakage reduction in red (left) and flood protection in blue (right). 103

Figure 20 Benefit Cost Ratio ($BCRs$) per option, with flood defence options highlighted in blue and leakage reduction options in red. Options are ranked from those offering the highest BCR to the lowest. BCRs, plotted on the y axis using a log scale for plot clarity. 104

Figure 21 Option with the greatest BCR per system (OIs). Drought options are indicated in red, with darker red highlighting drought mitigation options with the greatest BCRs. Flood defence options are indicated in blue, with darker blue highlighting flood protection options with the greatest BCRs. Systems with the most economically efficient options are annotated. Parish outlines are plotted for greater spatial context. 105

Figure 22 The contribution of the most efficient option (OIs) of each type towards reducing costs for those in poverty ($DPBs$) in absolute terms in Jamaican dollars. Drought options are indicated in orange, with darker orange highlighting drought mitigation options with the greatest benefit to those in poverty. Flood defence options are indicated in purple, with darker purple highlighting flood protection options with the greatest benefit to those in poverty. Systems with the most economically efficient options are annotated. Parish outlines are plotted for greater spatial context. 106

Figure 23 Bivariate plot showing the interaction between economic- and equity-orientated objectives for each system, spatially. Strong option performance solely with respect to equity appear in yellow. Strong option performance solely with respect to economic objectives appear in blue. Strong performance with respect to both objectives appear in red. Systems with optimal options that have a strong synergy between both objectives are annotated. Five quantiles are applied to the score of each option relative to each objective. Parish outlines are plotted for greater spatial context. 107

Figure 24 (a) Present day annual expected drought risk per utility (utility headquarter location plotted) measured in terms of the annual expected UWSD per utility as a fraction of a year-round disruption of their full customer base. (b) Change in future annual expected UWSD -

projected under the middle-of-the-road SSP3-RCP7.0 scenario - relative to the present-day annual expected UWSD. Areas that are not covered by the utility database used for our analysis are indicated with hatching. 120

Figure 25 (a) (a) BCR of optimal option identified per utility. BCR is measured as the mitigated tariff revenue at risk divided by the cost of each adaptation option. Utilities that were found not to benefit from adaptation are plotted in grey. Areas that are not covered by the utility database used for our analysis are indicated with hatching. (b) Present day annual expected risk, measured as the percentage of the region's customer base at risk (in the present day), is plotted in light pink. Future risk (calculated as a mean of considered future scenarios) is plotted in dark red. The residual risk that could be achieved by implementing all utility-optimal adaptation options is plotted in light blue. The residual risk that could be achieved only by adaptation options with BCRs>1 is represented in dark blue. Error bars indicate the range of results produced across considered future scenarios. Note that all future and residual risk values are taken as an average over the 2030–2060-time horizon projected for the adaptation analysis. 123

Figure 26 (a) optimal option type identified per utility. Utilities for which the adaptation options considered were ineffective at reducing risk are plotted in grey. Areas that are not covered by the utility database used for our analysis are indicated with hatching. (b) % fraction of all utilities where optimal option is identified to be storage (navy), desalination (pink) and leakage reduction (orange) per region. Error bars represent the uncertainty in the optimal option type selection, induced by uncertainty in each option type's cost. 125

List of Appendices

Appendix A: Co-author paper contribution statements 200

Appendix B: Supplement to chapter 4 209

Appendix C: Supplement to chapter 5 224

Appendix D: Supplement to chapter 6 232

1. Introduction

Resilient water supply infrastructure is fundamental for human life and wellbeing. Further to the number of people who are not yet connected to a safe, regulated water network, millions suffer from frequent and prolonged water supply disruptions (Hope and Rouse, 2013; Rouse, 2014). This problem is particularly acute in the developing world, where countries lack the finances for sufficient water infrastructure maintenance and development, and where extreme climatic and hydrological variability make the challenge of designing resilient water systems even greater (Grey *et al.*, 2013; Hall *et al.*, 2014; Gaupp, Hall and Dadson, 2015).

Drinking water infrastructure systems are designed to abstract, treat, and distribute water from sources (e.g., rivers, lakes, groundwater) to households, communities and businesses. Utility water supply to households is used for drinking (although utility water may not be safe to drink without treatment at home), cooking, bathing, washing dishes/laundry and toilet flushing. In this this thesis, resilient water infrastructure is characterised as the sustainable provision of water despite 1) variations in water resources (droughts) and 2) withstanding asset failures (e.g., due to flood inundation or extreme cyclonic winds). Figure 1 provides a schematic overview of a water supply system supplying a city, comprising both a river and well source, a river abstraction pumping station, a water treatment plant, storage tank, booster pumping station and a network of pipelines. Interactions between different hazards, including droughts, floods and extreme winds, and the system are annotated, with droughts impacting water availability at each source and flood and/or extreme winds impacting the physical conveyance and treatment infrastructure (Cromwell, Smith and Raucher, 2007). Floods and extreme winds can also impact the sources by increasing the runoff of contaminants from cities and agricultural land (ibid).

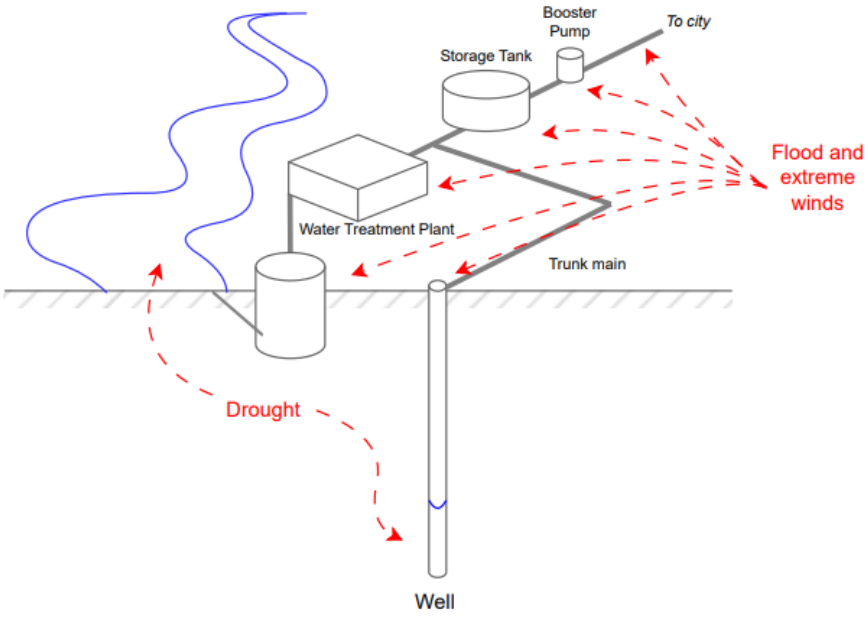


Figure 1 Schematic overview of a water supply system supplying a city. Interactions between climate hazards and system components annotated in red, with droughts impacts water resources and floods and/or extreme wind impacts assets. Water supply system schematic was redrawn and adapted based on (Summaryplanet, 2023).

Thus, water infrastructure is subject to different kinds of disruption triggered by different hazards. Another particular consideration for analysing risks to water infrastructure is that - because water is difficult and expensive to transport - the configuration of the infrastructure system is constrained by the proximity of water resources to communities. Indeed, water infrastructure in countries with low reservoir capacity is often made up of relatively small, decentralised systems fed by rivers, springs, and wells. Therefore, if the source supplying a particular community is affected, for example by drought conditions, there is not always enough redundancy in the system to allow substitution from other, unaffected sources (Zhang *et al.*, 2019). Further, because climatic extremes are often driven by large-scale weather systems (Berghuijs *et al.*, 2019; Gaupp *et al.*, 2020), it is likely that neighbouring systems are affected by the same event, resulting in more widespread disruptions.

Water security can be defined in a multitude of ways. Grey and Sadoff (2007, p. 548) define the concept as "the availability of an acceptable quantity and quality of water for health, livelihoods, ecosystems and production, coupled with an acceptable level of *water-related risks* to people, environments and economies". Thus, understanding the risks that multiple hazards impose on water infrastructure, and the spatial footprint of these hazards, can be used to prioritise adaptation strategies that improve infrastructure resilience and enhance water security

(Hall and Borgomeo, 2013). The more hazards to which a utility is exposed (and associated impact propagation channels) that are comprehended within a coherent risk analysis framework, the more efficiently utilities can allocate limited adaptation budgets to the most critical and vulnerable parts of the system. Benefit-Cost Ratio (BCR) analysis on new, proposed water supply infrastructure typically involves comparing the additional revenue that would be collected through tariffs over a new project's lifetime to the cost of its construction, operation, and maintenance (Altaf, 1994). However, this does not account for the benefits of projects that improve the resilience of potable water supply services for existing customers to climate shocks now and in the future. For example, reducing the costs of repairing physical asset damages (due to extreme winds and/or flood inundation) (Pant *et al.*, 2018) or reducing the frequency with which households must acquire water from paid sources outside utility services (e.g., bottled water (Heflin, Jensen and Miller, 2014) or tanker trucks (Garrick, O'Donnell, *et al.*, 2019)). This means that the focus of new utility investments can become skewed towards expanding network coverage and away from projects that improve the reliability of services for existing customers (Altaf, 1994). This is problematic because (climate-induced) service disruptions can significantly undermine efforts towards universal water access (Clasen, 2012), particularly for the most vulnerable groups of society (Pattanayak *et al.*, 2005). Further, in the context of effectively integrating climate adaptation strategies within national development plans and objectives (Oates, Conway and Calow, 2011; Conway and Mustelin, 2014), the identification of 'pro-poor' infrastructure investment strategies are increasingly called-for (Sohail *et al.*, 2007; Nagpal, Malik and Eldridg, 2018). However, decision-making approaches for strengthening the climate resilience of water supply services through an equity lens are not yet mainstreamed.

With the climate finance agenda rapidly gaining momentum, quantifying the climate adaptation benefits of infrastructure investments is becoming increasingly relevant for investors (CPI, 2020). Climate-resilient water supply utilities provide a material contribution towards adaptation goals by providing safe, reliable, and affordable drinking water services to households, even during climate extremes and despite conditions of increasing uncertainty under climate change. This makes the water sector an important channel for climate adaptation finance. Among many barriers in the deployment of capital for adaptation, one is the challenge that investors face in quantifying the contribution of projects towards the climate adaptation agenda (Ranger and Mullan, 2022), i.e., their effectiveness at reducing climate-related risks such as climate-induced water supply disruptions. This means that, although water supply

utilities are a strong candidate for climate finance investments, their potential to attract greater flows of finance has yet to be fully realised.

1.1. Research Gaps

A risk-based approach is widely applied in the water infrastructure planning literature (Hall *et al.*, 2012; Matrosov, Woods and Harou, 2013; Borgomeo *et al.*, 2015, 2018; Herman *et al.*, 2016; Huskova *et al.*, 2016; Guo, Westra and Maier, 2018). The approach can be divided into two steps. The first step is to analyse the likelihood of possible undesirable outcomes by "considering the full range of conditions [hazards] to which a system might be exposed, including those that exceed the 'design condition'…" (Hall and Borgomeo, 2013, p. 6). The second step is to identify an efficient, or priority adaptation strategy by comparing the costs and benefits (in terms of risk mitigated) of adaptation measures: "Analysis of risks provides essential evidence to enable choices between alternative courses of action" (Hall and Borgomeo, 2013, p. 9).

Despite the maturity of this research field, and the application of climate risk-based decision making for water utilities to a wide range of case studies around the world (Ghile *et al.*, 2014; Zhao *et al.*, 2018), three major research gaps were identified and developed in this thesis.

1.1.1. Lack of multi-hazard risk assessment framework for water supply systems

1.1.1.1. Multi-hazard risk approach

All infrastructure is vulnerable to a range of climate extremes, with floods and hurricanes imposing a heavy financial burden in terms of infrastructure damages (Hall *et al.*, 2019). Much work has investigated climate risks posed to transport (Koks *et al.*, 2019), energy (Byers *et al.*, 2020), digital communications (Oughton and Russell, 2020) and water infrastructure (Dobson et al., 2020), with some research considering the combined impacts of multiple hazards (Koks et al., 2019). Water infrastructure systems stands out compared to other infrastructure sectors because water resources can be more challenging to substitute during droughts, particularly if sources are exposed to synchronous shortages during the same drought events and networks have limited storage capacity and redundancy to buffer against shocks. This is in contrast to other infrastructure systems: for example, in the energy sector, where vulnerable energy supplied by hydropower could be substituted with alternatives such as solar and wind that wouldn't be affected by the same hazard; in the transport sector, drought- and flood-affected inland waterway networks might be substituted using roads and railways.

Multiple hazards disrupt water infrastructure in different ways. Hydrological droughts are among the most widely studied climate-related risks to water systems (Milly, Dunne and Vecchia, 2005). Depending on the capacity of water storage infrastructure, the system may be able to buffer against droughts, protecting dependent populations from water shortages (Grey and Sadoff, 2007; Hall *et al.*, 2014; Gaupp, Hall and Dadson, 2015). Flood inundation and extreme cyclonic wind poses a different kind of climate-related risk to water systems (Pant *et al.*, 2018; Thacker *et al.*, 2018; Stødle *et al.*, 2021). Non-climate related risks, such as earthquakes, have also been studied widely in the context of water supply systems (Bata, Carriveau and Ting, 2022), as well as operational risks including due to mechanical, electrical and chemical failures (Gheisi, Forsyth and Naser, 2016). In the event that critical water infrastructures assets fail due to flood or wind-related damage, they would have to be taken out of operation, causing disruptions (Pant *et al.*, 2018; Stødle *et al.*, 2021).

Although previous research has analysed the risks that droughts (Borgomeo *et al.*, 2015), extreme wind (Stødle *et al.*, 2021) and flooding (Pant *et al.*, 2018) independently pose to water supplies, no research has provided a multi-hazard risk framework for water infrastructure.

1.1.1.2. Spatially coherent climate extremes

Hydrological extremes are driven by large-scale weather systems and may cause flood inundation, excessive turbidity levels or the desiccation of water resources at multiple water supply systems simultaneously. The importance of understanding the risk of spatially extensive extremes causing widespread disruptions is briefly discussed by Hall and Borgomeo (2013, p. 8): "Understanding spatial correlation is essential to understand how risks aggregate with spatial scale. Countrywide risk estimates may conceal smaller locations of high risk, in particular when populations are concentrated in areas of higher risk". However, the vast majority of existing research on climate risks to infrastructure does not account for the temporal relationship between events across space. For example, this includes research that employs return period maps for flood risk analysis (Pant *et al.*, 2018) or studies for which the scope of analysis is restricted to a single water supply system (Borgomeo *et al.*, 2015), where correlated disruptions between systems would not be relevant for the analysis.

Understanding the risk of widespread disruptions caused by spatially coherent hydrological extremes is necessary for water infrastructure for two reasons. First, widespread disruptions could overwhelm the contingency plans that utilities have in-place to supply disrupted communities, e.g., provisioning of bottled water or tanker trucks. Therefore, if a large region

and population is affected simultaneously, this poses a greater burden than would be captured by considering disruption events independently. Although some research captures the likelihood of hydrological extremes occurring simultaneously at multiple infrastructure network locations - for example in the context of flood (Berghuijs *et al.*, 2019; Quinn *et al.*, 2019) and drought risk (Dobson *et al.*, 2020) - no research has analysed the risk of widespread disruptions to water supply networks caused by both extreme droughts, floods and cyclones.

Second, in the context of allocating scarce financial resources to the most vulnerable parts of the water utility systems, it would be beneficial to understand how multiple hazards interact with water infrastructure systems by integrating multiple disruption pathways, in a single modelling framework. Thus, developing a modelling approach to prioritise a portfolio of multiple adaptation option types, e.g., flood defences vs leakage reduction, it is important to quantify and compare adaptation needs across multiple relevant hazards. However, apart from a handful of local or regional studies that focus on the exposure of water supply infrastructure to hazards that affect both water availability and damage to the supply network (e.g., Espada et al., 2013), multi-hazard risk analysis studies of water supply systems are lacking.

Therefore, there is scope to build-upon this literature by 1) quantifying the risk of widespread disruptions caused by multiple hazards and 2) exploring its application for large-scale water infrastructure planning strategies.

1.1.2. No climate adaptation prioritisation framework that captures both utility and household cost of water supply disruptions

Analysing water infrastructure system risks can guide the implementation of new infrastructure projects that promote system resilience and expand network coverage. Multi-optimisation approaches are commonly used to evaluate proposed water infrastructure options with respect to various objectives (e.g., minimising the risk of water service disruption while minimising cost) (Reed *et al.*, 2013; von Lany *et al.*, 2013; Mortazavi-Naeini, Kuczera and Cui, 2014). Hereafter, this evaluation of adaptation options will be referred to as optioneering.

Literature on improving the resilience of existing networks involves evaluating options such as increasing water resource extraction, increasing raw and potable storage capacity and reducing leakage (Chang, Chen and Yeh, 2009; Mortazavi, Kuczera and Cui, 2012; Reed *et al.*, 2013; Borgomeo *et al.*, 2016). For example, Borgomeo et al. (2018) perform a stress-test of the Thames water system to 100 realisations of baseline, near future and far future rainfall,

using the severity of water use restrictions imposed on utility customers as a risk metric. They then apply a multi-objective optimisation algorithm to identify a desirable adaptation strategy. Comparing the economic costs associated with drought-related restrictions against the costs of a given water adaptation measure provides a powerful insight for utility managers to undertake cost-benefit analysis on adaptive infrastructure and select priority investment strategies.

However, this research is concentrated in high-income countries (Haasnoot *et al.*, 2013; Culley *et al.*, 2016; Hassanzadeh *et al.*, 2016; Kingsborough, Borgomeo and Hall, 2016; Turner *et al.*, 2016), that do not speak to the financial losses incurred during water service disruptions in low- and middle-income countries. Indeed, in these countries, the impacts of such failures are not only borne by utility companies, but more often spill over to utility customers (Altaf, 1994; Baisa *et al.*, 2010; Heflin, Jensen and Miller, 2014; Bakker, 2017). This is because, unless utilities have compulsory contingency and compensation plans in-place to replace disrupted services, dependent households are forced to acquire water from elsewhere during supply disruptions (Majuru, Suhrcke and Hunter, 2016). Other sources of water to which households may turn to during disruptions include unregulated sources such as springs and wells, bottled water and private tanker trucks, the latter being a key agent in supplying water during utility supply disruptions globally (Wutich, Beresford and Carvajal, 2016; Garrick, O'Donnell, *et al.*, 2019). The externalised costs on society to replace utility water supply are often far beyond what the relative cost of receiving water from a functioning piped water supply system: for example, in Ghana, water distribution by road tanker costs 14 times as much as distribution through the pipe networks (Rouse, 2013). This may represent a significant additional household expense for connected customers (i.e., beyond their regular tariff payments), particularly for low-income households (Pattanayak *et al.*, 2005). Indeed, the burden of water supply disruptions, as with other climate impacts, are typically disproportionately felt by lower income households, reinforcing the conditions that prevent households from moving out of poverty (Eakin *et al.*, 2016). Despite this, the importance of mainstreaming equity considerations in potable water sector planning processes (Grasham, Charles and Abdi, 2022), and wider development goals in climate adaptation planning (Klein, 2010; Conway and Mustelin, 2014), to the authors' knowledge, no existing frameworks investigate synergies and trade-offs between economic and equity objectives when comparing alternative climate adaptation infrastructures. Garcia et al. (2021) evaluates potential investments in drinking water infrastructure in coastal Bangladesh, comparing deep tube wells, desalination plants and piped systems. In this study, multi-objective optimisation is used to explore the trade-off between

two public policy goals: 1) maximising the population with access to improved water supplies (the greater good); and 2) maximising provision for the poorest (the greater need). Garcia et al. (2021) shows how different investment strategies are evaluated with respect to how many people they serve and the level of vulnerability in the served population. This research acknowledges the importance of water infrastructure development for improving water services to those in poverty and vulnerable situations and, indeed, shows how broader development objectives, such as equity, can be integrated into water infrastructure planning decisions. There is scope to build-upon this, among other existing water infrastructure planning frameworks, to integrate or mainstream equity considerations.

1.1.3. Limited quantification of global climate drought and adaptation benefits

Although every utility has a unique set of challenges and are subject to specific regulatory, operational and physical constraints, comparison across a large sample of utilities can be useful for identifying opportunities for investors with regional or global operations, e.g., Climate Funds and Multi-Lateral Development Banks. Most analysis of water scarcity at the global scale has focussed on meteorological and hydrological droughts, and their implications for agriculture (Vogel *et al.*, 2019; Gaupp *et al.*, 2020; Meza *et al.*, 2020). With increasing urbanisation and expanding piped water networks throughout the world, greater volumes of water are being allocated to municipal water supplies. The climate risk to these supplies is real (Muller, 2018; Chakrabarti, 2019), but under-researched. Maps of hydrological drought risk or water stress that have been developed previously (Gassert *et al.*, 2014), but these do not account for the influence of water utility infrastructure in mediating risk. These have previously been captured in more detailed, small-scale studies (Borgomeo *et al.*, 2014; Culley *et al.*, 2016), but not yet at the global scale.

Existing global analysis of drought risk to, and adaptation planning in, the municipal water sector is limited for two main reasons: 1) lack of consideration of utility scale of operations and infrastructure characteristics, 2) lack of rigorous, risk-based approaches for identifying benefits of adaptation options and optimal investments. Speaking to the first limitation around scale, previous studies typically adopt the catchment as the unit scale for drought analysis (McDonald *et al.*, 2011; Hanasaki *et al.*, 2013; Padowski and Gorelick, 2014; He *et al.*, 2021). However, this scale does not necessarily speak to the way that water is managed by utilities. Speaking to the second limitation around risk-based approaches, existing studies have indicated

the suitability of adaptation options across geographies based on qualitative criteria (Wada, Gleeson and Esnault, 2014; Larsen *et al.*, 2016; Buurman, Mens and Dahm, 2017; Greve *et al.*, 2018; Krueger, Rao and Borchardt, 2019; He *et al.*, 2021). However, there is scope to undertake more rigorous risk analysis procedures, inspired by approaches typically undertaken at a smaller scale (Brown *et al.*, 2012; Borgomeo *et al.*, 2014), to prioritise options based on their risk reduction potential.

1.2. Aims and objectives

The goal of this DPhil is to develop and apply a set of complementary frameworks for analysing climate risks to water supply systems to guide climate adaptation decision making. This overarching aim is divided into three research objectives.

1) To develop a framework for analysing widespread, multi-hazard climate risks to water supply systems.
2) To develop an approach for quantifying adaptation benefits for both water supply utilities and their customers, including those in poverty.
3) To develop a global framework for quantifying utility-level drought risk as a platform for optioneering across multiple drought adaptation options and performing BCR assessments.

The first two modelling frameworks are developed through a case study of the National Water Commission in Jamaica, where water supplies are vulnerable to multiple climatic extremes and subject to frequent disruptions. The third framework is applied at a global scale.

1.3. Structure

The thesis will consist of seven chapters.

Chapter 2: Literature review

This chapter provides an overarching literature review across relevant research fields.

Chapter 3: Case study: Jamaica's national water supply utility

This chapter provides an overview of the National Water Commission in Jamaica, that was the case study used for two frameworks developed in this thesis.

Chapter 4: A multi-hazard risk framework to stress-test water supply systems to climate-related disruptions.

This chapter presents a generally applicable framework for generating multi-hazard event sets to stress test water infrastructure systems and compare the risk of customer disruptions. The framework is applied to the case of the National Water Commission's system in Jamaica.

Becher, O., Pant, R., Verschuur, J., Mandal, A., Paltan, H., Lawless, M., Raven, E. and Hall, J., 2023. A Multi-Hazard Risk Framework to Stress-Test Water Supply Systems to Climate-Related Disruptions. Earth's Future, 11(1), p.e2022EF002946.

Co-author contributions: Prof Jim Hall from the University of Oxford assisted with the conceptualisation and structuring of the paper, as well as with review and feedback on the manuscript. Dr Jasper Verschuur and Dr Raghav Pant from the University of Oxford, Dr Arpita Mandal from the University of the West Indies and Homero Paltan from the World Bank assisted with review and feedback on the manuscript. Mark Lawless and Emma Raven provided JBA flood event and return period map data.

Chapter 5: Prioritising climate adaptation options to minimise financial and distributional impacts of water supply disruptions.

This chapter presents a framework for evaluating the effectiveness of climate adaptation options in reduced water supply infrastructure damages and utility customer spending on substituting disrupted services. The framework is applied to the case of the National Water Commission's system in Jamaica.

Becher, Verschuur, Pant and Hall (in review with Environmental Research: Infrastructure and Sustainability [manuscript status is 'Accept in Principle', conditional on meeting editorial

requests]), Prioritising climate adaptation options to minimise financial and distributional impacts of water supply disruptions.

Co-author contributions: Prof Jim Hall from the University of Oxford assisted with the conceptualisation and structuring of the paper, as well as with review and feedback on the manuscript. Dr Jasper Verschuur and Dr Raghav Pant from the University of Oxford assisted with review and feedback on the manuscript.

Chapter 6: Closing the climate adaptation gap for water supply utilities.

This chapter presents a framework for evaluating utility level drought risk and the effectiveness of drought mitigation options, including increasing desalination capacity, reducing leakage losses and increasing storage capacity. The framework is applied to ~5,600 utilities globally.

Becher, Verschuur, Smilovic, Pant, Tramberend and Hall (in review with Nature Communications Earth and Environment [manuscript status is 'Accept in Principle', conditional on meeting editorial requests]), Closing the climate adaptation gap for water supply utilities.

Co-author contributions: Prof Jim Hall from the University of Oxford assisted with the conceptualisation and structuring of the paper, as well as with review and feedback on the manuscript. Dr Mikhail Smilovic from the International Institute of Applied Systems Analysis provided the Community Water Model inputs and advised on their methodological application for the paper. Dr Raghav Pant and Dr Jasper Verschuur from the University of Oxford and Dr Sylvia Tramberend from the International Institute of Applied Systems Analysis assisted with review and feedback on the manuscript.

Chapter 7: Synthesis

This chapter summarises the work undertaken for this thesis and outlines key reflections on potential applications of the DPhil research in policy and practice, and outlines a set of recommendations for future research.

Chapter 8: Concluding remarks

This chapter provides a high-level conclusion of the work undertaken throughout this thesis and synthesises some key takeaways.

Appendices

The appendices include the author contributions and statements, followed by the Supplementary Information associated with chapters 4, 5 and 6.

1.4. Key terms

A selection of key terms are provided here to clarify some important concepts that are relevant throughout the thesis.

Climate Hazard: A hazard is the potential occurrence of climate-related acute physical events or chronic trends that may cause damage and loss (IEA, 2020).

Hazard event sets: Hazard event sets are defined as a wide set of observed and/or synthetically generated spatially coherent hazards events, characterised by their likelihood and severity (Guillod *et al.*, 2018).

Vulnerability: Infrastructure vulnerability can be defined as the degree to which a system or asset is susceptible to climate hazards of a given severity (IPCC, 2001).

Exposure: Exposure can be defined as the people and assets, and associated economic activity, that could be adversely affected by a given hazard (Peduzzi *et al.*, 2009).

Quantified risk analysis: quantified risk analysis involves estimating the probability and consequences of harmful events (hazards) (Hall and Leng, 2019).

Compound event: Where hazard impacts depend on multiple statistically dependent variables or events, it is known as a compound event (Leonard *et al.*, 2014).

Stress-testing: Stress testing involves analysing the potential impacts of a given hazard scenario on a system (Hall and Leng, 2019).

Resilience: Resilience is defined as "the ability of a system, community or society exposed to hazards to resist, absorb, accommodate, adapt to, transform and recover from the effects of a hazard in a timely and efficient manner" (UNISDR, 2009, p. 24).

Climate adaptation: Climate adaptation represents an adjustment in natural or human systems aimed at reducing the severity of, or risks associated with, adverse climate change impacts, as well as actions seeking to harness any beneficial opportunities that may arise under a changed future climate (IPCC, 2007).

Optioneering: Optioneering is evaluation of alternative options based on a prioritisation framework, typically with respect to their BCRs.

Equitable prioritisation: Among other definitions, equitable prioritisation is defined here as the prioritisation of the most in need (Sen, 1974).

2. Literature review

Climate risk is commonly defined as the interaction of hazard (i.e. the likelihood of a hazard occurring), exposure (i.e. the people, infrastructure, housing, and other tangible human assets located in hazard-prone areas) and vulnerability (i.e. the conditions determined by physical, social, economic and environmental factors or processes which increase the susceptibility of an individual, asset or system to the impacts of hazards) (UNDRR, 2022). Figure 2 provides a high-level overview of these risk drivers: climate hazards are driven by both natural variability and anthropogenic climate change; both exposure and vulnerability are driven by socioeconomic processes. Climate hazards are climate-related physical events associated with negative outcomes, characterised by their likelihood and severity. Societal exposure can be driven by population density, high value assets and economic activity. Whereas vulnerability is driven by the likelihood of adverse outcomes among exposed populations, assets and/or businesses for a given hazard severity.

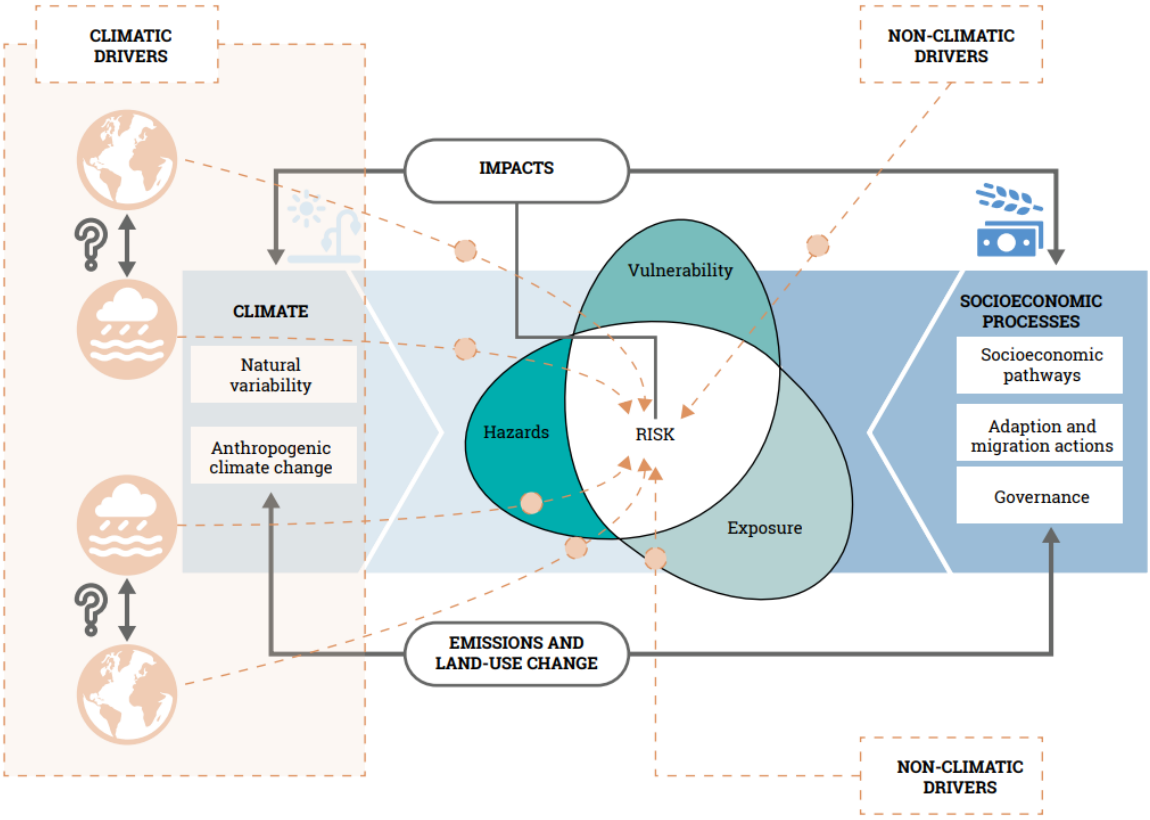


Figure 2 Conceptual overview of how risk is a function of climate hazards, social exposure and vulnerability. Figure taken from (Zscheischler et al., 2018).

Water supply utilities are exposed to a range of climate hazards, including extreme winds, flood inundation and drought. Due to the proximity of, and dependency on critical water supply infrastructure, such as surface water treatment plants and intakes, to waterbodies, floods and droughts are a major concern (Arrighi *et al.*, 2017; See, Nayan and Rahaman, 2017; Chau, Thanh and Toan, 2021; Barbetta *et al.*, 2022). Climate-related disruptions are prevalent across continents. Drought-related water supply restrictions have been implemented in low- and middle-income countries, in Tunisia, Mexico and Brazil (Romero, 2015; Tsui, 2022; Speakman Cordall, 2023), as well as in Europe, including in France, Italy and Spain (Tremlett, 2005; Giuffrida, 2022; O'Brien, 2023). Flood and/or cyclone-related disruptions have occurred in Japan, Peru, Indonesia and Puerto Rico (Collyns, 2017; De Córdoba, Campo-Flores and De Avila, 2017; GFDRR, 2018; IFRC, 2020). For example, an estimated 11 million people were affected by the crisis in Chennai, India, in 2019 when four of Chennai's drinking water supply reservoirs dried up (Kumar-Rao, 2019). Floods damaged treatment plants and canals in 2017 in Lima, Peru, disrupting water supplies for up to five days, with some treatment plants remaining inactive for 6 months (Collyns, 2017). The propagation pathways of drought versus floods and cyclones are different in nature. During droughts, where water supplies fall short of demand, water supply utilities must ration water across networks, either through implementing uniform water demand restrictions across their customer base (e.g., hosepipe bans), or cutting water supply to certain areas of the network during certain times of day (intermittent supply). Flood inundation and extreme winds can uproot trees and loosen debris, damaging water supply assets and/or electricity supplies and contaminating waterways, among other impacts (Collyns, 2017; Purwar, Sliuzas and Flacke, 2020; Lakhani, 2022).

Given utility exposure to climate-related water supply damage and disruption, risk quantification approaches can be valuable tools in targeting limited utility budgets across particularly vulnerable and exposed parts of utility networks. Further, in the context of the poor financial performance of water supply utilities globally (IBNET, 2022), attracting concessional financing provided by Multi-Lateral Development Banks and Donors can help utilities implement adaptation strategies that deliver more climate resilience services. Risk quantification and optioneering approaches can be used to estimate the impact of climate adaptation investments, to attract concessional loans from investors that are mandated to address the impacts of climate change.

“Risk-based decision making uses evidence of risks to inform individual and societal choices about which courses of action to adopt in future. It involves weighing up risks and costs, in the broadest possible sense, for a range of different actors” (Hall and Borgomeo, 2013, p. 3). The application of risk-based decision making is widely applied for prioritising climate adaptation options in the potable water supply sector (Borgomeo *et al.*, 2016; Kingsborough, Borgomeo and Hall, 2016). The workflow for analysing climate risk and adaptation needs for utilities can be outlined as follows: 1) identify the climate hazards to which the utility is exposed and future risk drivers under climate change; (2) identifying utility vulnerabilities and evaluate risks by combining the likelihood of the event with the modelled consequences; and (3) identify and prioritise suitable adaptation options to reduce present and future risk. The following literature review covers an overview of each step in the workflow of quantifying the risk reduction benefit of adaptation investments in the water supply utility sector. Section 2.1. covers an overview of the climate hazards to which utility infrastructure and services are exposed now and under climate change conditions. Section 2.2 provides an overview of how risk to utilities and their customers is quantified across different types of climate hazards that are associated with different risk propagation channels. Section 2.3 presents an overview of the range of adaptation approaches that are considered to adapt to different hazards and how options are prioritised in the literature both at the level of an individual utility and at a regional or global scale.

2.1. Climate hazards impacting potable water supply services

In many countries, climate hazards, such as floods, extreme cyclonic winds and droughts, pose a major risk of water supply failures. Understanding the probability and intensity of climate hazards is the first step of risk quantification. The modelled representation of a hazard plays an important role in how the spatial pattern of risk is captured and there is substantial variation between approaches. This section explores a range of modelling approaches adopted in the literature for drought, flood and extreme cyclonic wind risk analysis.

2.1.1. Drought hazard modelling

Droughts are among the more challenging hazards to define for risk analysis (Hall and Leng, 2019). They can be characterised based on multiple geophysical variables (e.g., precipitation, temperature, soil moisture, river flow, groundwater levels), across multiple dimensions (e.g., intensity, duration, and accumulated deficit). This has led to the creation of a wide range of drought definitions and indicators (Mukherjee, Mishra and Trenberth, 2018). Drought risk approaches often make use of such available indices, such as the Effective Drought Index, (Byun and Wilhite, 1999) and Standard (Effective) Precipitation Index (Spinoni *et al.*, 2019), are widely used for drought hazard and risk modelling. They involve collecting time series information on meteorological and/or hydrological data on a range of variables, including precipitation, soil moisture and streamflow (Van Loon *et al.*, 2016). Quantile thresholds are established, e.g., the median effective precipitation, to explore variations relative to this threshold (Bachmair, Kohn and Stahl, 2015). Some studies integrate these hazard maps with index-based information on vulnerability, to formulate a description of risk at large scales (Carrão, Naumann and Barbosa, 2016). Such approaches can provide useful insights into meteorological and hydrological variability across space (Hammond *et al.*, 2022) and how drought risks are projected to evolve over time under climate change conditions (Vicente-Serrano *et al.*, 2022). The problem with approaches that use arbitrary, quantile thresholds is that they do not necessarily predict the likelihood of physical drought impacts (Bachmair, Kohn and Stahl, 2015). The strength of hazard-impact relationships varies across sectors (Stagge *et al.*, 2015), however, in the potable water sector, this relationship is strongly mediated by the resilience of infrastructure systems (Tsakiris *et al.*, 2013). Further, they do not provide a platform to quantify the benefits of adaptation measures in quantified terms using tangible, meaningful metrics for decision makers.

In contrast to approaches that perform drought risk analysis by analysing meteorological or hydrological variations relative to a quantified threshold, more meaningful thresholds can be established to more accurately describe the risk, for example, of utilities being unable to serve populations fully. Such an approach may require detailed data on utility sources, infrastructure characteristics (including storage and leakage losses), and customer demand. As such, the approaches are typically undertaken for a single water supply system in context of sophisticated water resources model development (e.g., Kuczera, 1992). Further discussion on these approaches is provided in section 2.2.2.1. Once such a threshold is established, one can quantify the probability of relevant meteorological and/or hydrological variables, and associated consequences of, surpassing this threshold. For the analysis of drought hazard impact to water supply utilities, mass-balance approaches can be used to tangibly analyse the variation of water availability at utility sources relative to meet physical thresholds that are relevant for their operational performance during drought (Tsakiris *et al.*, 2013). This involves hydrological and/or hydrogeological modelling of utility sources. Process-based hydrological models simplify elements of the Earth's system to simulate the relationships between components of the water balance in space and time (Clark *et al.*, 2015). They use information about the land surface, such as topography, land use, upstream catchment size and soil type (Xiao *et al.*, 2019). They can be applied and calibrated at the catchment level using local data (Whitehead, Wilson and Butterfield, 1998) or at the regional or global level using data of lower quality and resolution but greater coverage (Burek *et al.*, 2020). Lower resolution outputs can provide only a limited insight into streamflow rates or groundwater level at point locations, e.g., across river abstraction points and/or borehole pumping stations. Highly granular studies have simulated water availability at a single source using (calibrated) process-based models (Mortazavi-Naeini *et al.*, 2019). On the other hand, many approaches rely on empirical data and statistical methods 1) to gap-fill incomplete records (Wheeler *et al.*, 2020) and 2) to generate synthetic time series data beyond the observed period (Borgomeo *et al.*, 2015). Indeed, weather and/or streamflow generators have become widely adopted tools for drought hazard modelling for risk analysis (Steinschneider and Brown, 2013; Herman *et al.*, 2016; Guo, Westra and Maier, 2018).

2.1.2. Flood and extreme wind hazard modelling

During periods of heavy precipitation, river discharge may exceed the capacity of river channels and surface infiltration rates, leading to fluvial and pluvial flood inundation events. Rainfall runoff models are used to quantify how precipitation is apportioned into different

components of the water balance - through interception, evapotranspiration and infiltration processes – to simulate how rainfall is converted into runoff (Robinson and Ward, 2017). These processes, and the subsequent apportionment between surface and subsurface runoff, determine the volume of water draining into streams and rivers during precipitation events (ibid). River routing models are used to represent how runoff is directed through a drainage network and, when combined with an inundation model, can be used to model flood inundation dynamics over the surrounding land based on the relationship between river flow and channel capacity (Lamb, Crossley and Waller, 2009; Yamazaki et al., 2011). These models are widely used for risk analysis applications (Hirabayashi et al., 2013; Winsemius et al., 2013). However, there are challenges with simulating flood inundation at scale, notably the limited resolution of digital elevation models (Manfreda, Di Leo and Sole, 2011), leading to inaccuracies and inconsistencies in model output (Trigg *et al.*, 2016).

Cyclonic wind speeds can be modelled using process based (weather) models (Casson and Coles, 2000; Lin et al., 2012) or parametric approaches (Vickery, Skerlj and Twisdale, 2000; Emanuel et al., 2006; Bloemendaal et al., 2020). The former typically being undertaken at smaller spatial scales, due to computational requirements (Lin et al., 2012), than the latter (Bloemendaal *et al.*, 2020). These approaches are used to understand the trajectory of cyclone paths, their rate of energy gain pre-landfall and rate of decay post-landfall, as well as associated intensity of precipitation and maximum wind speeds (Walsh *et al.*, 2016).

2.1.3. Hazard event sets

Time series information on water availability, flood inundation dynamics and wind speeds is often simplified for the purpose of risk analysis through extreme value analysis and the development of return period maps (Ward *et al.*, 2013). A return period is an estimated average time between events, such as floods, of a particular severity. The inverse of a return period approximates the annual exceedance probability. For example, if a flood of a given depth is estimated to occur once every 20 years, the probability of it occurring in a given year is ~1/20. The return period (or the likelihood) of a given event is found by statistically analysing a modelled or observed time series. For example, by ranking (r) annual maximum water level observed in a river channel over a period of n years - with 1 being highest water level and n being the lowest - the return period (RP) associated with a given water level and its associated annual exceedance probability can be formulated as in Equations 1 and 2, respectively:

$$RP = \frac{n+1}{r} \quad (1)$$

$$p = \frac{1}{RP} \quad (2)$$

This relationship can be generalised using Extreme Value Analysis, applied through a range of statistical models (e.g., Pearson, Gumbel and Generalised Extreme Value distributions) (Karim and Chowdhury, 1996). The most appropriate model can be selected based on the goodness-of-fit between the observed and the modelled distribution, with certain models being more appropriate for a given hazard (Karim and Chowdhury, 1996).

Droughts, floods and extreme winds can be driven by large-scale events. For example, large-scale climate dynamics, such as El Niño Southern Oscillation (ENSO) (Ward *et al.*, 2014), the North Atlantic Oscillation (NAO) (Trigo, Osborn and Corte-Real, 2002), high-pressure systems resulting in heatwaves and precipitation deficits (Ionita *et al.*, 2020) and atmospheric rivers (Paltan *et al.*, 2017), trigger floods and droughts across a wide range of spatial and temporal scales. For example, Figure 3 shows the extensive spatial pattern of three separate flood events in the UK and their associated damages to households, particularly relevant for the home insurance sector.

Risk assessments based on return period maps are widely accepted and commonly understood by governments, engineers and insurance underwriters. However, return period maps fail to inform users on the temporal relationship between events across space (Towe, Tawn and Lamb, 2018). As such, approaches that rely on return period maps do not capture the likelihood of events impacting multiple catchments, systems and/or assets simultaneously (Volpi, 2019). This can be problematic when aggregating impacts on spatially extensive, inter-connected infrastructure networks (Henriques and Spraggs, 2011; Dunant *et al.*, 2021). Assuming that the spatial impacts of a given hazard occur completely independently from each other (i.e., would not occur at the same time), would lead to an underestimation of risk (Lamb *et al.*, 2010; Duha Metin *et al.*, 2020). On the other hand, assuming that all the spatial impacts of a hazard would occur at the same time (i.e., assuming full dependence across hazard impacts), would lead to an over estimation of risk (ibid). The true risk must be derived by understanding the partial dependence across the spatial footprint of a given hazard (ibid). Statistical tools, such as copulas developed to quantify the joint probability of two or more variable are widely applied to capturing the covariance relationship of hazards at multiple locations (Jongman *et al.*, 2014; Gaupp *et al.*, 2020).

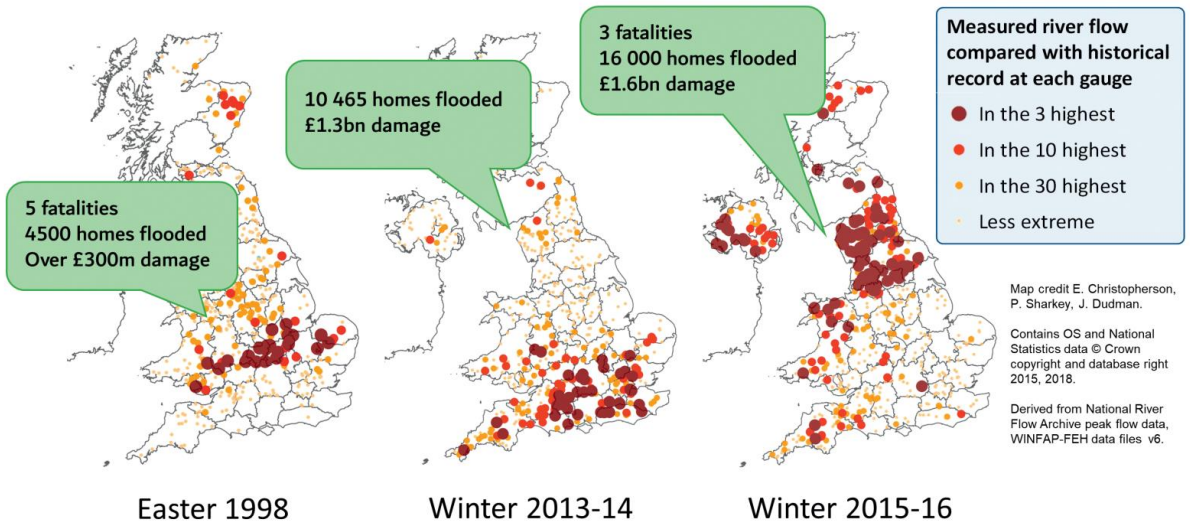


Figure 3 Shows the extremeness of river flows (relative to historic records) across river gauges in the UK during three widespread flood events, where the larger and darker point represent a more extreme flow. Figure taken and adapted from (Towe, Tawn and Lamb, 2018).

A growing field of research aims to understand correlated risks of extreme river flows. One approach for quantifying spatial dependence between flood events involves the application of statistical frameworks on gauged discharge data. For example, copulas (Timonina *et al.*, 2015; Serinaldi and Kilsby, 2017; Brunner *et al.*, 2020) and multivariate extreme frequency analysis (Heffernan and Tawn, 2004) are the two commonplace statistical methods. Bayesian networks can also be used to constrain the dependency structure between extreme river flows across vast spatial scales (Paprotny and Morales-Nápoles, 2017). However, many of these studies remain restricted to a hazard perspective and have not yet been used to quantify associated risks posed to infrastructure systems. Studies that have been applied to developing improved risk metrics for infrastructure systems include Speight et al., (2017), who combine a previously developed conditional dependence model (Heffernan and Tawn, 2004) with local detail on inundation and associated damages. This work identifies clusters of caravan sites which are both susceptible to flooding and vulnerable to defence failure. Lamb et al. (2019) develop a new fragility assessment of Britain's railway network in relation to bridge failure caused by flooding-induced scour, where flood events are simulated from a spatial joint probability model based on observed extreme river flows. This is used to quantify the probability of single or multiple bridge failures and second-order impacts on passenger journey disruptions and associated economic costs. Jongman et al. (2014) compute flood-induced asset losses aggregated to a river basin-level over time, using a copula model to determine the likelihood of simultaneous,

widespread losses to discuss the implications of widespread flood losses for the insurance sector.

Another limitation of return period maps is that they also fail to capture joint-dependency relationships between hazards. Many hazards are driven by interrelated, underlying drivers (Leonard *et al.*, 2014). For example, cyclones drive both extreme wind and extreme precipitation, which in turn triggers flood inundation and landslides. Where hazard impacts depend on multiple statistically dependent variables or events, it is known as a compound event (Leonard *et al.*, 2014). Using return period maps to describe risk posed by hazards that are co-dependent, e.g., extreme wind and flood inundation during cyclones, independently, could result in double-counting the damages to the same asset caused by two different hazards during the same event. Capturing the joint probability of inter-dependent hazards is necessary for quantifying compound risk accurately (Zscheischler *et al.*, 2018). This has been explored for a wide range of hazard combinations, for example, compound coastal and fluvial flood (Moftakhari *et al.*, 2017; Qiu *et al.*, 2022) and compound drought and extreme heat (Libonati *et al.*, 2022).

Quantifying the joint probability between hazard events across space and the cross-correlation between hazards using the statistical models outlined above, enables events using observed data allows the sampling of synthetic events for stress-testing applications (Herman *et al.*, 2016) (see section 2.2.2). Generating a large probabilistic hazard event set based on a wide range of potential climate / hydrological scenarios for stress-testing applications, enables the recognition of uncertainty in the system, particularly under conditions of non-stationarity under climate change (Milly et al., 2008). Note that a major difference in the characteristics of drought and flood/extreme wind events, is that droughts can manifest over longer timescales, and may only translate into water shortages through the cumulative effect of below normal flows on reservoir levels over many months. However, here, for simplicity, we refer to both droughts and flood/extreme wind events as hazard events.

2.1.4. Climate change impact on hazards

Anthropogenic climate change is the result of increasing levels of carbon dioxide (CO2) and other greenhouse gases (GHGs) (e.g., methane) in the atmosphere. The most easily observed and predictable outcome of rising GHG emissions, is its effect on driving increased global mean surface temperatures (Karl and Trenberth, 2003). As temperatures increase, the capacity of the atmosphere to store water increases (at a rate provided by the Clausius-

Clapeyron relationship), which is widely predicted to lead to more extreme precipitation events and higher pluvial and fluvial flood risk (Allan *et al.*, 2020). Driving temperatures also lead to increased evaporation rates from the land surface, driving droughts and hardening of the land surface, reducing the rate of soil infiltration (Várallyay, 2010), leading to higher rates of runoff and potential flooding. Although, the impacts of climate change on driving the risk of floods and droughts has been widely evidenced, including evidence of widespread, spatially extensive events (Milly *et al.*, 2002; Jongman *et al.*, 2014; Paltan *et al.*, 2018; Gaupp *et al.*, 2019), it is by no means a simple relationship. The relationship between climate change and hazard severity and frequency varies globally and the influence of climate change on hydrological events is associated with significant uncertainty (Trenberth, Fasullo and Shepherd, 2015; King, Reid and Saunders, 2023). Indeed, for cyclones, analysis on the impacts of climate change provide a mixed picture in terms of how cyclone intensity and frequency is influenced by climate change (Knutson *et al.*, 2020). The changing probabilities of compound extremes under climate change are increasingly explored in the literature (Zscheischler *et al.*, 2018). Again, the relationship between climate change and compound risk varies in space and across hazard types: for example, there is strong evidence of increasing risk of compound extreme heat and drought (Zscheischler and Seneviratne, 2017; Yin *et al.*, 2023); whereas for coastal and inland flooding, the picture is more nuanced (Wahl *et al.*, 2015; Ganguli *et al.*, 2020).

For risk analysis, climate change projections can be used to estimate shifts in likelihood of extreme events (Maraun, 2016). Global Climate Models (GCMs) provide insights into our changing climate and the water cycle (Manabe, Smagorinsky and Strickler, 1965; Nijssen *et al.*, 2001; Allen and Ingram, 2002). GCMs are modelled representations of the earth system, used to simulate how changes, such as greenhouse gas emissions (Manabe and Wetherald, 1967) or land use changes (Manabe and Wetherald, 1967; Zipper, Keune and Kollet, 2019), may impact the global meteorological and hydrological processes. Further projections are simulated under different scenarios of anthropogenic climate change and global development. Through an effort to coordinate and align global research on climate change risks within the scientific community, the Representative Concentration Pathways (RCPs) (van Vuuren et al., 2011) and Shared Socioeconomic Pathways (Riahi et al., 2017) have been developed. The RCP scenarios propose a set of four scenarios of radiative forcing: 2.6, 4.5, 7.0 and 8.5 W/m^2. The SSP scenarios outline five alternative scenarios of future global development trajectories, contingent on different levels of fossil-fuel dependency: SSP1, SSP2, SSP3, SSP4 and SSP5. These scenarios provide, to an extent, an envelope of optimistic and pessimistic outlooks of

anthropological climate change impacts to the global climate system. Thus, GCM projections under alternative RCP and SSP scenarios can be integrated with afrementioned drought, flood and extreme wind modelling approaches, to derive the re-distribution of probability associated with hazard events of a given severity. As GCMs are run at a global scale, their output may not accurately describe local trends and variations in absolute terms. Thus, GCMs can be regionally downscaled or bias corrected to provide more meaningful inputs for risk analysis (Lafon *et al.*, 2013). In essence, this involves finding appropriate scaling factors to convert outputs of GCMs to values that are more representative of/aligned with datasets that are considered to describe the context of interest more accurately. This can be done in a range of ways, e.g., based on a linear factor, i.e., by applying one scaling factor to the GCM output, or based on a non-linear (exponential) relationship (Lafon et al., 2013). Quantile scaling or mapping approaches are widely used and involves deriving the relationship between the same quantiles across the GCM output and more accurate datasets; these relationships are then applied to the GCM output as change factors (Hassanzadeh et al., 2019; Olschewski et al., 2023)

2.1.5. Summary

Multiple hazards impact water supply services. Droughts influence the availability of water supply utility sources. Floods and extreme winds may incur damage and disruption at critical assets. However, there is limited literature on the spatial interaction of hazard events, accounting for cross-correlation of different hazard events, on water supply utility networks. Hazard event sets are increasingly applied for both drought and flood/extreme wind analysis. Events sets enable a more accurate simulation of hazard impacts on interconnected infrastructure networks and, therefore, a more accurate way to aggregate risk across multiple locations at scale compared to traditional approaches that may rely on return period maps. They also enable the compound risk of inter-dependent hazards, such as cyclonic wind and flooding, to be accounted for, e.g., by preventing double-counting of damages and disruptions caused by different hazards during the same event. Increasing global temperatures under climate change are predicted to intensify the risk of floods and droughts, although the rate and direction of change is influenced by a range of complex processes, introducing substantial levels of uncertainty into projections. Climate change has also been shown to increase the intensity of tropical cyclones, however, the influence of climate change on extreme wind risk is less defined.

How these hazard inputs are overlayed with infrastructure networks and translated into risk estimates is outlined in the next section.

2.2. Climate risk to water supply utilities and their customers

The workflow for analysing climate risk to utilities can be outlined as follows: 1) identify the climate hazards to which the utility is exposed, (2) identifying utility vulnerabilities and future risk factors and (3) analyse and evaluate risks by combining the likelihood of the event with the modelled consequences. Although presented as a sequence, the stages can be undertaken in an integrated and iterative manner.

Different types of risk analysis have been performed for water supply utilities. Existing literature can be grouped based on the scale of analysis, the hazard considered and the type of impact quantified. This section provides an overview of a range of different approaches adopted for analysing climate risks to potential water supply systems. The section starts with an exploration of how water supply networks are modelled for risk analysis approaches, followed by a description of the risk transmissions pathways associated with different climate hazards, and how these are modelled in the literature, and ends with an overview of the impacts to different stakeholders that have been the focus of previous studies.

2.2.1. Infrastructure as a network to understand climate risk propagation

Apart from spatial dependencies between hazard events, discussed in the previous section, additional complexity arises in determining infrastructure risks by accounting for interconnections between assets and systems. Potable water supply networks are widely represented as a network of nodes and edges to either 1) model how single or multiple disruptions in one part of the network impact other parts of the network and dependent users (Buldyrev *et al.*, 2010) or 2) to simulate the flows of water from source to sink as a dynamic mass balance (Kuczera, 1992). Establishing a network representation of potable water infrastructure involves establishing nodes as the assets involved in water supply services, such as treatment, plants, pumping stations, storage tanker, reservoirs, river abstraction points and wells. Edges can be established to reflect connections/flow rates between assets, and connections/flow rates between assets and users. Characteristics of assets and nodes may include their volumetric capacity, operational rules, population served, asset type and leakage losses. Although the network structure is similar whether applied as a static graph model (Diao *et al.*, 2016; Thacker *et al.*, 2017; Pant *et al.*, 2018) or a dynamic mass balance simulation model (Borgomeo *et al.*, 2015; Mortazavi-Naeini *et al.*, 2019; Dobson *et al.*, 2020), the two applications are not typically integrated in the literature. The first category puts emphasis on analysing water network infrastructure and employs models that represent all the infrastructure

assets that make up the network. Pant et al. (2018)analyse a database of source nodes (abstraction points, water treatment plants, reservoirs) and sink nodes (demand centres) to build a detailed representation of the water distribution system in England and Wales, including links with the electricity nodes supplying water assets. The second category of literature puts emphasis on the dynamics of water resources that feed distribution systems and employ water network models to allocate water resources to the populations they serve (Watts *et al.*, 2012; Mens, Gilroy and Williams, 2015; Borgomeo, 2022). This enables scenarios of surface water and/or groundwater resource availability/quality to be compared to the demand, and hence exposure, of dependent populations.

A range of water system modelling approaches have been used in academic literature beyond those highlighted above. For example, agent-based modelling and integrated assessment models have been used for evaluating water allocation problems, common pool management policies and identifying nexus solutions across water, land and energy sectors (Madani and Dinar, 2012; Kahil *et al.*, 2018). Here, focus is placed on network models due to their widespread application for climate risk and adaptation needs analysis.

Due to the data requirements associated with asset level network modelling, risk analysis can be based on intersecting hazard layers with exposure and vulnerability layers that are aggregated over larger spatial scales (Carrão, Naumann and Barbosa, 2016). For example, for droughts, risk can be quantified at the catchment or basin scale (Gaupp, Hall and Dadson, 2015); for floods, risk can be quantified based on land use classifications (Winsemius *et al.*, 2013). An asset level network representation offers a number of advantages over such aggregated approaches. A major advantage of an asset level approach is that the vulnerability threshold for water service delivery can be based on the physical characteristics and associated operational thresholds of assets and systems (Tsakiris *et al.*, 2013). This allows modellers to analyse the likelihood that hazards, such as drought and floods/extreme winds, translate into impacts, such as water shortages and asset damages, respectively. Infrastructure vulnerability can be defined as the degree to which a system or asset is susceptible to climate hazards of a given severity (IPCC, 2001). Water supply infrastructure can be highly complex and spatially variable, even within the same catchment. Drought vulnerability can be strongly influenced by leakage rates, storage capacities and transfers (Borgomeo *et al.*, 2018). Flood and/or extreme wind vulnerability is influenced by the infrastructure asset type, protection level/condition and the level of network redundancy (Hall *et al.*, 2003; Koks *et al.*, 2019)

Further, risk exposure is often based on the population or economic activity that is dependent on a given asset or system. A network representation enables the propagation of a disrupted node or edge to the affected population or economic activity to be simulated. Thus, network representations of critical infrastructure systems enable 'systemic'[1] impacts of asset-level disruptions to captured and analysed. This also enables the quantification of benefits associated with flood defences or other structural risk mitigations interventions.

Figure 4 illustrates the importance of establishing vulnerability thresholds based on the physical characteristics of water supply networks for quantified risk analysis. The risk that droughts propagate through the system to result in water shortages in a given city, can be derived by balancing water availability at the system's sources, against a vulnerability threshold. In this case, the vulnerability thresholds can be defined as the variable demands of the city accounting for leakage losses and the supply provided by the well (Figure 4). Note that the effect this water shortage has the city population depends on the context: for example, in higher income countries, household demand is higher as water is used for non-essential uses, such as lawn irrigation (Verschuur *et al.*, 2023). In these contexts, water shortages are more likely to manifest as demand restrictions, such as hosepipe bans (Thames Water, 2013). In lower income contexts, water shortages are more likely to manifest as intermittent supply, as utilities aim to serve different parts of their networks at different times of day (Totsuka, Trifunovic and Vairavamoorthy, 2004). The risk that floods and/or extreme winds result in asset damages at a given site, can be derived through the use of vulnerability thresholds describe by damage curves (Huizinga, de Moel and Szewczyk, 2017; Miyamoto, 2019; FEMA, 2020). These relate the intensity of a hazard at a given asset to the fraction of damage. For example, in the case of a water treatment plant being impacted by a flood, as illustrated in Figure 4, the relevant vulnerability thresholds would be a flood depth-damage function describing the damage response of water treatment plants to flooding. The damage response, measured a fraction of complete physical destruction of the asset, can be used to derive 1) the financial costs of reconstruction, 2) the duration of reconstruction and 3) the level of service continuity (i.e., as a fraction of service during non-disrupted conditions) (FEMA, 2020). Note that whether a given level of asset damage would result in disruptions in the city population would depend on whether the network topography enables substitution from unaffected assets.

[1] "the potential knock-on effects to initially unaffected components of a system because of system interactions or dependencies (UNDRR, 2022)".

In the case illustrated in Figure 4, case, well supply could still function, enabling partial continuity of services.

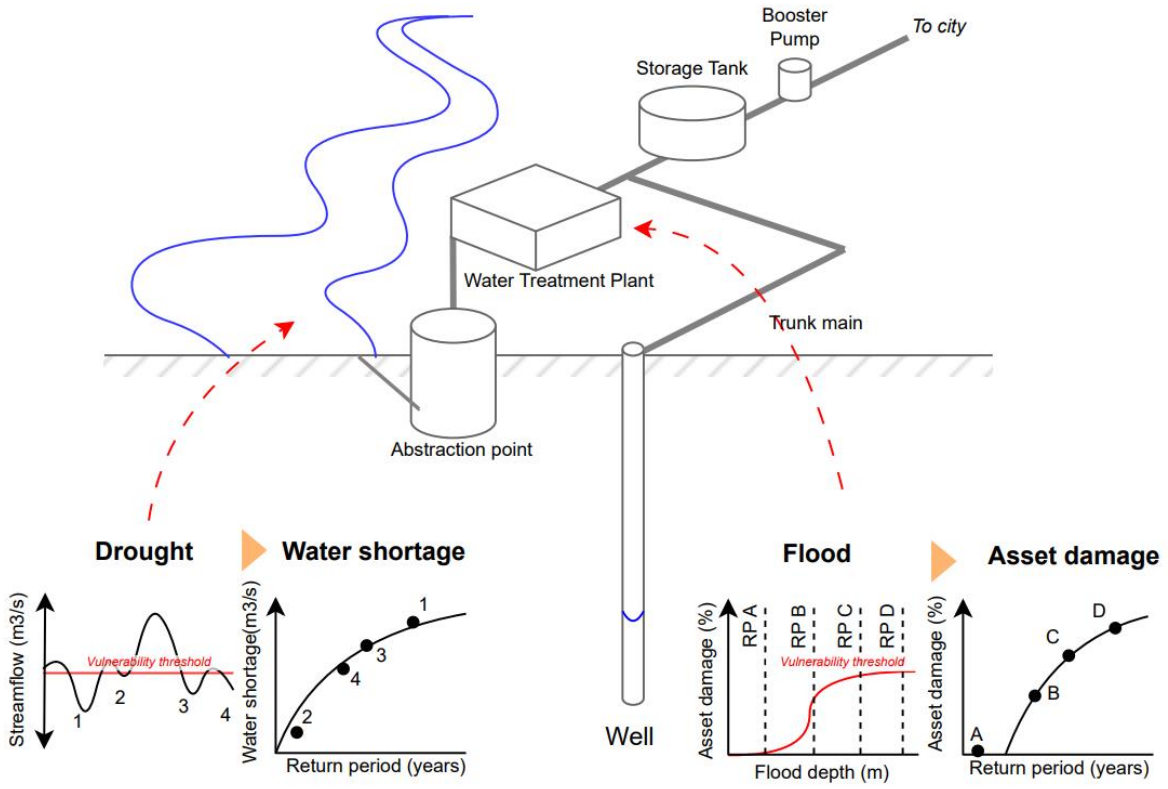


Figure 4 Conceptual representation of how floods and drought hazards interact with system-based vulnerability characteristics to result in asset damage and water shortage risk. Water supply system schematic was redrawn based on (Summaryplanet, 2023).

2.2.1.1. Quantifying annual expected risk

Annual expected risk represents the loss that would be incurred by a given hazard(s) per year, averaged over a given time horizon. Equation 3 below provides how risk can be formulated for a given hazard event. Where I_e is the physical damage cost estimate for an asset or system (j) for a given hazard event (e). F_j denotes the vulnerability function, e.g., damage curve ratio (Miyamoto, 2019; FEMA, 2020), which reflects the characteristics of the asset or system based on which the impact (in terms of fraction of maximum damage/disruption) for a given hazard intensity - e.g., flood depth (h_e) - can be estimated for flood hazard event (e). R_j denotes the impact of a maximum disruption, either in terms of population affected or reconstruction costs of that particular asset or system j.

$$I_{ej} = F_j(h_e)\, R_j \qquad (3)$$

In Equation 4, the annual expected risk across several events is estimated by numerically integrating the inverse cumulative probability function of the direct damage costs $Q_{\bar{I}}(p)$. $Q_{\bar{I}}(p)$ is obtained by ranking the damages for a range of events (conditioned under a given climate scenario) to estimate annual exceedance probabilities of damages (p) (see Equation 2).

$$\bar{I}_j = \int_0^1 Q_{\bar{I}}(p)dp \qquad (4)$$

2.2.2. Potable water supply climate-related disruption pathways

Water supply systems are subject to a range of hazards which can damage infrastructure assets and reduce the availability of the resources (water) which is being delivered by the supply network. A major difference in the characteristics of each risk is that droughts can manifest over longer timescales compared to floods and cyclone and may only translate into water shortages through the cumulative effect of below normal flows on reservoir levels over many months. Thus, multi-hazard risk assessments require direct comparison of damaging acute extreme events (e.g., floods, cyclones) and analysis of drought risks to water supplies, which can be chronic. The former of these processes involves the overlay of a given hazard (flood, cyclone) with a representation of the vulnerability of the water asset network to yield a probability distribution of asset failures and consequential disruptions of supply to water users (Laucelli and Giustolisi, 2015). Note that this is similar to approaches that would be adopted for other types of infrastructure networks. Analysis of drought risk involves hydrological modelling and mass balance calculations (supply vs. demand) in order to estimate the frequency and duration of water shortages for users (Borgomeo *et al.*, 2014).

2.2.2.1. Water shortage

As mentioned in the 'Drought hazard modelling' section, comparing the availability of supply, now and/or in the future, at water system source locations against operational failure thresholds (e.g., customer demand) provides a more direct means of analysing the risk that droughts translate into utility *water shortages*. Stress-testing approaches involve analysing the potential impacts of a given hazard scenario on a system (Hall and Leng, 2019). Compiling a wide set of observed and/or synthetically generated meteorological or hydrological drought events, characterised by their likelihood and severity (Guillod *et al.*, 2018), enables a range of uncertainty to be captured and the inclusion of very extreme (or worst-case) events in the stress-testing exercise. The response of the water supply system(s) to each event is modelled to reveal the likelihood of key vulnerability thresholds being breached, providing meaningful insights

for decision makers and utility operators (Borgomeo *et al.*, 2014; Culley *et al.*, 2016). This process can be repeated by projecting changes in event likelihood under future climate change conditions (Hallegatte *et al.*, 2012; Poff *et al.*, 2016). Common metrics used to describe water supply system responses include reliability and Levels of Service (LoS) (Thames Water, 2013). Reliability is typically used when describing the impacts of droughts on reservoir levels and is calculated as the number of months in which a system can meet a particular threshold, often set through communication with relevant stakeholders, divided by the total number of months in the simulation period (Brown *et al.*, 2012; Taner, Ray and Brown, 2019). In the UK, a relevant risk metric is LoS, which describe a range of four severities of water use restrictions, triggered by reservoir levels falling to different fractions of full storage capacity (Thames Water, 2013). Such service reliability-related metrics can be taken further to understand impacts to users, as discussed in section 2.2.3.

2.2.2.2. Asset damage

As mentioned in section 3.1.2, many studies that analyse the climate-related risk physical asset damages and disruptions adopt the approach of overlaying return period maps with spatial network information on asset networks, including asset types, interconnections, and their service areas (Pant *et al.*, 2018). Asset types have different levels of vulnerability, described by damage curves that relate the intensity of a given hazard (e.g., flood depth or wind speed) to the extent of damage that would be incurred by the asset (Huizinga, de Moel and Szewczyk, 2017; Miyamoto, 2019; FEMA, 2020). Interconnections describe level of network redundancy, for example, if one asset is out of service, another asset may be able to substitute, resulting in no disruptions to users. In the absence of known outlines of the users that are service by a given asset, service areas are typically estimated based on network concepts, such as Voronoi diagrams (Thacker *et al.*, 2017). By integrating these three categories of information, studies explore a range of consequences of asset damages: including direct financial damages to asset stock, disruptions to people, and wider economic impacts (Pant *et al.*, 2018; Koks *et al.*, 2019; Koks *et al.*, 2019). These impact channels are discussed further in section 3.2.3. Note that, as the risk analysis approaches are directly comparable across hazards that impose physical damages to assets (e.g., flood inundation and extreme wind), here those hazards are grouped collectively.

2.2.2.3. Spatial aggregation of risk across utility networks

By analysing the distribution of event probability and associated impacts, through a loss-probability distribution, one can integrate across this distribution to find the average loss that would be expected per year, or the annual expected loss. This annual expected loss estimate provides a meaningful metric for decision makers, and a useful starting point for evaluating future changes in risk under scenarios of climate change, economic and population growth and/or infrastructure resilience improvements.

Although the majority of past literature focusses on a single system for their analysis (Culley *et al.*, 2016), many water supply utilities operate more than one source across more than one water supply system. Here, water supply system is defined as a network of sources, storage, treatment plants, pumping stations, pipelines and other assets that all operate collectively to service the same population. As discussed in previous sections 3.1.3 and 3.2.2.3, hazard impacts can impact multiple locations simultaneously, so assessing infrastructure vulnerability requires an understanding of how assets operate as a network. Given this spatial interrelationship between extreme weather events and between infrastructure networks (e.g., the potential for substitution), understanding the potential risk of multiple asset failures is essential to accurately aggregate risk across critical infrastructure systems (Koks *et al.*, 2019), which conventional risk frameworks fail to capture. For example, assuming flood or wind-related impacts are independent, modellers are more likely to find that disrupted assets can be substituted by non-disrupted assets. On the other hand, assuming they are dependent, may lead to an over-estimation of risk, as the more assets that are disrupted simultaneously, the less easily their services can be substituted. Either of these assumptions limit our understanding of how climate hazards interact with water supply systems.

Thus, event-based, stress-testing approaches are particularly useful for analysing the spatial footprint of events across systems operated by one utility for multiple reasons. Firstly, utilities may wish to understand the risk of customers across multiple systems being affected by a single climatic event, so that they can design contingency plans (e.g., tanker truck provisions to affected communities) accordingly. Secondly, the viability of transfers between systems could be undermined if the water availability at both system sources is critically reduced by the same event (Zhang *et al.*, 2018; Dobson *et al.*, 2020).

Here an overview of how climate-induced disruptions are burdened by different stakeholders is provided.

2.2.3. Impacts to utilities and customers

The nature of impacts analysed for water shortages versus asset damages differ substantially. Asset damages result in direct reconstruction costs, which are burdened by utilities. As many utilities are state-owned enterprises, and heavily subsidised, this often directly impacts fiscal balance sheets. Further, asset damages can take assets offline partially or completely for days, weeks and months, depending on the severity of the damage. As droughts do not affect the physical stock of assets, they do not incur reconstruction costs for the utility or the government (Freire-González, Decker and Hall, 2017). Further, as they are typically more chronic in nature, their impacts may be felt in different ways, for example, utilities may introduce demand restrictions or intermittent supply windows, rather than complete shutting off water supply for specific communities (Aboelnga, 2019).

Both water shortage- and asset damage-related disruptions can leave customers, including households and businesses without crucial water supply. Utilities are also financially impacted by disruptions. For example, utilities may incur revenue shocks, as volumetric sales of water decrease (Armstrong *et al.*, 2022). They may also be subject to fines, as in the UK (Borgomeo *et al.*, 2018). Further, supply disruptions and intermittency are also associated with increased production costs, as intermittent system operations become more complex, and pressure variations in water supply distribution lead to a deterioration of asset performance, efficiency and resilience (Totsuka, Trifunovic and Vairavamoorthy, 2004). Customers may also incur economic and/or financial losses during disruptions through a range of channels: including through health impacts, such as diarrhoeal diseases (Ercumen *et al.*, 2015); welfare losses (Roibas, Garcia-Valiñas and Fernandez-Llera, 2019); and service replacement costs (Pattanayak *et al.*, 2005).

A wide range of literature has examined household impacts of water supply disruptions (Majuru, Suhrcke and Hunter, 2016). A substantial component of household coping costs during water supply disruptions is the costs of replacing disrupted supply. Alternative supplies include untreated sources (e.g., well/springs), tanker trucks, and bottled water (Ochungo *et al.*, 2019). Households may also invest in household storage tanks (Wunderlich *et al.*, 2021) or rainwater harvesting (Otaki *et al.*, 2022). Replacement costs for customers range widely across geographies. In high income countries, such as the UK, utilities are subject to fines and are required substitute disrupted services. For example, when the water treatment plant serving Monmouth, Wales, was flooded by Strom Dennis, the Welsh utility Dwr Cymru supplied

bottled water to all households affected (Sky News, 2020). As such, the financial burden of disruptions, is effectively absorbed by the utility. In other cases, particularly in lower income countries, utilities do not have the capacity to absorb the full costs of disruptions, and replacement costs spill over to dependent customers. Tanker trucks are a common form of service delivery during disruptions in low- and middle-income country contexts (Eakin *et al.*, 2016; Zozmann *et al.*, 2022). Relative to income, poorer households will have to pay more for the same replacement services from tanker trucks, incurring disproportionate impact on the most vulnerable. One study undertaken in Kathmandu, Nepal, analysed the costs of substituting disrupted water supply with alternative sources (including tanker trucks and bottled water) relative to household income, to found that this cost represented 56% and 34% of overall coping costs for poor and risk households, respectively (Pattanayak *et al.*, 2005). This highlights the importance of resilient water supply utilities for maintaining affordable services for protecting customers, particularly lower-income customers, from climate shocks.

Economic activities also depend on water supply services, with disruptions triggering cascading impacts on production and associated macro-economic flows. Much work has been done on the impacts of hydrological variability on economic growth (Brown *et al.*, 2011; Hall *et al.*, 2014) and the impacts critical infrastructure damage and disruption on reduced economic productivity at the business level and associated macro-economic consequences on the wider economy (Koks *et al.*, 2019). Regarding the impacts of utility water supply on business outputs, Sjöstrand et al., (2021) distributed an online survey to 1405 companies in Sweden on consequences of complete and unplanned water supply outages to find that 4.5% companies are affected at a cost of 300,000 SEK [30,000 USD] (in terms of lost value added). Jenkins et al., (2021) conduct drought risk analysis on 7 economic sectors in the UK, using a combined hydrological and water resource system model. Elasticity factors are used to convert variations in water supply into changes in losses in Global Value Added across sectors, with the annual expected loss estimated at 11.7 million GBP (15.3 million USD). A static input-output model is used to translate this into macroeconomic economic losses which is estimated to be far greater, at 30.2 million GBP [40 million USD].

The interplay between development and water supply services has been explored widely (Eakin *et al.*, 2016; Deshpande *et al.*, 2020). Although utilities often introduce pro-poor tariff structures and other measures to improve access to the most vulnerable, the challenging financial position of many utilities means that service delivery is worse in poorer communities

that are less able to pay tariffs (Cross and Morel, 2005). These distributional impacts of climate-related events across income groups have been explored widely in the literature (Winsemius *et al.*, 2018; Barbour *et al.*, 2022), including the disproportional impacts of water supply disruptions on poor households (Cook, Kimuyu and Whittington, 2016). However, rarely for climate-related utility water supply disruptions. This is because the literature on quantified climate risk analysis on potable water supply disruptions is skewed towards higher-income contexts, with a particular concentration of literature based on the UK water utility context (Borgomeo *et al.*, 2015; Dobson *et al.*, 2020; Jenkins *et al.*, 2021; Murgatroyd *et al.*, 2022).

2.2.4. Summary

Climate-related water supply service disruptions have multi-scale impacts across actors. Thus, the scale at which one adopts for risk analysis defines how meaningful and relevant the results will be to stakeholders. The different modelling approaches tailored to, and impact channels associated with, asset damage- and water shortage-related utility disruption has meant that the two modes of disruption are rarely considered together. For floods and extreme winds, return periods maps are typically overlaid with spatial information on asset-level vulnerability and network effects, to translate flood and extreme wind hazard severity into information on the risk of asset damage and disruption. For droughts, analysing the risk (frequency and severity) of water shortages for users involves meteorological and/or hydrological modelling and simulating the balance between supply and demand. The impacts of water supply utility disruptions on utilities, governments, households and the wider economy vary widely between and within countries. Particularly in low and middle-income countries, private tanker truck vendors can be a major form of alternative service delivery during climate-induced water supply disruptions. The financial, economic and welfare burden of substituting safe water supply, e.g., through purchase from tanker trucks, during utility disruptions is often disproportionately felt by lower income customers.

2.3. Adaptation strategies for water utilities

Climate adaptation represents an adjustment in natural or human systems aimed at reducing the severity of, or risks associated with, adverse climate change impacts, as well as actions seeking to harness any beneficial opportunities that may arise under a changed future climate (IPCC, 2007). There is currently a substantial gap between climate adaptation investment needs and the actual flows of investment (CPI, 2020). International investors, such as national and multinational development bank and donors, are a key source of finance for climate adaptation projects (ibid). Such institutions are subject to their own mandates and constraints and are under increasing levels of scrutiny in scaling-up finance for adaptation (Klein, 2010). This adaptation finance must be demonstrated as 'additional' to financing for broader development; however, it is essential that adaptation investments are wholly attuned to the needs of the recipient country and context (Conway and Mustelin, 2014). Indeed, a critical aspect of climate adaptation is that strategies must be aligned with national development priorities in order to be effective and sustainable (Hulme, O'Neill and Dessai, 2011; Conway and Mustelin, 2014). For many countries, climate adaptation may not be a priority for overall development (Christoplos, 2010). Frameworks that capture the contribution of investments towards both climate adaptation and other national development priorities can be useful for trading-off, or reconciling, the objectives of multiple stakeholders, including utilities, households, governments and investors (Ziervogel, Shale and Du, 2010).

Water utilities deliver a range of climate adaptation benefits to society by delivering water supply, wastewater and, in many cases, urban drainage services. Widely held perceptions of water as being a basic human right (Gleick, 1998) and the strong influence of governments on utility operations, including for autonomous, private utilities, leads to strong linkages between water sector development and national development. They, therefore, provide a key opportunity for scaling up climate adaptation investments that are sensitive and responsive to national development priorities. However, the water sector, in many ways, typifies the difficulty investors face in scaling up investments in adaptation projects in developing countries. Potable water supply is a basin human need, driving down tariff rates relative to the costs of services to ensure affordability (Andres *et al.*, 2021), posing a challenge around attracting investments in resilient infrastructure developments (OECD, 2011). This increases the vulnerability of networks to (e.g., climate-related) shocks, leading to increased rates of disruption, lower customer satisfaction and, hence, lower willingness to pay high tariff rates. Despite, the vastly diverse institutional, environmental, and social contexts in which utilities

operate, this trend, widely referred to as the vicious cycle of water utilities, speaks to the vast majority of cases globally. This has contributed to inability of utilities to support the expansion of services and, hence, the large, and widening, service gap in many countries (Hutton and Varughese, 2016). The vicious cycle is illustrated in Figure 5, where a high rate of network disruptions leads to decreased customer satisfaction, in turn leading to lower tariff income for utilities. Insufficient tariff revenue leaves utilities with lower creditworthiness, making it more difficult for them to invest in the infrastructure needed to reduce network disruptions. Figure 5 highlights the role of climate shocks in increasing the rate of network disruptions and, thus, how climate adaptation activities can help to negate or reverse this vicious cycle.

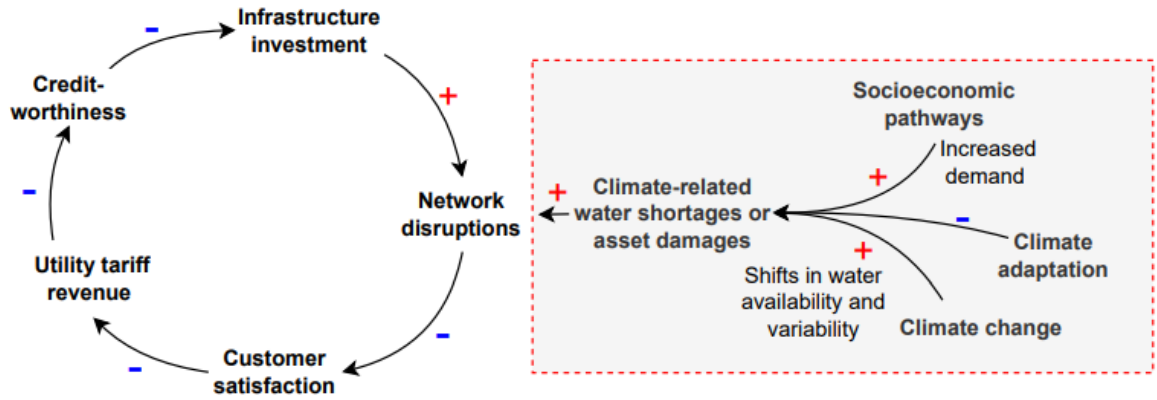


Figure 5 Interaction between climate-related disruptions, climate adaptation and the water utility vicious cycle.

Here, the method by which portfolios of options are evaluated in the literature, based on the ratio between their risk reduction potential and cost, is described. This is followed by an overview of the range of adaptation options that water utilities may consider in order to adapt to multiple climate hazards, now and in the future, are explored. The section is concluded with an overview of how consideration of the disproportionate impacts of water supply disruptions (as discussed in section 2.2.3) on more vulnerable population can be integrated into decision-making frameworks for future climate adaptation planning.

2.3.1. Adaptation optioneering approaches

Economic analysis to prioritise a portfolio of options involves comparing the costs of interventions against their expected risk reduction benefits. The expected risk reduction benefit is calculated as the difference between i) drought risk without the intervention and ii) the residual risk thanks to the intervention. Thus, the risk quantification process covered in the

previous section, can be used to explore the risk reduction effects of a range of adaptation options.

Figure 6 shows a graph that compares the costs and benefits of 30 potential investment drought adaptation plans for Thames Water utility in London, UK (Borgomeo *et al.*, 2018). Benefits of each option are derived by first, calculating the risk of droughts to the utility under present day and future conditions, without the integration of any adaptation measures. This is quantified based on hydrological modelling of water availability at the utility's reservoirs. Where water availability falls below certain thresholds, defined across different months of the year, different levels of restriction, or LoS (see section 3.2.2.1), are implemented across the utility's customer-base. The annual expected risk of each type of water use restriction is estimated by stress-testing the system with a wide set of synthetic drought events conditioned under an emissions scenario of RCP8.5 over a 30-year time horizon (the typical design life of water infrastructure assets). The most severe LoS thresholds are associated with an economic loss for customers that has been estimated by the utility to convey the risk of restriction events in cost terms (Thames Water, 2013). This process is then repeated, simulating the integration of a single or portfolio of adaptation options: ranging from reservoir capacity expansion, new sources, transfers, desalination and potable wastewater reuse options. By comparing the annual expected risk, in cost terms, before and after each adaptation option is integrated, one can derive their risk reduction benefit. This annual expected risk reduction benefit can be compared to each option's capital and operational costs, where both benefits and costs are discounted to Present Value and accrued over the time horizon of each option's design life. The graph in Figure 6 shows the residual risk after each option on the y-axis, measured in terms of the economic cost of restrictions, and the cost of each option on the x-axis. The dotted line in Figure 6, illustrates the annual reconstruction cost that Thames Water deems 'tolerable', and the red point thus identifies the option that would provide this residual level of risk at the least cost. This approach is useful for decision makers, e.g., in government or development banks, that aim to prioritise investments and quantify investment impact on reducing risk.

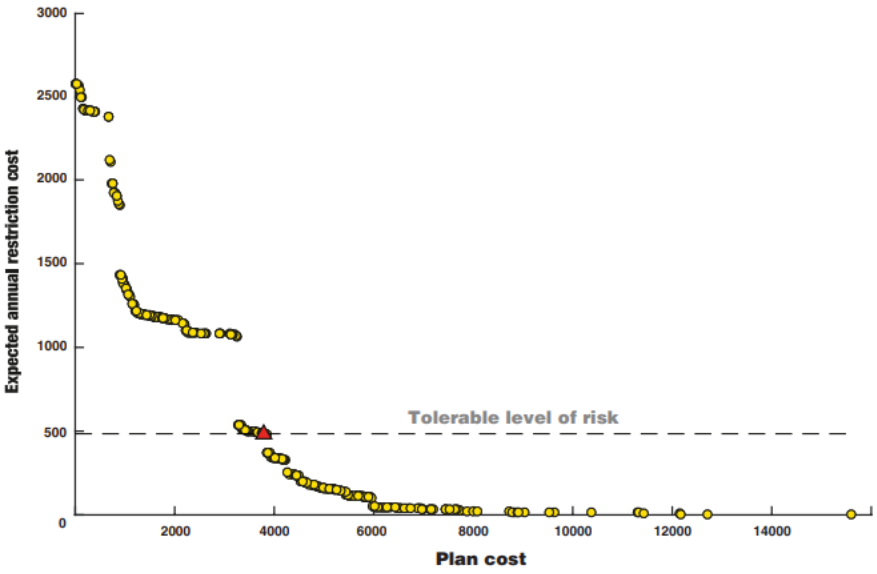


Figure 6 Graphical representation of the investment costs, in million British pounds, (x-axis) of a portfolio of 30 adaptation plans against the residual drought risk, in million British pounds, that would remain after each option's implementation (y-axis). Figure taken from (Borgomeo et al., 2018, p. 480).

Note that this tolerable residual risk concept differs from the evaluation of options with respect to BCRs. The tolerable residual risk threshold depends on the risk appetite of relevant stakeholders, whereas the BCR is a more general threshold describing the economic viability of a project. For decision-makers interested in the BCR of options, a graph of a plan's cost and benefit (i.e., the risk before plan implementation minus the residual risk after plan implementation) could be drawn. Options can then be prioritised based on cost-effectiveness.

Also note that although the example of adaptation to water shortages is highlighted above, a similar approach is adopted for other hazards, such as flooding and/or extreme wind (Panteli *et al.*, 2017; Thacker *et al.*, 2018). The annual expected risk of flood/extreme wind can be compared before and after a portfolio of relevant adaptation options (see section 2.3.1.4 for a range of potential options to consider for asset protection) to estimate their risk reduction benefit. Again, each option's capital and operational costs can be compared to its annual expected risk reduction benefits, discounted and accrued over its design life (Molinari *et al.*, 2021).

2.3.1.1. Quantifying Benefit-Cost Ratios

Considering adaptation options designed (d) to protect water supply systems from risk posed by events associated with a range of return periods $RP = \{20,50,100,200,500\}$ (in years) or design annual exceedance probabilities $P = \{1/20,1/50,1/100,1/200,1/500\}$. Assuming

that the discount rate of $r\%$ is applied over the adaptation planning horizon, the discounted cost ($DC_s(p_d)$) for applying the adaptation option of a given design annual exceedance probability ($p_d \in P$) to a water asset or system j is estimated at $t = 0$ and the sum of the subsequent asset level discounted annual maintenance costs $CM_{jt}(p_d)$ over the planning horizon. This is formulated in Equation 5.

$$DC_j(p_d) = \sum_{\forall j \in s} CI_{j0}(p_d) + \sum_{\forall j \in s} \sum_{t=0}^{t=T} \frac{CM_{jt}(p_d)}{(1+0.01r)^t} \qquad (5)$$

Similar to the costs, the risks before and after each adaptation option are estimated at annual timescales over the planning horizon, as outlined in Equation 6. Future risk (and hence risk reduction potential of adaptation options) are driven by climate change driven hazard events. The discounted benefit ($DB_s(p_d)$) of risk reduction due to the adaptation option is estimated by discounting the annual timeseries of marginal risk reduction ($\overline{D}_j - \overline{D}_t(p_d)$) over the planning horizon.

$$DB_j(p_d) = \sum_{t=0}^{t=T} \frac{\overline{D}_{jt} - \overline{D}_{jt}(p_d)}{(1+0.01r)^t} \qquad (6)$$

The effectiveness of the option is, thus, assessed in terms of the benefit-cost ratio ($BCR_s(p_d)$), given by Equation 7.

$$BCR_j(p_d) = \frac{DB_s(p_d)}{DC_s(p_d)} \qquad (7)$$

2.3.1.2. What scale for potable water risk analysis?

Despite the advanced research body on drought risk-based decision-making for water supply utilities, this has largely been undertaken for individual utilities (Herman *et al.*, 2015; Borgomeo *et al.*, 2016; Kingsborough, Borgomeo and Hall, 2016; Li *et al.*, 2019). On the other hand, analyses of global drought risk to utility water supply are often conducted at the catchment scale (McDonald *et al.*, 2011; Hanasaki *et al.*, 2013; Padowski and Gorelick, 2014; He *et al.*, 2021), rather than the utility scale. These catchment-scale studies do not capture utility-level characteristics that mediate the propagation of meteorological/hydrological droughts to utility water shortages, such as leakage losses and storage capacities. Further, utilities are not necessarily constrained to the water supply available from a single catchment. They may receive water from inter-basin transfers (McDonald *et al.*, 2014; Zhang *et al.*, 2018). They may also be supplied by non-conventional water sources, such as potable wastewater reuse and desalination (van Vliet *et al.*, 2021). Indeed, the total installed desalination plant

capacity increased by 60% between 2010 and 2020, and is expected to reach 120 million m^3/day by 2025 (GWI, 2022a), becoming an increasingly important adaptation option. Desalination is still predominately considered in highly drought and water stress-prone areas such as the Middle East and North Africa, but also even in traditionally water abundant contexts, such as London, UK. Further, previous global drought risk studies typically capture a snapshot of water stress and then overlay either solely quantify hydrological variability (e.g., through the SPI or SPEI) (Blauhut *et al.*, 2016; Peng *et al.*, 2020) or stress (He *et al.*, 2021). As such, they adopt the index-based approaches discussed in section 2.1.1, limiting the extent to which they speak to, or predict, the risk utility water shortages. As discussed in section 2.3.1.3, meaningful drought risk analysis ideally involves establishing a mass-balance simulation between the water supply available to utilities relative to a meaningful volumetric threshold, that is meaningful for the prediction of utility water shortages. Another limitation associated with previous studies on drought risk at the global scale is that they typically assume that water is allocated 'optimally' between users during droughts, with utility water supply classified as the most essential use in such frameworks (Padowski and Gorelick, 2014). However, in reality, this 'optimal' allocation of water resources between sectors requires sophisticated operational and physical infrastructure, which is only very rarely available. As such, this reallocation during drought does not always play out in reality. For example, during a drought in 2022, household taps ran dry in Monterrey, leaving people without sufficient water to bathe, wash dishes and flush the toilet and wholly dependent on tanker trucks (Guardian, 2022). Muller (2017, 2018) argues that some of the worst consequences of the infamous Cape Town Day Zero event could have been avoided through improved foresight from decision makers, including pre-emptive demand restrictions on agricultural water use to reallocate greater flows to public supply reservoirs.

The lack of utility-scale risk analysis limits the rigour with which previous studies have been able to identify adaptation benefits, i.e., in terms of risk reduction potential (Wada, Gleeson and Esnault, 2014; Larsen *et al.*, 2016; Buurman, Mens and Dahm, 2017; Greve *et al.*, 2018; Krueger, Rao and Borchardt, 2019; He *et al.*, 2021). Many optioneering studies are not preceded by risk analysis, but rather use qualitative assessments to explore the applicability/relevance of alternative adaptation options (Haddeland *et al.*, 2014; Wada, Gleeson and Esnault, 2014). For example, Wada et al., (2014) analyse the feasibility of six adaptation solutions - improving agricultural productivity, improving irrigation efficiency, reducing industry/domestic water use, reducing population growth, developing greater

reservoir storage, and increasing desalination capacity - in water stressed basins to reduce the population living in water stress by 2050. However, they assess the population exposure to water stress (based on river basin-aggregated water withdrawals: water availability ratios) without considering the unique (infrastructure) characteristics of the utilities that deliver water supply services. Further, adaptation options are explored in terms of their volumetric contribution to the water deficit, without considering their effectiveness at reducing society's vulnerability to hydrological variability and droughts. He et al., (2021) assess the level of water stress in 526 large cities (with populations >1 million) based on the spatial overlay of water scarcity layers, projected under a range of climate and growth scenarios, and spatial information on population incidence. The eligibility of a range of potential adaptation options, including desalination, groundwater exploitation, reservoir construction, inter-basin transfers, domestic virtual water trade and international water trade. Eligibility is determined based on the characteristics of each city, e.g., proximity to the coast (for desalination eligibility) and the rate of aquifer depletion (for groundwater exploitation eligibility). However, the eligibility of a given option only provides partial insight into how options should be prioritised: understanding the effect of options in reducing drought and water shortage risk is an essential basis for rigorous optioneering analysis.

Combining global hydrological modelling approaches with local utility information to quantify risk at the utility-scale, would enable the identification of the risk reduction potential of alternative adaptation actions. The following sections explore the range of adaptation strategies that are typically considered to mitigate the risks of water shortage- and asset damage-related water supply utility disruptions.

2.3.1.3. Water shortage adaptation strategies

Utilities can implement a range of strategies to minimise drought risk. These can be divided between demand- and supply-side measures. Demand-side measures focus on reducing water needs, e.g., through promoting water efficiency measures and behaviours. These can be incentivised through campaigns or implementing tariff structures that disincentivise water consumption, such as increasing block tariffs (Pinto, de Carvalho and Marques, 2021).Water use efficiency can also be promoted through raising awareness around water consumption, e.g., through awareness raising campaigns, and through implementing water saving devices/strategies in buildings (Stavenhagen, Buurman and Tortajada, 2018). They can also include reducing leakage losses, a major problem for utilities globally, particularly where water

infrastructure is relatively old and where utilities have not invested sufficient amounts in network maintenance and rehabilitation (Rouse and El Achi, 2019).

Supply-side measures focus on increasing the amount of available water, e.g., through source diversification, transfers, unconventional water sources (e.g., desalination and potable reuse) and storage capacity increase. Although necessary and widely implemented in many contexts to adapt to dwindling water resources and rising demand, they are often associated with negative outcomes for the environment and society. Storage infrastructure, such as reservoirs and dams, are essential for storing excess water during relatively wet periods, reducing flood risk (Boulange *et al.*, 2021), and enabling supply continuity during dry periods. Such infrastructure allows countries to harness their endowment of water resources and support economic growth (Grey and Sadoff, 2007; Hall et al., 2014). However, large infrastructure, such as reservoirs, often require land acquisition and displacement of prior land use (Gellert and Lynch, 2003). They also can have major impact to the environment: e.g., by reducing environmental flows and sediment flux downstream (Bussi et al., 2021). Other, e.g., nature-based, forms of storage such as conserved/restored/constructed wetlands, managed aquifer recharge and rainwater harvesting can provide less environmentally and socially harmful alternatives to reservoirs and dams (Dillon et al., 2019; World Bank, 2023). Indeed, in many contexts, in both developed and developing contexts, nature-based options may be preferred over hard infrastructure options. In lower-income countries, soft options are typically much cheaper and, therefore, more feasible (World Bank, 2019). In a higher-income context (e.g., in Spain, California, Israel and Singapore), many water supply systems are already heavily engineered with large volumes of storage, low leakage losses and access to desalination plants. Although further investment is need to maintain the assets, cope with population growth and restore the environment, these countries may be approaching the limit to which further investments in hard infrastructures would be cost effective (Del Moral Ituarte and Giansante, 2000) and only serve to heighten risk exposure (Di Baldassarre *et al.*, 2015, 2018). Further, social acceptability can be a limiting factor in infrastructure development, particularly reservoirs that displace communities. Although nature-based storage solutions, such as wetlands and managed Aquifer Recharge, could be an alternative storage solution, their effectiveness in risk reduction is highly context dependent, hindering their inclusion in optioneering frameworks at scale (Seddon *et al.*, 2020). Desalination plants are increasingly used to solve water deficit problems for coastal cities, however, the highly saline brine byproduct (Sharkh *et al.*, 2022) can have harmful consequences for the environment and/or

society, depending on the management approach (Einav, Harussi and Perry, 2003). Wastewater reuse for potable, industrial or irrigation water use is another form of unconventional water supply that could be considered, but challenges around social acceptability persist among other barriers (Lee and Jepson, 2020). Long-range water transfers can also be an effective solution for balancing supply and demand (McDonald *et al.*, 2014). However, the implementation of such schemes can be highly controversial in water scarce areas and face strong opposition from users of the proposed transfer source catchment (Hernández-Mora *et al.*, 2014). In contexts where there is competition between water users, such as between agriculture and cities, long term reallocation strategies might be considered (Molle and Berkoff, 2009; Garrick, De Stefano, *et al.*, 2019). A major challenge occurs in arid climates where temperatures are well suited to the cultivation of agriculture, but where water resources are severely limited (Martínez-Valderrama *et al.*, 2023), for example, including in Central Asia, the Middle East and North Africa and southern Europe. In these contexts, sustainable water futures may require substantial shifts in sectoral water allocation (Garrick, De Stefano, *et al.*, 2019).

2.3.1.4. Asset damage adaptation strategies

Asset damage strategies can either be targeted at various stages upstream of an exposed asset or directly on-site (APFM, 2020). Upstream strategies may include regulating peak flows through land use management and/or constructing small dams or building embankments and barriers along riverbanks (Ward *et al.*, 2017; APFM, 2020). Nature-Based Solutions for flood risk management (Dadson *et al.*, 2017) provide a viable alternative to hard infrastructure measures, particularly where utilities are not in a strong position to acquire debt in order to construct capital-intensive structural measures. For example, tree cover can shield structures from extreme winds (Gill *et al.*, 2007) and also buffer against floods, although their effectiveness is not easily quantified, contested and highly context specific (Carrick *et al.*, 2019; Buechel, Slater and Dadson, 2022). Conserving, restoring and/or constructing wetlands and sustainable urban drainage systems promote the infiltration of flood, replenishing aquifer levels, providing both flood and drought protection (Wilby and Keenan, 2012; Saurí and Palau-Rof, 2017; Dillon *et al.*, 2019). However, a barrier in their uptake and inclusion in large-scale optioneering studies the challenges in quantifying their (high context-specific) effectiveness in comparison to artificial protection structures (Morris *et al.*, 2018). Alternatively, flood barriers can be implemented directly at exposed structures and equipment, e.g., assets can be elevated or surrounded with flood walls, providing local protection against flood inundation (Mott MacDonald, 2018). Early warning systems, insurance and emergency recovery plans can also

be implemented as non-structural adaptation strategies for extreme events that induce physical asset damages, such as extreme winds and flooding (APFM, 2020).

2.3.2. Just adaptation frameworks

Over 2 billion people lack safely managed drinking water and 785 million do not even have basic water supply services, as defined by the SDGs (UNICEF, 2019). The majority of people without safe access are in low- and middle-income countries, highlighting the vast inequity in access to critical services among the global population (ibid). Further to inter-country differences, within countries, rural population are typically poorer, with lower rates of access to safe water services (ibid). Within cities, historic inequalities have left certain neighbourhoods underserved (Bakker, 2017); further, in rapidly developing cities, informal settlements expand at a faster rate to the expansion of services (Chitonge, 2014). However, access to services, as measured under the Sustainable Development Goal (SDG) framework, is not the end of the story. Even where services exist, most utilities in low- and middle-income countries do not operate continuous services, driven by a lack of adequate maintenance (Grasham *et al.*, 2022). Given that utilities are typically not achieving full cost recovery (generating sufficient revenue to cover costs), they typically prioritise higher income areas of their networks, where they can expect higher rates of return, leading to worse services in lower income areas of their network. As discussed in section 2.2.3, water supply disruptions, in turn, have disproportionate impacts on the poorest, most vulnerable customers, reinforcing cycles of poverty (Eakin *et al.*, 2016). Figure 7 illustrates the link between the vicious of utilities and poverty traps for low-income customers.

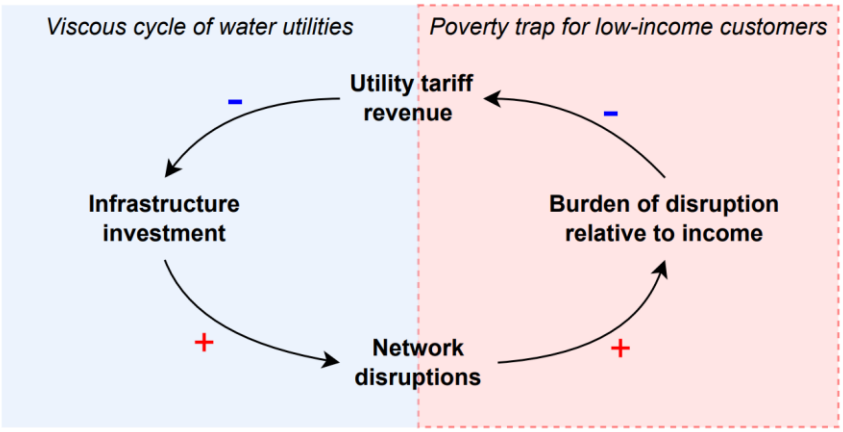


Figure 7 Conceptual representation of the interaction between utility vicious cycle and poverty trap for low-income customers.

Water sector development is inextricably linked to the national development priorities of a country, such as poverty alleviation (Ahmad, 2003). For example, governments strive to keep water tariff rates as low as possible to ensure affordability (Andres *et al.*, 2021). At the same time, utilities are heavily subsidised by governments to invest in infrastructure and expand networks to underserved communities (OECD, 2011). This link between water utilities and national development strategies, underlines the fact that decision-making in the water utility sector cannot be reduced to a one-dimensional problem of reducing costs. Thus, further to the optimisation approaches outlined in section 2.3.1, frameworks have been developed in the academic literature to evaluate proposed water infrastructure options based not only relevant financial information but also information on wider development objectives.

Garcia et al., (2021) develop such a framework in coastal Bangladesh, where water resources are threatened by groundwater contamination and extreme weather events (e.g., floods and cyclones). They pose a question around whether water investment plans to expand water access in a particular polder should "adopt a utilitarian approach to maximise for the greater good for the greatest number of people or a Rawlsian social justice approach which prioritises the "most in need" (Sen, 1974)?" (Garcia *et al.*, 2021, p. 1). Figure 8 shows the results of their analysis of how alternative investment plans (represented by the grey dashed lines) perform against two objectives: (1) maximizing overall access to improved water supplies and (2) maximizing access for the population with the lowest welfare. The red lines represent investment plans of three alternative budgets. Such information can be presented to decision makers to help them understand the benefits of different levels of investment, and different investment strategies, in terms of improving water supply overall and/versus to the most vulnerable people.

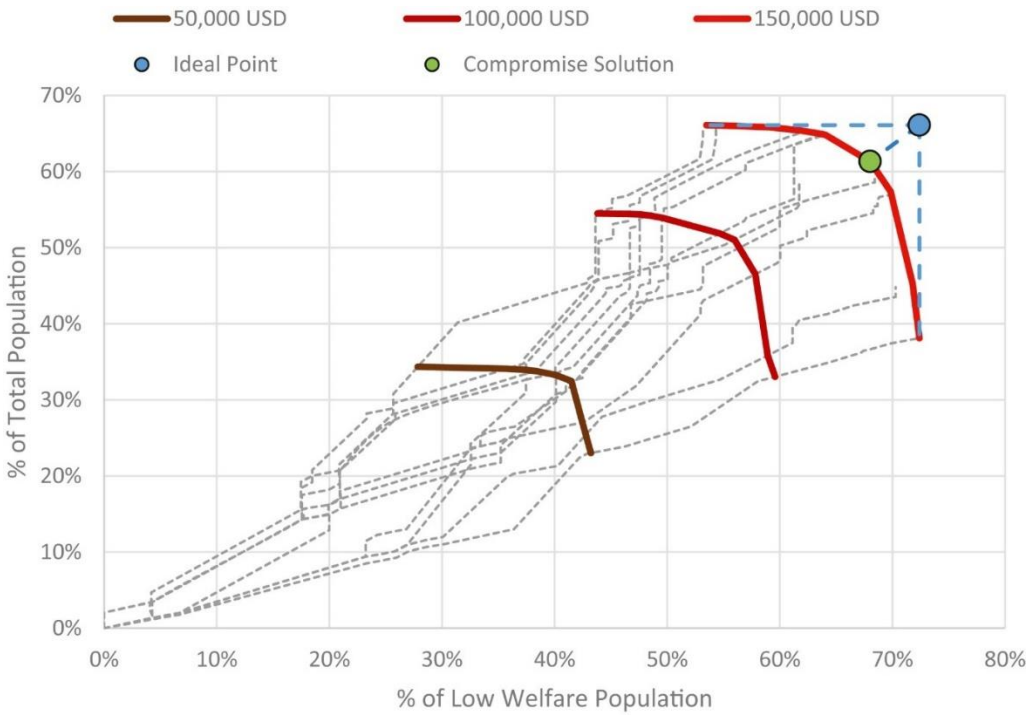


Figure 8 Illustration of decision-making framework tailored to analyse trade-offs between maximizing access to low welfare population and maximizing overall access. Figure taken from (Garcia et al., 2021, p. 11).

There is a large body of literature that focusses on the inequalities of water access and service reliability levels (Beard and Mitlin, 2021). There is also extensive research on the disproportionate impact of climate shocks on poorer parts of the population (Hallegatte and Rozenberg, 2017; Winsemius *et al.*, 2018). However, there is not much literature that combines climate risk and development objectives into decision making frameworks (Verschuur *et al.*, 2020), particularly for water supply utility optioneering. As discussed in section 2.2.3, climate impacts and water supply disruptions can have a disproportionate impact on the poorest households (McIntosh, 2003). Utilities play a key role in either reinforcing or counteracting these, often historical, inequalities. As such, optioneering frameworks that consider benefits through a poverty lens provide a valuable basis decision-making, particularly in developing countries.

2.3.3. Summary

Climate adaptation actions can be a critical input into reversing the vicious cycle of water supply utilities. Measures such as flood defences, leakage reduction, desalination and reservoir storage, can reduce the rate of climate-related network disruptions. However, many utilities are not in a strong financial position to invest in adaptation measures. Thus, limited utility budgets

must be carefully allocated to the measures that will achieve the greatest benefits, relative to cost. Optioneering studies are widely applied for this purpose. To prioritise drought adaptation options, the risk reduction potential of options, over each option's lifespan accounting for future risk projections (e.g., under climate change) such as increasing reservoir storage capacity, reduction leakage loss, inter-basin transfers and desalination are weighed against costs. The approach is similar for adaptation options that respond to asset damages (e.g., flood protection), however, consideration must be paid to the different impact channels that are defended through different types of adaptation option. Asset damage adaptation options, e.g., flood walls, protect against both operational disruption and physical damages. The misalignment between impact channels and modelling approaches as prevented the prioritisation of portfolios of assets that adapt to a range of hazards, i.e., including events that induce assets damages and water shortages, in the potable water sector.

Water utility investment plans are not only driven by a need to reduce the costs of disruptions, but also by the need increase the quality of services in underserved areas and maintain affordable services. This calls for multi-dimensional decision-making frameworks. Although the distributional impacts of climate shocks on vulnerable groups have been widely explored, and the need for decision making in the water sector has been widely cited, climate-based optioneering relative to both risk reduction and wider (e.g., pro-poor) development-agendas, is not yet mainstreamed.

3. The case of the Jamaican National Water Commission

The National Water Commission (NWC) in Jamaica is the focus for developing both frameworks introduced in chapters 4 and 5. Jamaica has achieved the highest rate of access to public water supplies of all the Caribbean islands (National Water Commission, 2020a). The NWC is the main water utility on the island, serving 90% of the potable water supply across the island and covering 70% of the total population (National Water Commission, 2020a). However, the utility is burdened by a very high rate of service disruptions, with ~1400 number of water supply disruptions recorded over the course of 2020, many of which are driven by climate extremes including as cyclones, floods and droughts (National Water Commission, 2020c).

Jamaica's water system is highly decentralised, as a function of its relatively sparsely distributed, rural population and large topographical variations, as well as low storage capacity. Indeed, there are only 2 major reservoirs dedicated to public supply (and 1 dedicated for agriculture) on the island, both serving the country's capital, Kingston. The remaining population served by the utility is supplied by relatively small-scale systems fed by 160 underground wells, over 116 river sources (via water treatment plants) and 147 springs (National Water Commission, 2020a). For reference, a map of the NWC network is provided in Figure 9. Because the surface water systems are the most vulnerable to drought and flood hazards (due to their proximity to and dependence on major water sources), the main priority going forward is to improve the resilience of surface water systems (National Water Commission, 2020c). However, the scope for greater groundwater extraction to supplement these surface water systems is limited as groundwater in Jamaica is vulnerable to pressures such as salinisation, groundwater depletion and contamination from sewage and industrial processes, and only receives a chlorination treatment step before distribution (Miller, Waite and Harlan, 2001). The contamination of aquifers is irreversible in the vast majority of cases, so when the salinity or nitrate levels have exceeded an acceptable threshold, wells must be abandoned. For example, in the capital, Kingston, where the majority of domestic sewage is stored in septic tanks as opposed to being treated, many wells can no longer be used due to contamination from sewage (Miller, Waite and Harlan, 2001).

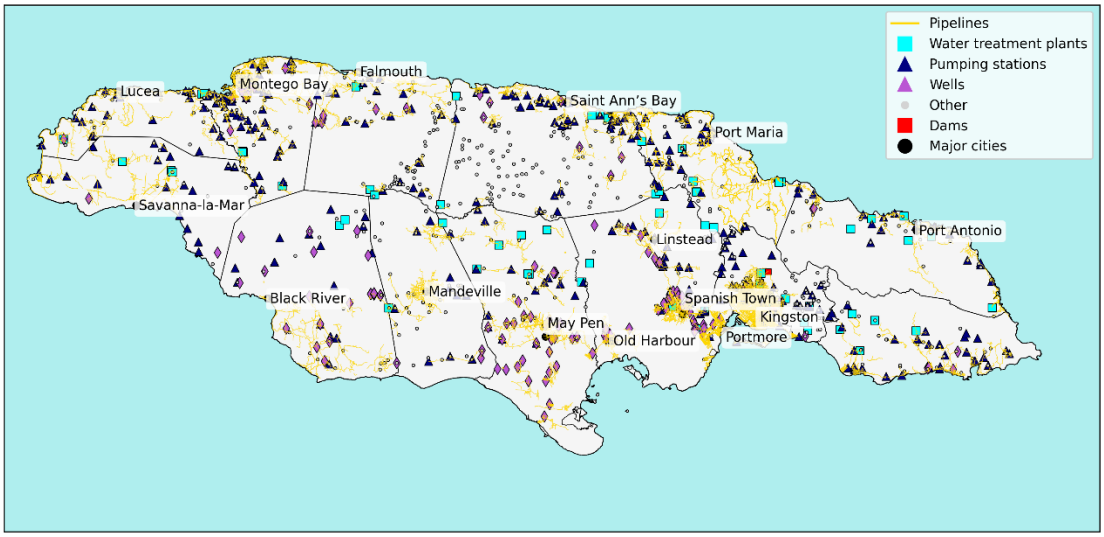


Figure 9 Potable asset data comprising 129 plants, 409 pumping stations, 169 wells and 2 dams that are owned and operated by the NWC Utility in Jamaica, the NWC (National Spatial Management Branch, 2010). The locations of major city locations are also provided (OpenStreetMap, 2018).

Jamaica's water infrastructure is intensely vulnerable to damage from flooding (Mandal and Maharaj, 2013; Burgess *et al.*, 2015; Glas *et al.*, 2015; Mandal *et al.*, 2016; Nandi *et al.*, 2016), droughts (Campbell, Barker and McGregor, 2011; Setegn *et al.*, 2014; Goyal, Madramootoo and Richards, 2015; Lester, 2015) and cyclones (He and Cha, 2021). During climate-induced disruptions, customers typically turn to other sources, such as tanker trucks or untreated well or spring supply. There are several health risks associated with water stored in household tanks (Lester, 2015) and unregulated sources, such as wells, which are becoming increasing contaminated by untreated sewage (Mandal *et al.*, 2011; Jamaica Water Resources Authority, 2020). Therefore, tanker trucked water offers an essential alternative replacement service during utility water supply disruptions, to protect the health of disrupted communities. For example, in 2019, a major drought hit the island, completely drying up many surface water sources and significantly reducing the operational capacity of many more (National Water Commission, 2020c). During such events, the NWC often transports water to disrupted communities via tanker trucks (JIS, 2015). However, due to the NWCs limited capacity, private tanker trucks are a major form of service delivery during disruptions, where "households often had [have] no choice but to purchase expensive water from private water trucks or drink untreated water, which increased the risk of disease" (Altink, 2022). Piped water supply disruptions typically hit poorer communities hardest, with private tanker trucks costing up to 8 times more than piped water supply tariffs (Jamaica Observer, 2018; National Water Commission, 2020a).

Improving the resilience and coverage of water supplies is a key element of Jamaica's development vision and trajectory (Planning Institute of Jamaica, 2018). The NWC aims to improve system reliability over the coming decade (for example, by reducing intermittent supply, which ranges between 12-20 hours per day across parishes) (National Water Commission, 2011) and is committed to contributing positively to national development by providing high quality, affordable potable water services to customers (MEGJC, 2019). However, the capacity for the utility to invest in better services is hindered by a high level of debt, with the highest debt-to-equity (leverage) ratio in the Caribbean (Burdescu *et al.*, 2020). In this context, it is essential that the adaptation strategy that the utility adopts is as cost-efficient as possible and minimises the financial losses burdened by the utility. This could be achieved by minimising costly damage to assets incurred, for example, due to flood inundation. Another priority adaptation strategy is maximise the utilisation of water resources that are currently being extracted through reducing leakage, which can reach levels of 70% in some systems, which would delivery co-benefits including reducing operational (energy and production) costs and lessening the risk of pipe bursts (National Water Commission, 2011).

Existing research on Jamaica's water system and water related-hazards includes studies of flood risk and damage assessments (Mandal and Maharaj, 2013; Burgess *et al.*, 2015; Glas *et al.*, 2015; Mandal *et al.*, 2016; Nandi *et al.*, 2016), analyses of hydrological variability for water scarcity planning in specific catchments (Setegn *et al.*, 2014; Goyal, Madramootoo and Richards, 2015) and small-scale, regional assessments of social vulnerability to water stress (Campbell, Barker and McGregor, 2011; Lester, 2015). Descriptions of the water resources management challenges on the island from a hydrological and institutional perspective are also provided (Miller, Waite and Harlan, 2001), but there is no research on risk-based water infrastructure planning or national-level extreme streamflow analysis on the island.

Therefore, Jamaica provides a suitable case study upon which to develop the proposed multi-hazard risk assessment and socially orientated risk-based adaptation framework, as set out in chapters 4 and 5. The climatic and water supply challenges faced in Jamaica are comparable to, and reflective of, utilities in many other low and middle-income countries, which are stuck in the trap of disrupted services, low-cost recovery and low adaptive capacity. As such, frameworks developed in chapters 4 and 5 are generally applicable to any national infrastructure setting with adequate data availability that face similar challenges.

4. A multi-hazard risk framework to stress-test water supply systems to climate-related disruptions

Chapter 4 presents a generally applicable framework for generating multi-hazard event sets to stress test water infrastructure systems and compare the risk of customer disruptions. The framework is applied to the case of the National Water Commission's system in Jamaica.

Paper: Becher, O., Pant, R., Verschuur, J., Mandal, A., Paltan, H., Lawless, M., Raven, E. and Hall, J., 2023. A Multi-Hazard Risk Framework to Stress-Test Water Supply Systems to Climate-Related Disruptions. *Earth's Future,* 11(1), p.e2022EF002946.

Co-author contributions: Prof Jim Hall from the University of Oxford assisted with the conceptualisation and structuring of the paper, as well as with review and feedback on the manuscript. Dr Jasper Verschuur and Dr Raghav Pant from the University of Oxford, Dr Arpita Mandal from the University of the West Indies and Homero Paltan from the World Bank assisted with review and feedback on the manuscript. Mark Lawless and Emma Raven provided JBA flood event and return period map data.

Key Points:

- We propose a spatial multi-hazard risk framework for analysing present-day and future climate risks to water users.
- We propose the use of customer disruption days (CDD) as a common metric for comparing different hazards impacts.
- The framework can be used by decision makers to prioritise investments across asset protection against flooding and cyclones, and drought mitigation options.

Abstract

Water utilities' supply systems are vulnerable to several climate-related hazards, including droughts, floods and cyclones. Here we propose a generally applicable framework for conducting multi-hazard risk assessments of water supply systems and use it to quantify the impact of present and future climate extremes on the national water supply network in Jamaica. The proposed framework involves stress-testing a model of the system with a large set of spatially coherent drought, cyclone and pluvial and fluvial flood events to calculate the number of water users whose supplies would be disrupted during an event, i.e. the Customer Disruption Days (CDD). We estimate the total multi-hazard annual expected disruption to be approximately 5 days per year per utility customer under present conditions. This is increases by a factor of between 2 and 2.5 when end-of-century climate scenarios are propagated through the model. Our analysis shows that more high probability drought events lead to greater CDD compared with asset damage events. However, extreme asset damage events, despite manifesting over shorter timescales (days) compared to drought events (months), can lead to more widespread CDD. This quantified risk framework would allow utility managers to compare the risk of both asset damage- and water shortage-induced disruptions via a common, decision-relevant metric. However, applications to other utilities would require tailored hazard modelling approaches. The proposed risk assessment is intended to inform prioritization of infrastructure investments, ranging from asset protection to drought mitigation projects, with the goal of enhancing water supply resilience in the face of a changing climate.

4.1. Introduction

Resilient water supply infrastructure is fundamental for human life and wellbeing (Hallegatte, Rentschler and Rozenberg, 2019). As adverse impacts of climate change threaten the Sustainable Development Goals (SDGs) (UN, 2021), the need for resilient water supply infrastructure is increasingly important if several SDG targets including clean water and sanitation (SDG 6) are to be met by 2030 (Krueger *et al.*, 2020). However, climate extremes, such as droughts and floods, pose a high risk to water supply infrastructure given their proximity to, and dependency on, water bodies (Brozović, Sunding and Zilberman, 2007; Grey and Sadoff, 2007; Hall et al., 2014; Matthews, 2016; Damania et al., 2017).

Water supply infrastructure systems are subject to multiple climate-related impacts on a range of different timescales, with different spatial extents and varying impacts. Droughts usually materialise over periods of weeks and months, but when demand exceeds supply for prolonged periods the consequence can be water shortages for water users. Floods, on other hand materialise over shorter timescales and can cause direct damage to water supply infrastructure assets, such as pumping stations and treatment works, resulting in water supply disruptions. Extreme winds, from cyclones can also directly damage water infrastructure assets and other infrastructures (e.g. power lines) upon which they depend. For example, an estimated 11 million people were affected by the crisis in Chennai, India, in 2019 when four of Chennai's drinking water supply reservoirs dried up (Kumar-Rao, 2019). In Chile, supplies to four million people were disrupted in 2017, when floods in the Andes led to the contamination of Santiago's water source (BBC, 2017). Floods damaged treatment plants and canals in 2017 in Lima, Peru, disrupting water supplies for up to five days, with some treatment plants remaining inactive for 6 months (Collyns, 2017). However, apart from a handful of local or regional studies that focus on the exposure of water supply infrastructure to hazards that affect both water availability and damage to the supply network (e.g., Espada et al., 2013), multi-hazard risk analysis studies of water supply systems are lacking.

The risks of natural hazards to water supplies can be particularly acute in low- and middle-income countries that face rapidly growing demand for urban water supplies, alongside under-investment in system maintenance and upgrade (Khatri, Vairavamoorthy and Porto, 2009; van der Bruggen, Borghgraef and Vinckier, 2010; Ray and Shaw, 2019). These countries are often also disproportionately impacted by extreme and chronic climatic events and have limited institutional capacity to adapt (Rouse, 2013). It is, therefore, essential that governments and

water service providers understand climate risks so that they can proactively respond to adverse impacts, particularly in the context of increasing likelihoods of extreme floods, droughts and cyclones due to anthropogenic climate change (Mirza, 2003). Understanding and quantifying the risks that multiple hazards pose to water supply infrastructure can be useful to identify adaptation strategies that improve resilience against a variety of threats (Hallegatte *et al.*, 2012; Hall and Borgomeo, 2013).

Water supply systems are networks of infrastructure designed to abstract water from sources (rivers, lakes, groundwater, desalination), treat it, store it and distribute it to customers (which can be households or businesses). Modelling these systems therefore requires representation of (i) the meteorological factors (rainfall, evapotranspiration etc.) that influence the amount of water in surface and groundwater bodies; (ii) flow of water across the landscape to points where it is withdrawn; (iii) the engineered infrastructure which stores, transfers, treats and distributes water to where it is used. Various representations of these systems have been adopted, typically based on mass balance in the water resource system (Watts *et al.*, 2012; Mens, Gilroy and Williams, 2015) to quantify the risk of source water supply or quality being inadequate to meet demand (Mortazavi-Naeini *et al.*, 2019). Given the computational expense associated with simulating complex pipe network systems, some researchers have resorted to spatial network graph representations for mapping connectivity between water infrastructure assets and the linkages of individual assets to their served populations (Brozović, Sunding and Zilberman, 2007; Diao *et al.*, 2016; Thacker *et al.*, 2017; Pant *et al.*, 2018; Yunzhu *et al.*, 2022).

There has been a growing attention to multi-hazard risk assessment for infrastructure systems, in recognition of the disproportionate impact that compound risks can have on society and the economy (E. E. Koks *et al.*, 2019; Moftakhari and Aghakouchak, 2019; Hillier *et al.*, 2020). This is challenging for all spatial infrastructure systems, because of the challenges of (i) properly accounting for the statistical dependence between different hazard types; (ii) the spatial characteristics of the hazard; and (iii) the different impact channels in a consistent way. Many existing studies either employ case studies of a single multi-hazard event (De Angeli *et al.*, 2022), or employ return period maps that fail to account for the correlation between events (Towe, Tawn and Lamb, 2018), preventing the risk of simultaneous asset failures being accurately captured. Similarly, approaches using return period maps would fail to account for the correlation between different hazards, such as the coincidence of extreme winds and flood inundation during cyclones. It would be preferable to employ a stress-testing approach using

a range of observed or synthetically generated hazard events to understand the risk of multiple assets failing simultaneously and causing widespread disruptions (Jongman *et al.*, 2014; Speight, Hall and Kilsby, 2017; Lamb *et al.*, 2019).

For water supply systems, the challenge is particularly complex because as well as being subject to a range of acute hazards which can damage infrastructure assets, in addition, drought hazards impact the availability of the resources (water) which is being delivered by the supply network. Thus, multi-hazard risk assessment involves combining analysis of damaging acute extreme events (e.g. floods, cyclones) with analysis of drought risks to water supplies. The former of these processes is common to approaches adopted for other infrastructure networks and involves a convolution of spatial functions that represent the hydrometeorological hazard (drought, flood, cyclone) at a range of different timescales, with a probabilistic representation of the resistance of the water infrastructure system network to yield a probability distribution over a set of asset failures and consequential disruptions of supply to water users (Laucelli and Giustolisi, 2015; Pant *et al.*, 2018; Thacker *et al.*, 2018). Analysis of drought risk, involves hydrological modelling and mass balance calculations (supply versus demand) in order to estimate the frequency and duration of water shortage for given numbers of water users (Borgomeo *et al.*, 2014; Ghile *et al.*, 2014; Whateley, Steinschneider and Brown, 2014; Taner, Ray and Brown, 2019).

Here we propose a modelling framework that seeks to address acute damage to water supply networks at the same time as computing the risk of water shortages. We construct a systems model, which represents asset-level vulnerability as well as domestic water usage, leakage losses and major storage dynamics in the supply network. The systems model is stress tested with a large set of spatially coherent drought, cyclone and pluvial and fluvial flood events, combining observed events with synthetic (combining statistical and model-based) events. For each disruptive event, the number of water users whose supplies may be disrupted per day is calculated, multiplied by the event's duration in days, i.e., the Customer Disruption Days (CDD). CDD provides a common decision-relevant metric with which to compare risks to water supply customers from the intersection of multiple hazards and system vulnerabilities. We examine the impact of present and future climate extremes under a selection of RCPs which can be altered for different applications of the framework. The framework presented can be generally applied to identify local hotspots of disruption, as well as quantify the potential extent of simultaneous disruptions affecting a national-scale potable supply network, essential for

contingency planning in preparation for widespread disruptive events. To the best of our knowledge, no implementable frameworks exist to compare the risk of widespread water supply disruptions caused by water shortages (drought-induced) and asset damages (e.g., flood or wind-induced) through a common risk metric.

The proposed risks framework is applied to the context of Jamaica, although it is applicable to any national infrastructure setting with adequate data availability. The National Water Commission (NWC) of Jamaica is the utility that supplies drinking water to 90% of Jamaica's total potable water supply (National Water Commission, 2020a). The water supply infrastructure in the Caribbean region is exposed to multiple climate hazards that burden the population with frequent and devastating disruptions (Medina, Kullmann and Felter, 2021). Jamaica's water infrastructure is intensely vulnerable to damage from flooding (Mandal and Maharaj, 2013; Burgess *et al.*, 2015; Glas *et al.*, 2015; Mandal *et al.*, 2016; Nandi *et al.*, 2016), droughts (Setegn et al., 2014; Goyal, Madramootoo and Richards, 2015; Campbell et al., 2011; Lester, 2015) and cyclones (He and Cha, 2021). Improving the resilience and coverage of water supplies is a key element of Jamaica's development vision and trajectory (Planning Institute of Jamaica, 2018), with the NWC aiming to significantly improve system reliability over the coming decade (National Water Commission, 2020a). This situation is comparable to many other low and middle-income countries, which need to expand access to water supplies, despite aging infrastructure assets and a limited budget, combined with a large vulnerability to various climatic hazards that will be exacerbated by climate change (Mirza, 2003; Srinivasan *et al.*, 2013).

4.2 Materials and Methods

4.2.1 Overview of generally applicable framework

Our proposed multi-hazard risk framework for analysing the propagation of present-day and future climate extremes through networks of water supply infrastructure to water users is shown in Figure 10. First, a methodology is developed that allows generating a large set of (synthetic) spatially coherent drought, cyclone and pluvial and fluvial flood events, both for the present-day situation for three climate change scenarios projected to the end of century epoch (RCP2.6, RCP4.5 and RCP8.5). Second, we create a spatial model of the national water supply infrastructure network, which combines the simplicity of a graph-based approach (where nodes characterise asset level vulnerability to physical damage and disruption duration for variations in wind speed and flood inundation), with elements of a simulation-based approach (domestic demand and leakage calculation, and storage dynamics). Third, we intersect the event set of multi-hazards with the water supply network to quantify of the number of users impacted per day multiplied by the duration of the event (hereafter Customer Disruption Days, or CDD). This framework can accommodate droughts which typically induce relatively long-term, chronic water deficits to the infrastructure network, cutting off water supply to users without causing structural damage to assets (Singh *et al.*, 2021). Alongside the acute effects of extreme rainfall, fluvial flooding and extreme winds during cyclones can damage infrastructure assets, completely cutting off customers that are dependent on these assets for their water supply during repair and restoration activities. Our proposed quantification of CDD applies across these different hazard types. Expressing risk in terms of expected annual CDD enables one to compare and contrast the risk from different disruption mechanisms, allowing the water utility to identify critical vulnerabilities and prioritise adaptation strategies across asset protection and drought mitigation options. We recognise that disruptions of different durations are not exactly equivalent: complete loss of supply for a shorter period during a flood event may be considered as a more severe impact than prolonged periods of intermittency, if storage tanks are able to buffer against the repeated supply cuts. Having said this, several months of water shortage-induced disruption, even if it manifests through intermittency, may burden households more than a shorter period of disruption if households have to rely on expensive tanker truck vendors. To illustrate these effects, we also report on the duration of water shortage and asset damage-induced disruptions.

Next, we describe the different components of the framework in detail and support our methods with the working example from Jamaica.

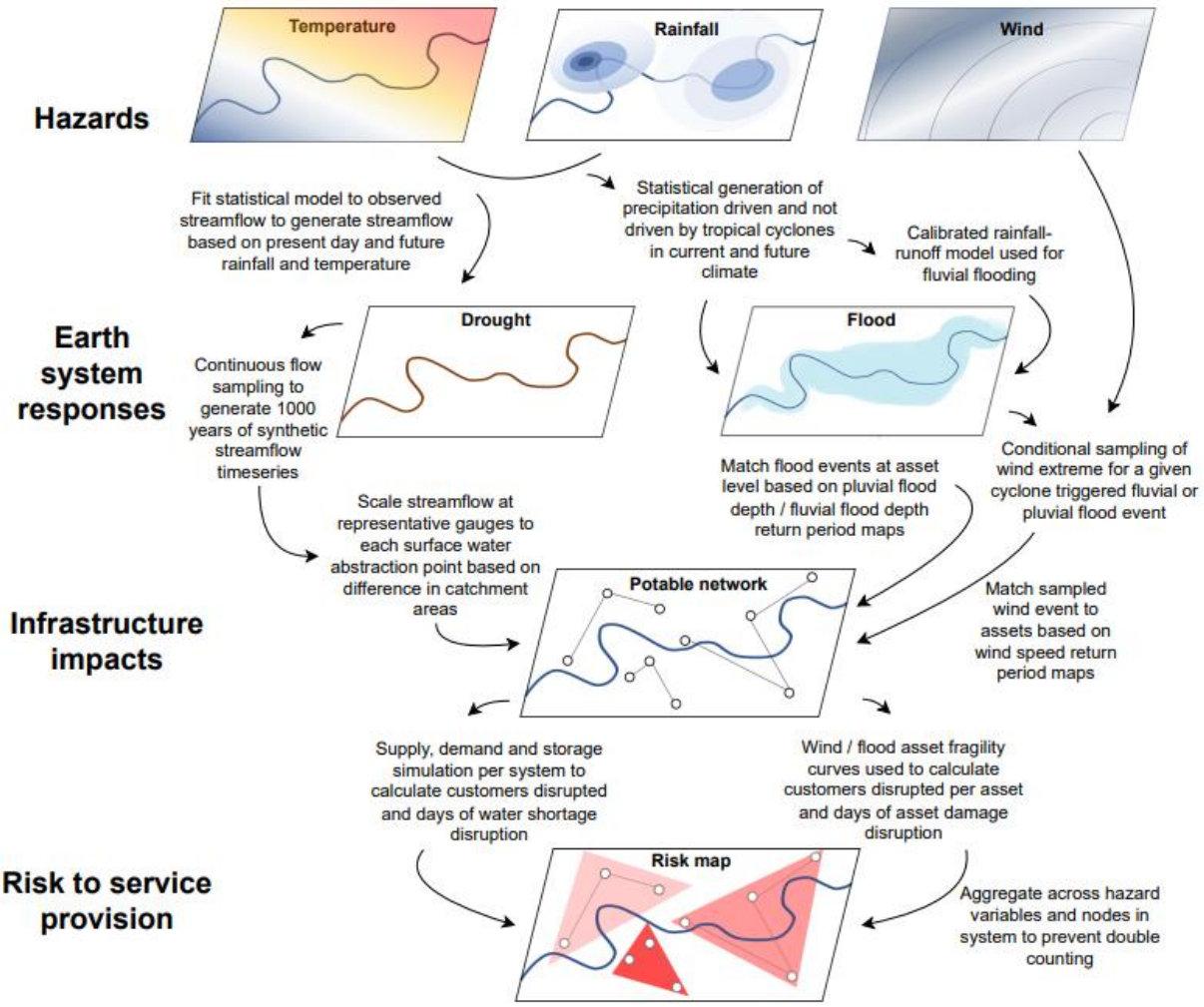


Figure 10 Methodological framework

4.2.2. Application of framework to Jamaica case study

The steps below are described to be as generic and transferrable as possible. Aspects of the analysis have been adapted to match the geographical characteristics and data availability of the case study.

4.2.2.1. Multi-hazard event set generation

We provide an overview of the methodology to sample (1) spatially coherent and co-varying pluvial and fluvial flood inundation and cyclone wind extremes, and (2) spatially coherent drought events. We separate the multi-hazard occurrence into these components as droughts are unlikely to be correlated with extreme flooding and tropical cyclones, as droughts are

induced by periods of below average rainfall, whereas floods and cyclones are associated with above average rainfall. However, we do acknowledge that flash floods may co-occur with droughts, the likelihood of this is not examined as part of this study. On the other hand, cyclone often cause flooding (both fluvial and pluvial) and extreme winds, the combination of which has to be explicitly considered to accurately quantify the risk. The methods of analysis have been adapted to the differing characteristics of floods and droughts - for drought risk analysis we constructed an empirical model from the extensive network of streamflow gauges to assess volumes of water available on monthly timesteps, whereas the flooding analysis is run on a daily timestep, to capture shorter periods of intense rainfall, and is coupled directly to the flood inundation model.

Statistical analysis and simulation of spatially coherent precipitation

The dependence (across variables and in space) of flooding and extreme wind is a consequence of the driving meteorological conditions. To obtain a complete gridded dataset of relevant variables we use data from the Climate Forecast System Reanalysis (1979-2010) (Saha et al., 2010). We generate a multivariate hierarchical spatial statistical model of precipitation across the island through a spatial relation network (Ledford and Tawn (1996)), based on the coefficient of tail dependence between rainfall Observation Points (Ops) (grid points). Supplementary Figure B1 shows the locations of rainfall and discharge Ops on the island. The multivariate model was sampled to generate 10,000 years of synthetic events.

Rainfall-runoff modelling and flood inundation modelling

This synthetic precipitation data is used to force 26 calibrated daily timestep rainfall-runoffs model to provide synthetic daily river discharge estimates for a number of Ops across the island. The discharge events are assigned a return period using extreme value theory. We have linked the sampled fluvial flood return periods to different hydrological basins (called Hydrological Accumulation Zones, HAZ), which are assumed to be suitable geographical areas for the spatial co-occurrence of fluvial flooding. We refer to Supplementary Text B1 for a description of the hydrological modelling procedure, calibration and validation procedure, and the HAZ and the link with the Ops. We use sampled patterns of return periods associated with discharge extremes per hydrological basin to look up simulated flood extent and depth for six return period maps (25, 50, 100, 200, 500, 1500) (provided at a 30m resolution), and use this to interpolate or extrapolate the flood depth at individual asset locations within the HAZs (JBA, 2020a). An illustration of a 100-year return period flood map is included in Supplementary

Figure B2. Scaling of the flood depth between two known flood depths (d_l, d_u) associated with two return periods (r_l, r_u) within the set of available $\boldsymbol{r} = \{11, 20, 50, 100, 200, 500, 1500\}$ is done through logarithmic scaling as shown in Equation 8. Logarithm scaling is used as it is as an approximation of a Weibull extreme value distribution, which is widely used to represent the relationship between flood depth and event frequency (Heo, Salas and Boes, 2001). Note that the same process was undertaken for surface water flooding, however, rather than coupling extreme discharge events to fluvial flooding return period maps, we directly couple extreme rainfall events to pluvial return period maps.

$$d_y = \begin{cases} 0 & if\ r_y \leq 11 \\ d_l + (d_u - d_l)\left(\frac{\log r_y - \log r_l}{\log r_u - \log r_l}\right) & if\ r_l \leq r_y \leq r_u, \forall r_l, r_u \in \boldsymbol{r}, r_u > r_l \\ d_{max} & if\ r_y \geq r_{max} = 1500 \end{cases} \quad (8)$$

The rainfall-runoff and flood inundation modelling is an extract of a global hydrological modelling procedure which was undertaken at each of approximately 170,000 points of interest covering the globe. This promotes the scalability of the approach and allows further application of the framework to other contexts beyond Jamaica. However, we acknowledge that, given that the approach we adopt has been designed with computation efficiency and scalability in mind, and given the complexity of Jamaica's mountainous geography, characterizing flood risk with 100% accuracy is near-impossible, even despite the high (30m) resolution of the flood maps. To validate the model's performance, we compare model outputs for historical events against a dataset of actual events (National Spatial Management Branch, 2010). We find that the model picks up all years of flooding over a 30-year period. We undertake a hit / miss analysis to find that the model determines whether a given parish is or is not affected by each flood event with ~72% accuracy, please see Supplementary Figure B3.

Co-variance with cyclone wind speeds

The co-dependence between extreme wind speed and flooding during cyclones is represented by estimating the dependence structure between the extreme wind speed and the severity of flooding across all Ops. We derive the co-dependence between extreme wind and flooding, as opposed to precipitation, as it provides directly links the relevant hazard variables. We find an asymptotic relationship between flood occurrence and extreme wind, as shown in Supplementary Figure B4, indicating that more extreme floods are associated with relatively less extreme winds. This is likely because larger flood events tend to occur with slower moving cyclones, as they provide sufficient time for precipitation to augment river flows and heighten

water levels (Lai *et al.*, 2020; Titley *et al.*, 2021), and slower moving cyclones are less intense in terms of wind speed (Eguchi, Hattori and Nomura, 2021).

The dependency relationship between flood and wind occurrence is used to sample synthetic co-occurring wind extremes using a copula. The Gaussian copula model (Sklar, 1959) is selected due to its suitability for representing the dependency between variables that have an asymptotic relationship (Renard and Lang, 2007). The return periods sampled at each location are transformed to extreme wind using return period maps of cyclone wind speed from Bloemendaal et al. (2020). A 100-year return period wind map is presented in Supplementary Figure B5. For each grid (0.25°), we fit a GEV function to static return period maps and use this to sample wind speed per return period sampled with the Copula, which we assign to the assets located within that grid.

Spatially coherent drought events

To estimate water shortage-induced disruptions, monthly streamflow variations (or reservoir volume variations, where relevant) are estimated for each point where water is abstracted for public water supplies. We only consider surface water-fed sources and assume that normal supply from groundwater sources is maintained during droughts. This is because, firstly, groundwater systems are significantly more resilient to drought in our case study context in Jamaica according to historic data provided by the utility, where only 20 out of the 1000 instances of droughts affecting utility water supply services between 2014 and 2021 are recorded from affected groundwater-fed systems (NWC, 2021a). Secondly, surface water sources make up the majority of supply in terms of population served (approximately ~65% of NWC customers (National Water Commission, 2011)).

Each abstraction point is assigned a streamflow record based on the closest upstream gauge with at least 10 years of monthly mean streamflow data including the year 2019 (Jamaica Water Resources Authority, 2020) (see Supplementary Figure B6). For abstraction points that are too far upstream to have an upstream gauge, the nearest undisturbed gauge is used, i.e., upstream of any abstraction point, where abstraction point data is provided by the National Spatial Management Branch of Jamaica (National Spatial Data Management Division, 2001). For each streamflow gauge, precipitation and temperature from ERA5 reanalysis data is extracted (1980 to 2020) (Hersbach *et al.*, 2020). Because we want to create synthetic streamflow time series now and in the future, streamflow needs to be linked to the meteorological drivers (temperature and precipitation) per gauge. RandomForest algorithms have been found to offer a high level

of accuracy and flexibility in the prediction of non-linear processes such as streamflow modelling (Li, Sha and Wang, 2019). We, therefore, train and apply a RandomForest model site-by-site to gap fill the available observed streamflow record and generate a consistent timeseries length across all considered gauges of 40-years (1980 to 2020). Note that we expanded the length of the historic epoch for the drought risk analysis compared to the flood and cyclone risk analysis to maximise the data input for training the Random Forest. The RandomForest regression model is trained to predict monthly streamflow measurements at each abstraction point using monthly ERA5 precipitation (with 1- and 2- month lags) and temperature timeseries. A sample of the first 80% of the observed streamflow record is used for training, such that model predictions can be tested on the last 20% of the sample, to prevent overfitting. The models are found to perform to a high level of accuracy, with the vast majority of models performing with R-squared values above 0.8 (see Supplementary Figure B7). The difference in catchment area feeding the streamflow gauge and the upstream or downstream location of the abstraction point is used to scale monthly streamflow, as is a common methodology in hydrology for estimating ungauged river reaches (Gianfagna *et al.*, 2015).

For the two reservoirs in the network, monthly mean reservoir volume data since 2006, provided by the NWC, is related to the nearest upstream streamflow variations with 1-, 2-, 3- and 4- monthly lags using a separate RandomForest model. The R^2 is found to be ~0.6 for both reservoirs. The lower accuracy of reservoirs relative to streamflow is likely due to the relatively short period of observed data and because data on the variations of human extractions are not driven by meteorological factors and therefore harder to capture. As our objective is to model the supply variations for each reservoir, as opposed to the combined effect of supply and abstraction variations, we propose this approach as the best possible approximation, given the data constraints. As can be observed from Supplementary Figure B8, the major reservoir level drops are, however, well captured by the modelled output.

The 40-year statistically derived streamflow timeseries scenarios is input into a multi-site, non-parametric streamflow generation model, as introduced and described in Wheeler et al. (2020). This model reproduces the temporal dependence of hydrological sequences at each location on a range of timescales, including long-memory persistence characterised by the Hurst coefficient (i.e., a measure of long-term autocorrelation within a time series), as well as reproducing the spatial dependence between each timeseries. Here, this model is applied to generate ten realisations of a 100-year timeseries that represent baseline climate conditions,

providing 1,000 years of synthetic streamflow record at all abstraction points on the island. As for the systems fed by the two major reservoirs, the RandomForest model used to derive reservoir volume variations based on the nearest upstream stream gauge records is applied to the synthetic streamflow generated for each of these gauges to generate 1,000-years of synthetic reservoir volume timeseries. Given the natural climate variability in the system, we deemed 10 realisations of 100 years satisfactory to have both a long enough time record as well as have a number of alternative realisations. However, this number of iterations should be adjusted to suit the contexts of future studies. We plot the empirical cumulative distribution function of the baseline synthetic flow generated against the input flow data for a selection of 6 gauges, to indicate the range of scenarios tested in comparison to input data in Supplementary Figure B8.

Climate change scenarios

We wish to explore system performance in possible future climate conditions. End of century climate scenarios are used in order to explore a wide range of possible uncertainties posed by climate change across the hazards considered. The possible future climate scenarios are explored using change factors from relevant climate model outputs.

Monthly precipitation and temperature outputs under RCP2.6, RCP4.5 and RCP8.5 (van Vuuren et al., 2014) climate change projections at 0.5-degree resolution are collected from the REGCM4 model. This model was chosen due to its strong performance across the Caribbean region: despite positive and negative biases, which were found to be particularly pronounced in the rainy season, the model captures temperate and precipitation patterns well over the spatial domain, providing a strong correlation with observed datasets (Martínez-Castro *et al.*, 2018). It was also recommended by the University of the West Indies Climate Group for the current project. and was recommended by the University of the West Indies Climate Group for the current project. Change factors are derived (Bennett *et al.*, 2011) by comparing precipitation and temperature across a range of quantiles in the historical climate model output (1960-2010) to the same quantiles in the projected climate output under each scenario between years 2070 and 2100 per grid cell. Applying change factors in this way is widely used to downscale climate data to retain complex projected changes, such as increased frequencies of rainfall extremes and/or prolonged drying trends, particularly when using reanalysis datasets (Chen et al., 2018). These change factors are applied to the ERA5 reanalysis precipitation and temperature data by matching each ERA5 grid cell with the overlapping climate model grid cells. Note that

CHIRPS, which is purely observation-based (Funk *et al.*, 2015), would offer a viable alternative data source for the precipitation component. Thus, future precipitation and temperature timeseries representing each climate scenario are fed into the RandomForest model to simulate future streamflow time series for the drought risk analysis.

For future flooding scenarios, projected precipitation and temperature under baseline and end of century RCP2.6, 4.5 and 8.5 conditions between 2070-2100, from the CNRM-CM5 model (Voldoire *et al.*, 2013), are used to force the precipitation generator and rainfall-runoff model (JBA, 2020b). Note that while the baseline Event Set used Climate Forecast System Reanalysis data (Saha *et al.*, 2010), temperature and precipitation data CNRM-CM5 (Voldoire et al., 2013) for the future Event Set on the basis that this particular model ranked well in comparison to CFSR for the same region. The change in return period associated with annual maximum rainfall and runoff under the different climate scenarios compared to the present-day, are derived for pluvial and fluvial flooding per OP, respectively. These change factors are then applied to the fluvial and pluvial event set (JBA, 2020b).

To model the effect of climate change on wind speed, we scale the sampled wind speed with the projected change in maximum wind speed for the North Atlantic Cyclone Basin at the end of the century (see Knutson *et al.*, 2020). It is assumed that the co-dependency between flood depth and wind speed during cyclones will stay the same under climate change conditions, due to the difficulty in predicting such changing relationships.

4.2.2.2. Constructing the water network and supply model

National water supply infrastructure is typically made up of multiple systems that serve a particular population or demand zone. Each system is, in turn, made up of a set of assets, typically including a source (i.e., a well or river intake), treatment plants, pumping stations, pipelines, storage tanks. We collected an asset-level database of all (1,208) potable assets, represented as points in space, owned and operated by the national utility in Jamaica, the NWC (National Spatial Management Branch, 2010), as shown in Figure 11.

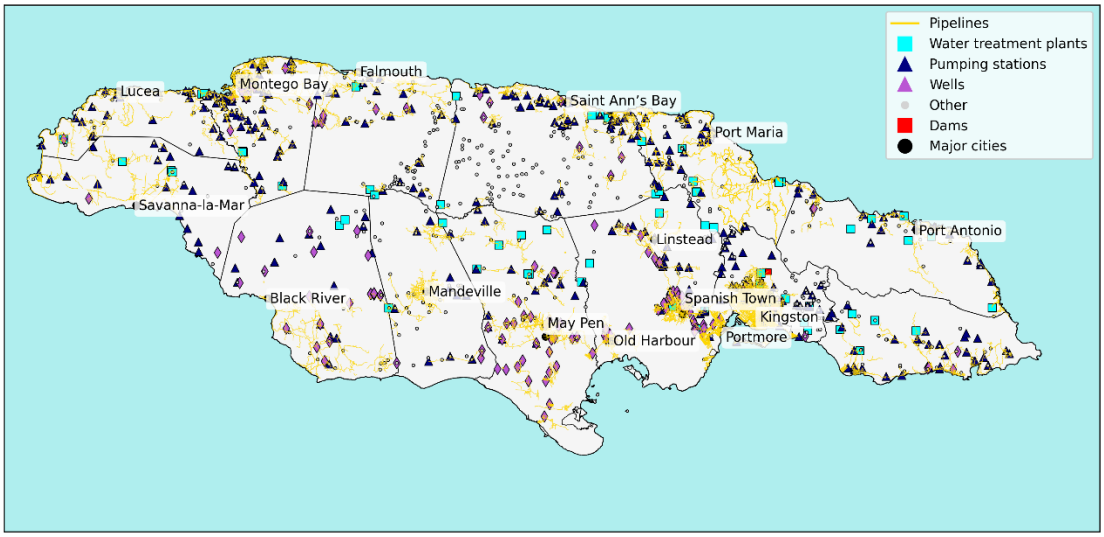


Figure 11 Potable asset data comprising 129 plants, 409 pumping stations, 169 wells and 2 dams that are owned and operated by the NWC Utility in Jamaica, the NWC (National Spatial Management Branch, 2010). The locations of major city locations are also provided (OpenStreetMap, 2018).

To estimate the population served per asset – and thus the number of people to whom water supply would be disrupted if a particular asset is damaged or does not have water – each asset should be assigned to the demand zone it serves, followed by a disaggregation of the population served by assets within each system. This disaggregation is accomplished based on the type of asset (source or sink, and type of source and sink asset) in the network. A more detailed description of the system representation of the infrastructure network and the coupling to demand zones and population is provided in Supplementary Text 2. A plot of the population served per asset and allocation to local system boundaries in Supplementary Figure B10. Network effects, or cascading effects, can be modelled whereby customers served by assets (sink assets) that are, in turn, served by upstream assets (source assets), are affected by disruptions to either of the assets. If multiple assets are simultaneously disrupted by an event, we consider the union of the affected areas to avoid double-counting. This follows-on from previous studies that account for interdependence between assets in a network when quantifying the extent of disruptions (Thacker, Pant and Hall, 2017).

4.2.2.3. Estimating impacts and quantifying risk

Asset damage-induced Customer Disruption Days

The level of damage to infrastructure assets is estimated by applying damage curves to the modelled flood depth and wind speed (Porter, 2021). Water asset-specific damage curves for flooding are obtained from (FEMA, 2020) and for extreme winds from existing sources

(Miyamoto International, 2019). For flooding, we use different damage curves for water treatment plants and pumping station provided (FEMA, 2020). The pumping station damage function is also applied to wells, intakes and pumping stations, whereas the water treatment plant damage curve is applied to only to water treatment plants. This should partially account for the different responses of different asset types to flood hazard severity: with pumping stations typically being more vulnerable to damage than water treatment plants for a given level of flood inundation. However, for wind, only one damage curve describing the repone of water treatment plants to wind speed was found (Miyamoto International, 2019) and applied to all the different asset types. The curves for water treatment plants with respect to flood inundation and wind speed curve are presented in Supplementary Figure B11, which also shows that we have considered a ±30% uncertainty range in the single curve estimates.

For physical asset failure-related disruptions, the Customer Disruption Days (CDD) is estimated by quantifying the number of days an asset is out of service multiplied by the number of customers impacted by that outage. The number of customers impacted is assumed to be equivalent to the number of customers the asset serves, assuming no redundancy or capacity for substitution between assets in a system (National Water Commission, 2020b). This assumption was based on a dataset of actual utility disruptions provided over the past 5 years, which highlighted which communities that be disrupted due to a given facility outage. By examining this database, we found that the redundancy in the water supply network in Jamaica is very low and, if we were to assume that assets connected via the piped network can substitute for each other, we would significantly overestimate the actual level of redundancy in the system. We therefore took the conservative assumption that there is no redundancy in the network, which was subsequently confirmed to be realistic with stakeholders. In contexts with more redundancy in the system, this simplifying assumption would have to be modified and more complex models of system dynamics would be required.

Flood and extreme wind events are associated with an intensity value (i.e., wind speed or flood depth) from which we derive the extent of damage using damage curves. The recovery period (i.e., amount of time that the asset would be out of operation) depends on the extent of damage and the speed of repair. This is estimated based on facility outage data provided by the utility which indicated that a relatively minor flood event (≤10% asset damage) would result in a day-long disruption, a more severe event (>30% asset damage) would result in a week-long disruption and a very extreme event may disrupt an asset for up to a month. For a given change

in the extent of asset damage, we derive the recovery duration by linearly interpolating between each of these thresholds. Such recovery durations are in line with commonly used restoration curves used in risk modelling (FEMA, 2020). The disruption duration (t_{ij}) caused by a given asset damage-induced event (i) is multiplied with the number of customers (c_j) served by the damaged asset (j) resulting in the number of CDD caused by asset damage-induced disruptions (Equation 9).

$$CDD_{ij} = t_{ij}\, c_j \qquad (9)$$

Water shortage-induced Customer Disruption Days

To determine where variations in surface water supply result in water shortages for customers, we intersect the supply estimates ('Spatially coherent drought events') against a drought threshold for each abstraction point. The drought threshold (DT_k) per surface water abstraction asset (k) is the sum of the volume of demand (d_k), leakage losses (l_k) and environmental flow threshold of Q10 of the streamflow input (s_k), i.e., the flow that is exceeded 90% of the time, to the abstraction point (NGPRP, 1974) (Equation 10):

$$DT_k = d_k + l_k + s_k \qquad (10)$$

A drought event can be defined as occurring when a single month (m), or several months in a year, have a positive hydrological deficit (HD) (Equation 11):

$$HD_{y,m,j} = min\left(s_{y,m,j} - DT_j\,(-1), 0\right) \qquad (11)$$

The drought threshold (DT_j) for each reservoir is estimated by matching the timings of disruptions in the receiving potable supply systems with the observed reservoir level variations.

Thus, a drought event affecting reservoir-fed systems can be defined as occurring when a single month (m), or several months in a year, experience a drop in reservoir levels $l_{y,m,j}$ relative to the DT_j (Equation 12):

$$HD_{y,m,j} = l_{y,m,j} - DT_j\,(-1) \qquad (12)$$

For the drought impact analysis, it is assumed that the fraction of required supply that is not met during a given month directly translates into a disruption to the fraction of the customers supplied by a given source. This disruption is assumed to be evenly spread across a given month such that it may be converted to a daily estimate and communicated in terms of CDD. This reflects that utility operators would likely reduce the daily time window for supply when a

deficit occurs rather than cutting water supply to household throughout the duration of a drought. Thus, drought events (i) can be characterised in terms of their duration (δ_i) (Equation 13) and total accumulated drought deficit (AD_i) (Equation 14) over the duration of the event (in months), per supply source (j):

$$\delta_{i,j} = \sum_{m=1}^{12} 1_{HD_{y,m,j}} \qquad (13)$$

Where $1_{HD_{y,m,j}}$ is an indicator function which is equal to 1 if $HD_{y,m,j} > 0$ otherwise it is 0.

$$AD_i = \sum_{m=1}^{12} HD_{y,m,j} \qquad (14)$$

Thus, monthly CDD can be derived by applying the fraction of unmet demand to the population served per system, multiplied by the number of days per month (d_m), per supply source (j) (Equation 15):

$$CDD_{ws}(i,j) = \frac{AD_{i,j}}{D_{i,j}\,\delta_{i,j}} C_j d_m \qquad (15)$$

Risk formulation

The simulation-based method described above yields a large sample of disruption events, which are ranked to yield an empirical distribution of losses, from which CDD estimates at different return periods were extracted. The expected annual CDD_{av} was obtained by integrating integrating the inverse cumulative probability function of the disruption, $QCDD$, over the annual exceedance probabilities of losses using quadrature, Equations 16 and 17.

$$p_i = \frac{1}{RP_i} \qquad (16)$$

$$CDD_{av,j} = \int_0^1 QCDD_j(p)dp \qquad (17)$$

The same process was undertaken for events simulated with and without the effects of climate change scenarios. These climate change scenarios should be interpreted as stress tests of the current system with different climatic conditions, and not as comprehensive future scenarios, as the effects of aging infrastructure on increasing leakage rates or land use changes are not considered in the climate stress tests.

4.3. Results

4.3.1. Utility-scale risk in terms of CDD

The distribution of computed CDD is presented in Figure 12. For water shortage risk (a, left in a 10 year return period event amounts to about 16 million CDD, which is equivalent to 2% of a maximum annual disruption of the full customer base. The climate change stress tests increase the expected impact in a 10 year event to 18 million (2.5%) (RCP2.6 and 4.5) while in RCP8.5 the CDD are projected to increase to 22 million (3%) in that event. One can see that asset damages induce a far greater tail risk (b, right) (i.e., a relatively greater risk during low probability events), whereas water shortages induce more frequent disruptions (i.e., a relatively greater risk during high frequency events). The maximum total potential disruption imposed by asset damages is far greater than for water shortages, because when an asset is damaged, the entire customer base that it serves is cut off from its normal supply, whereas for water shortages, disruptions could manifest through intermittent water shortages over a longer time period, though the most severe droughts lead to total inability to supply. Indeed, asset damage disruptions induce ~40 million CDD (5%) during a 1 in 200-year event, whereas, a water shortage of the same likelihood would induce to ~20 million CDD (2.5%). However, drought-induced disruptions are associated with a far higher probability. Indeed, even under baseline conditions, water shortage-induced disruptions occur approximately once every 3 years, which increases to a drought event occurring every year. A description on the uncertainty analysis used to estimate a likely upper and lower bound for the results (the shaded area in the plots) is presented in Supplementary Text B3. The frequency with which different asset types are exposed to each of the considered hazards and the spatial distribution of exposure per asset per hazard for a range of return periods can be found in the Supplementary Figure B12 and B13, respectively. The results are most sensitive to the assumed vulnerability functions for all hazards. This is illustrated by the uncertainty upper and lower bounds, which are derived from adjusted the drought threshold (i.e., the sum of demand, leakage and environment flow volume per system) and the fragility curve (i.e., the amount of damage incurred by a given flood depth or wind speed per asset) by a common factor of +/-30% (Supplementary Text B3: Uncertainty analysis).

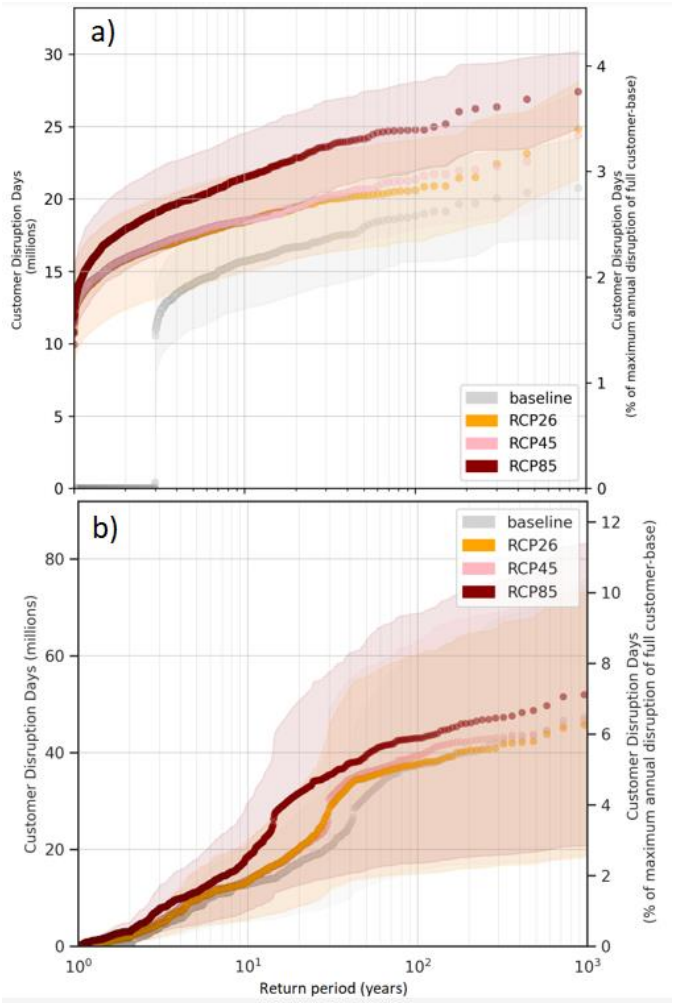


Figure 12 Customer Disruption Days (CDD) in millions and as a % of the maximum potential annual NWC customer-base disruption for water shortage (top, a) and asset damage (bottom, b) risks in Jamaica under baseline climate conditions (light grey) and end of century RCP2.6 (orange), 4.5 (pink) and 8.5 (dark red) up to a 1 in 1000-year event.

By integrating the distribution of losses over all events, we found the baseline total annual expected disruption to be ~10 million CDD per year (1.25% of the maximum potential annual NWC customer-base disruption) (Figure 13), equally apportioned between water shortage-induced and asset damage-induced disruptions. This is equivalent to roughly a tenth of the utility's customer base being disrupted once per year. This increases substantially under future conditions by the end of the century: under both RCP2.6 and 4.5, the annual expected disruption is equivalent to ~20 million CDD (2.5%), rising to over 25 million CDD under RCP8.5 where water shortages contribute more overall and wind damages contribute more to asset damage-related disruption risk. We note that in both the water shortage and asset damage risk analysis, there is a non-linear response between emissions and impacts: despite RCP2.6 being a low-emissions scenario compared to RCP4.5, there is little difference between impacts. This is driven by the hazard data inputs. The strength of the climate signal in the future annual

expected water shortage risk is because water shortages incur more CDD in the lower return period range of the risk curve compared to asset damages, which means that the annual expected disruption is more sensitive to changes; on the other hand, the increase in CDD during extreme water shortages and asset damages is roughly equivalent.

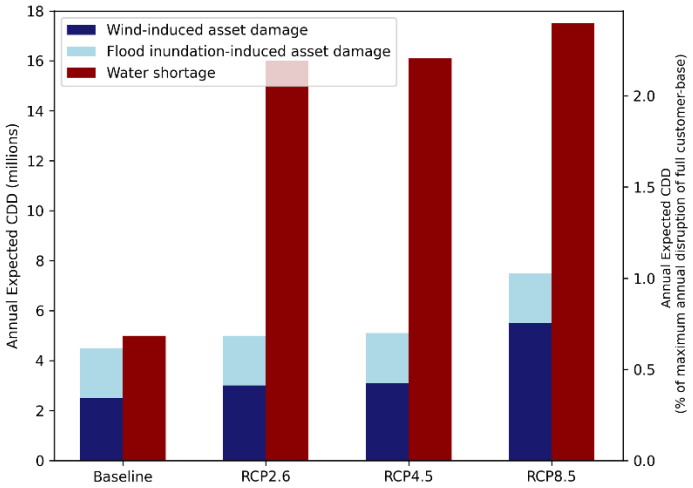
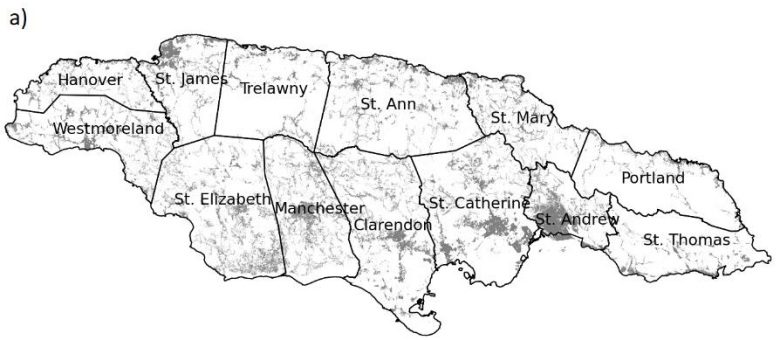


Figure 13 Annual expected CDD per climate change scenario and per hazard driver in CDD and as a % of the maximum potential annual NWC customer-base disruption: cyclone wind (dark blue), non-cyclone related flood (light blue) and water shortages (dark red).

4.3.2. Spatial correlation of CDD

We find there is considerable correlation in disruption co-occurrence across different parishes. Figure 14 shows cross-correlation plots for disruption incidence for large-scale droughts and flood or extreme wind-induced asset damage events. Note that this analysis can also be conducted at the system level, but parish-level aggregation is used for the purpose of plot simplicity. Supplementary Figure B14 also provides a spatial illustration of the mean correlation coefficient between each parish and all others.



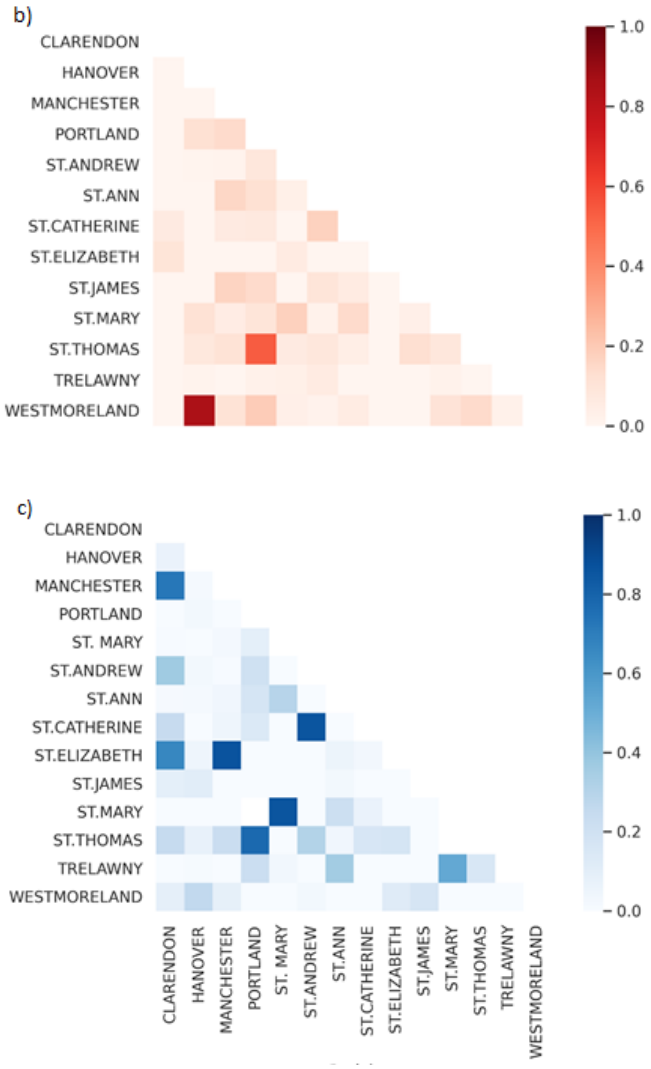


Figure 14 Likelihood of a water supply disruption in affecting two parishes simultaneously, (a) outlines the parish locations and distribution of households (OpenStreetMap, 2018). The correlation between parish-level CDD calculated as the correlation coefficient between the fraction of total parish customers disrupted multiplied by the days of disruption for water shortage events (red, b) and asset damage events (blue, c).

Because here we only consider surface water systems when calculating the risk of water shortages, and different parishes receive different fractions of total supply from surface water, this has a strong influence of the parish's vulnerability to modelled disruptions. For example, Hanover and Westmoreland are neighbouring parishes and are both highly dependent of surface water for potable supply. As would be expected, the pairs parishes that are most likely to undergo a simultaneous disruption – St. Catherine and St. Mary, St. Thomas and Portland and Westmoreland and Hanover – neighbour each other. The pairs of parishes that appear to be more resilient to simultaneous water shortages despite neighbouring each other have greater buffering capacity during droughts, for example through major surface water storage capacity

or greater portions of potable supply from groundwater. For example, the relatively low correlation between disruptions in Kingston and St. Andrew (here labelled St. Andrew) and St. Catherine is probably thanks to the buffering effects of the Hermitage and Mona reservoirs. Asset damage- and water shortage-induced disruptions in St Thomas and Portland parishes are both strongly correlated. This is because 1) they share a water source (Windsor Castle River), so droughts affecting this source will likely affect systems in both counties, 2) rivers in both counties originate in the Blue Mountain range, which would contribute to flood risk in the downstream, low-lying regions of both counties and 3) their geographical proximity would also mean that assets in both counties would be affected by the same cyclone events and of a similar hazard severity. Other pairs of parishes are not as hydrologically connected in terms of flood and drought risk.

Disruptions due to extreme asset damage-related events assets are found to be more widespread, which is to be expected because all systems are exposed and because wind damages during cyclones are likely to impact a greater spatial extent than the disruptions imposed by hydrological extremes like droughts and floods. Pairs of parishes that are most likely to experience simultaneous disruption include St. Elizabeth and Manchester, Portland and St. Thomas and St. Andrew and St. Catherine. Flood inundation from Black River, Yallahs and Rio Cobre rivers are likely to cause damages to assets belonging to potable systems in these pairs of parishes, respectively.

4.3.3. System level annual expected CDD and event duration

Figure 15 shows the annual expected population disrupted at the system level. Water shortages pose the greatest risk to systems with critical river-fed treatment plants that serve a large population, such as Logwood, Moravia, Annotto Bay, Spanish Town, Rio Cobre, Great River and Constant Spring potable supply systems. For example, Moravia has an annual expected CDD of ~60,000 and an annual expected drought duration of 3 months (Figure 15). Note that, if a 60,000 CDD event occurred within 3 months, this would result in an equivalent daily disruption rate of ~650 people during the course of the water shortage. Risk is high in these systems both because of the variability of the river systems supplying these systems and the water loss through leakage in these systems. For example, leakage loss from the Great River is roughly 60% of the maximum supply capacity of the Great River treatment plant. Systems that are most prone to asset damages include Mandeville, Rio Cobre, Spanish Town, Portmore and Annotto Bay, which contain assets that are prone to damage, are exposed to wind and flood

inundation and serve a large customer base. When combined, it becomes clear that the Moravia, Mandeville, Rio Cobre, Spanish Town, Portmore, Constant Spring and Annotto Bay systems are most at risk.

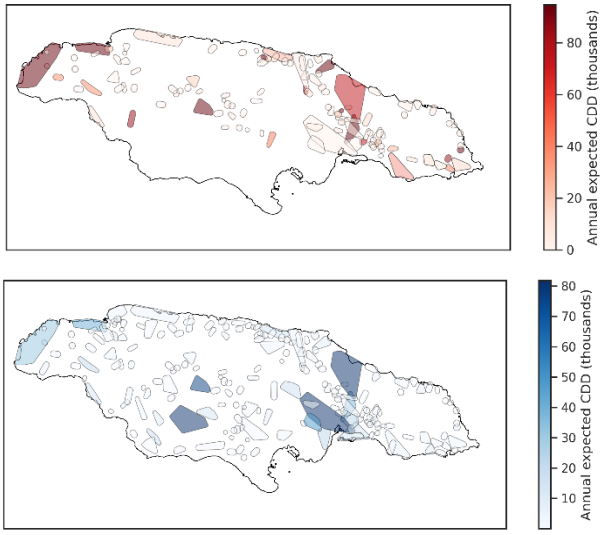


Figure 15 Annual expected CDD per system induced by water shortages (left) and asset damages (right). The baseline climate scenario output is shown as the spatial pattern is similar across climate scenarios.

We also provide the annual expected water shortage and asset damage duration estimates per system (Figure 16). We also see that the pattern of event duration does not necessarily match drought risk in terms of CDD, highlighting the combined effects of exposure (population served per system), event intensity and duration that are communicated via the CDD metric. Supplementary Figure B15 provides the overall distribution of annual expected water shortage and asset damage event durations per climate scenario.

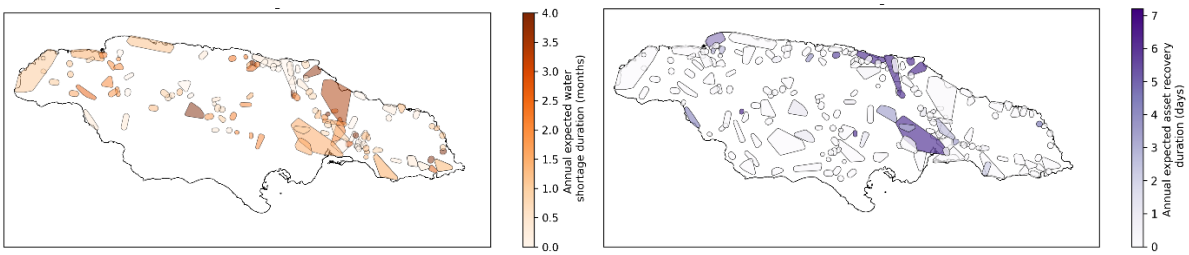


Figure 16 Annual expected drought-induced water shortage duration in months under baseline climate scenario (left) and asset damage disruption duration in days under baseline climate scenario (right).

4.4. Conclusions and Discussion

In this paper we have proposed and demonstrated a new framework for quantifying the impact of present and future climate extremes on national water supply utilities, applied in Jamaica. The framework integrates a large set of spatially coherent drought, cyclone and pluvial and fluvial flood events, combining observed events with synthetic events. The water utility's assets (pumping stations, treatment works, etc.) are combined in a system model, which also represented water supply to municipal water users, leakage and major storage capacities in the supply network. For each disruptive event, the number of water users impacted per day over the duration of the event, or the Customer Disruption Days (CDD), is computed.

In the application in Jamaica, we found that the annual expected disruption in the present day is equivalent to an average customer disruption of 5 days per year. Stress tests with future climate scenarios illustrate how this risk could double under RCP2.6 and 4.5 scenarios and by 2.5 under the RCP8.5 scenario by end of century. The division between CDD driven by water shortages (droughts) and asset damages (cyclones and pluvial and fluvial flooding) shows that for the most extreme events, asset damages induce more customer disruptions. Asset damage disruptions induce between 40 million CDD during a 1 in 200-year event across the considered climate change scenarios, compared to 20 million CDD for disruptions associated with water shortages. However, water shortages are a far more chronic problem in Jamaica, which occur not only as a result of hydrological extremes, but also due to the high rates of water loss through leakage. Indeed, even under baseline conditions, water shortage-induced disruptions occur roughly every 1 in 3 years, which is projected to rise to a supply interruption occurring every year across all considered scenarios of climate change.

The distinction between water shortage- and asset damage-driven disruptions provides decision-relevant insights into the types of adaptation infrastructure required for different systems. For example, for water shortages, leakage reduction would be a highly relevant adaptation intervention, whereas for asset damages, flood protection barriers and other interventions that would protect assets from extreme winds and flood inundation would be relevant.

The CDD metric provides a unifying metric of the overall magnitude of disruptions caused by the different drivers and enables comparison of short-lived wide-scale events (e.g., flooding during intense hurricane rainfall) with longer-term events which may just hit a small proportion of the population (e.g. low flows confined to one catchment). We recognise that, through this

unification, we lose detail on the multiple dimensions (duration versus intensity) in which a disruption propagates. This, in turn, prevents us from exploring the adaptive actions taken by utilities or households to mitigate disruptions during droughts, floods or storms, e.g., conservation behaviours or demand management strategies. Our results should therefore be regarded as an upper bound on the CDDs that may be expected. It also prevents us from exploring the tolerability of a given disruption. For example, households may be more willing to tolerate prolonged intermittent disruptions for water shortages, compared to total loss of supply from asset damages for a shorter period. Further work is required to look into the tolerability of different types of disruption and adaptation strategies in terms of their impacts of people, including those in low-income groups. Further work is also required to quantify and compare the financial consequences of climatic events to, for example, undertake cost benefit analysis on adaptive infrastructures. Asset repair works after flooding may be more costly for utilities than the financial losses incurred during disruptions through lost tariff revenue or the provisioning of alternative supplies (e.g., tanker trucks in the case of Jamaica).

Our proposed framework accounts for the spatial relationship between disruptions across space and between hazards, which not only provides a more accurate aggregate risk calculation, but also enables the utility to quantify the likelihood and the spatial pattern of widespread disruptions. This can be used to prepare sufficient emergency supply capacity (Hall and Borgomeo, 2013), as well as to explore potential adaptation options that promote resilience between multiple systems, such as inter-system transfers (Zhang *et al.*, 2018).

Although the overarching steps of our proposed framework are generally applicable, applications to other water utilities would require consideration of the performance of proposed flood, cyclone and drought hazard modelling approaches, and a strong contextual understanding through engagement with relevant stakeholders. In future applications, the analysis would be significantly improved by incorporating groundwater systems for the water shortage risk component and by utilising fragility and recovery curves that are country-specific for the asset damage risk component. Further, here we assume that all customers are served with water supplies at all times. However, in reality, this is often not the case. In subsequent applications, the uneven levels of intermittent supply that exist across water supply networks, even without the onset of climatic events, could be incorporated into the CDD metric. There is also scope to expand or tailor the range of hazards considered in this framework for different applications, such as coastal flooding, landslides and extreme cold.

Despite the limitations of the proposed framework, the overarching framework presented here is a useful step forward in the study of multi-hazard risks to water supply infrastructure. It also provides a platform for 1) promoting the need for investment in climate adaptation projects in the potable water sector and 2) prioritising strategic adaptation options across asset protection and drought mitigation projects, to enhance water supply system resilience in the face of a changing climate.

Acknowledgments

The authors declare no conflict of interest. O.B. acknowledges funding from the Engineering and Physical Sciences Research Council (EPSRC) under grant number EP/T517811/1.

The authors would like to thank the National Water Commission for both the data used in this work and for the meetings and ongoing communication throughout this work. We would also like to thank the Systemic Risk Assessment Team and the Planning Institute of Jamaica for supporting and coordinating this data exchange and ongoing collaboration between the Oxford team and the National Water Commission. We would also like to acknowledge JBA Consulting for the provision of flood risk maps and events sets, the University of the West Indies for the regional climate model output and the Jamaican Water Resources Authority for the streamflow data.

Open Research

ERA5 temperature, precipitation and wind data can be downloaded from the Copernicus data portal, https://cdsportal.copernicus.eu/ (registration required). All Jamaica-specific streamflow data can be downloaded from the Jamaican Water Resources Authority website, https://webmapjam.dyndns.pro/webmap/app/db/index.php (guest login provided). Data provided by colleagues at the National Water Commission - including the potable asset database, Water Supply Parish Plans, reservoir volume variations and historic drought disruption data - is sensitive information. Other users much contact the utility (https://www.nwcjamaica.com/) or the National Spatial Management Branch in Jamaica (https://jis.gov.jm/government/agencies/national-spatial-data-management-division/) to access this data. The climate model rainfall and temperature outputs were provided by the University of the West Indies Climate Group for the current project, https://www.mona.uwi.edu/physics/csgm/home and can be downloaded

here https://github.com/ICTP/RegCM or https://doi.org/10.5281/zenodo.4603556 (Filippo *et al.*, 2021). The synthetic streamflow generator used can be retrieved here https://github.com/kevingwheeler/SimulatedAnnealing. Derived data from the climate-conditioned synthetic streamflow workflow is provided on GitHub, https://github.com/obech/multi-hazard_risk_water_supply.git. Flood data supporting this research were made available by colleagues from JBA risk consulting (JBA, 2020a) under a license which restricts us from distributing the data to the public or research community. Please visit https://www.jbarisk.com/flood-services/maps-and-analytics/global-flood-maps/ for instructions on how users can gain access. Software used for the project include Python (https://www.python.org/) and QGIS (https://www.qgis.org/en/site/), both of which are open-access. The code for performing the main analysis and constructing the figures is available on GitHub, https://github.com/obech/multi-hazard_risk_water_supply.git.

5. Prioritising climate adaptation options to minimise financial and distributional impacts of water supply disruptions

Chapter 5 presents a framework for evaluating the effectiveness of climate adaptation options in reducing water supply infrastructure damages and utility customer spending on substituting disrupted services. The framework is applied to the case of the National Water Commission's system in Jamaica.

Paper: Becher, Verschuur, Pant and Hall (in review with *Environmental Research: Infrastructure and Sustainability* [manuscript status is 'Accept in Principle', conditional on meeting editorial requests]), Prioritising climate adaptation options to minimise financial and distributional impacts of water supply disruptions.

Co-author contributions: Prof Jim Hall from the University of Oxford assisted with the conceptualisation and structuring of the paper, as well as with review and feedback on the manuscript. Dr Jasper Verschuur and Dr Raghav Pant from the University of Oxford assisted with review and feedback on the manuscript.

Key points:

- Quantification of financial impacts of water utility failure for households provides the basis for multi-hazard, customer-oriented investment planning.
- In the case study in Jamaica, investment in leakage reduction tends to be more cost-effective for mitigating alternative water purchase losses during droughts.
- Investments in flood-proofing is most cost-effective in reducing climate risks to the water utility's assets.
- Leakage reduction programmes can be relied upon in most systems to deliver water security benefits, whilst flood protection needs to be carefully targeted.

Abstract

Climate-related disruptions to water supply infrastructure services incur direct financial losses to utilities (e.g., to repair damaged assets) and externalise a societal cost to domestic customers due to additional costs that they may incur e.g., to acquire water from alternative sources. The latter often represents an uncompensated social burden, which should be properly accounted for in investment planning. Here we present a new framework for quantifying the direct financial risks burdened by utilities and alternative water purchase losses incurred by domestic customers (including those in lowest income groups) during flood- and drought-induced utility water supply disruptions. This framework allows us to directly compare the benefit cost ratios of a large portfolio of asset protection and leakage reduction across supply systems for the island of Jamaica. A system-level optioneering analysis allowed us to identify the optimal adaptation option per system. We find that 34% of systems would benefit from flood defences and 53% would benefit from leakage reduction to adaptation to droughts. The cumulative annual benefit that could be achieved by implementing all system optimised adaptation options is estimated to be 720 million Jamaican dollars per year, representing a substantial saving for the utility and its customers, including low-income customers. We identify options that offer strong synergies between economic and equity (reduced cost of alternative water) objectives for both forms of adaptation. The proposed framework is established to support the business case for climate adaptation in the potable water utility sector and to prioritise flood and drought mitigation options. We take a first step towards mainstreaming equity considerations in potable sector optioneering by estimating the contribution of adaptation targeted towards reducing household costs for low-income customers.

5.1. Introduction

Formally operated and regulated water supply utility services offer a level of quality and affordability assurance that is not necessarily guaranteed by informal markets or, other unregulated supplies (WHO, 2014). The safety and affordability of water supplies are key components of Sustainable Development Goal (SDG) for clean water and sanitation (SDG6), highlighting the importance of utility operated water infrastructure for overall sustainable development (UN, 2021). However, many utilities in low- and middle-income countries are caught in a vicious cycle of disrupted services, non-revenue water and a low rate of reinvestment in infrastructure resilience (Rouse, 2013; Libey, Adank and Thomas, 2020). This has contributed towards a large, and widening, service gap in many countries (Hutton and Varughese, 2016; Rozenberg and Fay, 2019; OECD, 2022). Water supply systems in several low- and middle-income countries are also exposed and vulnerable to climate-related disruptions and climate change (Mirza, 2003a). Utilities in these countries have smaller budgets for emergency water supply deployments to affected communities during disruptions and lower creditworthiness, hindering their ability to attract loans for long-term infrastructure investments (Winpenny, 2015). Further, low income water utility customers suffer the most from disrupted services (Robak and Bjornlund, 2018), as they are less able to acquire water from other safe, alternative sources (Mitlin and Walnycki, 2020).

During climate shocks, utilities may suffer financial losses, for example, to repair assets after flood damage (Laucelli and Giustolisi, 2015; Pant *et al.*, 2018; Zorn *et al.*, 2020). However, the impacts of such failures are not only borne by utility companies, but often spill-over to utility customers (Altaf, 1994; Baisa *et al.*, 2010; Heflin, Jensen and Miller, 2014; Bakker, 2017). This is because, unless utilities have compulsory contingency and compensation plans in-place to replace disrupted services, dependent households are forced to acquire water from elsewhere during supply disruptions. Other sources of water to which households may turn to during disruptions include unregulated sources such as springs and wells, bottled water and utility tanker trucks, the latter being a key agent in supplying water during utility supply disruptions globally (Wutich, Beresford and Carvajal, 2016; Garrick, O'Donnell, *et al.*, 2019; Zozmann *et al.*, 2022). These externalised costs are often far beyond what the relative cost of operating functioning piped water supply system would be for the utility. For example, in Ghana, water distribution by road tanker costs 14 times as much as distribution through the pipe networks (Rouse, 2013). This may represent a significant additional household expense (i.e., beyond their regular tariff payments) for low-income households (Andres *et al.*, 2021).

Water supply utilities typically conduct cost-benefit analysis on new supply project proposals to attract funding from relevant investors (e.g., governments or multilateral development banks). This generally involves comparing the additional revenue that would be collected through tariffs over a new project's lifetime to the cost of its construction, operation, and maintenance. However, this does not account for the other benefits of projects that improve the overall resilience of potable water supply services to climate shocks now and in the future. For example, reducing the costs of repairing physical asset damages (due to extreme winds and/or flood inundation) (Pant *et al.*, 2018) or reducing the frequency with which households must acquire water from paid sources outside utility services (e.g., bottled water (Heflin, Jensen and Miller, 2014) or tanker trucks (Garrick, O'Donnell, *et al.*, 2019)). This means that the focus of new utility investments can become skewed towards expanding network coverage and away from projects that improve the reliability of services for existing customers (Altaf, 1994). This is problematic because intermittent supplies, including climate-induced service disruptions, can significantly undermine efforts towards water supply security (Griffin and Mjelde, 2000; Beard and Mitlin, 2021; Grasham *et al.*, 2022). Further, in the context of combined national development and climate adaptation objectives, the identification of 'pro-poor' investment strategies are increasingly called-for (Sohail *et al.*, 2007; Nagpal, Malik and Eldridg, 2018). However, despite the importance of mainstreaming equity considerations in risk analysis in the potable water sector (Grasham, Charles and Abdi, 2022), to the authors' knowledge, no decision-making framework exists to investigate synergies and trade-offs between climate adaptation and equity in this sector.

Risk-based decision making is a valuable method for the prioritisation of climate adaptation options in potable water networks (Hall and Borgomeo, 2013). To account for the uncertainty associated with quantifying the risk of extreme climatic events, now and in the future, decision-making under uncertainty approaches have been developed (Gober, 2014; Larson *et al.*, 2015; Jensen and Wu, 2016). One approach involves stress-testing a water supply system to a wide range of, climate-conditioned, synthetic events, simulating the impacts in terms of one or multiple relevant risk metrics (Brown *et al.*, 2012), e.g., Customer Disruption Days (CDD) (Becher *et al.*, 2023) or economic costs (Borgomeo *et al.*, 2018; Jenkins *et al.*, 2021). Iterating through the same calculation with the addition of one of multiple new adaptation options, the difference between the risk with and without any given option/portfolio of options is quantified as the option's benefit. Techniques are applied to optimise a given adaptation pathway across a long-list of options based on the priorities assumed to be most relevant for decision-makers

(Matrosov, Padula and Harou, 2013). This has been widely applied in various forms in research (Hall *et al.*, 2012; Matrosov, Woods and Harou, 2013; Borgomeo *et al.*, 2015, 2018; Herman *et al.*, 2016; Huskova *et al.*, 2016; Guo, Westra and Maier, 2018) as well as in policy and practice (e.g., in the UK for water supply planning (Hall *et al.*, 2012)). Hereafter we refer to this prioritisation process as optioneering, a term employed by practitioners to describe the practice of comparing alternatives to identify the preferred option for a given context (Mott MacDonald, 2022). While these approaches have been creatively advanced by researchers and widely applied by practitioners, to the authors' knowledge, no optioneering frameworks have been developed for the potable water sector that considers the risk reduction benefit of options across multiple climate hazards (e.g., droughts, floods). This would be valuable in contexts where utilities must develop adaptation action plans to respond to the threat of multiple hazards.

Climate resilient water supply utilities play a key role in maintaining affordable water prices and protecting poor populations from harmful climate extremes, both of which are SDG targets (UN, 2021). However, these benefits are not the focus of existing studies on water supply utility adaptation planning. Capturing the impacts of water supply disruptions to customers, in the context of distribution effects across income groups, would offer another dimension for decision-makers when comparing alternative adaptation options. This can be achieved by taking inspiration from two fields of literature that have not been previously integrated with the optioneering literature. 1) Literature that explores the economic, financial and welfare impacts of household water supply disruptions (Altaf, 1994; Baisa et al., 2010; Freire-González, Decker and Hall, 2017; Roibas, Garcia-Valiñas and Fernandez-Llera, 2019; Jenkins et al., 2021), including through the costly replacement from private tanker trucks (Bharti et al., 2020; Yoon et al., 2021). Economic and welfare impacts to customers can be captured through interviews and relatively small-scale surveys (Heflin, Jensen and Miller, 2014; Selelo et al., 2017; Sjöstrand et al., 2021). Such research is typically removed from the literature on climate risk to water supply systems, but rather focusses on non-climate related service intermittency (Grasham *et al.*, 2022). National scale studies have focussed on GDP losses across a range of economic sectors caused by water supply disruptions (Jenkins et al., 2021), however, there is limited research that estimates the costs of climate-related water supply disruptions to customers at the national scale. 2) Literature dedicated to the distributional impacts of climate extremes (Hallegatte et al., 2016; Hallegatte and Rozenberg, 2017). This literature places focus on the disproportionate welfare impacts burdened by lower income households, as they have a relatively lower financial capacity to cope and recover from shocks, resulting in a so-called

poverty trap (Borgomeo, Hall and Salehin, 2018). This can be obscured by research that focusses only on the economic costs of extremes, as higher income households typically have higher asset values at risk to shocks (Hallegatte et al., 2017). Although a major channel through which climate extremes impact households, including in vulnerable communities, is via disruptions to critical infrastructure services (e.g., transport, electricity and water supply), this has received far less attention (Verschuur *et al.*, 2023). There is scope to approach the traditional adaptation infrastructure optimisation problem from the perspective of the distributional costs of water supply disruptions to households.

Here, we present a framework that combines the multiple channels through which climate-induced water supply disruptions incur direct financial losses to utilities and to their customers. By stress-testing a model representation of Jamaica's potable water supply network to a large synthetic event set, comprising of both flood and drought events, the annual expected risk in terms of direct losses per water supply system is calculated. Direct losses considered include the losses incurred by utilities and the alternative water purchase losses incurred by domestic customers. Here, we define utility losses as flood-induced damages to the utility's asset-base, and alternative water purchase losses as the costs of replacing disrupted supply from tanker trucks. By repeating this process with and without the integration of a range of infrastructure interventions, comprising flood defence walls and leakage reduction strategies, we quantify the effect that each intervention has on the annual expected losses. The benefit of a range of adaptation options is, thus, quantified in terms of the reduced direct cost of disruptions. This is compared against each option's cost to compare options based on their benefit-cost ratios (BCRs). By estimating the reduction in alternative water purchase losses incurred by customers in poverty provided by each intervention, we identify 'pro-poor' adaptation strategies. The proposed framework quantifies the climate-related risk mitigated by adaptation projects to promote the business case for climate adaptation and enables optioneering analyses with respect to both economic objectives and 'pro-poor' agendas.

The proposed optioneering framework is applied to the national water utility in Jamaica, although it is applicable to any national infrastructure setting with adequate data availability. The National Water Commission (NWC) of Jamaica is a state-owned utility that supplies drinking water to 90% of Jamaica's total potable water supply to residential and non-residential customers (National Water Commission, 2020a). Jamaica's water infrastructure is intensely vulnerable to damage from flooding (Mandal and Maharaj, 2013; Burgess *et al.*, 2015; Glas *et*

al., 2015; Mandal *et al.*, 2016; Nandi *et al.*, 2016) and droughts (Setegn *et al.*, 2014; Goyal, Madramootoo and Richards, 2015). During climate-induced disruptions, customers typically turn to other sources, such as tanker trucks or untreated well or spring supply (Lester, 2015). There are several health risks associated with water stored in household tanks (ibid) and unregulated sources, such as wells, which are becoming increasing contaminated by untreated sewage (Mandal *et al.*, 2011; Jamaica Water Resources Authority, 2020). Therefore, tanker trucked water offers an essential alternative replacement service during utility water supply disruptions, to protect the health of disrupted communities.

The NWC aims to significantly improve system reliability over the coming decade (for example, by reducing intermittent supply) (NWC, 2015) and is an essential pillar of national development and poverty alleviation by providing high quality, affordable potable water services to customers (JIS, 2008; MEGJC, 2019). However, the capacity for the utility to invest in better services is hindered by a high level of debt, with the highest debt to equity (leverage) ratio in the Caribbean (Burdescu *et al.*, 2020). In this context, it is essential that the adaptation strategy that the utility adopts is as cost-efficient as possible. This situation is comparable to utilities in many other low and middle-income countries that are trapped in a cycle of disrupted services, low-cost recovery and low adaptive capacity. An important but often unrecognised driver of this downward spiral is utility's vulnerability to various climatic hazards, that will likely be further intensified under future climate change conditions (Mirza, 2003; Srinivasan *et al.*, 2013; Becher *et al.*, 2023).

5.2. Materials and Method

5.2.1. Overview

In a previous study, a spatial model of the national water supply infrastructure network in Jamaica was established and stress-tested with a large set of (synthetic) spatially coherent, climate conditioned drought and pluvial and fluvial flood events (Becher *et al.*, 2023) to conduct a climate risk analysis. The flood events were derived from synthetic daily rainfall-runoff simulations which generated a set of 14,000 events. Drought events were extracted from a synthetic monthly rainfall-runoff simulation. Both event sets were used to provide risk outputs up to a maximum return period of 1000 years. Assets are grouped into their respective systems, each serving a particular population or demand zone across the island (National Spatial Management Branch, 2010). In total there are ~240 systems in the potable water systems representation of Jamaica, with very little linkages and redundancies between each other. The water systems model represents both asset-level physical vulnerability as well as domestic water usage, leakage losses and major storage dynamics in the supply network to compute the number of domestic customers impacted (hereafter Customer Disruption Days (CDD)) per hazard event. Note that only surface water-fed systems were included in the drought risk analysis. The steps in calculating the physical vulnerability and CDD are summarised in Section 2.2 and a more detailed overview of 1) the method and conceptual framework presented in the previous work can be found in Supplementary Text C1 and Figure 1, 2) a table of datasets in Table S1 and 3) plots of the potable water asset database in Supplementary Figures C2.

Here, we employ the outputs from this probabilistic multi-hazard risk framework (Becher *et al.*, 2023) to quantify the effectiveness of a set of multi-hazard adaptation options for mitigating drought-induced water shortage and flood-induced asset damages and disruptions. First, we identify the ways in which climate-induced disruptions impose financial losses 1) to the utility through asset damages and 2) to the utility's water customers in Jamaica. Our previous analysis demonstrated that a significant threat of damage to the water utility's assets comes from flooding, so we focus upon the financial consequences of these losses for the utility. Domestic water users incur financial losses when their potable water supplies fail, so they have to purchase water from tanker truck suppliers. The value of these financial losses for people is estimated by applying the unit cost of acquiring water from utility vendors per person per day to the CDD risk output provided by Becher et al., (2023). We collect a set of adaptation options

comprising flood defence walls, proposed to protect all exposed assets against flood-induced damages and disruptions, and leakage reduction strategies, proposed to protect all systems against drought-induced water shortage disruptions. The costs and benefits of each flood protection and drought mitigation option are quantified on a system scale across range of flood and drought hazard intensity, respectively. Thus 10 options for 240 systems, provide a longlist of 2,400 options under consideration under this framework. By testing the system performance with and without each adaptation option, we investigate the benefit of each option, in terms of avoided utility damages and alternative water purchase losses. These are compared against each option's cost to compare options based on their BCRs. We go on to estimate the reduction in alternative water purchase losses burdened by customers in poverty provided by each intervention. This allows us to assess each climate adaptation option in terms its business case and its contribution towards 'pro-poor', or equity-related development agendas. This framework is illustrated in Figure 17.

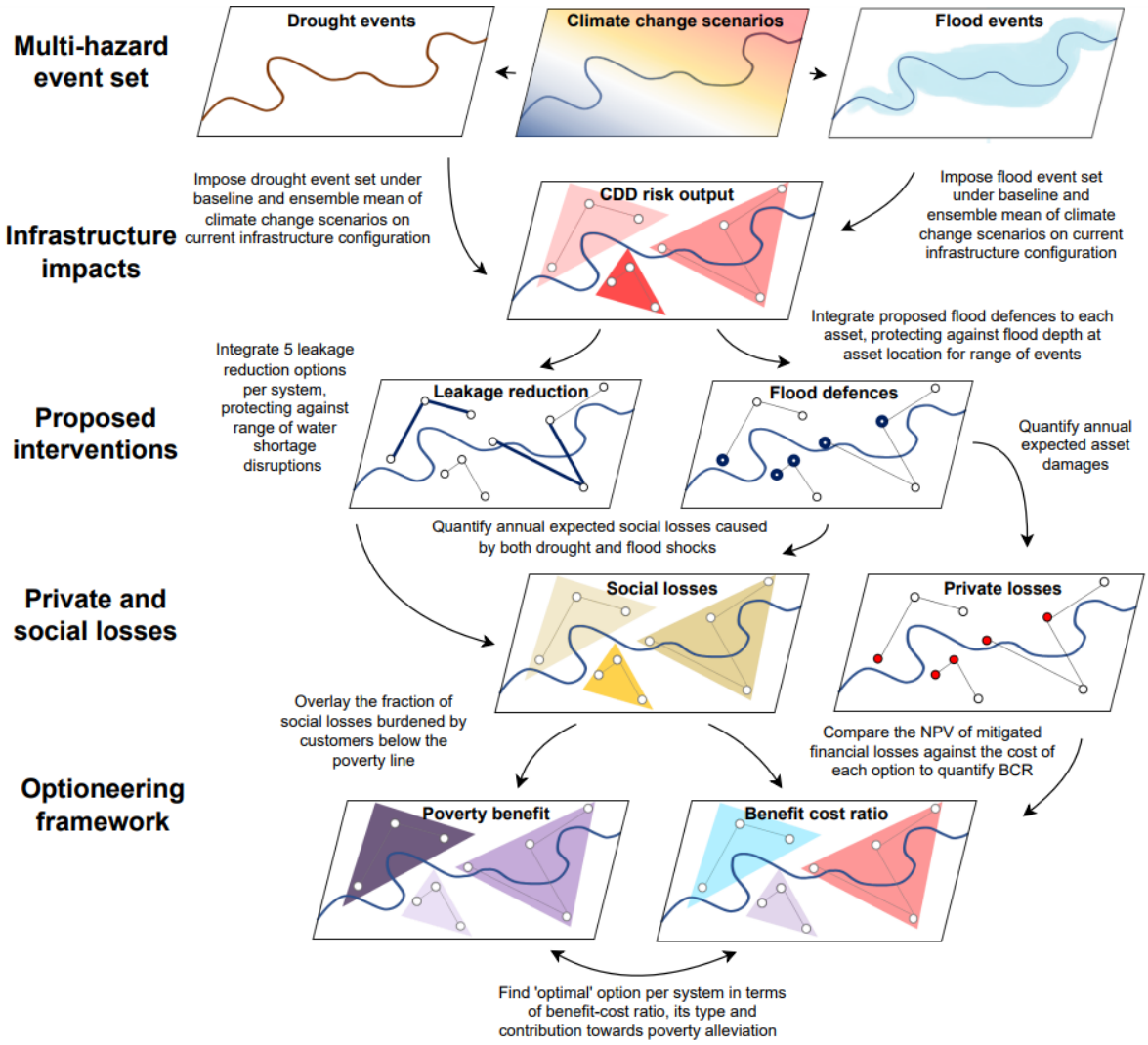


Figure 17 Methodological framework for multi-hazard optioneering with respect to economic and equity (poverty benefit) incentives.

Our methodological approach draws from two studies undertaken in Bangladesh that approached the problem of planning critical infrastructure networks through the lens of the distributional benefits of service provision among diverse populations. Garcia et al., (2021) explore the trade-offs between purely utilitarian (maximising the number of people with access) and poverty-orientated (maximising the number of people in poverty with access) objectives when planning the expansion of water supply infrastructure. Verschuur et al., (2023) quantify the impacts of cyclonic flooding on a range of critical infrastructure networks, including water supply, to understand distributional impacts on Multi-Dimensional Poverty (MDB), a description of poverty defined partially in relation to access to critical infrastructure services, such as water, sanitation and electricity. Thus, the permanent spatial distribution of MDB, extracted from census data (Bangladesh Bureau of Statistics, 2020), is

superimposed with temporary climate-induced critical service disruption to evaluate the benefit of coastal flood protection measures in vulnerable communities.

5.2.2. Application of multi-hazard risk approach for optioneering

The framework outlined in Becher et al., (2023) involves quantifying the impact of present-day and future climate extremes on the national water supply utility in Jamaica, in terms of CDD. Below, we summarise the overarching steps of this framework:

1. Generate a large set of spatially coherent drought and pluvial and fluvial flood events, representing both historic climate conditions, as well as future climate change conditions. We consider climate change projections up to the year 2080 under RCP2.6, RCP4.5 and RCP8.5 emissions scenarios.
2. Combine the water utility's assets (1,208 pumping stations, treatment works, etc.) in a system model, which also represents water supply to municipal water users, leakage and major storage capacities in the supply network. Assets are grouped into their respective systems, each serving a particular population or demand zone across the island (National Spatial Management Branch, 2010).
3. For flood events, intersect each hazard event with the water utility asset and identify the set of directly damaged (and hence failed) assets that will lead to disruptions across different water utility sub-systems.
4. For drought events, estimate the risk of system-level disruptions by estimating the unmet water demand due to loss of water supply resulting from lower stream flows, reservoir levels, and system leakages.
5. Quantify, for each disruptive event, the number of water users impacted per day during the event, multiplied by the duration of the event (in days), i.e., the CDD.
6. Compute the annual expected CDD per utility supply system by integrating the risk distribution using quadrature, or the trapezoidal rule.

The water system model and risk analysis methods are published and described in detail in Becher et al. (2023) and summarised in Supplementary Text C1. Here we employ the CDD risk outputs, projected up to the year 2080 under RCP2.6, RCP4.5 and RCP8.5 emissions scenarios in this aforementioned study (Becher *et al.*, 2023), to quantify the effect of a portfolio of climate adaptation options on mitigating future water shortage- and flood-induced asset damages and disruptions.

5.2.2.1. Direct climate-related damages and losses

Cost of damages incurred by the utility

Here we consider the costs required to repair assets following flood inundation as an important climate-related financial risk transmission pathway for the Jamaican water utility. Although in other utility contexts, financial losses may also include lost tariff revenue during disruptions, this is not relevant for the NWC (Personal Communication, 2022) as a lack of monitoring capabilities (e.g., sensors in the distribution network) limits the utility's ability to adjust tariff payments for customers based on temporary drops in consumption during disruptions. The level of damage to the utility's potable asset-base is modelled by applying damage curves, which quantify the ratio of the expected repair cost to the replacement cost of an asset against a given flood depth (Porter, 2021). Asset-specific damage curves for flooding for pumping stations (applied to wells, intakes and pumping station) and treatment plants were obtained from (FEMA, 2020), see Supplementary Table C2. Reservoirs were assumed to be invulnerable to flooding at the return periods considered here, given that design standards for reservoirs typically specify a very high protection level. We apply the fraction damage curve value for a given flood event to the cost of reconstruction for the different types of assets to estimate the cost of flood-induced damage, as shown in Equation 18.

$$PD_{ij} = F_j(d_i)\, R_j \qquad (18)$$

Where PD_j is the physical damage cost estimate for an asset (j) for the flood hazard event (i), F_j denotes the function that describes the damage curve ratio (Miyamoto, 2019; FEMA, 2020) for the asset from which we estimate the damage ratio value at given flood depth (d_i) for the flood hazard event i, R_j denotes the full reconstruction cost of that particular asset j (National Water Commission, 2011). Equation 18 was applied to estimate damages to wells, treatment plants, intakes and pumping stations. We collected replacement costs for these assets from the Jamaican water utility's supply plans (National Water Commission, 2011) and can be found in Supplementary Table C1.

Asset level direct damage risk across several events is estimated, as the annual average damage ($\overline{PD_j}$), by numerically integrating the inverse cumulative probability function of the direct damage costs $Q_{\overline{PD}}(p)$, obtained by ranking the damages for all events to estimate annual exceedance probabilities of damages (p), as formulated in Equation 19. The annual average damage is estimated for given flood events set under a given climate scenario.

$$\overline{PD_j} = \int_0^1 Q_{\overline{PD}}(p)dp \qquad (19)$$

The asset level annual average damages are aggregated at the system level, where each system serves a set of customers. For a given system s, the annual average damage ($\overline{PD_s}$) is estimated by adding up the annual average damages of all assets j within s (i.e., $j \in s$), as per Equation 20, as we can assume that direct physical damages to assets are independent of each other.

$$\overline{PD_s} = \sum_{j \in s} \overline{PD_j} \qquad (20)$$

Cost of disruptions externalised to society

There is a strong reliance on tanker trucks to replace disrupted water utility supplies in Jamaica, although most households are fitted with one or multiple water tanks which can be filled during periods of non-disrupted utility supply (Lester, 2015), providing a limited buffer against disruptions. Further, some households also turn to unregulated spring and well water sources during disruptions. However, in the absence of detailed data on the dynamics of tanker truck vendors and household storage tank capacity, we simplify complex, household-level responses to utility water supply disruptions, and assume that the only viable alternative form of water supply comes from tanker trucks. Thus, due to a disruptions from a hazard event i (flood or drought), costs of disruptions to society dependent on a water system s, the alternative water purchase losses (AWL_{is}), are assumed to be equivalent to the CDD associated with system s due to event i (CDD_{is}) multiplied by the unit cost (c) ($J per person per day) of tankering water for the utility (Jamaica Observer, 2018), see Equation 21.

$$AWL_{is} = CDD_{is}c \qquad \text{Equation (21)}$$

Note that, the framework can accommodate contexts where tariff revenue is elastic to the rate of disruption: disruption-induced lost tariff revenue can be estimated by multiplying the CDD metric by the daily lost tariff revenue per customer.

Similar to estimating an annual average damage (from Equation (19)), the annual average loss at the system level ($\overline{CRL_s}$) is estimated by integrating the inverse cumulative probability function of the alternative water purchase losses $Q_{\overline{SL}}(p)$, obtained by ranking the alternative water purchase losses for all events to estimate losses versus annual exceedance probabilities of losses (p). The annual average loss at the system level is estimated for a given hazard type (floods or drought) and for a given climate scenario, as shown in Equation 22.

$$\overline{AWL_s} = \int_0^1 Q_{\overline{SL}}(p)\ dp \qquad (22)$$

Relative burden to customers in poverty

To estimate the relative burden of public water supply disruptions to customers in poverty, we estimate the fraction of each NWC system's customer base that is below the Jamaican poverty line, defined as an income of 1.9 US dollars per day. Those in poverty are less likely to be able to afford a household connection to the utility networks and would likely source water from other unregulated (and potentially contaminated) sources such as springs and wells (Velleman, 2009). Indeed, approximately a third of Jamaica's poorest households obtain water from untreated wells and rivers (Haughton, 2015). In the absence of detailed system-level data on the income distribution of customers, our approach estimates the poverty-incidence of each NWC system as follows. The population residing in communities that intersect with the infrastructure network boundaries of each NWC system, are assumed to represent the population served by each system. Therefore, to find this representative population, we spatially overlay the system boundaries layer with a community level population (Statistical Institute of Jamaica, 2010) and poverty incidence layer (Planning Institute of Jamaica, 2011). All communities that intersect the NWC system boundaries were allocated to that system. Where a community intersects multiple NWC systems, the population within that community is split across systems in proportion to the population served by each system. Because the NWC water supply network covers ~70% of the full Jamaican population (National Water Commission, 2020a), the population allocated to each system based on the census data, exceeds the population that is served by the utility, according to the NWC Parish Water Supply Plans. Thus, we assume that each NWC system's customer-base would first be made up of the full population allocated to that system that is not in poverty. Where the above poverty population, allocated to a NWC system is less than the NWC customers served by that system, the remaining customers are assumed to be made up of the population is in poverty.

The fraction of each NWC system's customer base in poverty can be estimated as follows:

$$f_s = \frac{TB_s}{(TA_s + TB_s)} \qquad (23)$$

Where f_s denotes the fraction of people in poverty, TA_s denotes the number of customers estimated to be above the poverty line within the service area of NWC system (s) and TB_s denotes the number of customers estimated to be below the poverty line within the service area of NWC system.

Our results, illustrated in Supplementary Figure C3, are sensitive to both the input data on poverty incidence and population, which are both outdated by ~10 years. However, the consistent use of the input datasets across systems provides an indication of the relative measure of how climate-related water infrastructure disruptions affect utility customers in poverty. Household survey data (e.g., (World Bank, 2012)), that captures information on whether households have access to piped water supply connection and their rate of income, could be usefully applied in future applications of this framework.

To summarise the risk per NWC water system s due to each hazard (flooding and drought) in total ($\overline{TR}_s$) and the total poverty risk ($\overline{TPR}_s$) affecting the fraction of people in poverty (f_s) are estimated as following:

$$\overline{TR}_s = \begin{cases} \overline{PD}_s + \overline{CRL}_s, & \text{if hazard} = \text{flood} \\ \overline{SL}_s, & \text{if hazard} = \text{drought} \end{cases} \quad (24)$$

$$\overline{TPR}_s = \overline{TR}_s \, f_s \quad (25)$$

5.2.2.2. Multi-hazard, multi-objective optioneering framework

Flood defence walls and leakage reduction are selected as the two most relevant types of intervention for our optioneering study, as they are targeted specifically for climate adaptation interventions. We test different levels of flood wall and leakage reduction designs meant to prevent floods and drought events of return periods varying from 20, 50, 100, 200, 500 years at each NWC water system level.

The optioneering involves doing a cost benefit analysis (CBA), where the costs of adaptation options were weighed against the benefits, or avoided risks, that would be accumulated (i.e., the net present value) over the lifetime of a project. We assume the lifetime of a project would be 50 years and simulate the time window from project implementation in 2030 (the date of the Capital Improvement Plan) to the year 2080. We assume a linear rate of change in annual expected risk between the baseline $\overline{TPR}_s$ (2010) and future $\overline{TPR}_s$ under each climate change scenario, where future risk is averaged across three RCP scenarios (RCP2.6, 4.5 and 8.5) projected to the end of century epoch (year 2080). A discount rate of 10% was applied, following the approach that is taken by the utility in their Water Supply Parish plans (National Water Commission, 2011).

For a selected adaptation option of flood defence walls or leakage reduction the planning of the chosen option is done over an annual timescale denoted by $t = 0, \ldots, T$, where $t = 0$ is the

start of the implementation of the option and $t = T = 50$ years is the option lifetime. Here we assume different options (flood defence walls or leakage reduction) would provide a level of protection up to given (flood or drought) event severity. For example, a flood defence wall designed to a 1/100 return period standard will provide protection against flood events up to 1/100-year return periods. Then we apply the CBA to assess the effectiveness of each option.

As noted above we apply different adaptation options to present floods or drought risks for the set of design return periods $RP = \{20,50,100,200,500\}$ (in years) or design annual exceedance probabilities $P = \{1/20,1/50,1/100,1/200,1/500\}$. Assuming that the discount rate of $r\%$ is applied over the adaptation planning horizon, the discounted cost ($DC_s(p_d)$) for applying the adaptation option of a given design annual exceedance probability ($p_d \in P$) to a water system s is estimated in terms of the sum of initial investment costs of all assets j within system s ($CI_{j0}(p_d)$) at $t = 0$ and the sum of the subsequent asset level discounted annual maintenance costs $CM_{jt}(p_d)$ over the planning horizon. This is formulated in Equation 26.

$$DC_s(p_d) = \sum_{\forall j \in s} CI_{j0}(p_d) + \sum_{\forall j \in s} \sum_{t=1}^{t=T} \frac{CM_{jt}(p_d)}{(1+0.01r)^t} \qquad (26)$$

Using Equations 19-25, these reduced risks are obtained by rerunning the risks analysis with the hazard event sets, where now the risks to a system are only estimated for more severe events with exceedance probabilities $\leq p_d$. This is formulated as following:

$$\overline{TR}_s(p_d) = \begin{cases} \sum_{\forall j \in s} \int_0^{p_d} Q_{\overline{PD}}(p)dp + \int_0^{p_d} Q_{\overline{SL}}(p)dp \ , & \text{if hazard} = \text{flood} \\ \int_0^{p_d} Q_{\overline{SL}}(p)dp \ , & \text{if hazard} = \text{drought} \end{cases} \qquad (27)$$

Similar to the costs, the risks before and after the adaptation option are estimated at annual timescales over the planning horizon. As noted above, future risks are driven by climate change driven hazard events. The discounted benefit ($DB_s(p_d)$) of risk reduction due to the adaptation option is estimated by discounting the annual timeseries of marginal risk reduction ($\overline{TR}_{st} - \overline{TR}_{st}(p_d)$) over the planning horizon, as shown in Equation 28.

$$DB_s(p_d) = \sum_{t=0}^{t=T} \frac{\overline{TR}_{st} - \overline{TR}_{st}(p_d)}{(1+0.01r)^t} \qquad (28)$$

The effectiveness of the option is assessed in terms of the benefit-cost ratio ($BCR_s(p_d)$), given by Equation 29.

$$BCR_s(p_d) = \frac{DB_s(p_d)}{DC_s(p_d)} \qquad (29)$$

The optioneering solution (for a given type of hazard protection) involves finding the optimal intervention (OI_s) per system, identified as the option where the ratio between benefit and cost (BCR) is greatest.

$$OI_s = \max_{\forall p_d \in P} \left(BCR_s(p_d) \right) \qquad (30)$$

Choice of flood defences

For flood defences adaptation, we look across all assets exposed to flooding. First, we find the flood depths to which each asset exposed for a range of return periods of 20-, 50-, 100-, 200- and 500-years. The costs of flood walls built up to a range of heights to protect a number of water treatment plants and pumping stations in the UK are extracted from a report provided by a Mott MacDonald report (Mott MacDonald, 2018) which were converted to Jamaican dollars. Extracted costs are used to produce cost curves of flood walls built to a range of heights, with upper and lower uncertainty bounds. This curve is provided in the Supplementary Table C4. Thus, we derive a cost curve for the initial investment costs ($CI_{j0}(p_d)$) of flood defence walls to protect against the aforementioned range of return periods for all exposed assets, which are estimated at 0.5% of the original value of the flood wall (Aerts, 2018), also provided in Supplementary Table C4.

As water systems are made up of a network of assets that depend on each other to varying degrees, implementing flood defences at an individual site may still result in the same risk of customer disruptions as not implementing any defences if multiple assets in a system are exposed to the same events. Here, we adopt a systems perspective and calculate the costs, and the risk reduction potential of, protecting each system from events across the return periods of 20-, 50-, 100-, 200- and 500-years. Note that this requires assuming that a given flood event would cause disruptions at all exposed (have a probability of flooding > 0) and vulnerable (experience damage given a flood inundation depth above some level) assets within in a given system simultaneously, which is a reasonable assumption given their spatial proximity to each other. This assumption has also been adopted in other studies (e.g., Koks *et al.*, 2019). As such, for a given system to be protected from a 1 in 100-year flood, we sum the costs of defences at all exposed assets that would be required to mitigate damage and disruptions in a system equivalent to a 1 in 100-year event.

Choice of leakage reduction options

Leakage reduction is proposed to improve the reliability of services to the currently served population. Further, the NWC considers leakage reduction a priority intervention, due to the significant losses of up to 70% that current affect the distribution networks (National Water Commission, 2011). In the capital improvement plan, the budget proposed for leakage reduction is provided as a national total volume leakage loss mitigated (NWC, 2015). We use this to derive a unit cost per m^3 of leakage loss reduction. In reality, the marginal cost of reducing leakage increases as the target loss rate becomes more ambitious: for example, it is cheaper to reduce leakage from 50 to 40% than from 20 to 10%. In developed countries such as in the UK, 15% leakage is considered to be an economic loss rate, beyond which costs become unjustifiable (GWI, 2022b). In cases where water infrastructure is thought to be most advanced globally, such as Singapore and Israel, leakage rates can be as low as ~5% (ibid). Here, we assume a linear cost curve and adopt 30% as the economic loss rate for Jamaica, based on the utility's infrastructure planning documents (National Water Commission, 2011).

To determine the effect of reducing leakage on water shortage risk per system, we run iterations of the water balances established in (Becher *et al.*, 2023), provided in Supplementary Text C2, with a range of leakage reduction scenarios from the original loss rate to a minimum rate of 30% (National Water Commission, 2011). The water balances involve simulating monthly supply inputs for all surface water systems across the NWC network against a system specific drought threshold. The drought threshold per surface water abstraction asset is the sum of the volume of demand, leakage losses and an environmental flow threshold. The sensitivity of each system's risk of drought-induced water shortages with respect to changes in leakage loss rates was derived. This was used to find the reduction in leakage that would be required to achieve a protection level equivalent to each of the five customer disruption events associated with return periods of 20-, 50-, 100-, 200- and 500-years. The cost of each leakage reduction option $CI_{j0}(p_d)$, is found by applying the unit cost (m^3) to the volume of leakage loss mitigated by each option. Note that, in some cases, reducing leakage by the minimum amount (i.e., to 30%) would not be enough to protect systems against a very extreme event, in which case the option was removed from consideration.

Following the same approach as that defined by Equations 26-30, we derive investment costs and mitigated alternative water purchase losses associated with each leakage reduction option at each system, and, comparing the two, identify the optimal option per system.

Mitigated alternative water purchase losses for customers in poverty

We use the estimated alternative water purchase losses that would be burdened by those in poverty ($\overline{TPR_s}$) to simulate the mitigated poverty losses as the Discounted Poverty Benefit $DPB_s(p_d)$ achieved by alternative interventions as follows.

$$DPB_s(p_d) = \sum_{t=0}^{t=T} \frac{\overline{TPR}_{st} - \overline{TPR}_{st}(p_d)}{(1+0.01r)^t} \qquad (31)$$

Uncertainty analysis

The uncertainty range is derived firstly by determining whether results are more sensitive to 1) the variation in cost and benefit of each option provided by each of the considered RCP scenarios or 2) perturbations of the combined vulnerability (damage fraction and leakage). For flood defences, the latter source of uncertainty dominates, whereas for leakage reduction, the former source of uncertainty dominates. We also factor cost estimates by +/-25% for both flood defences and leakage reduction options.

5.3. Results and Discussion

In this section, we present the overall distribution of costs and benefits for each option type, across all considered protection levels across all systems. We go on to explore results at the system level. We explore the BCRs of system-optimised options, to identify the optimal options tailored for each individual system and to explore to what extent economic (in terms of BCR) and equity (in terms of costs alleviated for those in poverty) objectives align or diverge for different systems. We also provide the spatial distribution in utility damages induced by flooding per climate change scenario in Supplementary Figure C5.

5.3.1. Benefit cost ratio of all options

Figure 18 provides an overview of the distribution of discounted benefits versus discounted costs of all considered leakage reduction (top) and flood defence (bottom) options, while also showing the discounted benefits associated with poverty reduction. For leakage reduction, we see that for many options, the benefit is zero or close to zero. In these cases, the sensitivity of a system's water shortage risk to changes in leakage losses is low. We also observe a strong relationship between an option's overall benefit and its benefit to those in poverty. This is because the BCRs of leakage reduction options are defined as the mitigated alternative water purchase losses they offer, of which their benefit to those in poverty is a fraction. However, this relationship is not always clear: some options offer a high BCR but a low benefit to those in poverty, which signifies a relatively low poverty incidence among these options' beneficiaries (i.e., the customer base of the relevant NWC system).

The differences between costs and benefits for different options is lower for leakage reduction compared to those for flood defences. This is because the contribution of leakage towards water shortage risks across NWC systems is relatively pervasive in comparison to relatively localised flooding impacts. In the lower subplot of Figure 18, we see the costs and benefits of flood defences. In absolute terms, the costs and benefits of flood defences are both much higher than for leakage reduction options, given the contribution of avoided asset damages and that interventions require construction of protective walls around all exposed assets in a system. We also note that, although there is still an observable relationship between overall option benefit and the benefit for those in poverty (indicated by the shade of blue of each point), it is not as clear as for leakage reduction options. This is because the contribution of avoided utility damages vastly outweighs that of avoided alternative water purchase losses in the calculation of overall benefits of flood protection, and the riskiest assets from a damage perspective are

not necessarily the same from a disruption perspective. This mismatch between damage and disruption risk hotspots aligns with prior studies on flood risk to critical infrastructure services (e.g., Koks *et al.*, 2019). The range of uncertainty associated with each point is indicated by error bars.

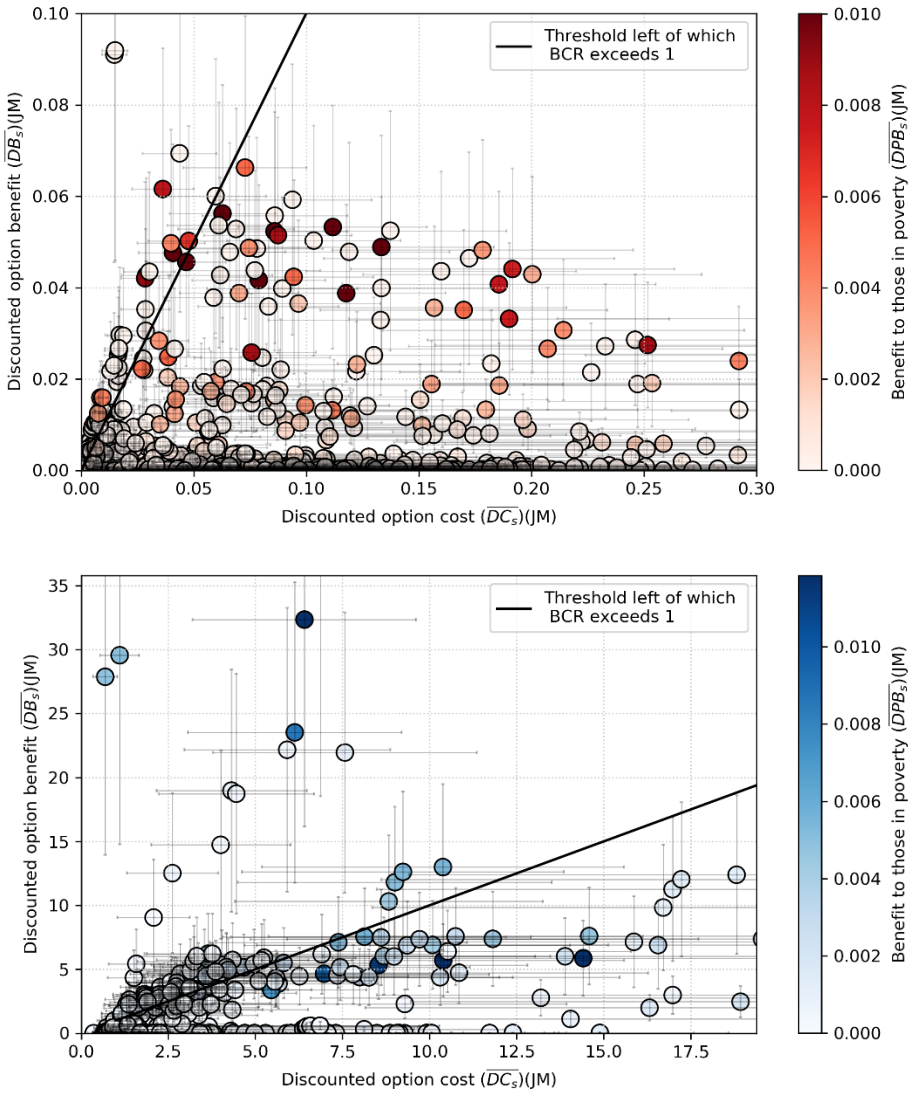


Figure 18 Cost ($\overline{DC_s}$) and benefit ($\overline{DB_s}$) per leakage reduction option (top) and flood defence option (bottom), where points are coloured by the tankering costs saved by customers in poverty. All costs are provided in million Jamaican dollars. Vertical lines associated with each scatter point represent the uncertainty associated with the benefit of each option. The black line illustrated the 1-1 relationship between benefit and cost; therefore, options are associated with BCRs that are >1.

5.3.2. System level optioneering

Adopting a systems perspective, Figure 19 compares the BCRs across all protection levels considered, for drought (left) and flood (right) options. Please note that we only show the 44

systems which have a BCR > 1 for either leakage reduction or flood defences. We see that the BCRs for leakage reduction options generally increase with an increased protection level, whereas the variation between BCRs for flood defences do not show this trend. This suggests that the increase in flood defence costs co-varies with the increase in avoided damages as the protection level of an option increases, such that the ratio between benefit and cost does not necessary increase. Whereas the change in drought risk, and therefore the additional required investment in leakage reduction, is relatively smaller for a given increase in protection level, which means that incremental investments in leakage reduction tend to pay-off more than for flood defences. This means that the losses incurred by high frequency drought events increase to a lesser extent relative to flood events, such that protecting against a 1-in-500-year protection level would not cost significantly more than protecting up to a lower protection level. System 137 is the only system where both types of option with a BCR > 1, highlighting the benefit of adopting a multi-hazard perspective for optioneering to capture all climate vulnerable systems that would benefit from adaptation (Koks *et al.*, 2019; Dong *et al.*, 2020; Luo *et al.*, 2023; Verschuur, Koks, *et al.*, 2023).

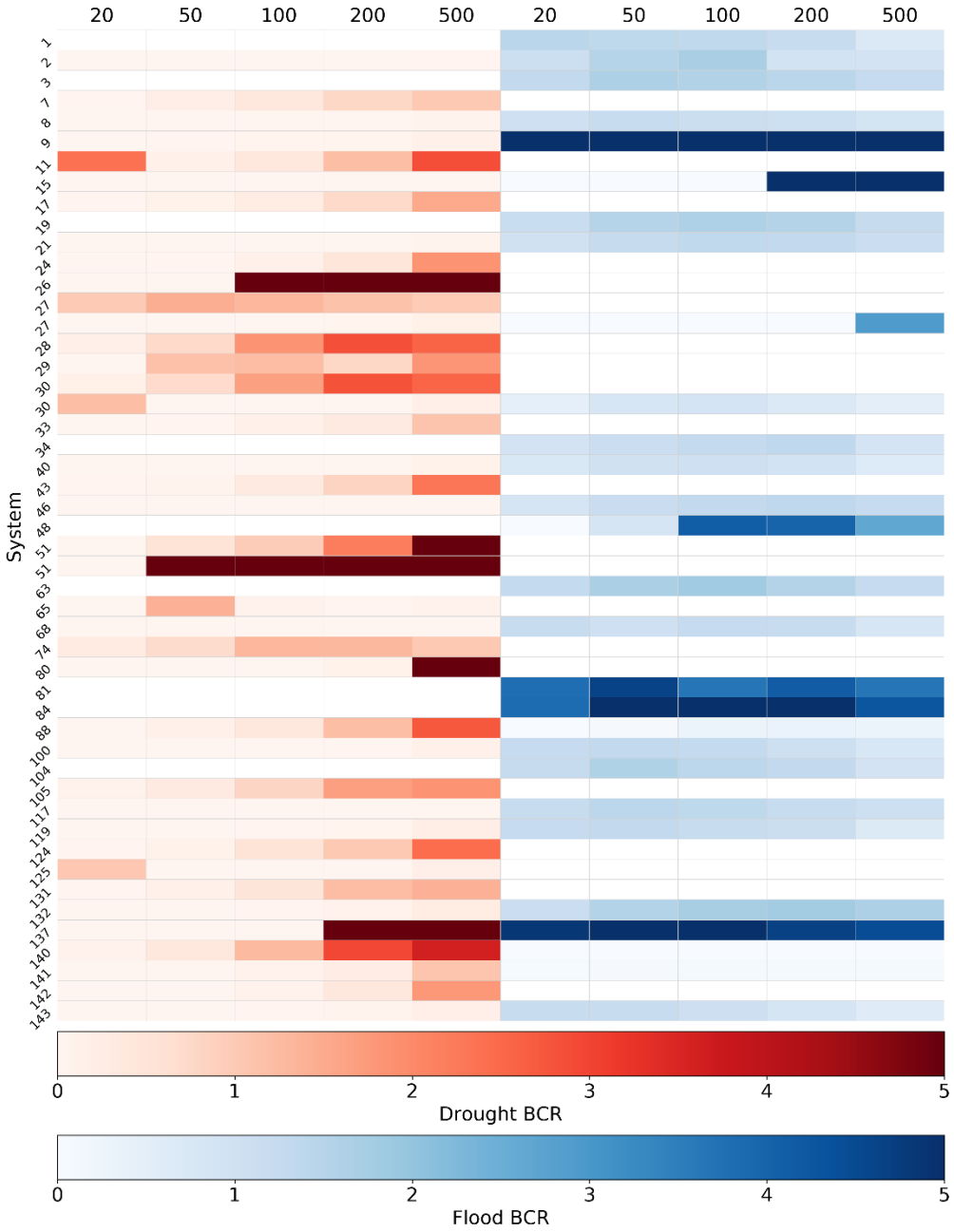


Figure 19 BCRs across all considered protection levels (20-, 50-, 100-, 200- and 500-year return period) $BCR_s(p_d)$ for leakage reduction in red (left) and flood protection in blue (right).

Figure 20 provides the BCR per option, broken down between leakage reduction options (red, left) and flood defences (blue, right). Note that here we plot the optimal option per system with a BCR greater than 0. High leakage losses are pervasive across NWC systems, driving the risk of water shortage-induced alternative water purchase losses (Becher *et al.*, 2023). In contrast, flood impacts are relatively localised (ibid). Thus, as presented in Figure 20, we find a greater number of leakage reduction options that offer BCRs > 1 compared to flood defence options, and a greater variation between BCRs asset protection options than for drought mitigation.

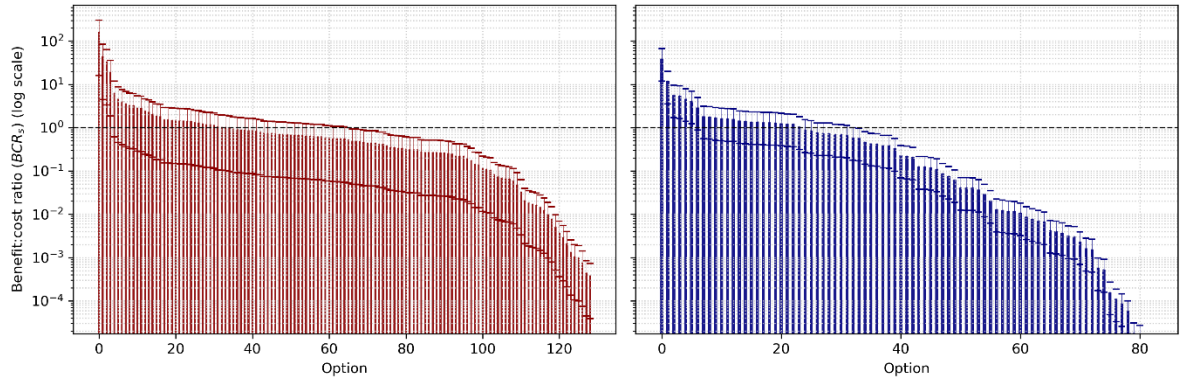


Figure 20 Benefit Cost Ratio (BCR_S) per option, with flood defence options highlighted in blue and leakage reduction options in red. Options are ranked from those offering the highest BCR to the lowest. BCRs, plotted on the y axis using a log scale for plot clarity.

In Figure 21, we represent the system-optimised option's BCR spatially (across leakage reduction and flood defence). Leakage reduction options are indicated in red, with darker red highlighting options with the greatest BCRs. Flood defence options are indicated in blue, with darker blue highlighting flood protection options with the greatest BCRs. Notable system numbers are provided for ease of comparison between plots (e.g., Figure 19 and Figure 21). We can see that the options with the greatest BCRs for leakage reduction are targeted at system 137, 80 and 26 systems. Likewise, the flood defence strategies with the greatest BCRs are targeted at 9, 84, 81 systems. Two hotspots are identified in systems serving areas near Ocho Rios and Kingston, where adaptation options for protection against both drought and flood hazards would be particularly efficient from a cost-benefit perspective. There isn't a distinct spatial clustering between flood and drought optimal BCRs, which one would expect if BCR was only influence by hazard risk. This is not surprising, given the influence of local system characteristics on risk and, in turn, on the cost-effectiveness of risk mitigation strategies (Becher *et al.*, 2023). Flood risk is driven by the spatial pattern of flood inundation depth for a range of return periods in relation to asset location, vulnerability (defined by asset-specific flood depth-damage curves) and criticality (population served per asset) or reconstruction costs for disruption and damage risk, respectively. Drought risk is driven by population served by a system and the availability of water supply relative to demand (ibid), where demand reflects household consumption after leakage losses across the distribution network, and supply inputs capture the dynamics major storage reservoirs (ibid). Note that we do not consider the difference in water consumption rates for customers across income groups (Hussien, Memon and Savic, 2016), but rather assume a constant rate of demand, following the approach take in the utility's Parish Water Supply plans (National Water Commission, 2011).

We find that system-optimised flood defence and leakage reduction options with benefit-cost ratios (BCRs) > 0 are 83 and 128, respectively, and the number of options >1 amount to 24 and 26, respectively. The total annual benefit that could be achieved by implementing all system-optimised flood defences with BCRs > 0 is estimated to be approximately 0.72 billion Jamaican dollars per year. This saving represents a significant portion of annual total repair costs, which ranges from 1 to 3 billion Jamaica dollars per year (NWC, 2019). However, flood defence walls are relatively capital intensive and can have unintended consequences (Jongman, 2018), such as exacerbating flooding in surrounding areas. Further, they would not address turbidity risks, one of the most common forms of flood-related disruption for the utility (NWC, 2021a). A broader range of option types, such as integrated, nature-based approaches (including rainwater harvesting and flood retention), could be included within the framework in future applications (FAO, 2019). The difficulty in quantifying the effectiveness of nature-based solutions for flood risk reduction at scale prevented their inclusion here (Turkelboom *et al.*, 2021). Whereas, for leakage reduction, the total benefit would be 1.5 million Jamaican dollars per year. We note that reducing leakage, e.g., through pressure management and network rehabilitation, could also help to reduce energy consumption and strengthen network resilience to drought and operational risks, such as pipe bursts (Ahopelto and Vahala, 2020). This is particularly challenging in the hard-to-reach, rural communities in the centre of the island, which have a higher poverty incidence (see Figure S3): extreme pressure variations due to steep topography pose a challenge to the functioning of distribution networks in these systems, increasing the risk of leaks and mechanical failures (Lambert, 2000). Indeed, J$ 828.8 million project is being channels into reducing the level of Non-Revenue Water by 30% in the next 5 years and reduce maintenance by correcting for leaks (NWC, 2015).

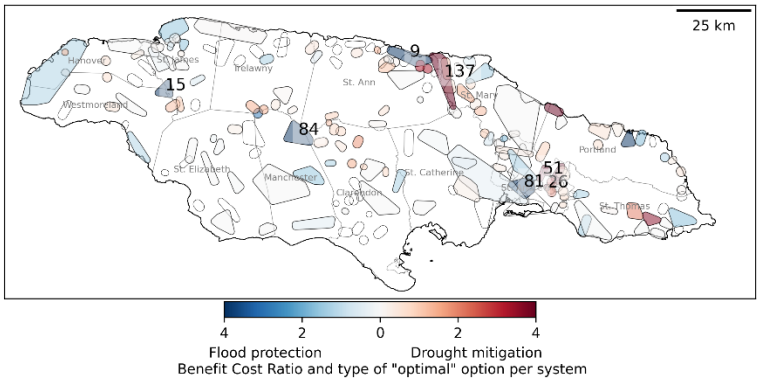


Figure 21 Option with the greatest BCR per system (OI_s). Drought options are indicated in red, with darker red highlighting drought mitigation options with the greatest BCRs. Flood defence options are indicated in blue, with darker blue highlighting flood protection options

with the greatest BCRs. Systems with the most economically efficient options are annotated. Parish outlines are plotted for greater spatial context.

In Figure 22 we provide the spatial contribution of the most economically efficient options towards equity-related objectives, defined in terms of absolute annual alleviated expenditure for those in poverty. Drought options are indicated in orange, with darker orange highlighting drought mitigation options with the greatest benefit to those in poverty. Flood defence options are indicated in purple, with darker purple highlighting flood protection options with the greatest benefit to those in poverty. Systems with the most economically efficient options are annotated. The total annual benefit to those in poverty that would be achieved by implementing these options is estimated at 0.08 million Jamaican dollars per year, with optimal flood defences and leakage reduction options offering a comparable contribution towards this annual benefits. Although this is relatively small number in absolute terms, from the perspective of a low-income household, could represent a significant saving relative to income (Hallegatte and Rozenberg, 2017).

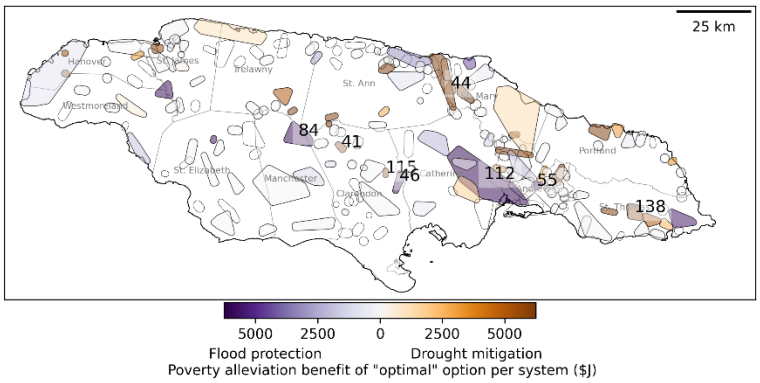


Figure 22 The contribution of the most efficient option (OI_s) of each type towards reducing costs for those in poverty (DPB_s) in absolute terms in Jamaican dollars. Drought options are indicated in orange, with darker orange highlighting drought mitigation options with the greatest benefit to those in poverty. Flood defence options are indicated in purple, with darker purple highlighting flood protection options with the greatest benefit to those in poverty. Systems with the most economically efficient options are annotated. Parish outlines are plotted for greater spatial context.

However, the options with the greatest BCRs do not always align with the options that provide the greatest overall benefit to those in poverty. Figure 23 explores the interaction between economic- and equity-orientated objectives spatially, offering an integrated perspective for decision-makers. Options were ranked (or scored) based on their economic performance (BCR) and performance with respect to equity (alternative water purchase losses alleviated for those in poverty). Options with a high score based on both economic and equity performance, offer

a strong the synergy between economic and equity objectives and are represented in red in Figure 23. System-optimised options with a high score for only one objective, are more subject to trade-offs. Those options with a high score for equity and low for economy, appear in yellow. Options with a high score for economy and low for equity, appear in blue.

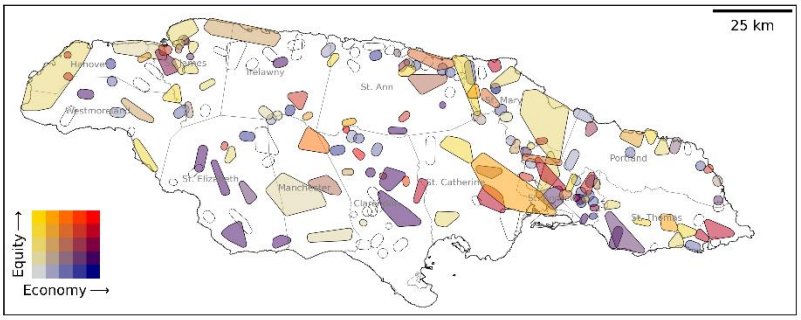


Figure 23 Bivariate plot showing the interaction between economic- and equity-orientated objectives for each system, spatially. Strong option performance solely with respect to equity appear in yellow. Strong option performance solely with respect to economic objectives appear in blue. Strong performance with respect to both objectives appear in red. Systems with optimal options that have a strong synergy between both objectives are annotated. Five quantiles are applied to the score of each option relative to each objective. Parish outlines are plotted for greater spatial context.

5.4. Assumptions, limitations, uncertainty and merits

A new optioneering framework for potable sector planning has been presented to quantifying direct social and utility losses incurred during climate-induced water supply disruptions. This framework allows us to directly compare the benefit cost ratios of a portfolio of protection works, proposed to reduce asset-level vulnerability to flooding, and leakage reduction options, proposed to reduce system vulnerability to drought impacts. It also allows us to estimate the alternative water purchase losses burdened by customers in poverty that would be alleviated due to potable water sector investments.

In the application of the framework to the water supply utility in Jamaica, we find that system-optimised flood defence and leakage reduction options with benefit-cost ratios (BCRs) > 0 are 83 and 128, respectively, and the number of options >1 amount to 24 and 26, respectively. The total annual benefit that could be achieved by implementing all system-optimised flood defences with BCRs > 0 is estimated to be between ~0.72 billion Jamaican dollars per year. Whereas implementing all leakage reduction options would provide a total benefit of ~1.5 million Jamaican dollars per year. The total annual benefit to those in poverty that would be achieved by implementing these options is estimated at 0.08 million Jamaican dollars per year. Overall, asset protection interventions are cost effective for mitigating utility damages, whereas drought mitigation options are more effective at mitigating alternative water purchase losses. Drought mitigation options are more cost effective than flood defences, however, investment-worthy flood defences have a greater effect in absolute terms. Both option types have a comparable benefit to those in poverty, in absolute terms.

We acknowledge that, in such a budget constrained environment, non-climate-related factors would likely dominate investment decision-making process of utilities, such as the drive to expand networks to unconnected communities. Nonetheless, an optioneering framework, such as that presented here, can help utilities to more efficiently allocate limited budgets. The framework proposed also serves to demonstrate the positive impacts of adaptation investments, which can be useful for attracting grants and concessional finance. These issues are becoming more prominent in Low- and Middle-Income Countries and Small Island Developing states in particular, so methodological innovation, as described in this paper, is needed.

The framework presented here introduces an alternative quantified measure of the direct financial returns that could be realised by investing in more climate resilient municipal water supply infrastructure. Typically, cost benefit analysis on new water supply infrastructure is

done based on the projected tariff revenues that would be gained by increasing the volumetric sale of water, which are limited due to intense political pressures to keep tariffs low. Further, benefits defined as such would be zero for projects that improve the climate resilience of supplies to existing customers and, therefore, do not increase tariff revenue. Our approach supports both the economic business case for climate adaptation projects in the water sector at the same time as demonstrating the benefit of infrastructure for society. In the context of an infrastructure investment landscape that is increasingly motivated to generated positive equity-related impacts, including alignment with the Paris agreement, and increasing the portion of green investments, such analytical frameworks could help water supply utilities.

The wider applicability of certain aspects of the approach should be re-evaluated before employing this framework in other utility contexts. Firstly, here, we simplify the definitions of utility and alternative water purchase losses: in applications of the framework in the other contexts, the components of these losses must be considered. For example, many utilities lose tariff revenue during drought-induced water shortages, due to a lower overall volumetric sale of water; in such cases, lost tariff revenue should be incorporated into the definition of utility losses. Further, we focus on direct financial losses, omitting more intangible, indirect losses such as the time spent on acquiring alternative water sources and associated losses in productivity, or wellbeing impacts.

Another aspect of our approach to consider is that utilities typically consider a much broader portfolio of infrastructure investments than those considered in this paper, such as piped network construction or supply capacity enhancement projects. We do not consider options that are proposed to increase the number of household connections to the network as, although they may also improve the resilience of the whole network, it is not possible to disentangle this from the potentially negative effects of increasing demands on the relevant system and greater societal exposure to disruption. Rather, we limit the scope to two option types to focus solely on climate adaptation options.

Third, due to data constraints, here we do not account for the uneven levels of service, or rate of intermittency, provision across systems. This information could, in subsequent applications, be harnessed to better understand the vulnerability of households to disruptions of different durations. For example, the less frequently households receive water supply, the less likely they would be able to store a sufficient amount to buffer against disruptions. Given that many poor communities are often burdened with greater levels of supply intermittency, the uneven

levels of vulnerability that that exist across water supply networks, could be incorporated into the underlying CDD risk metric in future applications.

Despite the limitations of the framework, our approach highlights the importance for stakeholders to capture multiple hazards and financial risk transmission pathways when evaluating what measures would offer the most cost-beneficial climate resilience outcomes. The framework also takes an early step towards mainstreaming equity considerations in potable sector decision making, to identify co-benefits and trade-offs between economic- and equity-related objectives.

5.5. Concluding remarks

Through the delivery of potable water supply services, utilities are responsible for among the most basic services for humanity. Climate-related disruptions to water supply services incur direct financial losses to utilities (e.g., to repair damaged assets) and temporarily deprive households, imposing a financial burden in acquiring water from alternative water sources. This uncompensated social burden, which is particularly high for low-income households that have less disposable income to spend on alternative supply (e.g., from tanker trucks), should be properly accounted for in investment planning.

Here we show that adapting potable water supply networks to climate change would not only mitigate annual asset damage losses for utilities, but also improve the affordability of safe water services and protect the most vulnerable domestic customers from climate extremes. This speaks to a wider discussion on the interlinkages between climate adaptation and overall development and demonstrates the synergy between the two in the case of potable water investments.

We draw three main policy recommendations from this work. 1) It is necessary to capture the costs of customer disruptions, and distributional effects across income groups, in water supply optioneering frameworks to promote the multi-dimensional benefits of climate adaptation options. 2) Utilities that are exposed to multiple climate extremes require an integrated approach for directly comparing the risk mitigation potential of options that adapt to different hazards. 3) Leakage reduction is a low-regret, climate adaptation option for water supply utilities to improve overall operational efficiency and reduce the risk of customer disruptions.

Acknowledgments

The authors declare no conflict of interest. O.B. acknowledges funding from the Engineering and Physical Sciences Research Council (EPSRC) under grant number EP/T517811/1.

The authors would like to thank the National Water Commission for both the data used in this work and for the meetings and ongoing communication throughout this work. We would also like to thank the Systemic Risk Assessment Team and the Planning Institute of Jamaica for supporting and coordinating this data exchange and ongoing collaboration between the Oxford team and the National Water Commission.

Open Research

Data provided by colleagues at the National Water Commission - including Water Supply Parish Plans and Capital Improvement Plan - is sensitive information. Other users much contact the utility (https://www.nwcjamaica.com/) or the National Spatial Management Branch in Jamaica (https://jis.gov.jm/government/agencies/national-spatial-data-management-division/) for this data. Software used for the project include Python (https://www.python.org/) and QGIS (https://www.qgis.org/en/site/), all of which are open-access.

6. Closing the climate adaptation gap for water supply utilities

This chapter presents a framework for evaluating utility level drought risk and the effectiveness of drought mitigation options, including increasing desalination capacity, reducing leakage losses and increasing storage capacity. The framework is applied to ~5,600 utilities globally.

Paper: Becher, Verschuur, Smilovic, Pant, Tramberend and Hall (in review with Nature Communications Earth and Environment [manuscript status is 'Accept in Principle', conditional on meeting editorial requests]), *Closing the climate adaptation gap for water supply utilities.*

Co-author contributions: Prof Jim Hall from the University of Oxford assisted with the conceptualisation and structuring of the paper, as well as with review and feedback on the manuscript. Dr Mikhail Smilovic from the International Institute of Applied Systems Analysis provided the Community Water Model inputs and advised on their methodological application for the paper. Dr Raghav Pant and Dr Jasper Verschuur from the University of Oxford and Dr Sylvia Tramberend from the International Institute of Applied Systems Analysis assisted with review and feedback on the manuscript.

Key points:

- Undertaking drought risk assessments of water supply utilities globally enables the evaluation of adaptation actions in the potable water sector.
- Leakage reduction is found to be the most cost-effective adaptation action for most utilities compared to desalination and increased storage, reducing both operational costs and water shortage risks.
- Although implementing the considered adaptation actions would have a material impact on reducing present and future climate risks to society, challenges around attracting the necessary financing, among other challenges, persist.

Abstract

Many drinking water utilities face immense challenges in supplying sustainable, drought-resilient services to households. Here we propose a quantified framework to perform drought risk analysis of ~5600 potable water supply utilities and evaluate the benefit of adaptation actions. We identify global hotspots of present-day and mid-century drought risk under future scenarios of climate change and demand growth (namely, SSP1-2.6, SSP3-7.5, SSP5-8.0). We estimate the mean rate of unsustainable or disrupted utility supply at 15% (interquartile range, 0-26%) and project a global increase in risk of between 30-45% under future scenarios. Implementing the most cost-effective adaptation action identified per utility would mitigate additional future risk by 75-80%. However, implementing the subset of cost-effective options that generate sufficient tariff revenue to provide a benefit-cost ratio that is greater than 1 would only achieve 5-20% of this benefit. The results underline the challenge of attracting the financing required to close the climate adaptation gap for water supply utilities.

6.1. Introduction

Water supply utilities are responsible for among the most basic services for humanity. However, their services to households are highly vulnerable to the impacts of climate change due to their dependence on variable water resources (Danilenko, 2010). Drought-induced disruptions to water supplies (hereafter: 'drought-induced disruptions') are commonplace in many parts of the world, particularly in low- and middle-income countries where water supply infrastructure typically lacks the capacity and storage to buffer against hydrological variability (Grey and Sadoff, 2007; Hall *et al.*, 2014; Bakker, 2017; Salehi, 2022). Drought-induced disruptions are also of increasing concern in higher income countries where water supply infrastructure depends upon over-exploited water resources and may compete with agricultural and industrial (including energy) water users (Van Loon *et al.*, 2016; Di Baldassarre *et al.*, 2018; Flörke, Schneider and McDonald, 2018). The risk of water shortages is expected to increase under future climate change and growth scenarios (Pokhrel *et al.*, 2021), requiring the development of more resilient utility-operated infrastructure networks.

However, many utilities, particularly in low- and middle-income countries, are caught in a vicious cycle of disrupted services, low rates of tariff collection and inadequate investments in infrastructure resilience (Briscoe, 1999; Rouse, 2013; Libey, Adank and Thomas, 2020). Utilities that cannot cover operational and reinvestment costs typically have lower creditworthiness, hindering their ability to attract finance and leading to a reliance on government subsidies (Winpenny, 2015). This has contributed to a large and widening infrastructure gap in many countries (Hutton and Varughese, 2016; OECD, 2022). Previous studies estimate the gap between current water infrastructure investment flows and what would be required to keep pace with population growth and the SDG targets (Ward *et al.*, 2010; Hutton, 2013; Winpenny, 2015). However, the level of investment required to improve the climate resilience of utility water supplies at a global scale has rarely been the focus of previous studies. Failure to integrate climate risk and resilience considerations into water sector development may undermine the financial performance of utilities, hindering the extension of network of services, and could lock-in development in hazard-prone areas, leading to maladaptation.

Drought risk analysis methods provide useful insights into meteorological and hydrological variability across space under present day conditions (Carrão, Naumann and Barbosa, 2016; Hammond *et al.*, 2022) and future scenarios of climate change and socioeconomic growth

(Milly *et al.*, 2008; Pokhrel *et al.*, 2021; Duan *et al.*, 2022; Vicente-Serrano *et al.*, 2022). Research on drought-related risk analysis for water supply utilities has largely been undertaken for individual utilities (Herman *et al.*, 2015; Borgomeo *et al.*, 2016; Kingsborough, Borgomeo and Hall, 2016; Li *et al.*, 2019). On the other hand, analyses of global drought risk to water supply are often conducted at the catchment scale, without capturing utility-level characteristics such as leakage losses, service areas and desalination inputs (McDonald *et al.*, 2011; Hanasaki *et al.*, 2013; Padowski and Gorelick, 2014; He *et al.*, 2021). Capturing this information is essential for understanding the propagation of meteorological/hydrological droughts through utility networks to dependent households. The appropriate scale for water management has long been discussed and debated (Vörösmarty *et al.*, 2015), however, so far, the utility-scale has not been considered as an appropriate unit for analysis. This has limited the rigour with which previous studies have identified adaptation benefits for the potable water sector, i.e., in terms of risk reduction potential (Wada, Gleeson and Esnault, 2014; Larsen *et al.*, 2016; Buurman, Mens and Dahm, 2017; Greve *et al.*, 2018; Krueger, Rao and Borchardt, 2019; He *et al.*, 2021). As such, there is large research gap in understanding utility-level climate risks, which could help inform policymakers in terms of terms of the most appropriate solutions to mitigate this as well as the finance required to fund adaptation needs of utilities globally.

Here, we propose a risk assessment framework to 1) perform drought risk analysis of 5600 water supply utilities at present and under future climate change and 2) evaluate the benefit (i.e., risk reduction potential) of alternative infrastructure investment opportunities to identify the optimal option type per utility. A model of each utility's water balance that represents supply inputs (from surface water, renewable groundwater, and desalination), municipal water usage and leakage losses is established to estimate the risk of utilities being unable to supply their full customer base sustainably in terms of Unsustainable Water Supply Days (UWSD). UWSD describes number of days in a year during which a utility's supply is unsustainable (i.e., utilities tap non-renewable groundwater supply) or disrupted due to drought (i.e., demand restrictions and/or reallocation strategies are implemented). We estimate tariff revenue at risk by multiplying UWSD by the daily equivalent tariff revenue per capita per utility. By comparing the impact of three alternative infrastructure interventions on UWSD and associated tariff revenue at risk against each option's cost, discounted over its lifespan, we estimate the Benefit-Cost Ratio (BCR) of each option type and identify the most cost-effective intervention per utility.

We find a global UWSD of 510 million per year, increasing by 30-45% under future scenarios between 2030-2060. Countries with the highest risk utilities include Botswana, Ecuador, Mexico, Morocco, South Africa and Pakistan. 90% of the utilities currently at risk from water shortages would benefit from at least one of the three adaptation options considered, with desalination, leakage reduction and increased storage capacity offering the most cost-effective option for 10%, 60% and 30% of all considered utilities, respectively. Implementing the most cost-effective climate adaptation action per utility would mitigate ~75-80% of future increase in risk levels. However, implementing options with benefit-cost ratios>1 would achieve only ~5-20% of this benefit and would skew benefits towards higher-income countries that typically set higher tariff rates.

6.2. Results

6.2.1. Present-day and future utility drought risk

Water supply at a utility level depends on the renewable sources available within their utility service area (renewable groundwater, river discharge and lake and reservoir storage), long-range water transfers and desalination inputs. Here we quantify the balance between available supply inputs at the utility level against customer demand, accounting for distribution losses through leakage (see 6.4. Methods). We quantify the risk of utilities being unable to supply their full customer base sustainably, i.e., without tapping non-renewable groundwater or implementing demand restrictions/reallocation strategies, in terms of UWSD. We employ this modelling approach to infer utility-level changes in risk under three future scenarios (SSP1-2.6, SSP3-7.0 and SSP5-8.5), combining plausible futures in terms of climate change and socioeconomic growth between 2030-2060.

48% of all modelled utilities are subject to present day drought-induced water supply disruptions (i.e., have an annual expected risk greater than zero). We estimate the global UWSD risk to be of 510 million per year, and the median utility to be subject to 15 days of unsustainable or disrupted supply under present day conditions. Figure 24a provides the annual expected UWSD as a fraction of each utility's customer base multiplied by the number of days in a year. Countries with the highest risk utilities include Botswana, Ecuador, Mexico, Morocco, South Africa and Pakistan.

Overall, variations in water stress and total hydrological supply variability are the main drivers of risk. The influence of water stress and hydrological variability on risk varies across regions and climate scenarios, as shown in Supplementary Figure D1. Note that our UWSD metric captures the tapping of non-renewable groundwater as a disruption, which explains why risk hotspots align with areas that depend on unsustainable groundwater abstraction, for example in North Africa, Western USA, Mexico, Northern India, the Arabian Peninsula, and Northern China (Gleeson *et al.*, 2012). Further, our UWSD metric does not capture contingency measures implemented by managers during or preceding acute drought-induced disruptions, such as imposing demand restrictions and reallocating flows from farms to cities; practices that are well-established in California (Mini, Hogue and Pincetl, 2015), for example. This is to reflect the reality that, in many parts of the world, these practices are not mainstreamed or fail-safe (Marston and Cai, 2016) due to a lack of enabling physical and operational infrastructure.

Thus, our results reflect the worst-case scenario should no inter-sectoral/reallocation or demand management measured be implemented. Further, demand restrictions can come at a cost to the quality of life of customers, particularly during drought conditions which are often coupled with extreme heat events (Libonati *et al.*, 2022). Thus, our results on the effect of adaptation actions on reducing UWSD, also reflect the benefit of adaptation actions on the livability of cities.

Between 70-85% of at risk utilities are subject to increased risk in the future, depending on the choice of future scenario, increasing the global total UWSD by 30-45% between 2030-2060. Figure 24b shows the hotspots of elevated risk under the middle-of-the-road SSP3-7.0 projection. Future risk is driven by increasing hydrological variability and water stress (Supplementary Figure D1), the latter, in turn, being driven by decreasing supply and increasing demand. Countries with utilities that are projected to be subject to the highest relative increase in risk in the future include the Philippines, Mozambique, India, Kenya, Australia and France, driven mainly by increasing water stress. However, there is substantial variation in future risk changes within countries (e.g., in China and India). This highlights the importance of local, utility-level characteristics (such as leakage losses and access to desalination) in determining their sensitivity to regional shifts in climate and hydrological processes. Figure 24b shows a clear negative risk signal in parts of Europe and Japan, where population growth is expected to decline across scenarios by the mid-century under SSP7.0 (World Bank, 2021). The change in future UWSD outputs per utility across other considered future scenarios is provided Supplementary Figure D2. The variation across considered future scenarios is driven more by changes in demand than by changes in supply, see Supplementary Figure D3, with population growth inputs being the dominant source of uncertainty, see Supplementary Figure D4.

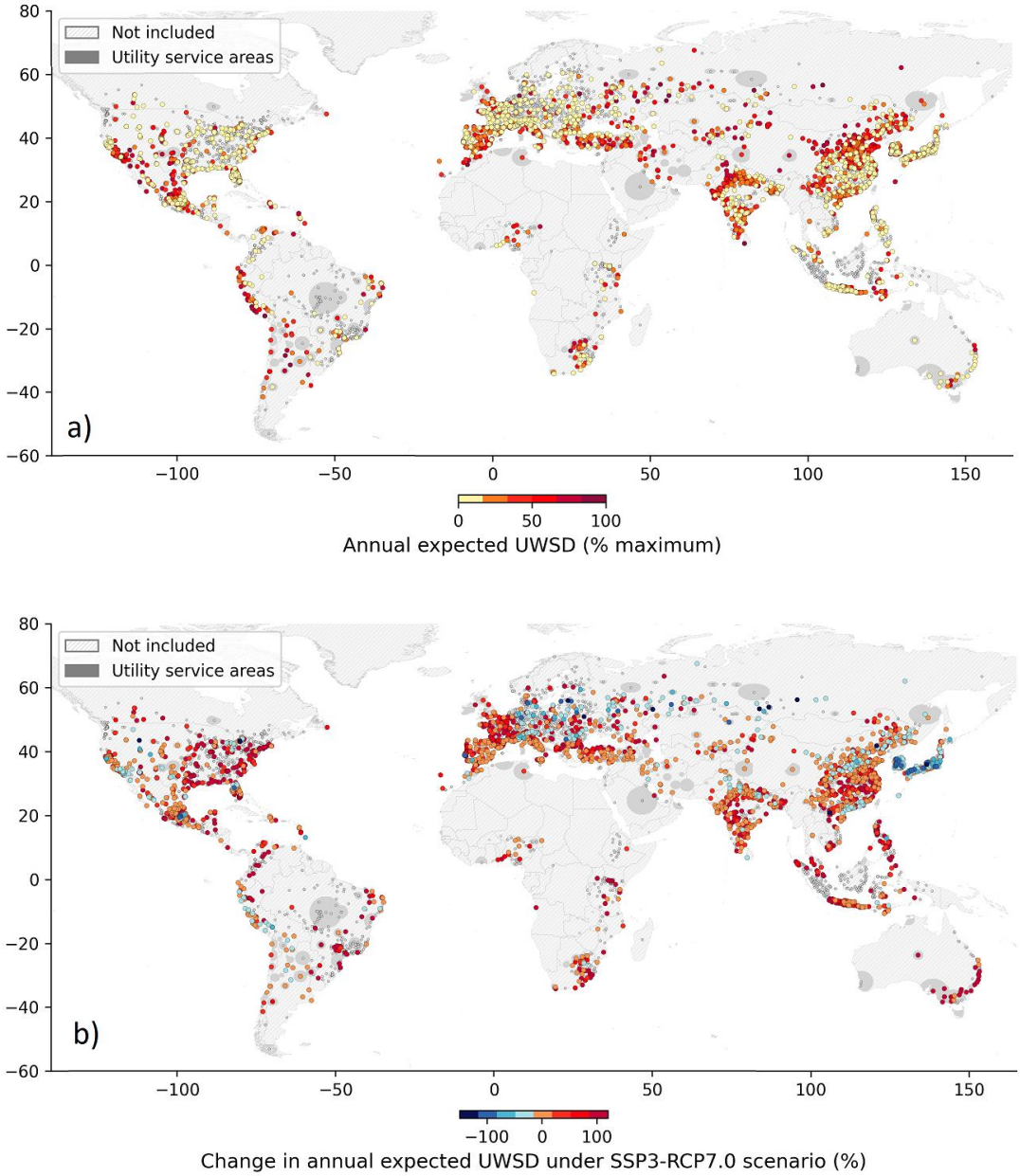


Figure 24 (a) Present day annual expected drought risk per utility (utility headquarter location plotted) measured in terms of the annual expected UWSD per utility as a fraction of a year-round disruption of their full customer base. (b) Change in future annual expected UWSD - projected under the middle-of-the-road SSP3-RCP7.0 scenario - relative to the present-day annual expected UWSD. Areas that are not covered by the utility database used for our analysis are indicated with hatching.

6.2.2. The business case for adaptation

We utilise the risk framework to evaluate the impact of alternative climate adaptation actions, including increasing water storage capacity, desalination, and reducing leakage losses, on

UWSD and associated tariff revenue at risk. By comparing the financial benefits (in terms of reduced tariff revenue at risk) and costs associated with a given option, discounted over its lifespan, we estimate the benefit-cost ratio (BCR) of each option type and identify the most cost-effective intervention per utility.

90% of the utilities currently at risk from water shortages would benefit from the three adaptation options considered. The remaining 10% of utilities are too water stressed and/or are too far from the coast for desalination to benefit from the maximum protection level we define for each adaptation option type considered in this study. For example, we assume that the maximum desalination and reservoir storage adaptation option design capacities cannot exceed 90% of utility customer demand, and leakage rates cannot fall below 10% of demand, see Supplementary Table D1. For these utilities, alternative adaptation options would have to be implemented, such as reducing upstream or customer demand. Implementing the most cost-effective, or utility-optimal, adaptation intervention for each of these utilities (i.e., the option with the greatest BCR) (Figure 25a) would mitigate ~75-80% of additional future risk levels. The total estimated net cost (including reduced operational costs thanks to leakage reduction) of all utility-optimal adaptation options equates to 20 billion USD per year, discounted over a 30-year asset lifetime, equating to a quarter of the total annual tariff revenue received by benefitting utilities. As we have only considered the costs of implementing 1 optimal option (where utilities would typically seek to implement a portfolio of climate adaptation options to reduce risk to what managers and operators deem to be a tolerable level (Borgomeo *et al.*, 2016)) and employ an incomplete utility database, this cost does not represent the global climate adaptation investment needs in water supply utilities.

Overall risk and the risk mitigation potential of adaptation varies across regions, as summarised in Figure 25b. South Asia and Sub-Saharan Africa regions have the greatest fraction of water supply utility customers at risk, with climate change and growth projections driving the greatest relative increase in risk in the Middle East and North Africa and Sub-Saharan Africa. Water stress is the main driver of risk in South Asia and Sub-Saharan Africa, leading to the greatest fraction of customers at risk on a chronic basis, see Supplementary Figure D1. Future projections point to declining water supply trends and increasing demand in the Middle East and North Africa and Sub-Saharan Africa, as shown in Supplementary Figure D3, increasing levels of stress and driving the relative increase in risk. Further details on the specifics of each scenario of climate change and socioeconomic growth are provided in Methods, under section

‘6.4.3.1. Drought hazard data’. Residual risk after implementing the optimal adaptation option falls to or below present-day risk in East Asia and the Pacific, Europe and Central Asia and Latin America and the Caribbean. Whereas in the Middle East and North Africa, Sub-Saharan Africa, North America and South Asia, the optimal option is insufficient to reduce risk to present day levels: a portfolio of adaptation options would be required to mitigate future increase in risk. This is substantially influenced by regional trends in population growth: regions including the Middle East, North Africa, Sub-Saharan Africa and South Asia have high population growth up to mid-century, making it far harder for considered adaptation options to mitigate future risk below current levels. We find that implementing all utility-optimal options would also achieve a total reduction in tariff revenue at risk of 5-25 billion USD per year and, therefore, an overall BCR of 0.4-1.4. However, implementing options with BCRs>1 would achieve only ~5-20% of this benefit. Options with BCRs>1 are concentrated in countries that set higher tariff rates, which is typically the case in higher income countries such as in North America and Europe and Central Asia, which is why the difference between residual risk and residual risk only with options with BCR>1 is lower than for other regions in Figure 25b. See the relationship between unit tariff rates and GDP per capita in Supplementary Figure D5.

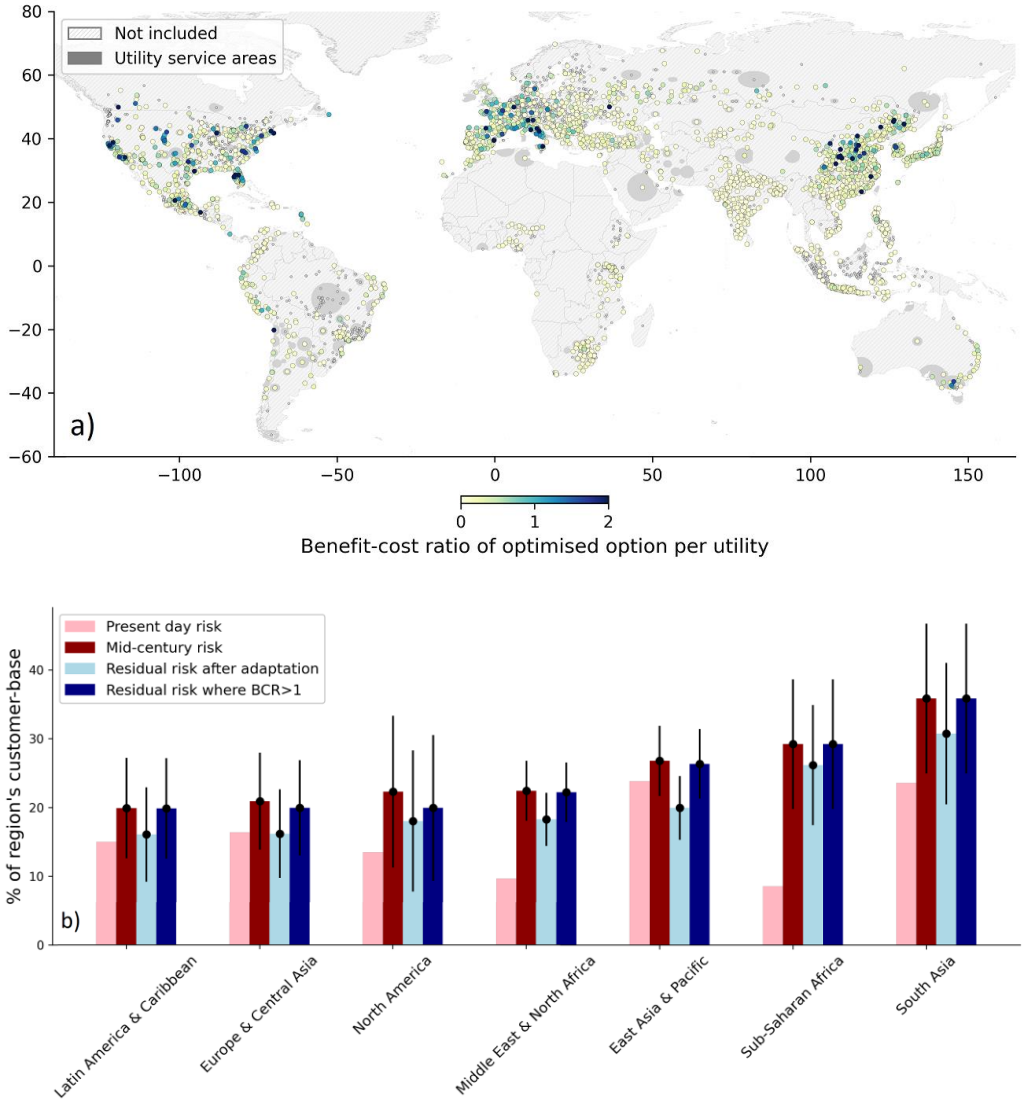


Figure 25 (a) (a) BCR of optimal option identified per utility. BCR is measured as the mitigated tariff revenue at risk divided by the cost of each adaptation option. Utilities that were found not to benefit from adaptation are plotted in grey. Areas that are not covered by the utility database used for our analysis are indicated with hatching. (b) Present day annual expected risk, measured as the percentage of the region's customer base at risk (in the present day), is plotted in light pink. Future risk (calculated as a mean of considered future scenarios) is plotted in dark red. The residual risk that could be achieved by implementing all utility-optimal adaptation options is plotted in light blue. The residual risk that could be achieved only by adaptation options with BCRs>1 is represented in dark blue. Error bars indicate the range of results produced across considered future scenarios. Note that all future and residual risk values are taken as an average over the 2030–2060-time horizon projected for the adaptation analysis.

6.2.3. Optimal adaptation investment

Figure 3a shows the optimal option type identified per utility and Figure 3b shows the regional breakdown of preferred option types. Error bars in Figure 3b reflect the uncertainty around in our selection of each utility's optimal option type. By factoring option costs for each option type in turn (e.g., desalination) by -/+25%, we compare the minimum/maximum BCR against the maximum/minimum BCR of the other two options (e.g., storage and leakage reduction) (factoring costs by +/-25%) and quantify the change in proportion of utility-optimal option type selected per region. For 60% of all utilities, leakage reduction would be the most cost-effective means of reducing water shortages. Indeed, the average utility loses more than a quarter of its distributed water supply (GWI, 2022b) and improving efficiency would reduce the costs of disruptions as well as the costs of operations. This makes reducing leakage a highly attractive and necessary climate adaptation investment opportunity, despite frequently being overlooked in research (Wada, Gleeson and Esnault, 2014; He *et al.*, 2021) and (in some cases) practice, in favour of supply-side options. Leakage reduction is most frequently selected to be the most cost-effective option across all but two regions, particularly in Latin America and the Caribbean and South Asia where leakage rates are relatively high (GWI, 2022b). For most coastal utilities, and 10% of utilities overall, desalination is the highest priority option, even when the high operation costs of desalination are considered. Desalination is the preferred option for the highest fraction of utilities in the Middle East and North Africa, aligning with trends in the market growth for desalination in this region (GWI, 2022a), as well as East Asia and Pacific, North America and Europe and Central Asia. Greater storage is found to be a priority option in the remaining 30% of cases, where there is sufficient excess surface water available to capture and distribute. Reservoir storage is the preferred option for the highest fraction of utilities in the Middle East and North Africa and Sub-Saharan Africa, where countries typically have relatively low reservoir storage volumes per capita (Lehner *et al.*, 2011).

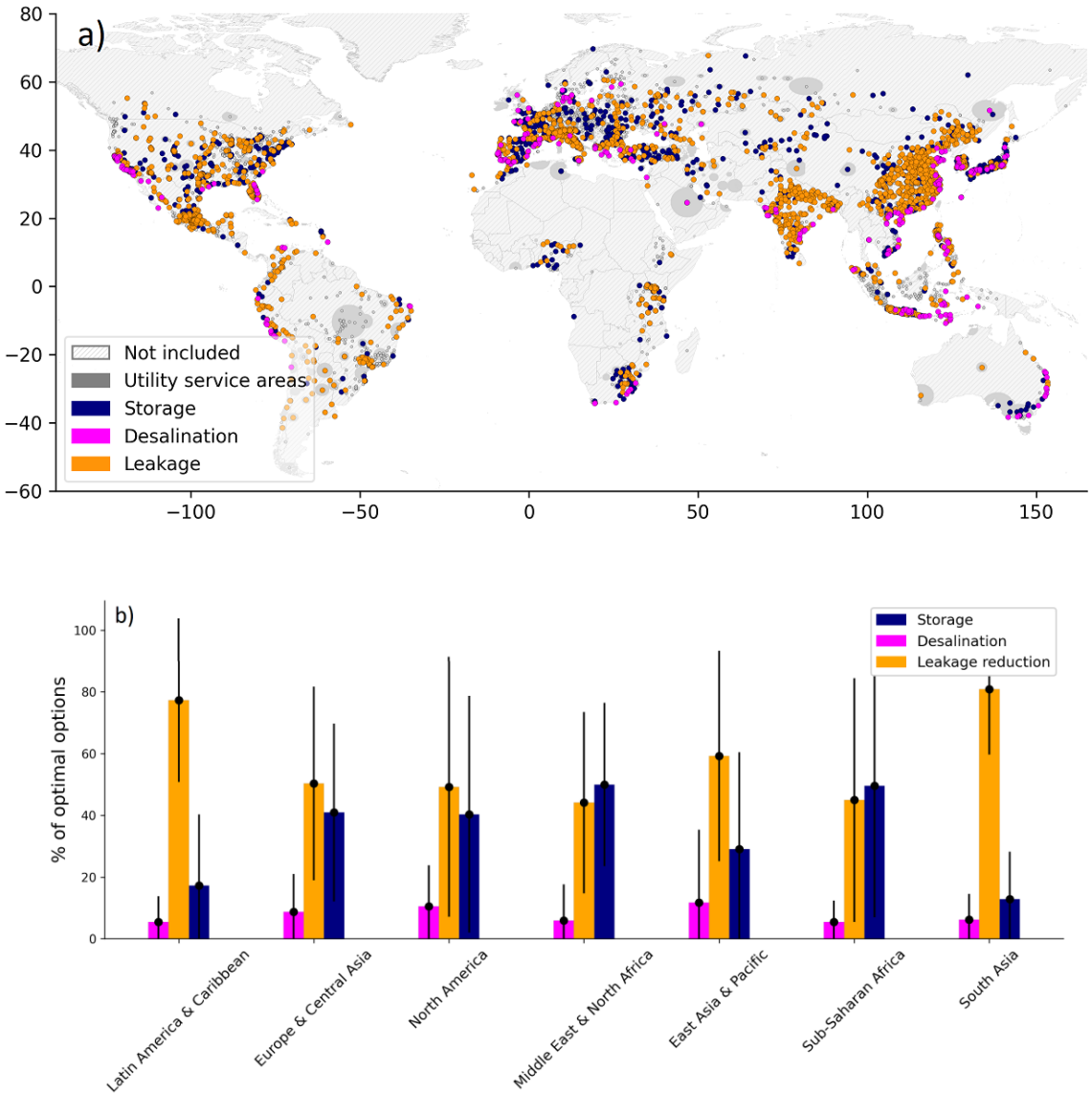


Figure 26 (a) optimal option type identified per utility. Utilities for which the adaptation options considered were ineffective at reducing risk are plotted in grey. Areas that are not covered by the utility database used for our analysis are indicated with hatching. (b) % fraction of all utilities where optimal option is identified to be storage (navy), desalination (pink) and leakage reduction (orange) per region. Error bars represent the uncertainty in the optimal option type selection, induced by uncertainty in each option type's cost.

6.3. Discussion

By implementing a generalised framework to perform drought risk analysis under present-day and future scenarios, we have, for the first time, evaluated the risk-reducing benefits and costs of alternative climate adaptation investments for water supply utilities at a global scale.

Our drought risk formulation reflects the combined effects of chronic water stress, hydrological variability and utility-level characteristics (e.g., leakage losses). Previous global drought risk studies typically employ drought hazards metrics that describe hydrological variability (e.g., through the Standard (Effective) Precipitation Index (Spinoni *et al.*, 2019)) or water stress, and intersect this with a geospatial representation of the exposed population (Blauhut *et al.*, 2016), (He *et al.*, 2021). The problem with such approaches is that drought hazards metrics do not necessarily predict the likelihood of physical drought impacts (Bachmair, Kohn and Stahl, 2015). The strength of hazard-impact relationships varies across sectors (Stagge *et al.*, 2015), however, in the potable water sector, this relationship is strongly mediated by the resilience of infrastructure systems. Further, they do not provide a platform to quantify the benefits of adaptation measures in quantified terms using tangible, meaningful metrics for decision makers. As such, existing global analysis of drought risk to, and adaptation planning in, the municipal water sector is limited for two main reasons: 1) lack of consideration of utility scale of operations and infrastructure characteristics, 2) lack of rigorous, risk-based approaches for identifying benefits of adaptation options and optimal investments. Speaking to the first limitation around scale, previous studies typically adopt the catchment as the unit scale for drought analysis (McDonald *et al.*, 2011; Hanasaki *et al.*, 2013; Padowski and Gorelick, 2014; He *et al.*, 2021). However, this scale does not necessarily speak to the way that water is managed by utilities. Speaking to the second limitation around risk-based approaches, existing studies have indicated the suitability of adaptation options across geographies based on qualitative criteria (Wada, Gleeson and Esnault, 2014; Larsen *et al.*, 2016; Buurman, Mens and Dahm, 2017; Greve *et al.*, 2018; Krueger, Rao and Borchardt, 2019; He *et al.*, 2021) or a long term balance between supply and demand (Ward *et al.*, 2010). However, there is scope to undertake more rigorous risk analysis procedures, inspired by approaches typically undertaken at a smaller scale (Brown *et al.*, 2012; Borgomeo *et al.*, 2014), to prioritise options based on their risk reduction potential.

Our analysis is designed to illustrate the distribution of present-day risks to municipal water supplies and how this may increase in the future, serving as a platform for adaptation planning.

We use UWSD to estimate tariff revenue at risk, however, we acknowledge the wider economic and social implications associated with water supply disruptions. For example, utilities often serve industrial and commercial entities, disruptions to which may result in economic losses (Freire-González, Decker and Hall, 2017; Jenkins *et al.*, 2021). Further, customers often suffer financial, welfare and productivity losses in acquiring water supplies from alternative sources (e.g., bottled water or tanker trucks) (Baisa *et al.*, 2010; Buck *et al.*, 2016). The costs of acquiring alternative water sources are often far beyond what the relative cost of receiving water from a functioning piped water supply system: for example, in Ghana, water distribution by road tanker costs 14 times as much as distribution through the pipe networks (Rouse, 2013). This may represent a significant additional household expense, particularly for low-income households (Pattanayak *et al.*, 2005). Indeed, one study undertaken in Kathmandu, Nepal, analysed the costs of substituting disrupted water supply with alternative sources (including tanker trucks and bottled water) relative to household income, to found that this cost represented 56% and 34% of overall coping costs for poor and rich households, respectively (Pattanayak *et al.*, 2005). Further work is required to quantify the impacts of these losses across geographies, particularly for customers in low-income groups. We note that residential per capita demand is typically higher in more developed countries, as greater volumes of water are used for non-essential purposes, such as lawn irrigation (FAO, 2020). Therefore, in high-income countries, disruption events would include demand restrictions for less essential purposes, such as hosepipe bans. Whereas, in lower income countries with lower residential demand per capita, disruption events are likely to entail cuts in relatively more essential water uses, such as for drinking, cooking and hygiene applications.

We find desalination to be the most cost-effective option for 10% of utilities for which an adaptation option was found to be effective. Indeed, the market for desalination is expected to double over the next decade (GWI, 2022a). Desalination is typically less disruptive in many places compared to reservoir construction or expansion. Further, desalination provides an opportunity to tap an infinite source of supply that is not susceptible to droughts, unlike reservoir storage and leakage reduction measures. However, for many utilities, or systems within utilities, desalination would not be considered cost-efficient due to geographical location far from the sea (Stillwell, King and Webber, 2010). Further, best practices would deem it necessary to address network efficiencies before investing in desalination plants, given the environmental impacts of brine effluents and high energy requirements, particularly in countries located on semi-enclosed seas with less potential for brine dispersal and with a high

reliance on fossil fuel-dependent energy generation (Lattemann and Höpner, 2008; Panagopoulos, 2021). For quantified risk and optioneering assessments, it is challenging to capture intangible constraints, such as social acceptance, to the implementation of certain options. However, given the large and growing number of people dependent on desalinated water with, unlike treated wastewater, very little social objection, we don't consider social acceptance to present a major barrier to coastal utilities adapting to climate change through harnessing supply through desalination, particularly in water stressed areas. We estimate operational energy costs based on national average data; however, many desalination plants have dedicated, stand-alone energy generation capacity. Thus, we do not account for the full costs of supplying electricity for desalination in our model. On the other hand, minimising leakage offers a two-fold benefit of adaptation and mitigation, by improving network resilience to droughts and reduction energy consumption. It is, therefore, an important first step for utilities with high inefficiencies. However, reducing leakage losses can only go so far in future-proofing utilities to future water stress and drought risk and the costs involved in reducing leakage are also highly variable and context-specific. Desalination and reservoir storage can offer utilities a chance to not only improve resilience of existing networks, but also to expand network access to serve unserved areas.

The analysis has illustrated that there is a strong business case for adaptation intervention in many, but not all locations, in terms of the avoided tariff revenue loss thanks to reduced UWSD. In places where interventions are justified in narrow CBA terms, further examination is required around the barriers to investment. In places where adaptation is not justified on narrow cost-benefit terms, there needs to be wider consideration of the economic and social case for resilient water supplies (e.g., reduced health care costs and time savings for customers (Hutton, 2013)), and exploration of appropriate forms of finance, e.g., concessionary finance.

Water supply utilities face financing problems in most parts of the world (Winpenny, 2015). Reflecting on the growing importance of climate adaptation finance in the banking sector, there is an opportunity to capitalise on this alignment between investor-incentives and climate adaptation needs in the water sector (Ranger and Mullan, 2022). Climate-resilient water supply utilities provide a material contribution towards adaptation goals by managing variable hydrology and climatic uncertainty to provide safe, reliable, and affordable drinking water services to households. This makes the water sector an important channel for climate adaptation finance. However, a range of barriers exist in attracting the necessary financing for capital

improvements and in incentivising adequate levels of network maintenance, the latter being particularly important for achieving long term reductions in leakage losses to strengthen the operational, financial and climate resilience of services. There is no single solution in overcoming these barriers; financing must be coupled with technical capacity building and institutional reform activities. However, risk quantification and adaptation optioneering frameworks can be useful tools for decision makers to allocate limited financial resources and demonstrate the impacts of investments towards the global climate adaptation agenda. Quantifying the impact of investments towards the climate adaptation agenda can also be effective in unlocking concessional financing from the proliferation of climate funds and investors. Our analytical framework provides a valuable starting point for quantifying the contribution of adaptation investments in the potable water sector towards the global climate adaptation agenda.

Although we focus on three hard infrastructure options (leakage reduction, reservoir storage and desalination) these options are not necessarily the most cost-effective in all cases. Soft options, such as inter-sectoral reallocation (Marston and Cai, 2016; Garrick, De Stefano, *et al.*, 2019) or drought mitigation-targeted nature-based solutions, such as managed aquifer recharge (Dillon *et al.*, 2019), could be the focus for future developments of this framework. Indeed, in many contexts, in both developed and developing contexts, soft options may be preferred over hard infrastructure options. In lower-income countries, soft options are typically cheaper and, therefore, more feasible (World Bank, 2019). In a higher-income country context, many countries already have heavily engineered water supply systems, with large volumes of storage, low leakage losses and access to desalination plants. These countries may be approaching the limit to which further investments in hard infrastructures would be cost-effective (Del Moral Ituarte and Giansante, 2000). Further, social acceptability can also be a limiting factor in infrastructure development, particularly reservoirs that displace communities; managed aquifer recharge could be an alternative storage solution to be explored in future applications of the framework.

Of course, the viability of utility service provision may be influenced by climate-related variations in water quality as well as quantity. Water pollution and turbidity can exceed the treatability limits for the production of municipal water supply (Mortazavi-Naeini *et al.*, 2019; Nyman *et al.*, 2021) or may require increased dilution (Van Vliet, Florke and Wada, 2017), potentially leading to water supply disruptions. The reason we do not consideration of water

quality risk in the current framework is that utilities' treatability thresholds are highly variable globally. For example, in many countries water treatment processes are highly sensitive to turbidity compared to others where processes are more robust to turbidity variations. The risks that climate-related water quality variations pose to water supply services globally could be explored in future research.

As with any large-scale and long-term assessment, we acknowledge a range of uncertainties associated with the analysis and affecting the results. The accuracy of the global hydrological model is constrained by the quality of underlining global datasets and limited availability of data for calibration. Further, hydrological projections under climate change up to the end of the century are subject to uncertainties in climate models' ability to accurately predict population growth, land use changes and future changes in precipitation and evapotranspiration (Burek *et al.*, 2020). We have applied a general rule that utilities would source water from within their service area, unless sources are explicitly included in the utility water sources database (McDonald *et al.*, 2014). Further, we assume that all water supply within the utility service area would be available, whereas, in reality, utility water supply is constrained by the capacities of their respective infrastructures. Another source of uncertainty is in the comprehensiveness of the utility dataset, which is particularly limited in certain geographies such as in Africa, though this also reflects the lesser coverage of utilities in the less developed world.

Despite these limitations, our approach provides an early step towards achieving a global drought risk assessment of household water supply under present day and future climate change scenarios, adopting the utility scale as the unit for analysis to support climate adaptation planning. The framework provides a platform for identifying the most cost-effective infrastructure investment pathways for water supply utilities by quantifying the contribution of adaptation towards metrics that would be relevant to both investors and/or utilities.

Given the exposure and vulnerability of their services to climate hazards, water utilities are uniquely positioned to deliver climate adaptation services for society. However, the importance of water utilities in strengthening society's climate resilience is disproportionate relative to the attention received at the high-level dialogue on climate finance. With the climate agenda rapidly gaining momentum, there is an opportunity to strengthen recognition that the utilities receive in high level dialogue on climate finance and channel greater financial flows towards closing the climate adaptation gap for water supply utilities.

6.4. Methods

6.4.1. Overview

Here, we combine a global hydrological modelling approach with local utility information, which allows us to identify global water utility climate risk and adaptation needs and explore the cost-effectiveness of alternative adaptation option types. We do this by constructing a water balance between supply and demand for 5,600 water supply utilities under present day and three future scenarios available from the ISIMIP (Warszawski *et al.*, 2014) and CMIP6 (O'Neill *et al.*, 2016) experiments (combined RCP (van Vuuren *et al.*, 2011) and SSP (Riahi *et al.*, 2017) scenarios between 2030-2060). This allows us to quantify the risk of utilities being unable to supply their full customer base sustainably. We define this risk in terms of Unsustainable Water Supply Days (UWSD); a metric describing number of customer-days where utility supply is unsustainable (i.e., renewable water supply is insufficient to meet a utility's demand) or disrupted (i.e., where implementing demand restrictions and/or reallocation strategies become necessary). Our metric effectively combines the concept of reliability (percentage of time an asset/system is operational) (Brown *et al.*, 2012) and exposure (people affected) (He *et al.*, 2021) into one metric, improving its relevance for decision-making. Apart from the UWSD, we also quantify the associated tariff revenue at risk caused by each supply disruption based on utility-level tariff rates. We use this framework to simulate the potential benefit of three alternative infrastructure interventions: namely, increased water storage capacity, desalination, and leakage reduction. By simulating the impact of alternative adaptation interventions on the risk over each project's lifespan, we quantify the cost effectiveness of each option type in terms of mitigated UWSD and mitigated tariff revenue loss. Per utility, the optimal (or most cost-effective) option type is identified, as well as the overall benefit that it achieves.

6.4.2. Water supply utility database

Global data on the headquarter location, population served (P) and non-revenue water rates of approximately 5,600 water supply utilities offering water supply services was provided by Global Water Intelligence (GWI, 2022b). We estimate that this covers over 60% of the global population supplied by piped water estimated by the Joint Monitoring Programme for the year 2019 (JMP, 2017). The data coverage is not uniform across the globe, as shown in the map of the rate of utility data coverage per country in Supplementary Figure D6.

A water supply utility is a formal entity that supplies water to households, commercial or industrial entities. Utility water supply contrasts to sourcing water supply from a private well, tanker truck, surface water, springs or other sources, for private, industrial or agricultural use. Some utilities will both abstract, treat and distribute water, whereas others will operate across only a fraction of the value chain: for example, bulk water utilities, may abstract water to serve a network of utilities that go on to distribute this to households. In structures such as this, as we are concerned with the balance of water supply and demand, we group bulk water providers with their respective distributors.

6.4.2.1. Water utility demand

Monthly water consumption (C) per person, at present and in the future, is assigned to each utility based on the national average data (FAO, 2020). A database of per capita water consumption was collected per country for 124 countries, for which a regression formulation is set up using GDP per capita, percentage urbanization, temperature, precipitation and regional dummy as explanatory variables to gap fill domestic water consumption for countries without data (Verschuur *et al.*, 2023). This regression formulation is also used to construct future per capita water demand projections using different combinations of SSP-RCP scenarios as adopted in this work (Verschuur *et al.*, 2023). Results of this are provided in Supplementary Figure D7. Although utilities may also service industrial and commercial customers, we neglect this additional demand based on the assumption that municipal customers would be prioritised during drought events. We estimate the future population served per utility by scaling in proportion to national population growth estimates provided by each SSP scenario (Richters *et al.*, 2022). This is a simplification of the complex ways in which utilities will respond to future demographic and economic changes in reality: some may expand at a faster or slower rate relative to growth; some new utilities may appear in new locations; and others may be abandoned. We also do not account for the possibility of utilities that operate seasonally, but rather assume that all utilities operate year-round.

Leakage effectively increases the demands on the utility, as operators must account for the network losses incurred during distribution to meet customer consumption needs. Therefore, the demand is quantified as the sum of the customer consumption need and the network losses (L, %). However, the data we collected provides the non-revenue water rate which is the combination of leakage losses and the rate of tariff collection (GWI, 2022b). Here, we assume

that the non-revenue water rate is equivalent to the leakage loss rate, which is a significant limitation, but necessary due to data limitations.

Thus, the demand (m^3) that is exposed to drought for each utility is provided by the following equation.

$$D = P\,C\,(1 + L) \qquad (32)$$

Note that we assume that monthly demand is constant throughout the year.

6.4.2.2. Water utility service areas

To map out the area each utility serves (SA), we first map the utility headquarter locations which we assume aligns with its central demand location. We then map the global population at 1km resolution (Lloyd *et al.*, 2019) and overlay this with the rate of piped water access globally. Piped water access was derived by disaggregating JMP rural versus urban estimates (UNICEF and WHO, 2020) based on a gridded urbanization layer (Gao and O'Neill, 2020). The SA per utility was mapped by iteratively expanding radially from the utility headquarters, incorporating more of the population served, until the utility customer base was equal to the population within the SA. Note that the grid cell size of the population layer (1km) based on which the service areas were assigned, is smaller than that of the hydrological model (CWatM). Where utility service areas do not cover an entire CWatM grid cell, we factor the supply input to the utility by the fraction of the grid cell covered. For overlapping utility service areas, we extract the total supply within overlapping grid cells, and apportion it between the utilities in proportion to the population served. We do not determine how utility service area size may change and assume they remain fixed under future population growth scenarios.

6.4.2.3. Desalination

We use an up-to-date database of desalination plants used for municipal supply(GWI, 2022a) to find the fraction of demand that is supplied by desalination (DS) per utility by assigning each desalination plant to each utility based on its main service city.

The component of demand that is vulnerable, or the Vulnerable Demand (VD) (m^3) per utility, is therefore provided by the following equation.

$$VD = D - DS \qquad (33)$$

6.4.3. Drought risk method

To estimate drought risk, a simulation of the supply (monthly river discharge, groundwater volume and reservoir storage) available to each utility was balanced against the volume of customer demand that would be vulnerable to drought-induced water shortages. Where shortages occur, the utility customer base that would be disrupted is approximated in terms of UWSD by applying the fraction of demand unmet, to the population served. This section outlines the simulation of supply (including how demands are abstracted for other sectors), followed by the utility-level water balance and the risk analysis procedure.

6.4.3.1. Drought hazard data

The Community Water Model (CWatM) is a state-of-the-art hydrological model which simulates river discharge, renewable groundwater supply, and lake and reservoir storage among other variables, at a monthly time step, at a 0.5-degree resolution (Burek *et al.*, 2020). Please find a full description and validation of CWatM in Burek et al., (2020), with a brief description provided in Supplementary Text D1. Each utility was allocated the total available reservoir storage within the perimeter of the utility's SA. We select the cell recording the maximum river discharge per month within the utility SA most frequently over the timeseries, for the river discharge input. Available renewable groundwater supply is found as the average monthly volume totaled across the area of coverage of the utility (6.4.2.2. Water utility service area). Note that, abstracting from non-renewable groundwater is common practice for many utilities (Gleeson *et al.*, 2012). Therefore, discounting non-renewable groundwater from the supply available utilities does not reflect current practices. Rather, we provide a formulation of risk as a basis for decision making; under the assumption that utilities would wish to adapt away from unstainable practices such as non-renewable groundwater abstraction.

We collect the historical model output (1980-2010) and investigate the effects of potential climate change scenarios and socioeconomic growth projections (2030-2060) across SSP1-2.6, SSP3-7.0 and SSP5-8.5 on supply (river discharge, reservoir storage and renewable groundwater supply) each across the 5 climate models included in the CMIP6 experiment (O'Neill *et al.*, 2016): GFDL-ESM4 (Dunne *et al.*, 2020), IPSL-CM6A-LR (Boucher *et al.*, 2020), MPI-ESM1-2-HR (Gutjahr *et al.*, 2019), MRI-ESM2-0 (Yukimoto *et al.*, 2019), UKESM1-0-LL (Sellar *et al.*, 2019). This selection of future projections is simulated under different scenarios of anthropogenic climate change and global socio-economic development as per phase 6 of the Coupled Model Intercomparison Project (CMIP6) (Eyring *et al.*, 2016).

The RCP scenarios represent transient scenarios for increasing radiative forcing (van Vuuren *et al.*, 2011) and SSP scenarios outline five alternative scenarios of future global development trajectories, contingent on different levels of fossil-fuel dependency (Riahi *et al.*, 2017). SSP1-2.6 represents an optimistic, sustainability-focused scenario of growth with a strong reduction of CO2 emissions and an associated increase in temperature of 1.8°C by the end of the 21st century (Meinshausen *et al.*, 2020). SSP3-7.0 represents a medium-high emissions scenario where the average temperatures projected to rise by 3.6°C (Meinshausen *et al.*, 2020). SSP5-8.5 represents a fossil fuel-driven scenario of the future where, by 2100, in which average global temperature would increase by 4.4°C (Meinshausen *et al.*, 2020). All considered scenarios of supply were run through the same balance to quantify the change in likelihood of drought-induced utility UWSD in the future relative to the present-day.

6.4.3.2. City water sources

We account for long-range water transfers through the use of a database which identifies the major water source for a set of ~1,600 sources for ~270 large cities (McDonald *et al.*, 2014). We assigned the major city water sources to each utility based on the matching utility headquarter location. Supplementary Figure D8 provides the spatial distribution of utilities for which a long-distance transfer has been identified. The water sources database indicates whether the source is a desalination plant, a river extraction point, a storage dam or reservoir or a borehole. Thus, the relevant supply type (i.e., river discharge, renewable groundwater or lake and reservoir storage) from CWatM was extracted from the coordinates of the source and combined with the total supply available within each utility service area. Note that we extract the supply within the grid cell that overlaps with the source location.

6.4.3.3. Allocation rules

Here we assume that in each month, the supply available in each utility's SA is available and prioritised above all other sectors. However, in preceding months, we assume that upstream abstractions are fully met by all dependent sectors, such as irrigation. We adopt this approach to capture the risk that over abstraction, now and in the future, poses to essential water utility services. As such, the model of water supply variations is run based on a global, standard, allocation scheme (Wada, Wisser and Bierkens, 2014), however, locally, utility sources are prioritised for municipal use. This introduces an inconsistency into the method because municipal demand inputs are integrated with the model of supply (see Huang *et al.*, 2018), but also input as a function of utility characteristics (see 6.4.2.1. Water utility demand). Note that

the inconsistency between CWatM municipal demand versus our utility level demand inputs would have a negligible effect on the results, as the model adopts a return flow assumption of 90% for municipal withdrawals.We adopt a threshold for environmental flows of Q10 of monthly river discharge under baseline climate conditions (the flow exceeded 90% of the time), a widely held assumption (Pastor *et al.*, 2014).

6.4.3.4. Drought risk formulation

We simulate a monthly balance between total supply (m^3/month) (S) - which is the sum of available reservoir storage (RS), river discharge (RI) and available renewable groundwater supply (GW), minus the environmental flow (EF) - and the demand that is vulnerable to droughts (m^3/month), accounting for leakage losses (VD). Where the supply is not sufficient to meet the demand, a Deficit (m^3/month) (Def), greater than zero is recorded.

$$S_m = RS_m + RI_m + GW_m - EF_m \quad (34)$$

$$Def_m = \max(VD_m - S_m, 0) \quad (35)$$

This formulation is used to derive the first benefit metric, $UWSD$ (customer-days for a given month). To estimate the population that would be disrupted by a given monthly deficit, we apply the equivalent fraction of demand that is unmet by the supply to the population served by a given utility and multiply by the number of days in a month (d).

$$UWSD_m = \frac{Def_m}{VD_m} P\ d_m \quad (36)$$

We merge the utility-level tariff rates (per m^3) from the IBNet tariff benchmarking database (IBNET, 2022) with the GWI utility database. Where utilities included in the GWI utility database were missing from IBNet, we apply the national average. Thus, equivalent tariff revenue loss (USD/month) (TRL) (Equation 37) associated with the reduced volumetric sale of water was estimated, where t is the tariff rate for water supply (GWI, 2022b; IBNET, 2022). Given the strong relationship between tariff rates and GDP per capita (r^2, 0.6), see Supplementary Figure D5, we use this linear relationship to estimate future tariff rates based national level data on GDP and population under alternative SSPs (Richters *et al.*, 2022).

$$TRL_m = UWSD_m \frac{c\ t}{d_m} \quad (37)$$

For many utilities, tariff revenue would not be elastic to the volumetric sale of water, and disruptions may not be reflected in the utility's balance sheet. This depends on the type of

arrangement the utilities have in place; for example, whether they are able to monitor disruptions and adjust tariff charges accordingly. Some utilities may also be subject to fines, which are not considered here. However, the proposed approach enables a globally consistent estimation of potential financial losses incurred by utilities during droughts.

The monthly disruption is summed per year, and the annual average loss at the utility level in terms of $\overline{UWSD_u}$ (customer-days/year) and $\overline{TRL_u}$ (USD/year) is estimated by integrating each loss across the annual exceedance probabilities of events (p_y) (based on drought event ranks) and for a given climate scenario, as shown in Equation 38 and 39, respectively.

$$\overline{UWSD_u} = \int_0^1 UWSD_y \, dp_y \qquad (38)$$

$$\overline{TRL_u} = \int_0^1 TRL_y \, dp_y \qquad (39)$$

6.4.4. Climate adaptation optioneering

We use the above drought risk to understand the marginal benefits - i.e., the reduction in risk for a given dollar investment - of alternative infrastructure investments. Thus, the costs of climate adaptation can be weighed against the benefits. This is done by rerunning the risk analysis using model runs provided between 2030-2060 under future scenarios by simulating the effect of a range of 10 designs of each adaptation option type $P = \{min \ldots max\}$. The range of designs are established based on equal intervals between the minimum and maximum design limits of each option type, as outlined in Supplementary Table D1. We discount benefits and costs over the asset's lifetime (conservatively assumed to be 30 years, although typically much longer) to derive cost effectiveness in terms of mitigated UWSD and lost tariff revenue. Thus, assuming that the date of implementation would be 2030, we simulate the benefits that would accumulate over the duration of 30 years.

We assume that if any location within a utility's service area is within 50km from the coastline, it would be eligible for seawater desalination. This transfer distance threshold is established based on the database of water transfers for major cities (McDonald *et al.*, 2014); Supplementary Figure D9 illustrates the distribution of transfer distances for major cities. We acknowledge that there are other factors that would play into the eligibility of a utility for desalination, such as the continuity of electricity supply and the source of electricity in the context of green energy transitions (Panagopoulos, 2021). Here we limit the eligibility criteria to proximity of a utility's service area to the coastline due to data limitation. Note that we only

consider coastal desalination, as opposed inland to brackish water desalination due to data limitations on the presence of brackish water globally.

6.4.4.1. Adaptation option costs

The costs of each option are provided in Supplementary Table D2. For desalination, we apply the cost per volumetric unit to the equivalent change in capacity (Kahil *et al.*, 2018). We also estimate the length of the pipeline (t) that would be required to transfer desalinated water to the headquarter location of the utility and the pumping station required for the transfer (Kahil *et al.*, 2018). We then re-run Equation 33 with the new desalination capacity to simulate the effect of increasing desalination capacity on risk.

For leakage, we first estimate the volume of leakage lost per km of network, where both the rate of leakage and the length of the network are provided in the utility database (GWI, 2022b). Where the length of the piped water network is missing from the database, we employ a linear regression model ($r^2 = 0.8$) to predict values based on the population served, population density of the service area, per capita consumption (Verschuur *et al.*, 2023), piped water access rate (JMP, 2017) and national GDP per capita (Richters *et al.*, 2022). We thus estimate the length of pipeline that would have to be renovated for a given reduction in leakage and apply the unit cost per m (Kahil *et al.*, 2018), as provided in Supplementary Table D2. We then re-run Equation 32 with the new leakage rate to simulate the effect of reducing leakage losses on risk.

For storage, we simulate the river discharge balance as follows in Equation 40-44, to estimate the volume that would be available for storage per month for each utility. We then cumulatively sum the available river discharge to simulate the maximum available additional supply (m^3) (STM) that could be available per month, given sufficient storage capacity. We then define a range of additional storage capacities (m^3) (SC_a) as a fraction (a_s) of the maximum capacity that could be stored, with the maximum fraction being defined based on that which would provide a maximum release equivalent to 90% of demand. We then re-run Equation 34 with the additional reservoir storage (m^3/month) (ST_m) to simulate the effect of increasing storage capacity on risk.

$$SW_m = RI_m - EF_m - VD_m \frac{RI_m}{GW_m + RS_m + RI_m} \quad (40)$$

$$STM_m = \min(SW_{m-1} + SW_m, 0) \quad (41)$$

$$SC_a = a_s \max(STM_m) \quad (42)$$

$$ST_m = \min(STM_m, SC_a) \quad (43)$$

$$RS_m = RS_m + ST_m \quad (44)$$

We estimate the change in energy costs before and after the implementation of leakage reduction, storage capacity and desalination capacity increase measures, as well as other general O&M costs, such as equipment maintenance. To quantify this change in energy costs, we first estimate the overall annual energy requirement per utility (kWh/year) (ER_u), Equation 45.

$$ER_u = D_u\, v + DS_u\, e + n\,(DS_u + D_u) \quad (45)$$

Where v, e and n are the global average unit energy consumption in kWh/m^3/year required to treat water (not from desalination), to desalinate water and to transport water, respectively (GWI, 2022b) and D and DS represent the annual volume of non-desalinated and desalinated water treated per year (m^3/year) per utility.

Thus, the change in energy costs (EC) associated with each type of adaptation option are as follows. Note that the change in energy costs associated with leakage reduction would be negative and result in efficiency gains.

$$EC_{u,l} = ER_u\,(L - l)\,c_n \quad (46)$$

$$EC_{u,d} = (d_u\,(e + n))\,c_n \quad (47)$$

$$EC_{u,s} = (s_u\,(v + n))c_n \quad (48)$$

Where l denotes the new leakage rate (fraction), d denotes the increase in desalination capacity (m^3) and s denotes the increase in storage capacity (m^3). c_n is the average unit cost of energy for industrial users ($/kWh) per nation (Global Petrol Prices, 2022).

As noted above we apply different adaptation options to present and future drought risks for the set of designs $P = \{min \ldots max\}$. Assuming that the discount rate of r, estimated at 6%, is applied over the adaptation planning horizon, the discounted cost (USD) ($DC_s(p)$) for applying the adaptation option of a given design (p) to a utility is estimated in terms of the sum of initial investment costs (USD) ($CI_{j0}(p)$) at $t = 0$ and the sum of the subsequent asset level discounted annual operations and maintenance costs, accounting for increased/decreased energy costs, $CM_{jt}(p)$ over the planning horizon. This is formulated in Equation 49.

$$DC_s(p_d) = \sum_{\forall j \in s} CI_{j0}(p) + \sum_{\forall j \in s} \sum_{t=1}^{t=T} \frac{CM_{jt}(p)}{(1+r)^t} \qquad (49)$$

6.4.4.2. Benefit of adaptation

Due to the chosen adaptation option the risks to each utility will be reduced because of level of protection provided by the option. These new risks associated with the protection level provided by each adaptation option ($\overline{UWSD}_u(p)$, $\overline{TRL}_u(p)$) are obtained by rerunning the risks analysis, using Equations 33-39.

Similar to the costs, the risks before and after the adaptation option are estimated at annual timescales over the planning horizon. As noted above the future risks are driven by the climate change driven hazard events and the customers dependent on the water systems. The discounted benefit (USD) ($DB_s(p)$) of risk reduction due to the adaptation option is estimated by discounting the annual timeseries of marginal risk reduction ($\overline{TRL}_{st} - \overline{TRL}_{st}(p)$) over the planning horizon, Equation 50.

$$DB_u(p) = \sum_{t=0}^{t=T} \frac{\overline{TRL}_u - \overline{TRL}_u(p)}{(1+r)^t} \qquad (50)$$

The effectiveness of the option is assessed in terms of the benefit-cost ratio ($BCR_u(p)$), given by Equation 51.

$$BCR_s(p) = \frac{DB_u(p)}{DC_u(p)} \qquad (51)$$

The optioneering solution involves finding the optimal intervention (OI_u) per utility, identified as the option where the ratio between benefit and cost (BCR) is greatest, see Equation 52.

$$OI_u = max\big(BCR_u(p)\big) \qquad (52)$$

Acknowledgments

The authors declare no conflict of interest. The corresponding author acknowledges funding from the Engineering and Physical Sciences Research Council (EPSRC) under grant number EP/T517811/1. The authors would like to thank Global Water Intelligence (GWI) for access to the utility database. This first author would also like to thank the International Institute for Applied Systems Analysis, as this work was the focus of IIASA's Young Scientist Summer Programme.

Data and software availability

CWatM and global population data are openly available at the following source data link: https://www.isimip.org/impactmodels/details/257/?query=cwatm. The utility data is provided by GWI(GWI, 2022b) under a license which restricts us from distributing the data to the public or research community. Software used for the project include Python (https://www.python.org/) and QGIS (https://www.qgis.org/en/site/), both of which are open-access.

Code availability

Code involved in performing the analysis will be made available at the corresponding author's github.

Acknowledgments

The authors declare no conflict of interest. The corresponding author acknowledges funding from the Engineering and Physical Sciences Research Council (EPSRC) under grant number EP/T517811/1. The authors would like to thank Global Water Intelligence (GWI) for access to the utility database. This first author would also like to thank the International Institute for Applied Systems Analysis, as this work was the focus of IIASA's Young Scientist Summer Programme.

7. Conclusions

The synthesis of this thesis covers a brief summary of the findings for every research question posed, followed by a discussion focusing on a number of overarching points of reflection and recommendations for future research.

7.1. Summary of findings

The goal of this DPhil is to develop and apply a set of complementary frameworks for analysing climate risks to water supply systems to guide decision making and promote the quantification of adaptation benefits. This overarching aim is divided into three research objectives.

1) To develop a framework for analysing widespread, multi-hazard climate risks to water supply systems.
2) To develop an approach for quantifying adaptation benefits for both water supply utilities and their customers, including those in poverty.
3) To develop a global framework for quantifying utility-level drought risk as a platform for optioneering across multiple drought adaptation options and performing BCR assessments.

A brief summary of the findings for every research question is provided.

7.1.1. Magnitude of multi-hazard climate risk faced by the NWC

In chapter 4, a generally applicable framework for conducting multi-hazard risk assessments of water supply systems is proposed and used to quantify the impact of present and future climate extremes on the national water supply network in Jamaica. The proposed framework involves constructing a systems model and stress testing it with a large set of spatially coherent drought, cyclone and pluvial and fluvial flood events. For each disruptive event, the number of water users whose supplies may be disrupted per day is calculated, multiplied by the event's duration in days, that is, the Customer Disruption Days (CDD) metric. CDD provides a common decision-relevant metric with which to compare risks to water supply customers from the intersection of multiple hazards and system vulnerabilities, addressing acute damage to water supply networks at the same time as computing the risk of water shortages. The impact of present and future climate extremes under a selection of RCPs are examined. The total multi-hazard annual expected disruption was estimated to be approximately 5 days per year per utility customer under present conditions, increasing by a factor of between 2 and 2.5 when end-of-century climate scenarios are propagated through the model. The analysis shows that more high probability drought events lead to greater CDD compared with asset damage events. However, extreme asset damage events, despite manifesting over shorter timescales (days) compared to drought events (months), can lead to more widespread CDD. The framework presented can be generally applied to identify local hotspots of both asset damage- and water shortage-induced

risk, as well as quantify the potential extent of simultaneous disruptions affecting a national-scale potable supply network, essential for contingency planning in preparation for widespread climatic events.

7.1.2. Priority climate adaptation options for the NWC and its customers

Chapter 5 introduces a new framework for quantifying the direct financial losses incurred by utilities and their customers (including those in lowest income groups) during flood- and drought-induced water supply disruptions in Jamaica. Using the outputs on annual expected risk in terms of CDD from chapter 4, annual expected financial losses per water supply system are calculated as the sum of those that would be burdened by 1) the NWC and 2) their domestic customers, the latter being equivalent to the costs of replacing disrupted supply from private tanker trucks. By repeating this process with and without a range of flood defence walls and leakage reduction options, the benefit that each intervention offers in terms of reducing annual expected losses is quantified. This is compared against each option's cost to derive their Benefit Cost Ratios (BCRs). The number of system-optimised flood defence and leakage reduction options that would benefit were found to be 83 and 128 respectively, and the number of options with BCRs>1 amount to 24 and 26, respectively. The total annual benefit that could be achieved by implementing all system optimised flood defences is estimated to be between ~0.72 billion Jamaican dollars per year, whereas implementing all leakage reduction options would provide a total benefit of ~1.5 million Jamaican dollars per year. Overall drought mitigation options are found to be more cost effective for reducing costs to households in poverty compared to flood defences. The proposed framework quantifies the climate-related risk mitigated by adaptation projects to promote the business case for climate adaptation and enables optioneering analyses with respect to both economic objectives and 'pro-poor' agendas.

7.1.3. Global distribution of drought adaptation needs and priority actions

In chapter 6, a novel risk assessment framework is proposed to perform a drought risk and optioneering analysis of water supply utilities at scale. The proposed risk framework is proposed to 1) perform drought risk analysis of 5600 water supply utilities at present and under future climate change and 2) evaluate the benefit of alternative infrastructure investment opportunities (enhanced reservoir storage, leakage loss reduction and desalination) to identify the priority option type per utility. A model of each utility's water balance that represents supply inputs, municipal water usage and leakage losses is established to estimate the risk of utilities being unable to supply their full customer base sustainably in terms of Unsustainable

Water Supply Days (UWSD). By comparing the impact of three alternative infrastructure interventions on UWSD and associated tariff revenue at risk against each option's cost, the BCR of each option type is estimated to identify the most cost-effective adaptation option per utility. Hotspots of present-day and future drought risk in terms of Unsustainable Water Supply Days (UWSD) under three future scenarios are identified, where UWSD is quantified in terms of the annual expected customer-days served by unsustainable or disrupted water supply services. The global UWSD is found to be of 510 million per year and the median utility to be subject to 15 days of unsustainable or disrupted supply, increasing by 30-45% under future scenarios. Implementing the most cost-effective adaptation action per utility would mitigate additional future risk by ~75-80%. However, implementing options with BCRs>1 would achieve only ~5-20% of this benefit, underling the challenge of closing the climate adaptation gap for water supply utilities.

7.2. Discussion & Limitations

Several points for discussion have been identified. These points are not all-encompassing but rather serve to summarise some of the major take-aways and limitations across the chapters.

7.2.1. Risk metrics for understanding multi-hazard water supply disruptions

Water supply infrastructure systems are exposed to multiple climate-related impacts on a range of different timescales, with varying spatial extents and different impact channels. Droughts usually materialise over periods of weeks and months, but when demand exceeds supply for prolonged periods, they can manifest in water shortages for users. Floods and/or extreme winds, on other hand materialise over shorter timescales and can cause direct damage to water supply infrastructure assets, such as pumping stations and treatment works, resulting in water supply disruptions.

In chapter 4, the CDD metric is introduced. The CDD metric provides a unifying metric of the overall magnitude of disruptions caused by the different drivers and enables comparison of short-lived wide-scale events (e.g., flooding during intense hurricane rainfall) with longer-term events which may just hit a small proportion of the population (e.g., low flows confined to one catchment). Through this unification, detail on the multiple dimensions (duration versus intensity) in which a disruption propagates is lost. This, in turn, prevents exploration of the adaptive actions taken by utilities or households to mitigate water supply disruptions, including during droughts, floods or storms, such as installing household storage tanks. For example, households may be more willing to tolerate prolonged intermittent disruptions for water shortages, compared to total loss of supply from asset damages for a shorter period. Results should therefore be regarded as an upper bound on the CDDs that may be expected. It also prevents the exploration of a disruption's tolerability.

Chapter 6 introduces the Unsustainable Water Supply Days (UWSD) metric, to capture the risk that utilities are unable to continue sustainable water supply services. This captures the number of days in a year during which a utility's supply is unsustainable (i.e., utilities tap non-renewable groundwater supply) or disrupted due to drought (i.e., demand restrictions and/or reallocation strategies are implemented). The metric is limited in conveying the risk of water shortages to some extent. For example, some utilities may have vast supplies of fossil-aquifer supply, providing a near infinite level of drought resilience. Further, it should be highlighted that many utilities systematically implement demand restrictions every year during dry summer

months, and consider this as normal operational procedure, rather than a disruption. However, it provides a consistent description of drought risk at a global scale, combining global hydrological models with local utility characteristics.

Despite limitations, the metrics introduced provide some notable improvements compared to previous efforts. For example, in providing a consistent characterisation of disruptions that can be compared across multiple water systems at scale, communicating both duration and severity (e.g., population impacted) via a single metric. Further, in their potential application for integrating social vulnerability considerations in climate adaptation planning.

7.2.2. The importance of climate resilient water supply utilities for society and development

A crucial aspect of climate adaptation is that strategies must be aligned with national development priorities (Hulme, O'Neill and Dessai, 2011; Conway and Mustelin, 2014). As utilities are largely state-owned, and water supply is widely perceived to be a public good, the adaptation strategy of water supply utilities is deeply intertwined with the development trajectory of developing countries. For example, where poverty alleviation is a key priority for many countries, water utilities may be mandated to serve poor communities and maintain affordable services. Where water supply disruptions are substituted by costly private tanker trucks, investing in resilience can provide a material saving for households, particularly those in low-income groups. In chapter 5, an alternative quantified measure of the financial benefits that would be realised through adaptive action is proposed, based on mitigating the costs of purchasing water from private tanker trucks in Jamaica. The proposed approach supports both the business case for climate adaptation projects in the water sector at the same time as demonstrating the benefit of infrastructure for society. However, it should be noted that the approach represents only a fraction of the impacts of water supply disruptions to utilities, government, households and the wider economy.

Further, the wider applicability of certain aspects of the approach proposed in chapter 5 should be re-evaluated before employing this framework in other utility contexts. For example, the range of financial costs that would be burdened by the NWC versus its customers are simplified: many utilities lose tariff revenue during drought-induced water shortages, due to a lower overall volumetric sale of water. In such cases, lost tariff revenue should be incorporated into the definition of utility losses. Further, social losses are characterised as the direct financial

losses that customers would burden should they resort to private tanker truck supply during all climate related NWC disruptions. The true dynamics of tanker truck markets are simplified by assuming that the unit cost of water from private tankers would remain uniform across the island and over time, and the reality that many customers would resort to raw water supply from springs or wells rather than tanker trucks is not captured. The framework does not account the intangible, indirect losses such as the time spent on acquiring alternative water sources and associated losses in productivity, or any physical or emotional health impacts. However, this was beyond the scope of the analysis, which is presented as an early step towards mainstreaming equity consideration in adaptation optioneering frameworks to identify co-benefits and trade-offs between economic- and equity-related objectives.

7.2.3. Generalising the quantification of adaptation benefits

Climate-resilient water supply utilities provide a material contribution towards adaptation goals by managing variable hydrology and climatic uncertainty to provide safe, reliable, and affordable drinking water services to households. This makes the water sector an important channel for climate adaptation finance. Adaptation investments are undertaken by a wide range of sources globally, including from Development Finance Institutions. For institutions that are mandated to boost finance for climate adaptation, the greater the impact of an investment towards climate adaptation, the greater the level of concessionality[2], which can be essential to attract the necessary financing (e.g., including from the private sector) for infrastructure projects in 'hard to invest' sectors, such as the water utility sector. However, a key barrier in the deployment of capital for adaptation is the methodological challenge investors face in quantifying the contribution of projects toward climate-resilient outcomes or adaptation benefits (Ranger and Mullan, 2022). This means that, although water supply utilities would be a strong candidate for climate finance investments, their potential has yet to be fully realised.

In chapter 6, an analytical framework for quantifying the contribution of adaptation investments in the potable water sector towards the global climate adaptation agenda is developed. This provides a material advance compared to previous global studies on drought risk, which are often conducted at the catchment scale and involve overlaying water stress or SPI risk maps with population density or city location layers. Previous studies do not capture utility-level characteristics such as leakage losses, service areas and desalination inputs, limited

[2] The degree to which interest-rates on a loan is set below the market rate, e.g., by blending with grants.

the rigour with which previous studies have identified adaptation benefits for the potable water sector (as discussed in section 2.3.1). Capturing this information is essential for understanding the propagation of meteorological/hydrological droughts through utility networks to dependent households. The framework generalises risk and adaptation benefits across utilities with vastly different constraints and characteristics.

Chapter 6 illustrated that, where the business case of adaptation action is defined in terms of the avoided tariff revenue loss thanks to reduced UWSD, there would be substantial barriers to investment. There are a number of limitations associated with this approach. Firstly, assuming that tariff revenue would decrease in proportion to the number days of disrupted or unsustainable supply is not realistic for many utilities. Secondly, wider economic and welfare losses are not considered. Indeed, places where interventions are justified in narrow CBA terms, further examination is required around the barriers to investment. In places where adaptation is not justified on narrow cost-benefit terms, there needs to be wider consideration of the economic and social case for resilient water supplies. For example, utilities often serve industrial and commercial entities, disruptions to which may result in economic losses (Freire-González, Decker and Hall, 2017; Jenkins *et al.*, 2021). Customers also often suffer financial, welfare and productivity losses in acquiring water supplies from alternative sources (e.g., bottled water or tanker trucks). Capturing these wider costs of disruption would provide a stronger business case for adaptation. For example, tanker trucks are a major form of service delivery during disruptions in Jamaica, and in many parts of the world (Baisa *et al.*, 2010; Buck *et al.*, 2016). In these contexts, weighing the mitigated costs of purchasing from private tanker trucks against adaptation investment costs, as in chapter 5, can provide a customer-orientated reflection of quantified financial benefits of adaptation, supporting the business case for adaptation investment.

As such, chapter 6 addresses a gap in our collective understanding utility-level climate risks, which could help inform policymakers identify the most appropriate solutions to mitigate this, as well as attract the finance required to address adaptation needs of utilities. However, it should be interpreted in the context that, through generalising complex processes, the true costs and benefits of adaptation actions in the potable water sector are greatly simplified.

7.2.4. Adaptation finance to reverse the vicious cycle?

Adaptation investments are undertaken by a wide range of sources globally, including governments, donors and banks, and receives a growing amount of attention in high level

policy dialogue (CPI, 2020). In recent years, the number of climate and climate adaptation focussed investors and donors has proliferated, with different sectoral and geographical remits (Smith *et al.*, 2011). Establishing a strong narrative around how water adaptation investments contribute towards global climate adaptation agenda can support utilities in attracting more concessional loans and grants. This investment can be channelled towards their infrastructure needs, such that they can move towards a more financially and operationally resilient service and negate the vicious cycle. Although investments need to be coupled with wider sector reform activities in many cases: in particular, to foster a regulatory environment that incentivises an adequate and a sustained rate of reinvestment in network maintenance e.g., through engaging the private sector through results-based financing (Rodriguez *et al.*, 2014).

7.2.5. Summary of peer review

This section provides a summary of the peer review process and changes made in response for each on each paper.

7.2.5.1. Review: A multi-hazard risk framework to stress-test water supply systems to climate-related disruptions

The paper presented in Chapter 4 of this thesis underwent two rounds of review. The first round recommended that major corrections should be made before consideration for publication in *Earth's Future*, whereas the second round recommended minor corrections. The edits made in response to reviewer feedback were accepted by the journal and the paper is now published. These edits are summarised below.

- We added more key details of modelling methodologies, particularly around the drought and flood risk approaches.
- We added justification of the different timescales, meteorological input data and modelling approaches used in the flood versus the drought hazard modelling approaches.
- We added greater discussion around the application of future climate scenarios as a sensitivity analysis, rather than as a coupled simulation of a complete future scenario set, given the lack of consideration around demand changes.
- We clarified that the risk results represent a worst-case scenario, where no adaptive action (e.g., buffering disruptions through household storage tanks) is taken by households.
- We added further discussion on the challenges around comparing water shortage- and asset damage-related disruptions as equivalents.

- We outlined how the tolerability of the different types of disruption could be accounted for in future applications, e.g., through accounting for the volume of household storage tanks.
- We reported on the duration of water shortage and asset damage-induced disruptions.

7.2.5.2. Review: Prioritising climate adaptation options to minimise financial and distributional impacts of water supply disruptions

The paper presented in Chapter 5 of this thesis underwent two rounds of review. The first round recommended that major corrections should be made before consideration for publication in *Environmental Research: Infrastructure and Sustainability*, whereas the second round recommended minor corrections. The edits made in response to the second round of reviewer feedback have recently been submitted to the journal and await further instructions on next steps. These edits are summarised below.

- We altered the original poverty incidence calculation to better account for the fact that populations in poverty are less likely to be customers of the utility compared to populations above the poverty line.
- We added some instruction on the potential use of alternative sources that could be used, where available, to improve the accuracy of the estimation of system-level poverty incidence.
- We added some discussion around how realistic the implementation of each type of option is, as well as the proposed method being used in decision making, in the Jamaican context.
- We highlighted that alternative option types could be the focus of future studies.
- We added some further assessment of the spatial pattern of the BCR results.
- We added further clarity around the assumption that tanker trucks provide the main mode of alternative service delivery during disruptions.
- We originally defined costs to households as 'social losses' and entitled the manuscript as 'Prioritising socially responsible climate adaptation options for water utilities'. This was seen to suggest a wider inclusion of economic and welfare-related impacts compared to the actual scope of the work. We agreed with this feedback and, as such, changed the defined terminology and title to 'alternative water purchase losses' and 'Prioritising climate adaptation options to minimise financial and distributional impacts of water supply disruptions', respectively.
- We added some key policy implications to conclude the paper.
- We added some further details about the papers' innovations relative to the state-of-the-art.

7.2.5.3. Review: Closing the climate adaptation gap for water supply utilities

The paper presented in Chapter 6 of this thesis underwent one round of review in which reviewers recommended that major corrections should be made before consideration for publication in *Nature Communications: Earth and Environment*. The edits made in response to the reviewer feedback were accepted by the journal and the manuscript status is currently considered ready for publication in principle, pending the action of further editorial requirements. These edits made in response to reviewer feedback are summarised below.

Here we list some of the main edits that have been made to-date in response to reviewer feedback.

- We added definitions of water supply utility and RCP and SSP scenarios.
- We added further details about the drivers of regional differences in risk and future changes in risk, and about the drivers of optimal option type selection.
- We provided a detailed definition of a water supply utility.
- We further explained the limitations around assuming that further utility customer bases will change in proportion to national population growth, given that future changes in utility customer bases will be driven by a range of other factors.
- We supplemented the literature review in the introduction and discussion sections to highlight the innovation of this work relative to prior research.
- We edited sections in the introduction and discussion as follows to further highlight the key motivations for, and implications of, this work.
- We added more discussion around the key sources of uncertainty into the discussion.
- We briefly discussed the distributional costs of disruptions to households across income groups, whereby lower income households may burden disproportionately greater costs relative to income.
- We discussed the omission of climate-related water quality-related disruptions and identified this as a potential area for future research.
- We added some commentary around the potential concerns associated with desalination, particularly around the discharge of brine and substantial energy requirements.
- We shortened the supplementary materials by removing non-essential content.
- We also improved the accessibility of the abstract by removing and/or clarifying specialist terms.

7.3. Recommendations for future research

There is vast potential to develop the frameworks presented here to quantify the risk of climate-related water supply disruptions and adaptation needs for water supply utilities. Here eight key topics for future research are outlined.

7.3.1. Use of alternative datasets for climate risk analysis

Large-scale modelling studies of climate risks to infrastructure is often difficult to validate. Indeed, the lack of validation of the results produced by the global framework presented in chapter 6 is a major limitation. At smaller scales, interviews and/or survey techniques have been used to gain a more immediate insight into both the rate and subsequent impacts of water supply disruptions at the household or business level (Sjöstrand *et al.*, 2021; Huberts *et al.*, 2023). Empirical datasets, such as that which can be scraped from social media sites, can provide a valuable source for disaster impact analysis at scale (Takahashi, Tandoc and Carmichael, 2015). However, the application of such alternative data sources for understanding exposure and vulnerability to critical infrastructure (including water supply) disruptions, is not as developed as for other applications, such as understanding public awareness and sentiment towards climate change (Lineman *et al.*, 2015; Berglez and Al-Saqaf, 2021). This could be the subject of future study both on the extent of disruptions, major disruption modes and the impacts of disruption to households.

7.3.2. Water-energy nexus

A major channel of disruption for water supply utilities is via disruption to electricity supply. Much work has been done to capture the risks posed to water assets, indirectly, from disrupted energy infrastructure networks (Pant *et al.*, 2018). However, not from a multi-hazard perspective. The term ‘system-of-systems’ has been coined to describe the modelling of dependency linkages across different types of infrastructure networks (e.g., transport, telecoms, energy, water and wastewater) (Thacker, Pant and Hall, 2017). Further, many energy generation assets, notably hydropower and thermoelectric power, are dependent on water for their operations (van Vliet *et al.*, 2016; Paltán *et al.*, 2021). As droughts undermine both the availability and, in many cases, the temperature of flows, they also undermine energy supply (ibid). The CDD metric introduced in chapter 4 is enables the direct comparison of vulnerability and risks outcomes in a consistent measure across interdependent sectors. Future applications and developments of the proposed frameworks, including the multi-hazard risk framework

proposed in chapter 4, could adopt a ‘system-of-systems’ approach to quantify multi-hazard risks to interconnected energy and water networks.

7.3.3. Inclusion of a wider range of hazards for multi-hazard framework

A wider range of extreme climate hazards impact water supply systems than those considered in the proposed frameworks. For example, landslides and extreme heat: landslides triggered by extreme precipitation can damage assets and contaminate water sources (Collyns, 2017); extreme heat can lead to equipment malfunction (Bondank, Chester and Ruddell, 2018) and drive demand, increasing the risk of water shortages (Fiorillo *et al.*, 2021). Further developments of the framework could explore a wider range of hazards and their impact channels.

7.3.4. Global multi-hazard risk analysis for water supply utilities

The thesis presents two spatial domains for risk analysis: one is focussed on the extent of the water supply network in Jamaica; the other is applied to 5600 water supply utilities across the world. At this stage, it was not possible to undertake detailed, multi-hazard risk and adaptation needs analysis for all utilities globally. This is because, asset-level information on water assets (including treatment plants, pumping stations) is not yet available, and would be required for flood and cyclone risk analysis. However, global analyses can be highly effective for advocacy around the investment needed to ensuring water supply utilities are resilient to climate shocks now and in the future (Koks *et al.*, 2019; Verschuur, Koks and Hall, 2020). Integrating other hazards such as flooding and tropical cyclones could also be the focus of future studies, including their impact on turbidity levels, which is an important but relatively under researched climate risk transmission pathway for utilities.

7.3.5. Accounting for water quality deterioration during extremes

Droughts and utility water shortages are not only driven by changes in water quantity. The application of water of poor quality for a particular purpose, may be associated with greater water requirements for dilution. A developing body of work has been conducted on the influence that water quality has on water quantity (Van Vliet, Florke and Wada, 2017). Although this has been largely approached from a chronic perspective, by exploring long term trends in water scarcity driven by water quality (Jones *et al.*, 2022), the impact of climate extremes on water quality could be explored in future applications. For example, floods result in greater rate of surface runoff from the land, leading to the contamination of water with

organic pollution (e.g., nitrates and phosphorus from fertiliser and sewers) and turbidity (Whitehead *et al.*, 2009). On the other hand, drought can lead to an intensification of contaminants and increasing water temperatures, which may trigger algal blooms and eutrophic conditions (van Vliet and Zwolsman, 2008; Whitehead *et al.*, 2009). These contaminants can exceed the treatability limits for the production of household water supply (Mortazavi-Naeini *et al.*, 2019; Nyman *et al.*, 2021), potentially leading to water disruptions for users.

7.3.6. Development of decision-relevant metrics

This research adopts two risk metrics, CDD and UWSD, that both obscure the dimensions of a disruption in terms of its severity and duration. This supports the ease of application of each metric for decision makers, however, fail to speak to the tolerability of a disruption for households. For example, a shorter, complete outage might be more tolerable than a longer, partial outage. Further, in the conversion of CDD to a financial cost for domestic customers in chapter 5, other losses that customers burden due to piped water supply disruptions, such as productivity and health losses, are not captured. Future work could focus on how to incorporate these dimensions, particularly into the global utility optioneering framework, to reflect the tolerability of water supply outages for local populations and their unique responses, across income groups. For example, how do people substitute piped water supply during disruptions and what are the associated productivity, or any physical or emotional health impacts of disruptions across different geographies? Understanding the duration of disruption that can be tolerated by households, can be used to establish thresholds to distinguish between tolerable and intolerable CDDs and further guide the efficient allocation of capital towards adaptation, for further development of the framework presented in chapters 4 and 5, respectively. Although chapter 5 takes a first step towards integrating the spatial distribution of poverty into an adaptation optioneering framework in the potable water sector, this could be advanced further to understand how income shocks translate into consumption, and hence welfare, shocks. For chapter 6, future work could be focussed on improving the depth and breadth of the global risk analysis, for example, by including higher resolution and more comprehensive information on utility water sources, locally calibrated hydrological data and non-renewable groundwater stores.

7.3.7. Co-benefits of leakage reduction

High leakage rates are a pervasive problem for utilities globally, arising from inadequate maintenance over the infrastructure's lifetime (Rouse and El Achi, 2019). Addressing leakage

losses can contribute significantly to both the financial performance and drought resilience of utilities (ibid). Yet, as discussed by Rouse and El Achi (2019, p. 319), "Water distribution tends to be the 'Cinderella' of water supply", and the problem of leaking infrastructure receives disproportion levels of attention from governments, donors and academia relative to the challenge/opportunity it presents. Indeed, leakage reduction was found to be an important adaptation option in both chapters 5 and 6, from a drought mitigation perspective. However, there are a multitude of other benefits to investing in leakage reduction. For example, leakage reduction programmes typically comprise of pipe replacement/maintenance and the installation of pressure management equipment (e.g., valves) which can substantially improve overall network performance and reduce the risk of pipe bursts (Lambert, 2000; Rouse and El Achi, 2019). Leakage reduction programmes also typically include the installation of analytical devices such as sensors to monitor flows and identify areas with the greatest loss rate, they can help utility operators improve their tariff collection rates by setting accurate household bills and addressing illegal connections (McIntosh, 2003). In particular, networks of sensors can optimise system operations and enable transparency of flows in the network, with the additional benefit of identifying illegal connections, where relevant. Other benefits include reduced water production costs and energy consumption requirements, the latter being a substantial but under-recognised source of emissions (Rothausen and Conway, 2011), providing an effective strategy to escape the vicious cycle on multiple levels. Further work could elaborate on the adaptation optioneering frameworks to capture the full range of benefits of leakage reduction for boosting the operational and financial performance of utilities.

7.3.8. Alternative adaptation solutions

As water supply utilities face financing problems in most parts of the world, utilities would typically consider a much broader portfolio of infrastructure investments than those considered in this paper; particularly options associated with lower capital costs. Although three hard infrastructure options are considered, it is not necessarily true that these represent the most cost-effective option in all cases. A broader range of options, such as inter-sectoral reallocation, wastewater reuse (for irrigation and/or potable purposes), demand restrictions, institutional reforms and/or nature-based solutions, could be the focus for future developments of this framework.

8. Closing remarks

Through the delivery of sanitation, drainage/flood protection and potable water supply services, water utilities are responsible for among the most basic services for humanity. Lacking access to water supply services, either temporarily (during disruptions) or permanently, imposes a financial, physical and emotional burden on households. This burden is particularly high for low-income households, that have less disposable income to spend on safe alternative water supplies. The close relationships between governments and water utilities (applicable for both private and public water utilities) make decision-making in the water sector inextricably linked to wider national government decision making. Thus, infrastructure developments in the water sector are often steered by a wide range of objectives, including pro-poor development agendas.

Utility potable water supply services are dependent on the reliant operation of a network of infrastructure assets that are exposed and vulnerable to a range of climate hazards. In the face of climate change and increasingly uncertain operating conditions, water supply utilities are under immense pressure to protect their assets and water resources to supply customers. The Jamaican national water utility is one such utility, where the physical and operational challenges they face due to climate extremes and change are compounded by a lack of available financing for infrastructure resilience. The well-known vicious cycle of disrupted services, low-cost recovery and low adaptive capacity is a trend that speaks to the vast majority of water utilities globally, particularly in low and middle-income countries. This undermines the ability of utilities to provide a reliable service to existing customers, let alone to expand access to vital services. However, global development targets in the water sector do not typically speak to service reliability, but rather focus on the expansion of safe water access. Failure to integrate climate risk and resilience considerations into water sector development may undermine the financial performance of utilities, hindering the extension of network of services to unserved populations, and could lock-in development in hazard-prone areas, leading to maladaptation.

Given the exposure and vulnerability of their services to climate hazards, water utilities are uniquely positioned to deliver climate adaptation services for society. However, the importance of water utilities in strengthening society's climate resilience is disproportionate relative to the attention received at the high-level dialogue on climate finance. As it stands today, far greater flows of capital are directed to climate mitigation projects aimed at preventing or reducing GHG emissions, which typically offer greater returns for investors, compared to climate adaptation flows into the water sector, which offer direct protection of society against extremes.

Climate adaptation actions can promote the resilience of services 1) despite variations in water resources (droughts) and 2) to withstand asset failures (e.g., due to flood inundation or extreme cyclonic winds). However, a range of barriers exist in attracting the necessary financing for capital improvements and in incentivising adequate levels of network maintenance, the latter being particularly important for achieving long term reductions in leakage losses to strengthen the operational, financial and climate resilience of services. There is no single solution in overcoming these barriers; financing must be coupled with technical capacity building and institutional reform activities. However, risk quantification and adaptation optioneering frameworks can be useful tools for decision makers to allocate limited financial resources and demonstrate the impacts of investments towards the global climate adaptation agenda. Quantifying the impact of investments towards the climate adaptation agenda can also be effective in unlocking concessional financing from the proliferation of climate funds and investors. However, the coherent assessment of risk and adaptation needs in the context of multiple hazards, distributional disruption impacts through a range of channels and wide-ranging adaptation option types poses a challenge for many utilities and investors.

This DPhil thesis introduced a set of interdisciplinary frameworks, that can be applied at both national and global scales to support investment decision making in the water supply sector. The frameworks also serve to demonstrate the impact of infrastructure investments in the water supply sector through the lens of climate adaptation, economic efficiency and wider development aims. With the climate agenda rapidly gaining momentum, there is an opportunity to strengthen recognition that the utilities receive in high level dialogue on climate finance and channel greater financial flows towards closing the climate adaptation gap for water supply utilities.

9. References

Aboelnga, H. (2019) *Running Out of Water: Cities shifting from 24×7 to intermittent water supply, IWA*. Available at: https://iwa-network.org/running-out-of-water-cities-shifting-from-24x7-to-intermittent-water-supply/ (Accessed: 30 June 2023).

Aerts, J. C. J. H. (2018) 'A review of cost estimates for flood adaptation', *Water (Switzerland)*. doi: 10.3390/w10111646.

Ahmad, Q. K. (2003) 'Towards poverty alleviation: The water sector perspectives', *International Journal of Water Resources Development*. doi: 10.1080/0790062032000089356.

Ahopelto, S. and Vahala, R. (2020) 'Cost-benefit analysis of leakage reduction methods in water supply networks', *Water (Switzerland)*. doi: 10.3390/w12010195.

Albiac, J. *et al.* (2003) 'The economic unsustainability of the Spanish National Hydrological Plan', *International Journal of Water Resources Development*. doi: 10.1080/0790062032000122961.

Allan, R. P. *et al.* (2020) 'Advances in understanding large-scale responses of the water cycle to climate change', *Annals of the New York Academy of Sciences*. doi: 10.1111/nyas.14337.

Allen, M. R. and Ingram, W. J. (2002) 'Constraints on future changes in climate and the hydrologic cycle', *Nature*. doi: 10.1038/nature01092.

Altaf, M. A. (1994) 'The economics of household response to inadequate water supplies: evidence from Pakistan', *Third World Planning Review*. doi: 10.3828/twpr.16.1.m1wk8611v47009u3.

Altink, H. (2022) *Drought and water inequity in Caribbean Small Island Developing States*. Available at: https://www.york.ac.uk/igdc/news/news/2022/drought-and-water-inequity/ (Accessed: 6 March 2023).

Andres, L. A. *et al.* (2021) *Troubled Tariffs : Revisiting Water Pricing for Affordable and Sustainable Water Services*. Available at: https://openknowledge.worldbank.org/handle/10986/36661.

De Angeli, S. *et al.* (2022) 'A multi-hazard framework for spatial-temporal impact analysis', *International Journal of Disaster Risk Reduction*. doi: 10.1016/j.ijdrr.2022.102829.

APFM (2020) *Integrated Flood Risk Management Cascade*. Available at:

https://www.floodmanagement.info/integrated-flood-risk-management-cascade/ (Accessed: 9 July 2023).

Armstrong, A. *et al.* (2022) 'Intra-seasonal rainfall and piped water revenue variability in rural Africa', *Global Environmental Change*. doi: 10.1016/j.gloenvcha.2022.102592.

Arrighi, C. *et al.* (2017) 'Flood impacts on a water distribution network', *Natural Hazards and Earth System Sciences*. doi: 10.5194/nhess-17-2109-2017.

Bachmair, S., Kohn, I. and Stahl, K. (2015) 'Exploring the link between drought indicators and impacts', *Natural Hazards and Earth System Sciences*. doi: 10.5194/nhess-15-1381-2015.

Baisa, B. *et al.* (2010) 'The welfare costs of unreliable water service', *Journal of Development Economics*. doi: 10.1016/j.jdeveco.2008.09.010.

Bakker, K. (2017) *Privatizing Water, Privatizing Water*. doi: 10.7591/9780801463617.

Di Baldassarre, G. *et al.* (2018) 'Water shortages worsened by reservoir effects', *Nature Sustainability*. doi: 10.1038/s41893-018-0159-0.

Bangladesh Bureau of Statistics (2020) 'Bangladesh Population and Housing Census 2011', *Bangladesh Bureau of Statistics (Bbs) Statistics and Informatics Division (Sid) Ministry of Planning Government of the People'S Republic of Bangladesh*.

Barbetta, S. *et al.* (2022) 'Assessment of Flooding Impact on Water Supply Systems: A Comprehensive Approach Based on DSS', *Water Resources Management*. doi: 10.1007/s11269-022-03306-x.

Barbour, E. J. *et al.* (2022) 'The unequal distribution of water risks and adaptation benefits in coastal Bangladesh', *Nature Sustainability*. doi: 10.1038/s41893-021-00846-9.

Bata, M. T. H., Carriveau, R. and Ting, D. S. K. (2022) 'Urban water supply systems' resilience under earthquake scenario', *Scientific Reports*. doi: 10.1038/s41598-022-23126-8.

BBC (2017) *Chile floods: Millions without water in capital Santiago*. Available at: https://www.bbc.co.uk/news/world-latin-america-39099313.

Beard, V. A. and Mitlin, D. (2021) 'Water access in global South cities: The challenges of intermittency and affordability', *World Development*. doi: 10.1016/j.worlddev.2021.105625.

Becher, O. *et al.* (2023) 'A Multi-Hazard Risk Framework to Stress-Test Water Supply

Systems to Climate-Related Disruptions', *Earth's Future*. doi: 10.1029/2022EF002946.

Bennett, J. C. *et al.* (2011) 'Performance of quantile-quantile bias-correction for use in hydroclimatological projections', in *MODSIM 2011 - 19th International Congress on Modelling and Simulation - Sustaining Our Future: Understanding and Living with Uncertainty*.

Berghuijs, W. R. *et al.* (2019) 'Growing Spatial Scales of Synchronous River Flooding in Europe', *Geophysical Research Letters*. doi: 10.1029/2018GL081883.

Berglez, P. and Al-Saqaf, W. (2021) 'Extreme weather and climate change: social media results, 2008–2017', *Environmental Hazards*. doi: 10.1080/17477891.2020.1829532.

Bharti, N. *et al.* (2020) 'Dynamics of urban water supply management of two Himalayan towns in India', *Water Policy*. doi: 10.2166/wp.2019.203.

Blauhut, V. *et al.* (2016) 'Estimating drought risk across Europe from reported drought impacts, drought indices, and vulnerability factors', *Hydrology and Earth System Sciences*. doi: 10.5194/hess-20-2779-2016.

Bloemendaal, N. *et al.* (2020) 'Generation of a global synthetic tropical cyclone hazard dataset using STORM', *Scientific Data*. doi: 10.1038/s41597-020-0381-2.

Bondank, E. N., Chester, M. V. and Ruddell, B. L. (2018) 'Water Distribution System Failure Risks with Increasing Temperatures', *Environmental Science and Technology*. doi: 10.1021/acs.est.7b01591.

Borgomeo, E. *et al.* (2014) 'Risk-based water resources planning: Incorporating probabilistic nonstationary climate uncertainties', *Water Resources Research*. doi: 10.1002/2014WR015558.

Borgomeo, E. *et al.* (2015) 'Assessing water resource system vulnerability to unprecedented hydrological drought using copulas to characterize drought duration and deficit', *Water Resources Research*. doi: 10.1002/2015WR017324.

Borgomeo, E. *et al.* (2016) 'Trading-off tolerable risk with climate change adaptation costs in water supply systems', *Water Resources Research*. doi: 10.1002/2015WR018164.

Borgomeo, E. *et al.* (2018) 'Risk, Robustness and Water Resources Planning Under Uncertainty', *Earth's Future*. doi: 10.1002/2017EF000730.

Borgomeo, E. (2022) ‘Water Resource System Modelling for Climate Adaptation’, in *Springer Climate*. doi: 10.1007/978-3-030-86211-4_17.

Borgomeo, E., Hall, J. W. and Salehin, M. (2018) ‘Avoiding the water-poverty trap: insights from a conceptual human-water dynamical model for coastal Bangladesh’, *International Journal of Water Resources Development*. doi: 10.1080/07900627.2017.1331842.

Boucher, O. *et al.* (2020) ‘Presentation and Evaluation of the IPSL-CM6A-LR Climate Model’, *Journal of Advances in Modeling Earth Systems*. doi: 10.1029/2019MS002010.

Boulange, J. *et al.* (2021) ‘Role of dams in reducing global flood exposure under climate change’, *Nature Communications*. doi: 10.1038/s41467-020-20704-0.

Briscoe, J. (1999) ‘The financing of hydropower, irrigation and water supply infrastructure in developing countries’, *International Journal of Water Resources Development*. doi: 10.1080/07900629948718.

Brown, C. *et al.* (2011) ‘Hydroclimate risk to economic growth in sub-Saharan Africa’, *Climatic Change*. doi: 10.1007/s10584-010-9956-9.

Brown, C. *et al.* (2012) ‘Decision scaling: Linking bottom-up vulnerability analysis with climate projections in the water sector’, *Water Resources Research*. doi: 10.1029/2011WR011212.

Brozović, N., Sunding, D. L. and Zilberman, D. (2007) ‘Estimating business and residential water supply interruption losses from catastrophic events’, *Water Resources Research*. doi: 10.1029/2005WR004782.

van der Bruggen, B., Borghgraef, K. and Vinckier, C. (2010) ‘Causes of water supply problems in urbanised regions in developing countries’, *Water Resources Management*. doi: 10.1007/s11269-009-9529-8.

Brunner, M. I. *et al.* (2020) ‘Spatial Dependence of Floods Shaped by Spatiotemporal Variations in Meteorological and Land-Surface Processes’, *Geophysical Research Letters*. doi: 10.1029/2020GL088000.

Buck, S. *et al.* (2016) ‘Measuring welfare losses from urban water supply disruptions’, *Journal of the Association of Environmental and Resource Economists*. doi: 10.1086/687761.

Buechel, M., Slater, L. and Dadson, S. (2022) ‘Hydrological impact of widespread afforestation

in Great Britain using a large ensemble of modelled scenarios', *Communications Earth and Environment*. doi: 10.1038/s43247-021-00334-0.

Buldyrev, S. V. *et al.* (2010) 'Catastrophic cascade of failures in interdependent networks', *Nature*. doi: 10.1038/nature08932.

Burdescu, R. *et al.* (2020) *A Benchmark for the Performance of State-Owned Water Utilities in the Caribbean, A Benchmark for the Performance of State-Owned Water Utilities in the Caribbean*. doi: 10.1596/33251.

Burek, P. *et al.* (2020) 'Development of the Community Water Model (CWatM v1.04) - A high-resolution hydrological model for global and regional assessment of integrated water resources management', *Geoscientific Model Development*. doi: 10.5194/gmd-13-3267-2020.

Burgess, C. P. *et al.* (2015) 'A macro-scale flood risk model for Jamaica with impact of climate variability', *Natural Hazards*. doi: 10.1007/s11069-015-1712-z.

Bussi, G. *et al.* (2021) 'Impact of dams and climate change on suspended sediment flux to the Mekong delta', *Science of the Total Environment*. doi: 10.1016/j.scitotenv.2020.142468.

Buurman, J., Mens, M. J. P. and Dahm, R. J. (2017) 'Strategies for urban drought risk management: a comparison of 10 large cities', *International Journal of Water Resources Development*. doi: 10.1080/07900627.2016.1138398.

Byers, E. A. *et al.* (2020) 'Drought and climate change impacts on cooling water shortages and electricity prices in Great Britain', *Nature Communications*. doi: 10.1038/s41467-020-16012-2.

Byun, H. R. and Wilhite, D. A. (1999) 'Objective quantification of drought severity and duration', *Journal of Climate*. doi: 10.1175/1520-0442(1999)012<2747:OQODSA>2.0.CO;2.

Campbell, D., Barker, D. and McGregor, D. (2011) 'Dealing with drought: Small farmers and environmental hazards in southern St. Elizabeth, Jamaica', *Applied Geography*. doi: 10.1016/j.apgeog.2010.03.007.

Carrão, H., Naumann, G. and Barbosa, P. (2016) 'Mapping global patterns of drought risk: An empirical framework based on sub-national estimates of hazard, exposure and vulnerability', *Global Environmental Change*, 39, pp. 108–124.

Carrick, J. *et al.* (2019) 'Is planting trees the solution to reducing flood risks?', *Journal of*

Flood Risk Management. doi: 10.1111/jfr3.12484.

Casson, E. and Coles, S. (2000) 'Simulation and extremal analysis of hurricane events', *Journal of the Royal Statistical Society. Series C: Applied Statistics*. doi: 10.1111/1467-9876.00189.

Chakrabarti, M. (On P. (2019) *Nearly 11 Million People In The Indian City Of Chennai Are Almost Out Of Water*. Available at: https://www.wbur.org/onpoint/2019/08/01/india-chennai-water-shortage-crisis-infrastructure.

Chang, L. C., Chen, Y. W. and Yeh, M. S. (2009) 'Optimizing system capacity expansion schedules for groundwater supply', *Water Resources Research*. doi: 10.1029/2008WR006835.

Chau, T. K., Thanh, N. T. and Toan, N. T. (2021) 'Primarily results of a real-time flash flood warning system in Vietnam', *Civil Engineering Journal (Iran)*. doi: 10.28991/cej-2021-03091687.

Chitonge, H. (2014) 'Cities Beyond Networks: The Status of Water Services for the Urban Poor in African Cities', *African Studies*. doi: 10.1080/00020184.2014.887743.

Christoplos, I. (2010) 'Understanding Climate Change Adaptation: Lessons from Community-based Approaches', *Climate and Development*. doi: 10.3763/cdev.2010.0028.

Clark, M. P. *et al.* (2015) 'A unified approach for process-based hydrologic modeling: 1. Modeling concept', *Water Resources Research*. doi: 10.1002/2015WR017198.

Clasen, T. F. (2012) 'Millennium Development Goals water target claim exaggerates achievement', *Tropical Medicine and International Health*. doi: 10.1111/j.1365-3156.2012.03052.x.

Collyns, D. (2017) *How can Peru prepare to withstand more devastating floods and landslides?*, *The Guardian*. Available at: https://www.theguardian.com/global-development-professionals-network/2017/apr/13/peru-prevent-floods-landslides-climate-change (Accessed: 9 March 2021).

Conway, D. and Mustelin, J. (2014) 'Strategies for improving adaptation practice in developing countries', *Nature Climate Change*. doi: 10.1038/nclimate2199.

Cook, J., Kimuyu, P. and Whittington, D. (2016) 'The costs of coping with poor water supply in rural Kenya', *Water Resources Research*. doi: 10.1002/2015WR017468.

De Córdoba, J., Campo-Flores, A. and De Avila, J. (2017) *Puerto Rico After Maria: No Power, No Cell Signal, and Lots of Damage*, *Wall Street Journal*. Available at: https://www.wsj.com/articles/puerto-rico-braces-for-more-flooding-as-maria-dumps-more-rain-1505998033 (Accessed: 22 March 2021).

CPI (2020) *Tracking climate finance per sector*. Available at: https://www.climatepolicyinitiative.org/tracking-climate-finance-by-sector/.

Croke, B. F. W. and Jakeman, A. J. (2004) 'A catchment moisture deficit module for the IHACRES rainfall-runoff model', *Environmental Modelling and Software*. doi: 10.1016/j.envsoft.2003.09.001.

Cromwell, J., Smith, J. and Raucher, R. (2007) *Implications of Climate Change for Urban Water Utilities*. Washington DC, USA: Association of Metropolitan Water Agencies.

Cross, P. and Morel, A. (2005) 'Pro-poor strategies for urban water supply and sanitation services delivery in Africa', *Water Science and Technology*. doi: 10.2166/wst.2005.0223.

Culley, S. *et al.* (2016) 'A bottom-up approach to identifying the maximum operational adaptive capacity of water resource systems to a changing climate', *Water Resources Research*. doi: 10.1002/2015WR018253.

Dadson, S. J. *et al.* (2017) 'A restatement of the natural science evidence concerning catchment-based "natural" flood management in the UK', *Proceedings of the Royal Society A: Mathematical, Physical and Engineering Sciences*. doi: 10.1098/rspa.2016.0706.

Damania, R. *et al.* (2017) *Uncharted Waters: The New Economics of Water Scarcity and Variability*, *Uncharted Waters: The New Economics of Water Scarcity and Variability*. doi: 10.1596/978-1-4648-1179-1.

Danilenko, A. (2010) 'Climate Change and Urban Water Utilities: Challenges and Opportunities', *P-notes*.

Deshpande, A. *et al.* (2020) 'Mapping geographical inequalities in access to drinking water and sanitation facilities in low-income and middle-income countries, 2000-17', *The Lancet Global Health*. doi: 10.1016/S2214-109X(20)30278-3.

Diao, K. *et al.* (2016) 'Global resilience analysis of water distribution systems', *Water Research*. doi: 10.1016/j.watres.2016.10.011.

Dillon, P. *et al.* (2019) ‘Sixty years of global progress in managed aquifer recharge’, *Hydrogeology Journal*. doi: 10.1007/s10040-018-1841-z.

Dobson, B. *et al.* (2020) ‘The Spatial Dynamics of Droughts and Water Scarcity in England and Wales’, *Water Resources Research*. doi: 10.1029/2020WR027187.

Dong, X. *et al.* (2020) ‘Vulnerability of urban water infrastructures to climate change at city level’, *Resources, Conservation and Recycling*. doi: 10.1016/j.resconrec.2020.104918.

Duan, W. *et al.* (2022) ‘Changes in temporal inequality of precipitation extremes over China due to anthropogenic forcings’, *npj Climate and Atmospheric Science*. doi: 10.1038/s41612-022-00255-5.

Duha Metin, A. *et al.* (2020) ‘The role of spatial dependence for large-scale flood risk estimation’, *Natural Hazards and Earth System Sciences*. doi: 10.5194/nhess-20-967-2020.

Dunant, A. *et al.* (2021) ‘Multihazards Scenario Generator: A Network-Based Simulation of Natural Disasters’, *Risk Analysis*. doi: 10.1111/risa.13723.

Dunne, J. P. *et al.* (2020) ‘The GFDL Earth System Model Version 4.1 (GFDL-ESM 4.1): Overall Coupled Model Description and Simulation Characteristics’, *Journal of Advances in Modeling Earth Systems*. doi: 10.1029/2019MS002015.

Eakin, H. *et al.* (2016) ‘Adapting to risk and perpetuating poverty: Household’s strategies for managing flood risk and water scarcity in Mexico City’, *Environmental Science and Policy*. doi: 10.1016/j.envsci.2016.06.006.

Eguchi, Y., Hattori, Y. and Nomura, M. (2021) ‘Comparative assessment of validity of gradient wind models for a translating tropical cyclone’, *SN Applied Sciences*. doi: 10.1007/s42452-021-04406-w.

Einav, R., Harussi, K. and Perry, D. (2003) ‘The footprint of the desalination processes on the environment’, *Desalination*. doi: 10.1016/S0011-9164(02)01057-3.

Emanuel, K. *et al.* (2006) ‘A statistical deterministic approach to hurricane risk assessment’, *Bulletin of the American Meteorological Society*. doi: 10.1175/BAMS-87-3-299.

Ercumen, A. *et al.* (2015) ‘Upgrading a Piped Water Supply from Intermittent to Continuous Delivery and Association with Waterborne Illness: A Matched Cohort Study in Urban India’, *PLoS Medicine*. doi: 10.1371/journal.pmed.1001892.

Espada, R. J., Apan, A. and Mcdougall, K. (2013) 'Using spatial modelling to develop flood risk and climate adaptation capacity metrics for vulnerability assessments of urban community and critical water supply infrastructure', *Spatial Modelling of Flood Risk and Climate Adaptation Capacity Metrics 49th ISOCARP Congress 2013*.

Eyring, V. *et al.* (2016) 'Overview of the Coupled Model Intercomparison Project Phase 6 (CMIP6) experimental design and organization', *Geoscientific Model Development*. doi: 10.5194/gmd-9-1937-2016.

FAO (2019) *National Water Sector Policy and Implementation Plan 2019*. Available at: https://www.fao.org/faolex/results/details/en/c/LEX-FAOC204202/.

FAO (2020) *AQUASTAT*. Available at: http://www.fao.org/aquastat/statistics/query/index.html (Accessed: 10 March 2021).

FEMA (2020) *Multi-hazard Loss Estimation Methodology Hazus-MH Flood Technical Manual*. Available at: https://www.fema.gov/sites/default/files/2020-09/fema_hazus_flood-model_technical-manual_2.1.pdf.

Filippo, G. *et al.* (2021) *ictp-esp/RegCM: Paper Release (4.7.1)*. Available at: https://doi.org/10.5281/zenodo.4603556.

Fiorillo, D. *et al.* (2021) 'Assessing the Impact of Climate Change on Future Water Demand using Weather Data', *Water Resources Management*. doi: 10.1007/s11269-021-02789-4.

Flörke, M., Schneider, C. and McDonald, R. I. (2018) 'Water competition between cities and agriculture driven by climate change and urban growth', *Nature Sustainability*. doi: 10.1038/s41893-017-0006-8.

Freire-González, J., Decker, C. and Hall, J. W. (2017) 'The Economic Impacts of Droughts: A Framework for Analysis', *Ecological Economics*. doi: 10.1016/j.ecolecon.2016.11.005.

Funk, C. *et al.* (2015) 'The climate hazards infrared precipitation with stations - A new environmental record for monitoring extremes', *Scientific Data*. doi: 10.1038/sdata.2015.66.

Ganguli, P. *et al.* (2020) 'Projected Changes in Compound Flood Hazard From Riverine and Coastal Floods in Northwestern Europe', *Earth's Future*. doi: 10.1029/2020EF001752.

Gao, J. and O'Neill, B. C. (2020) 'Mapping global urban land for the 21st century with data-driven simulations and Shared Socioeconomic Pathways', *Nature Communications*. doi:

10.1038/s41467-020-15788-7.

Garcia, O. R. *et al.* (2021) 'For the greater good or the greater need? Optimizing rural drinking water supply infrastructure to account for heterogenous groundwater quality and household poverty in coastal Bangladesh', *Unpublished*.

Garrick, D., O'Donnell, E., *et al.* (2019) *Informal Water Markets in an Urbanising World : Some Unanswered Questions*. Available at: https://documents.worldbank.org/en/publication/documents-reports/documentdetail/358461549427540914/informal-water-markets-in-an-urbanising-world-some-unanswered-questions.

Garrick, D., De Stefano, L., *et al.* (2019) 'Rural water for thirsty cities: A systematic review of water reallocation from rural to urban regions', *Environmental Research Letters*. doi: 10.1088/1748-9326/ab0db7.

Gassert, F. *et al.* (2014) 'Aqueduct Global Maps 2.1: Constructing decision-relevant global water risk indicators', *World Resources Institute*.

Gaupp, F. *et al.* (2019) 'Increasing risks of multiple breadbasket failure under 1.5 and 2 °C global warming', *Agricultural Systems*. doi: 10.1016/j.agsy.2019.05.010.

Gaupp, F. *et al.* (2020) 'Changing risks of simultaneous global breadbasket failure', *Nature Climate Change*. doi: 10.1038/s41558-019-0600-z.

Gaupp, F., Hall, J. and Dadson, S. (2015) 'The role of storage capacity in coping with intra- and inter-annual water variability in large river basins', *Environmental Research Letters*. doi: 10.1088/1748-9326/10/12/125001.

Gellert, P. K. and Lynch, B. D. (2003) 'Mega-projects as displacements', *International Social Science Journal*. doi: 10.1111/1468-2451.5501002.

GFDRR (2018) *Resilient Water Supply And Sanitation Services: The Case of Japan*. Available at: https://reliefweb.int/report/japan/resilient-water-supply-and-sanitation-services-case-japan-0 (Accessed: 25 June 2023).

Ghaffour, N., Missimer, T. M. and Amy, G. L. (2013) 'Technical review and evaluation of the economics of water desalination: Current and future challenges for better water supply sustainability', *Desalination*. doi: 10.1016/j.desal.2012.10.015.

Gheisi, A., Forsyth, M. and Naser, G. (2016) 'Water Distribution Systems Reliability: A Review of Research Literature', *Journal of Water Resources Planning and Management*. doi: 10.1061/(asce)wr.1943-5452.0000690.

Ghile, Y. B. *et al.* (2014) 'Bottom-up climate risk assessment of infrastructure investment in the Niger River Basin', *Climatic Change*. doi: 10.1007/s10584-013-1008-9.

Gianfagna, C. C. *et al.* (2015) 'Watershed area ratio accurately predicts daily streamflow in nested catchments in the Catskills, New York', *Journal of Hydrology: Regional Studies*. doi: 10.1016/j.ejrh.2015.09.002.

Gill, S. E. *et al.* (2007) 'Adapting cities for climate change: The role of the green infrastructure', *Built Environment*. doi: 10.2148/benv.33.1.115.

Giuffrida, A. (2022) *More than 100 towns in Italy's Po valley asked to ration water, Guardian*.

Glas, H. *et al.* (2015) 'A GIS-based flood risk tool for Jamaica: The first step towards a multi-hazard risk assessment in the Caribbean', in *International Multidisciplinary Scientific GeoConference Surveying Geology and Mining Ecology Management, SGEM*. doi: 10.5593/sgem2015/b22/s11.080.

Gleeson, T. *et al.* (2012) 'Water balance of global aquifers revealed by groundwater footprint', *Nature*. doi: 10.1038/nature11295.

Gleick, P. H. (1998) 'The human right to water', *Water Policy*, 1(5), pp. 487–503.

Global Petrol Prices (2022) *Electricity prices for businesses*. Available at: https://www.globalpetrolprices.com/electricity_prices/.

Gober, P. (2014) 'Decision making under uncertainty: A new paradigm for water resources planning and management', in *Modern Water Resources Engineering*. doi: 10.1007/978-1-62703-595-8_8.

Gosling, S. N. and Arnell, N. W. (2016) 'A global assessment of the impact of climate change on water scarcity', *Climatic Change*. doi: 10.1007/s10584-013-0853-x.

Goyal, M. K., Madramootoo, C. A. and Richards, J. F. (2015) 'Simulation of the Streamflow for the Rio Nuevo Watershed of Jamaica for Use in Agriculture Water Scarcity Planning', *Journal of Irrigation and Drainage Engineering*. doi: 10.1061/(asce)ir.1943-4774.0000802.

Grasham, C. F. *et al.* (2022) 'Equitable urban water security: beyond connections on premises',

Environmental Research: Infrastructure and Sustainability. doi: 10.1088/2634-4505/ac9c8d.

Grasham, C. F., Charles, K. J. and Abdi, T. G. (2022) '(Re-)orienting the Concept of Water Risk to Better Understand Inequities in Water Security', *Frontiers in Water*. doi: 10.3389/frwa.2021.799515.

Greve, P. *et al.* (2018) 'Global assessment of water challenges under uncertainty in water scarcity projections', *Nature Sustainability*. doi: 10.1038/s41893-018-0134-9.

Grey, D. *et al.* (2013) 'Water security in one blue planet: Twenty-first century policy challenges for science', *Philosophical Transactions of the Royal Society A: Mathematical, Physical and Engineering Sciences*. doi: 10.1098/rsta.2012.0406.

Grey, D. and Sadoff, C. W. (2007) 'Sink or Swim? Water security for growth and development', *Water Policy*. doi: 10.2166/wp.2007.021.

Griffin, R. C. and Mjelde, J. W. (2000) 'Valuing water supply reliability', *American Journal of Agricultural Economics*. doi: 10.1111/0002-9092.00035.

Guardian (2022) *'It's plunder': Mexico desperate for water while drinks companies use billions of litres*. Available at: https://www.theguardian.com/global-development/2022/jul/28/water-is-the-real-thing-but-millions-of-mexicans-are-struggling-without-it.

Guillod, B. P. *et al.* (2018) 'A large set of potential past, present and future hydro-meteorological time series for the UK', *Hydrology and Earth System Sciences*. doi: 10.5194/hess-22-611-2018.

Guo, D., Westra, S. and Maier, H. R. (2018) 'An inverse approach to perturb historical rainfall data for scenario-neutral climate impact studies', *Journal of Hydrology*. doi: 10.1016/j.jhydrol.2016.03.025.

Gutjahr, O. *et al.* (2019) 'Max Planck Institute Earth System Model (MPI-ESM1.2) for the High-Resolution Model Intercomparison Project (HighResMIP)', *Geoscientific Model Development*. doi: 10.5194/gmd-12-3241-2019.

GWI (2022a) *GWI DesalData*.

GWI (2022b) *GWI WaterData*.

Haasnoot, M. *et al.* (2013) 'Dynamic adaptive policy pathways: A method for crafting robust

decisions for a deeply uncertain world', *Global Environmental Change*. doi: 10.1016/j.gloenvcha.2012.12.006.

Haddeland, I. *et al.* (2014) 'Global water resources affected by human interventions and climate change', *Proceedings of the National Academy of Sciences of the United States of America*. doi: 10.1073/pnas.1222475110.

Hall, J. and Borgomeo, E. (2013) 'Risk-based principles for defining and managing water security', *Philosophical Transactions of the Royal Society A: Mathematical, Physical and Engineering Sciences*. doi: 10.1098/rsta.2012.0407.

Hall, J. W. *et al.* (2003) 'A methodology for national-scale flood risk assessment', *Proceedings of the Institution of Civil Engineers: Water and Maritime Engineering*. doi: 10.1680/wame.2003.156.3.235.

Hall, J. W. *et al.* (2012) 'Towards risk-based water resources planning in England and Wales under a changing climate', *Water and Environment Journal*. doi: 10.1111/j.1747-6593.2011.00271.x.

Hall, J. W. *et al.* (2014) 'Coping with the curse of freshwater variability', *Science*. doi: 10.1126/science.1257890.

Hall, J. W. *et al.* (2019) *Adaptation of Infrastructure Systems: Background Paper for the Global Commission on Adaptation*. Available at: https://gca.org/reports/adaptation-of-infrastructure-systems/.

Hall, J. W. and Leng, G. (2019) 'Can we calculate drought risk… and do we need to?', *Wiley Interdisciplinary Reviews: Water*. doi: 10.1002/wat2.1349.

Hallegatte, S. *et al.* (2012) 'Investment Decision Making Under Deep Uncertainty: Application to Climate Change', *Policy Research Working Paper*.

Hallegatte, S. *et al.* (2016) *Shock Waves: Managing the Impacts of Climate Change on Poverty*, *Shock Waves: Managing the Impacts of Climate Change on Poverty*. doi: 10.1596/978-1-4648-0673-5.

Hallegatte, S. *et al.* (2017) *Unbreakable: Building the Resilience of the Poor in the Face of Natural Disasters*, *Unbreakable: Building the Resilience of the Poor in the Face of Natural Disasters*. doi: 10.1596/978-1-4648-1003-9.

Hallegatte, S., Rentschler, J. and Rozenberg, J. (2019) *Lifelines: The Resilient Infrastructure Opportunity*, *Lifelines: The Resilient Infrastructure Opportunity*. doi: 10.1596/978-1-4648-1430-3.

Hallegatte, S. and Rozenberg, J. (2017) 'Climate change through a poverty lens', *Nature Climate Change*. doi: 10.1038/nclimate3253.

Hammond, J. C. *et al.* (2022) 'Going Beyond Low Flows: Streamflow Drought Deficit and Duration Illuminate Distinct Spatiotemporal Drought Patterns and Trends in the U.S. During the Last Century', *Water Resources Research*. doi: 10.1029/2022WR031930.

Hanasaki, N. *et al.* (2013) 'A global water scarcity assessment under Shared Socio-economic Pathways - Part 2: Water availability and scarcity', *Hydrology and Earth System Sciences*. doi: 10.5194/hess-17-2393-2013.

Hassanzadeh, E. *et al.* (2016) 'A risk-based framework for water resource management under changing water availability, policy options, and irrigation expansion', *Advances in Water Resources*. doi: 10.1016/j.advwatres.2016.05.018.

Hassanzadeh, E. *et al.* (2019) 'Quantile-based downscaling of rainfall extremes: Notes on methodological functionality, associated uncertainty and application in practice', *Advances in Water Resources*. doi: 10.1016/j.advwatres.2019.07.001.

He, C. *et al.* (2021) 'Future global urban water scarcity and potential solutions', *Nature Communications*. doi: 10.1038/s41467-021-25026-3.

He, X. and Cha, E. J. (2021) 'Risk-Informed Decision-Making for Predisaster Risk Mitigation Planning of Interdependent Infrastructure Systems: Case Study of Jamaica', *Journal of Infrastructure Systems*. doi: 10.1061/(asce)is.1943-555x.0000634.

Heffernan, J. E. and Tawn, J. A. (2004) 'A conditional approach for multivariate extreme values', *Journal of the Royal Statistical Society. Series B: Statistical Methodology*. doi: 10.1111/j.1467-9868.2004.02050.x.

Heflin, C., Jensen, J. and Miller, K. (2014) 'Understanding the economic impacts of disruptions in water service', *Evaluation and Program Planning*. doi: 10.1016/j.evalprogplan.2014.05.003.

Henriques, C. and Spraggs, G. (2011) 'Practical Paper Alleviating the flood risk of critical

water supply sites: Asset and system resilience', *Journal of Water Supply: Research and Technology - AQUA*. doi: 10.2166/aqua.2011.070.

Heo, J. H., Salas, J. D. and Boes, D. C. (2001) 'Regional flood frequency analysis based on a Weibull model: Part 2. Simulations and applications', *Journal of Hydrology*. doi: 10.1016/S0022-1694(00)00335-8.

Herman, J. D. *et al.* (2015) 'How Should Robustness Be Defined for Water Systems Planning under Change?', *Journal of Water Resources Planning and Management*. doi: 10.1061/(asce)wr.1943-5452.0000509.

Herman, J. D. *et al.* (2016) 'Synthetic Drought Scenario Generation to Support Bottom-Up Water Supply Vulnerability Assessments', *Journal of Water Resources Planning and Management*. doi: 10.1061/(asce)wr.1943-5452.0000701.

Hernández-Mora, N. *et al.* (2014) 'Interbasin water transfers in Spain: Interregional conflicts and governance responses', in *Globalized Water: A Question of Governance*. doi: 10.1007/978-94-007-7323-3_13.

Hersbach, H. *et al.* (2020) 'The ERA5 global reanalysis', *Quarterly Journal of the Royal Meteorological Society*. doi: 10.1002/qj.3803.

Hillier, J. K. *et al.* (2020) 'Multi-hazard dependencies can increase or decrease risk', *Nature Climate Change*. doi: 10.1038/s41558-020-0832-y.

Hirabayashi, Y. *et al.* (2013) 'Global flood risk under climate change', *Nature Climate Change*. doi: 10.1038/nclimate1911.

Hope, R. and Rouse, M. (2013) 'Risks and responses to universal drinking water security', *Philosophical Transactions of the Royal Society A: Mathematical, Physical and Engineering Sciences*. doi: 10.1098/rsta.2012.0417.

Huang, Z. *et al.* (2018) 'Reconstruction of global gridded monthly sectoral water withdrawals for 1971-2010 and analysis of their spatiotemporal patterns', *Hydrology and Earth System Sciences*. doi: 10.5194/hess-22-2117-2018.

Huberts, A. *et al.* (2023) 'Making scarcity "enough": The hidden household costs of adapting to water scarcity in Mexico City', *PLOS Water*. doi: 10.1371/journal.pwat.0000056.

Huizinga, J., de Moel, H. and Szewczyk, W. (2017) *Global flood depth-damage functions.*

Methodology and the database with guidelines, Joint Research Centre (JRC).

Hulme, M., O'Neill, S. J. and Dessai, S. (2011) 'Is weather event attribution necessary for adaptation funding?', *Science*. doi: 10.1126/science.1211740.

Huskova, I. *et al.* (2016) 'Screening robust water infrastructure investments and their trade-offs under global change: A London example', *Global Environmental Change*. doi: 10.1016/j.gloenvcha.2016.10.007.

Hussien, W. A., Memon, F. A. and Savic, D. A. (2016) 'Assessing and Modelling the Influence of Household Characteristics on Per Capita Water Consumption', *Water Resources Management*. doi: 10.1007/s11269-016-1314-x.

Hutton, G. (2013) 'Global costs and benefits of reaching universal coverage of sanitation and drinking-water supply', *Journal of Water and Health*. doi: 10.2166/wh.2012.105.

Hutton, G. and Varughese, M. (2016) *The Costs of Meeting the 2030 Sustainable Development Goal Targets on Drinking Water, Sanitation, and Hygiene*, *The Costs of Meeting the 2030 Sustainable Development Goal Targets on Drinking Water, Sanitation, and Hygiene*. doi: 10.1596/k8543.

IBNET (2022) *IBNET English | The International Benchmarking Network*. Available at: https://www.ib-net.org/.

IEA (2020) *Climate hazard assessment*. Available at: https://www.iea.org/reports/climate-resilience-policy-indicator/climate-hazard-assessment (Accessed: 10 July 2023).

IFRC (2020) *Indonesia: West Kalimantan & South Sulawesi Floods*. Available at: https://go.ifrc.org/emergencies/4641 (Accessed: 25 June 2023).

Ionita, M. *et al.* (2020) 'On the curious case of the recent decade, mid-spring precipitation deficit in central Europe', *npj Climate and Atmospheric Science*. doi: 10.1038/s41612-020-00153-8.

IPCC (2001) 'Climate Change 2001: Impacts, Adaptation, and Vulnerability. Contribution of Working Group II to the Third Assessment Report. Cambridge University Press, Cambridge, UK', *Ipcc*.

IPCC (2007) *Climate Change 2007 synthesis report: Contributions of working group I, II and III to the fourth assessment report of intergovernmental panel on climate change*, *Climate*

Change 2007 Mitigation of Climate Change.

Jamaica Observer (2018) *$60,000 water bill*. Available at: https://www.jamaicaobserver.com/news/-60-000-water-bill-kitson-town-st-catherine-residents-dig-deep-to-access-basic-commodity_138054 (Accessed: 6 April 2022).

Jamaica Water Resources Authority (2020) *Water Information System*. Available at: http://webmapjam.dyndns.pro/webmap/app/db/index.php (Accessed: 16 January 2021).

JBA (2020a) *Global 30m flood maps*. Available at: https://www.jbarisk.com/flood-services/maps-and-analytics/global-flood-maps/ (Accessed: 2 January 2021).

JBA (2020b) *JBA Global Flood Model Technical Report*.

Jenkins, K. *et al.* (2021) 'An Integrated Framework for Risk-Based Analysis of Economic Impacts of Drought and Water Scarcity in England and Wales', *Water Resources Research*. doi: 10.1029/2020WR027715.

Jensen, O. and Wu, X. (2016) 'Embracing Uncertainty in Policy-Making: The Case of the Water Sector', *Policy and Society*. doi: 10.1016/j.polsoc.2016.07.002.

JIS (2008) *Access to Clean Water Essential to Poverty Alleviation*. Available at: https://jis.gov.jm/access-to-clean-water-essential-to-poverty-alleviation/ (Accessed: 30 May 2023).

JIS (2015) *NWC Spends $100 Million to Truck Water*. Available at: https://jis.gov.jm/nwc-spends-100-million-to-truck-water/.

JMP (2017) *Data*. Available at: https://washdata.org/data (Accessed: 10 March 2021).

Jones *et al.* (2022) 'Current wastewater treatment targets are insufficient to protect surface water quality', *Commun Earth Environ*, 3(221).

Jongman, B. *et al.* (2014) 'Increasing stress on disaster-risk finance due to large floods', *Nature Climate Change*. doi: 10.1038/nclimate2124.

Jongman, B. (2018) 'Effective adaptation to rising flood risk', *Nature Communications*, 9(1986). doi: https://doi.org/10.1038/s41467-018-04396-1.

Kahil, T. *et al.* (2018) 'A Continental-Scale Hydroeconomic Model for Integrating Water-Energy-Land Nexus Solutions', *Water Resources Research*. doi: 10.1029/2017WR022478.

Karim, A. and Chowdhury, J. U. (1996) ‘Comparison of Five Distributions for Suitability in Flood Frequency Analysis in Bangladesh’, in. doi: 10.1007/978-94-011-0389-3_37.

Karl, T. R. and Trenberth, K. E. (2003) ‘Modern Global Climate Change’, *Science*. doi: 10.1126/science.1090228.

Keller, A., Sakthivadivel, R. and Seckler, D. (2000) ‘Water scarcity and the role of storage in development’, *Water Management*.

Khatri, K., Vairavamoorthy, K. and Porto, M. (2009) ‘Challenges for urban water supply and sanitation in developing countries’, in *Water for a Changing World - Developing Local Knowledge and Capacity - Proceedings of the International Symposium on Water for a Changing World*. doi: 10.1201/9780203878057.ch7.

King, A. D., Reid, K. J. and Saunders, K. R. (2023) ‘Communicating the link between climate change and extreme rain events’, *Nature Geoscience*, (16), pp. 552–554.

Kingsborough, A., Borgomeo, E. and Hall, J. W. (2016) ‘Adaptation pathways in practice: Mapping options and trade-offs for London’s water resources’, *Sustainable Cities and Society*. doi: 10.1016/j.scs.2016.08.013.

Kirshen, P. (2007) ‘Adaptation Options and Costs in Water Supply’, *Africa*.

Klein, R. J. T. (2010) *Mainstreaming Climate Adaptation into Development: A Policy Dilemma*. Washington DC, USA. Available at: https://elibrary.worldbank.org/doi/10.1596/9780821379943_CH01.

Knutson, T. *et al.* (2020) ‘Tropical cyclones and climate change assessment part II: Projected response to anthropogenic warming’, *Bulletin of the American Meteorological Society*. doi: 10.1175/BAMS-D-18-0194.1.

Koks, E. *et al.* (2019) ‘Understanding Business Disruption and Economic Losses Due to Electricity Failures and Flooding’, *International Journal of Disaster Risk Science*. doi: 10.1007/s13753-019-00236-y.

Koks, E. E. *et al.* (2019) ‘A global multi-hazard risk analysis of road and railway infrastructure assets’, *Nature Communications*. doi: 10.1038/s41467-019-10442-3.

Krueger, E. H. *et al.* (2020) ‘Balancing security, resilience, and sustainability of urban water supply systems in a desirable operating space’, *Environmental Research Letters*. doi:

10.1088/1748-9326/ab6c2d.

Krueger, E., Rao, P. S. C. and Borchardt, D. (2019) ‘Quantifying urban water supply security under global change’, *Global Environmental Change*. doi: 10.1016/j.gloenvcha.2019.03.009.

Kuczera, G. (1992) ‘Water supply headworks simulation using network linear programming’, *Advances in Engineering Software*. doi: 10.1016/0965-9978(92)90084-S.

Kumar-Rao, A. (2019) ‘India’s water crisis could be helped by better building, planning’, *National Geographics. Environment.*

Lafon, T. *et al.* (2013) ‘Bias correction of daily precipitation simulated by a regional climate model: A comparison of methods’, *International Journal of Climatology*. doi: 10.1002/joc.3518.

Lai, Y. *et al.* (2020) ‘Greater flood risks in response to slowdown of tropical cyclones over the coast of China’, *Proceedings of the National Academy of Sciences of the United States of America*. doi: 10.1073/pnas.1918987117.

Lakhani, N. (2022) *This article is more than 8 months old Puerto Rico battles blackout and lack of safe water in wake of Hurricane Fiona*, *Guardian*. Available at: https://www.theguardian.com/world/2022/sep/19/puerto-rico-hurricane-fiona-power-water (Accessed: 25 June 2023).

Lamb, R. *et al.* (2010) ‘A new method to assess the risk of local and widespread flooding on rivers and coasts’, *Journal of Flood Risk Management*. doi: 10.1111/j.1753-318X.2010.01081.x.

Lamb, R. *et al.* (2019) ‘A Probabilistic Model of the Economic Risk to Britain’s Railway Network from Bridge Scour During Floods’, *Risk Analysis*. doi: 10.1111/risa.13370.

Lamb, R., Crossley, M. and Waller, S. (2009) ‘A fast two-dimensional floodplain inundation model’, *Proceedings of the Institution of Civil Engineers: Water Management*. doi: 10.1680/wama.2009.162.6.363.

Lambert, A. (2000) ‘What Do We Know About Pressure: Leakage Relationships in Distribution Systems?’, in *IWA Conference on Systems Approach to Leakage Control and Water Distribution SystemManagement*.

von Lany, P. H. *et al.* (2013) ‘Applying Optimisation and Uncertainty Analysis to Help

Develop an Integrated Water Resources Plan for South East England', *Water Resources Management*. doi: 10.1007/s11269-012-0121-2.

Larsen, T. A. *et al.* (2016) 'Emerging solutions to the water challenges of an urbanizing world', *Science*. doi: 10.1126/science.aad8641.

Larson, K. L. *et al.* (2015) 'Decision-making under uncertainty for water sustainability and urban climate change adaptation', *Sustainability (Switzerland)*. doi: 10.3390/su71114761.

Lattemann, S. and Höpner, T. (2008) 'Environmental impact and impact assessment of seawater desalination', *Desalination*. doi: 10.1016/j.desal.2007.03.009.

Laucelli, D. and Giustolisi, O. (2015) 'Vulnerability Assessment of Water Distribution Networks under Seismic Actions', *Journal of Water Resources Planning and Management*. doi: 10.1061/(asce)wr.1943-5452.0000478.

Ledford, A. W. and Tawn, J. A. (1996) 'Statistics for near independence in multivariate extreme values', *Biometrika*. doi: 10.1093/biomet/83.1.169.

Lee, K. and Jepson, W. (2020) 'Drivers and barriers to urban water reuse: A systematic review', *Water Security*. doi: 10.1016/j.wasec.2020.100073.

Lehner, B. *et al.* (2011) *Global Reservoir and Dam Database, Version 1 (GRanDv1): Reservoirs, Revision 01, NASA Socioeconomic Data and Applications Center (SEDAC)*.

Leonard, M. *et al.* (2014) 'A compound event framework for understanding extreme impacts', *Wiley Interdisciplinary Reviews: Climate Change*. doi: 10.1002/wcc.252.

Lester, S. (2015) 'Assessing urban household vulnerability at the climate-water-health nexus in Jamaica: A case study of the August Town community', *Caribbean Geography*.

Li, J. *et al.* (2019) 'Water supply risk analysis of Panjiakou reservoir in Luanhe River basin of China and drought impacts under environmental change', *Theoretical and Applied Climatology*. doi: 10.1007/s00704-018-2748-2.

Li, X., Sha, J. and Wang, Z. L. (2019) 'Comparison of daily streamflow forecasts using extreme learning machines and the random forest method', *Hydrological Sciences Journal*. doi: 10.1080/02626667.2019.1680846.

Libey, A., Adank, M. and Thomas, E. (2020) 'Who pays for water? Comparing life cycle costs of water services among several low, medium and high-income utilities', *World Development*.

doi: 10.1016/j.worlddev.2020.105155.

Libonati, R. *et al.* (2022) 'Assessing the role of compound drought and heatwave events on unprecedented 2020 wildfires in the Pantanal', *Environmental Research Letters*. doi: 10.1088/1748-9326/ac462e.

Lin, N. *et al.* (2012) 'Physically based assessment of hurricane surge threat under climate change', *Nature Climate Change*. doi: 10.1038/nclimate1389.

Lineman, M. *et al.* (2015) 'Talking about climate change and global warming', *PLoS ONE*. doi: 10.1371/journal.pone.0138996.

Lloyd, C. T. *et al.* (2019) 'Global spatio-temporally harmonised datasets for producing high-resolution gridded population distribution datasets', *Big Earth Data*. doi: 10.1080/20964471.2019.1625151.

Van Loon, A. F. *et al.* (2016) 'Drought in the Anthropocene', *Nature Geoscience*. doi: 10.1038/ngeo2646.

Luo, T. *et al.* (2023) 'A framework to assess multi-hazard physical climate risk for power generation projects from publicly-accessible sources', *Nature Communications Earth & Environment*.

Madani, K. and Dinar, A. (2012) 'Cooperative institutions for sustainable common pool resource management: Application to groundwater', *Water Resources Research*. doi: 10.1029/2011WR010849.

Majuru, B., Suhrcke, M. and Hunter, P. R. (2016) 'How do households respond to unreliable water supplies? a systematic review', *International Journal of Environmental Research and Public Health*. doi: 10.3390/ijerph13121222.

Manabe, S., Smagorinsky, J. and Strickler, R. F. (1965) 'SIMULATED CLIMATOLOGY OF A GENERAL CIRCULATION MODEL WITH A HYDROLOGIC CYCLE 1', *Monthly Weather Review*. doi: 10.1175/1520-0493(1965)093<0769:scoagc>2.3.co;2.

Manabe, S. and Wetherald, R. T. (1967) 'Thermal Equilibrium of the Atmosphere with a Given Distribution of Relative Humidity', *Journal of the Atmospheric Sciences*. doi: 10.1175/1520-0469(1967)024<0241:teotaw>2.0.co;2.

Mandal, A. *et al.* (2011) 'Assessment of the groundwater quality of the wells of the Kingston

Basin, Kingston, Jamaica', in *Advances in Geosciences: Volume 23: Hydrological Science (HS)*. doi: 10.1142/9789814355339_0023.

Mandal, A. *et al.* (2016) 'Rainfall-runoff simulations using the CARIWIG Simple Model for Advection of Storms and Hurricanes and HEC-HMS: Implications of Hurricane Ivan over the Jamaica Hope River watershed', *Natural Hazards*. doi: 10.1007/s11069-016-2380-3.

Mandal, A. and Maharaj, A. (2013) 'Flooding in Jamaica with assessment of riverine inundation of Port Maria, St Mary', *Bulletin de la Societe Geologique de France*. doi: 10.2113/gssgfbull.184.1-2.165.

Manfreda, S., Di Leo, M. and Sole, A. (2011) 'Detection of Flood-Prone Areas Using Digital Elevation Models', *Journal of Hydrologic Engineering*. doi: 10.1061/(asce)he.1943-5584.0000367.

Maraun, D. (2016) 'Bias Correcting Climate Change Simulations - a Critical Review', *Current Climate Change Reports*, (2), pp. 211–220.

Marston, L. and Cai, X. (2016) 'An overview of water reallocation and the barriers to its implementation', *Wiley Interdisciplinary Reviews: Water*. doi: 10.1002/wat2.1159.

Martínez-Castro, D. *et al.* (2018) 'The performance of RegCM4 over the Central America and Caribbean region using different cumulus parameterizations', *Climate Dynamics*. doi: 10.1007/s00382-017-3863-y.

Martínez-Valderrama, J. *et al.* (2023) 'Complex Policy Mixes are Needed to Cope with Agricultural Water Demands Under Climate Change', *Water Resources Management*. doi: 10.1007/s11269-023-03481-5.

Matrosov, E. S., Padula, S. and Harou, J. J. (2013) 'Selecting Portfolios of Water Supply and Demand Management Strategies Under Uncertainty-Contrasting Economic Optimisation and "Robust Decision Making" Approaches', *Water Resources Management*. doi: 10.1007/s11269-012-0118-x.

Matrosov, E. S., Woods, A. M. and Harou, J. J. (2013) 'Robust Decision Making and Info-Gap Decision Theory for water resource system planning', *Journal of Hydrology*. doi: 10.1016/j.jhydrol.2013.03.006.

McDonald, R. I. *et al.* (2011) 'Urban growth, climate change, and freshwater availability',

Proceedings of the National Academy of Sciences of the United States of America. doi: 10.1073/pnas.1011615108.

McDonald, R. I. *et al.* (2014) 'Water on an urban planet: Urbanization and the reach of urban water infrastructure', *Global Environmental Change*. doi: 10.1016/j.gloenvcha.2014.04.022.

McIntosh, A. C. (2003) *Asian Water Supplies, Asian Development Bank and International Water Association*.

Medina, D., Kullmann, C. and Felter, G. (2021) *Revisiting Resilience in the Caribbean - Water Supply and Sanitation, Revisiting Resilience in the Caribbean - Water Supply and Sanitation*. doi: 10.1596/36409.

MEGJC (2019) *National Water Sector Policy and Implementation Plan*. Available at: https://megjc.gov.jm/docs/policies/national_water_sector_policy_2019.pdf.

Meinshausen, M. *et al.* (2020) 'The shared socio-economic pathway (SSP) greenhouse gas concentrations and their extensions to 2500', *Geoscientific Model Development*. doi: 10.5194/gmd-13-3571-2020.

Mekonnen, M. M. and Hoekstra, A. Y. (2016) 'Sustainability: Four billion people facing severe water scarcity', *Science Advances*. doi: 10.1126/sciadv.1500323.

Mens, M. J. P., Gilroy, K. and Williams, D. (2015) 'Developing system robustness analysis for drought risk management: An application on a water supply reservoir', *Natural Hazards and Earth System Sciences*. doi: 10.5194/nhess-15-1933-2015.

Messager, M. L. *et al.* (2016) 'Estimating the volume and age of water stored in global lakes using a geo-statistical approach', *Nature Communications*. doi: 10.1038/ncomms13603.

Meza, I. *et al.* (2020) 'Global-scale drought risk assessment for agricultural systems', *Natural Hazards and Earth System Sciences*. doi: 10.5194/nhess-20-695-2020.

Miller, N., Waite, L. and Harlan, A. (2001) *Water resources assessment of Jamaica*. Washington, DC.

Milly, P. C. D. *et al.* (2002) 'Increasing risk of great floods in a changing climate', *Nature*. doi: 10.1038/415514a.

Milly, P. C. D. *et al.* (2008) 'Climate change: Stationarity is dead: Whither water management?', *Science*. doi: 10.1126/science.1151915.

Milly, P. C. D., Dunne, K. A. and Vecchia, A. V. (2005) 'Global pattern of trends in streamflow and water availability in a changing climate', *Nature*. doi: 10.1038/nature04312.

Mini, C., Hogue, T. S. and Pincetl, S. (2015) 'The effectiveness of water conservation measures on summer residential water use in Los Angeles, California', *Resources, Conservation and Recycling*. doi: 10.1016/j.resconrec.2014.10.005.

Mirza, M. M. Q. (2003) 'Climate change and extreme weather events: Can developing countries adapt?', *Climate Policy*. doi: 10.3763/cpol.2003.0330.

Mitlin, D. and Walnycki, A. (2020) 'Informality as Experimentation: Water Utilities' Strategies for Cost Recovery and their Consequences for Universal Access', *Journal of Development Studies*. doi: 10.1080/00220388.2019.1577383.

Miyamoto (2019) *Overview of Engineering Options for Increasing Infrastructure Resilience, Report: Supporting Material and Reference Information*.

Moftakhari, H. and Aghakouchak, A. (2019) 'Increasing exposure of energy infrastructure to compound hazards: Cascading wildfires and extreme rainfall', *Environmental Research Letters*. doi: 10.1088/1748-9326/ab41a6.

Moftakhari, H. R. *et al.* (2017) 'Compounding effects of sea level rise and fluvial flooding', *Proceedings of the National Academy of Sciences of the United States of America*. doi: 10.1073/pnas.1620325114.

Molinari, D. *et al.* (2021) 'Cost–benefit analysis of flood mitigation measures: a case study employing high-performance hydraulic and damage modelling', *Natural Hazards*. doi: 10.1007/s11069-021-04814-6.

Molle, F. and Berkoff, J. (2009) 'Cities vs. agriculture: A review of intersectoral water re-allocation', *Natural Resources Forum*. doi: 10.1111/j.1477-8947.2009.01204.x.

Del Moral Ituarte, L. and Giansante, C. (2000) 'Constraints to drought contingency planning in Spain: The hydraulic paradigm and the case of Seville', *Journal of Contingencies and Crisis Management*. doi: 10.1111/1468-5973.00128.

Morris, R. L. *et al.* (2018) 'From grey to green: Efficacy of eco-engineering solutions for nature-based coastal defence', *Global Change Biology*. doi: 10.1111/gcb.14063.

Mortazavi-Naeini, M. *et al.* (2019) 'Assessment of Risks to Public Water Supply From Low

Flows and Harmful Water Quality in a Changing Climate', *Water Resources Research*. doi: 10.1029/2018WR022865.

Mortazavi-Naeini, M., Kuczera, G. and Cui, L. (2014) 'Application of multiobjective optimization to scheduling capacity expansion of urban water resource systems', *Water Resources Research*. doi: 10.1002/2013WR014569.

Mortazavi, M., Kuczera, G. and Cui, L. (2012) 'Multiobjective optimization of urban water resources: Moving toward more practical solutions', *Water Resources Research*. doi: 10.1029/2011WR010866.

Mott MacDonald (2018) *Appendix 4.1.A – Mott MacDonald flood risk assessment - Wessex Water*.

Mott MacDonald (2022) *Safeguarding England's water future*. Available at: https://www.mottmac.com/article/72981/strategic-water-resource-options-planning.

Mukherjee, S., Mishra, A. and Trenberth, K. E. (2018) 'Climate Change and Drought: a Perspective on Drought Indices', *Current Climate Change Reports*. doi: 10.1007/s40641-018-0098-x.

Muller, M. (2017) 'Understanding the origins of Cape Town's water crisis', *Civil Engineering*.

Muller, M. (2018) 'Cape Town's drought: Don't blame climate change', *Nature*. doi: 10.1038/d41586-018-05649-1.

Murgatroyd, A. *et al.* (2022) 'Strategic analysis of the drought resilience of water supply systems', *Philosophical Transactions of the Royal Society A: Mathematical, Physical and Engineering Sciences*. doi: 10.1098/rsta.2021.0292.

Nagpal, T., Malik, A. and Eldridg, M. (2018) *Mobilizing Additional Funds for Pro-Poor Water Services*. Available at: https://www.globalwaters.org/sites/default/files/mobilizing_additional_funds_for_pro-poor_water_services_3.pdf.

Nandi, A. *et al.* (2016) 'Flood hazard mapping in Jamaica using principal component analysis and logistic regression', *Environmental Earth Sciences*. doi: 10.1007/s12665-016-5323-0.

National Spatial Data Management Division (2001) *Licensed Sources*. Available at: https://jis.gov.jm/government/agencies/national-spatial-data-management-division/.

National Spatial Data Management Division (2021) ‘River and catchment outlines’.

National Spatial Management Branch (2010) *Potable water facilities*. Available at: https://jis.gov.jm/government/agencies/national-spatial-data-management-division/.

National Water Commission (2011) *Draft WS PPLan – October 12 2011*. Available at: https://www.nwcjamaica.com/uploads/Advertisements/PPLan - PDF/Draft. St. Elizabeth WS PPLan - October 12 2011.pdf.

National Water Commission (2020a) *About*. Available at: https://www.nwcjamaica.com/about.php.

National Water Commission (2020b) *Disruptions & Advisories*. Available at: https://www.nwcjamaica.com/Advisories.

National Water Commission (2020c) *NWC DROUGHT AFFECTED SYSTEMS*. Available at: https://www.nwcjamaica.com/articles/4-20190522-NWC-Drought-Affected-Systems-May-22-2019.pdf.

NGPRP (1974) *Instream needs sub-group report*. Helena, Mont., USA.

Nijssen, B. *et al.* (2001) ‘Hydrologic sensitivity of global rivers to climate change’, *Climatic Change*. doi: 10.1023/A:1010616428763.

NWC (2015) *Capital Improvement Plan*.

NWC (2019) *AUDITED FINANCIAL STATEMENTS AS Of MARCH 31*. Available at: https://www.nwcjamaica.com/financial_statement.php (Accessed: 6 April 2022).

NWC (2021a) ‘FACILITIES LOGGED ON FDIMS AFFECTED BY TURBIDITY AND DROUGHT CONDITIONS’.

NWC (2021b) ‘Reservoir volume variations’.

Nyman, P. *et al.* (2021) ‘Probability and Consequence of Postfire Erosion for Treatability of Water in an Unfiltered Supply System’, *Water Resources Research*. doi: 10.1029/2019wr026185.

O’Brien, C. (2023) *France Prepares For Water Restrictions As 2023 Drought Continues*, *Forbes*. Available at: https://www.forbes.com/sites/chrisobrien/2023/02/28/france-prepares-for-water-restrictions-as-2023-drought-continues/?sh=7fbe36035c0f (Accessed: 25 June

2023).

O'Neill, B. C. *et al.* (2016) 'The Scenario Model Intercomparison Project (ScenarioMIP) for CMIP6', *Geoscientific Model Development*. doi: 10.5194/gmd-9-3461-2016.

Oates, N., Conway, D. and Calow, R. (2011) 'The "mainstreaming" approach to climate change adaptation: insights from Ethiopia's water sector', *ODI Background Notes*.

Ochungo, E. A. *et al.* (2019) 'The Implication of Unreliable Urban Water Supply Service: The Case of Vendor Water Cost in Langata Sub County, Nairobi City, Kenya', *Journal of Water Resource and Protection*. doi: 10.4236/jwarp.2019.117055.

OECD (2011) 'Private sector participation in water infrastructure – checklist for public action', in. doi: 10.1787/9789264120525-13-en.

OECD (2022) *Financing a Water Secure Future*. Available at: https://www.oecd-ilibrary.org/environment/financing-a-water-secure-future_a2ecb261-en.

Olschewski, P. *et al.* (2023) 'An ensemble-based assessment of bias adjustment performance, changes in hydrometeorological predictors and compound extreme events in EAS-CORDEX', *Weather and Climate Extremes*. doi: 10.1016/j.wace.2022.100531.

OpenStreetMap (2018) *OpenStreetMap*.

Otaki, Y. *et al.* (2022) 'Demand-side water management using alternative water sources based on residential end-use', *Water Practice and Technology*. doi: 10.2166/wpt.2022.031.

Oughton, E. J. and Russell, T. (2020) 'The importance of spatio-temporal infrastructure assessment:Evidence for 5G from the Oxford–Cambridge Arc', *Computers, Environment and Urban Systems*. doi: 10.1016/j.compenvurbsys.2020.101515.

Padowski, J. C. and Gorelick, S. M. (2014) 'Global analysis of urban surface water supply vulnerability', *Environmental Research Letters*. doi: 10.1088/1748-9326/9/10/104004.

Paltan, H. *et al.* (2017) 'Global Floods and Water Availability Driven by Atmospheric Rivers', *Geophysical Research Letters*. doi: 10.1002/2017GL074882.

Paltan, H. *et al.* (2018) 'Global implications of 1.5 °c and 2 °c warmer worlds on extreme river flows', *Environmental Research Letters*. doi: 10.1088/1748-9326/aad985.

Paltán, H. A. *et al.* (2021) 'Increased water risks to global hydropower in 1.5 °C and 2.0 °C

Warmer Worlds', *Journal of Hydrology*. doi: 10.1016/j.jhydrol.2021.126503.

Panagopoulos, A. (2021) 'Water-energy nexus: desalination technologies and renewable energy sources', *Environmental Science and Pollution Research*. doi: 10.1007/s11356-021-13332-8.

Pant, R. *et al.* (2018) 'Critical infrastructure impact assessment due to flood exposure', in *Journal of Flood Risk Management*. doi: 10.1111/jfr3.12288.

Panteli, M. *et al.* (2017) 'Power System Resilience to Extreme Weather: Fragility Modeling, Probabilistic Impact Assessment, and Adaptation Measures', *IEEE Transactions on Power Systems*. doi: 10.1109/TPWRS.2016.2641463.

Paprotny, D. and Morales-Nápoles, O. (2017) 'Estimating extreme river discharges in Europe through a Bayesian network', *Hydrology and Earth System Sciences*. doi: 10.5194/hess-21-2615-2017.

Pastor, A. V. *et al.* (2014) 'Accounting for environmental flow requirements in global water assessments', *Hydrology and Earth System Sciences*. doi: 10.5194/hess-18-5041-2014.

Pattanayak, S. K. *et al.* (2005) 'Coping with unreliable public water supplies: Averting expenditures by households in Kathmandu, Nepal', *Water Resources Research*. doi: 10.1029/2003WR002443.

Peduzzi, P. *et al.* (2009) 'Assessing global exposure and vulnerability towards natural hazards: The Disaster Risk Index', *Natural Hazards and Earth System Science*. doi: 10.5194/nhess-9-1149-2009.

Peng, J. *et al.* (2020) 'A pan-African high-resolution drought index dataset', *Earth System Science Data*. doi: 10.5194/essd-12-753-2020.

Pinto, F. S., de Carvalho, B. and Marques, R. C. (2021) 'Adapting water tariffs to climate change: Linking resource availability, costs, demand, and tariff design flexibility', *Journal of Cleaner Production*. doi: 10.1016/j.jclepro.2021.125803.

Planning Institute of Jamaica (2018) *Jamaica voluntary national review on the implementation of the 2030 Agenda for Sustainable Development June 2018*.

Poff, N. L. *et al.* (2016) 'Sustainable water management under future uncertainty with eco-engineering decision scaling', *Nature Climate Change*. doi: 10.1038/nclimate2765.

Pokhrel, Y. *et al.* (2021) 'Global terrestrial water storage and drought severity under climate change', *Nature Climate Change*. doi: 10.1038/s41558-020-00972-w.

Porter, K. (2021) 'A Beginner's Guide to Fragility, Vulnerability, and Risk', in *Encyclopedia of Earthquake Engineering*. doi: 10.1007/978-3-642-36197-5_256-1.

Purwar, D., Sliuzas, R. and Flacke, J. (2020) 'Assessment of cascading effects of typhoons on water and sanitation services: A case study of informal settlements in Malabon, Philippines', *International Journal of Disaster Risk Reduction*. doi: 10.1016/j.ijdrr.2020.101755.

Qiu, J. *et al.* (2022) 'Quantitative Stress Test of Compound Coastal-Fluvial Floods in China's Pearl River Delta', *Earth's Future*. doi: 10.1029/2021EF002638.

Quinn, N. *et al.* (2019) 'The Spatial Dependence of Flood Hazard and Risk in the United States', *Water Resources Research*. doi: 10.1029/2018WR024205.

Ranger and Mullan (2022) *Mobilizing Finance for Adaptation at COP27: Next Steps in Aligning Finance and Investment with Climate-Resilient Development Goals*. Available at: https://gca.org/mobilizing-finance-for-adaptation-at-cop27-next-steps-in-aligning-finance-and-investment-with-climate-resilient-development-goals/.

Ray, B. and Shaw, R. (2019) 'Water Insecurity in Asian Cities', in. doi: 10.1007/978-981-10-8947-3_2.

Reed, P. M. *et al.* (2013) 'Evolutionary multiobjective optimization in water resources: The past, present, and future', *Advances in Water Resources*. doi: 10.1016/j.advwatres.2012.01.005.

Renard, B. and Lang, M. (2007) 'Use of a Gaussian copula for multivariate extreme value analysis: Some case studies in hydrology', *Advances in Water Resources*. doi: 10.1016/j.advwatres.2006.08.001.

Riahi, K. *et al.* (2017) 'The Shared Socioeconomic Pathways and their energy, land use, and greenhouse gas emissions implications: An overview', *Global Environmental Change*. doi: 10.1016/j.gloenvcha.2016.05.009.

Richters *et al.* (2022) 'NGFS Climate Scenarios Data Set', *Zenodo*. doi: https://zenodo.org/record/7198430.

Robak, A. and Bjornlund, H. (2018) 'Poor water service quality in developed countries may

have a greater impact on lower-income households', *Water International*. doi: 10.1080/02508060.2018.1446613.

Robinson, M. and Ward, R. (2017) 'Hydrology: Principles and Processes', *Water Intelligence Online*. doi: 10.2166/9781780407296.

Rodriguez, D. *et al.* (2014) *Applying Results-Based Financing in Water Investments*, *Applying Results-Based Financing in Water Investments*. doi: 10.1596/20041.

Roibas, D., Garcia-Valiñas, M. A. and Fernandez-Llera, R. (2019) 'Measuring the Impact of Water Supply Interruptions on Household Welfare', *Environmental and Resource Economics*. doi: 10.1007/s10640-018-0255-7.

Romero, S. (2015) *Taps Start to Run Dry in Brazil's Largest City*, *New York Times*. Available at: https://www.nytimes.com/2015/02/17/world/americas/drought-pushes-sao-paulo-brazil-toward-water-crisis.html (Accessed: 25 June 2023).

Rothausen, S. G. S. A. and Conway, D. (2011) 'Greenhouse-gas emissions from energy use in the water sector', *Nature Climate Change*. doi: 10.1038/nclimate1147.

Rouse, M. (2013) 'Institutional Governance and Regulation of Water Services: The Essential Elements - Second Edition', *Water Intelligence Online*. doi: 10.2166/9781780404516.

Rouse, M. (2014) 'The worldwide urban water and wastewater infrastructure challenge', *International Journal of Water Resources Development*. doi: 10.1080/07900627.2014.882203.

Rouse, M. and El Achi, N. (2019) 'A road map to sustainable urban water supply', in *Water Science, Policy, and Management: A Global Challenge*, pp. 309–328.

Rozenberg, J. and Fay, M. (2019) *Beyond the Gap: How Countries Can Afford the Infrastructure They Need while Protecting the Planet*, *Beyond the Gap: How Countries Can Afford the Infrastructure They Need while Protecting the Planet*. doi: 10.1596/978-1-4648-1363-4.

Saha, S. *et al.* (2010) 'The NCEP climate forecast system reanalysis', *Bulletin of the American Meteorological Society*. doi: 10.1175/2010BAMS3001.1.

Salehi, M. (2022) 'Global water shortage and potable water safety; Today's concern and tomorrow's crisis', *Environment International*. doi: 10.1016/j.envint.2021.106936.

Saurí, D. and Palau-Rof, L. (2017) 'Urban drainage in barcelona: From hazard to resource?',

Water Alternatives.

Seddon, N. *et al.* (2020) 'Understanding the value and limits of nature-based solutions to climate change and other global challenges', *Philosophical Transactions of the Royal Society B: Biological Sciences*. doi: 10.1098/rstb.2019.0120.

See, K. L., Nayan, N. and Rahaman, Z. A. (2017) 'Flood Disaster Water Supply: A Review of Issues and Challenges in Malaysia', *International Journal of Academic Research in Business and Social Sciences*. doi: 10.6007/ijarbss/v7-i10/3406.

Selelo, L. R. *et al.* (2017) 'The effects of extended water supply disruptions on the operations of SMEs', *Southern African Business Review*.

Sellar, A. A. *et al.* (2019) 'UKESM1: Description and Evaluation of the U.K. Earth System Model', *Journal of Advances in Modeling Earth Systems*. doi: 10.1029/2019MS001739.

Sen, A. (1974) 'Choice, Orderings and Morality', *Social Choice and Welfare*.

Serinaldi, F. and Kilsby, C. G. (2017) 'A Blueprint for Full Collective Flood Risk Estimation: Demonstration for European River Flooding', *Risk Analysis*. doi: 10.1111/risa.12747.

Setegn, S. G. *et al.* (2014) 'Modeling hydrological variability of fresh water resources in the Rio Cobre watershed, Jamaica', *Catena*. doi: 10.1016/j.catena.2014.04.005.

Singh, C. *et al.* (2021) 'Losses and damages associated with slow-onset events: urban drought and water insecurity in Asia', *Current Opinion in Environmental Sustainability*. doi: 10.1016/j.cosust.2021.02.006.

Sjöstrand, K. *et al.* (2021) 'The value of water—estimating water-disruption impacts on businesses', *Water (Switzerland)*. doi: 10.3390/w13111565.

Sklar, A. (1959) '{F}onctions de {R}eparition a n {D}imensions et {L}eurs {M}arges', *Publications de l'Institute de Statistique de l'Universite de Paris*.

Sky News (2020) *Storm Dennis: Monmouth stockpiles bottled water after floods hit treatment plant*. Available at: https://news.sky.com/story/storm-dennis-drinking-water-supplies-hit-as-flooding-shuts-down-monmouth-water-treatment-plant-11937225 (Accessed: 10 July 2023).

Smith, J. B. *et al.* (2011) 'Development and climate change adaptation funding: Coordination and integration', *Climate Policy*. doi: 10.1080/14693062.2011.582385.

Sohail, M. *et al.* (2007) ‘Pro-poor concessions for sustainable water services’, *Proceedings of the Institution of Civil Engineers: Civil Engineering*. doi: 10.1680/cien.2007.160.2.56.

Speakman Cordall, S. (2023) *Water ban in drought-stricken Tunisia adds to growing crisis*, *Guardian*. Available at: https://www.theguardian.com/global-development/2023/apr/05/water-ban-in-drought-stricken-tunisia-adds-to-growing-crisis (Accessed: 25 June 2023).

Speight, L. J., Hall, J. W. and Kilsby, C. G. (2017) ‘A multi-scale framework for flood risk analysis at spatially distributed locations’, *Journal of Flood Risk Management*. doi: 10.1111/jfr3.12175.

Spinoni, J. *et al.* (2019) ‘A new global database of meteorological drought events from 1951 to 2016’, *Journal of Hydrology: Regional Studies*. doi: 10.1016/j.ejrh.2019.100593.

Srinivasan, V. *et al.* (2013) ‘The impact of urbanization on water vulnerability: A coupled human-environment system approach for Chennai, India’, *Global Environmental Change*. doi: 10.1016/j.gloenvcha.2012.10.002.

Stagge, J. H. *et al.* (2015) ‘Modeling drought impact occurrence based on meteorological drought indices in Europe’, *Journal of Hydrology*. doi: 10.1016/j.jhydrol.2015.09.039.

Stavenhagen, M., Buurman, J. and Tortajada, C. (2018) ‘Saving water in cities: Assessing policies for residential water demand management in four cities in Europe’, *Cities*. doi: 10.1016/j.cities.2018.03.008.

Steinschneider, S. and Brown, C. (2013) ‘A semiparametric multivariate, multisite weather generator with low-frequency variability for use in climate risk assessments’, *Water Resources Research*. doi: 10.1002/wrcr.20528.

Stillwell, A. S., King, C. W. and Webber, M. E. (2010) ‘Desalination and Long-Haul Water Transfer as a Water Supply for Dallas, Texas: A Case Study of the Energy-Water Nexus in Texas’, *Texas Water Journal*.

Stødle, K. *et al.* (2021) ‘Dependent infrastructure system modeling: A case study of the St. Kitts power and water distribution systems’, *Reliability Engineering and System Safety*. doi: 10.1016/j.ress.2020.107421.

Summaryplanet (2023) *Water distribution systems*. Available at:

https://www.summaryplanet.com/engineering/Water-distribution-systems.html (Accessed: 7 July 2023).

Takahashi, B., Tandoc, E. C. and Carmichael, C. (2015) 'Communicating on Twitter during a disaster: An analysis of tweets during Typhoon Haiyan in the Philippines', *Computers in Human Behavior*. doi: 10.1016/j.chb.2015.04.020.

Taner, M. Ü., Ray, P. and Brown, C. (2019) 'Incorporating Multidimensional Probabilistic Information Into Robustness-Based Water Systems Planning', *Water Resources Research*. doi: 10.1029/2018WR022909.

Thacker, S. *et al.* (2017) 'Geographic Hotspots of Critical National Infrastructure', *Risk Analysis*. doi: 10.1111/risa.12840.

Thacker, S. *et al.* (2018) 'Evaluating the Benefits of Adaptation of Critical Infrastructures to Hydrometeorological Risks', *Risk Analysis*. doi: 10.1111/risa.12839.

Thacker, S., Pant, R. and Hall, J. W. (2017) 'System-of-systems formulation and disruption analysis for multi-scale critical national infrastructures', *Reliability Engineering and System Safety*. doi: 10.1016/j.ress.2017.04.023.

Thames Water (2013) *Water Resources Management Plan 2015–2040*. Reading, UK.

Timonina, A. *et al.* (2015) 'Structured Coupling of Probability Loss Distributions: Assessing Joint Flood Risk in Multiple River Basins', *Risk analysis : an official publication of the Society for Risk Analysis*. doi: 10.1111/risa.12382.

Titley, H. A. *et al.* (2021) 'Key factors influencing the severity of fluvial flood hazard from tropical cyclones', *Journal of Hydrometeorology*. doi: 10.1175/JHM-D-20-0250.1.

Totsuka, N., Trifunovic, N. and Vairavamoorthy, K. (2004) 'Intermittent urban water supply under water starving situations', in *People-Centred Approaches to Water and Environmental Sanitation: Proceedings of the 30th WEDC Conference*.

Towe, R., Tawn, J. and Lamb, R. (2018) 'Why extreme floods are more common than you might think', *Significance*. doi: 10.1111/j.1740-9713.2018.01209.x.

Tremlett, G. (2005) *Spanish forced to ration water*, *Guardian*. Available at: https://www.theguardian.com/world/2005/jun/06/spain.gilestremlett (Accessed: 25 June 2023).

Trenberth, K. E., Fasullo, J. T. and Shepherd, T. G. (2015) 'Attribution of climate extreme events', *Nature Climate Change*. doi: 10.1038/nclimate2657.

Trigg, M. A. *et al.* (2016) 'The credibility challenge for global fluvial flood risk analysis', *Environmental Research Letters*. doi: 10.1088/1748-9326/11/9/094014.

Trigo, R. M., Osborn, T. J. and Corte-Real, J. M. (2002) 'The North Atlantic Oscillation influence on Europe: Climate impacts and associated physical mechanisms', *Climate Research*. doi: 10.3354/cr020009.

Tsakiris, G. *et al.* (2013) 'A System-based Paradigm of Drought Analysis for Operational Management', *Water Resources Management*. doi: 10.1007/s11269-013-0471-4.

Tsui, K. (2022) *Drought in Mexico leads to water rationing, theft, Washington Post*. Available at: https://www.washingtonpost.com/world/2022/08/03/mexico-drought-water-shortages-theft-rationing/ (Accessed: 25 June 2023).

Turkelboom, F. *et al.* (2021) 'How does a nature-based solution for flood control compare to a technical solution? Case study evidence from Belgium', *Ambio*. doi: 10.1007/s13280-021-01548-4.

Turner, S. W. D. *et al.* (2016) 'Risk-based water resources planning in England and Wales: challenges in execution and implementation', *Urban Water Journal*. doi: 10.1080/1573062x.2014.955856.

UN (2021) *The sustainable development goals report 2019, United Nations publication issued by the Department of Economic and Social Affairs*.

UNDRR (2022) *Global Assessment Report On Disaster Risk Reduction (GAR)*. Available at: https://www.undrr.org/gar/gar2022-our-world-risk-gar.

UNICEF, W. (2019) *Progress on household drinking water, sanitation and hygiene 2000-2017. Special focus on inequalities*. New York.

UNICEF and WHO (2020) *WHO/UNICEF Joint Monitoring Programme (JMP)*. Available at: https://washdata.org/data.

UNISDR (2009) *2009 UNISDR Terminology on Disaster Risk Reduction*. Geneva, Switzerland. Available at: https://www.un.org/en/development/desa/policy/untaskteam_undf/thinkpieces/3_disaster_risk

_resilience.pdf.

Várallyay, G. (2010) ‘The impact of climate change on soils and on their water management’, *Agronomy Research*.

Verschuur *et al.* (2023) ‘Future domestic wastewater production projections for the 21st century’, *(in prep)*.

Verschuur, J. *et al.* (2020) ‘Prioritising resilience policies to reduce welfare losses from natural disasters: A case study for coastal Bangladesh’, *Global Environmental Change*. doi: 10.1016/j.gloenvcha.2020.102179.

Verschuur, J., Koks, E. E., *et al.* (2023) ‘Multi-hazard risk to global port infrastructure and resulting trade and logistics losses’, *Communications Earth and Environment*. doi: 10.1038/s43247-022-00656-7.

Verschuur, J., Becher, O. R. E., *et al.* (2023) *Welfare and Climate Risks in Coastal Bangladesh : The Impacts of Climatic Extremes on Multidimensional Poverty and the Wider Benefits of Climate Adaptation*. Washington DC, USA. Available at: https://documents.worldbank.org/en/publication/documents-reports/documentdetail/099317103212330683/idu00c1858fc0d675043970a2740e6b087827d9f.

Verschuur, J., Koks, E. E. and Hall, J. W. (2020) ‘Port disruptions due to natural disasters: Insights into port and logistics resilience’, *Transportation Research Part D: Transport and Environment*. doi: 10.1016/j.trd.2020.102393.

Vicente-Serrano, S. M. *et al.* (2022) ‘Global drought trends and future projections’, *Philosophical Transactions of the Royal Society A: Mathematical, Physical and Engineering Sciences*. doi: 10.1098/rsta.2021.0285.

Vickery, P. J., Skerlj, P. F. and Twisdale, L. A. (2000) ‘Simulation of Hurricane Risk in the U.S. Using Empirical Track Model’, *Journal of Structural Engineering*. doi: 10.1061/(asce)0733-9445(2000)126:10(1222).

van Vliet, M. T. H. *et al.* (2016) ‘Power-generation system vulnerability and adaptation to changes in climate and water resources’, *Nature Climate Change*. doi: 10.1038/nclimate2903.

van Vliet, M. T. H. *et al.* (2021) ‘Global water scarcity including surface water quality and

expansions of clean water technologies', *Environmental Research Letters*. doi: 10.1088/1748-9326/abbfc3.

Van Vliet, M. T. H., Florke, M. and Wada, Y. (2017) 'Quality matters for water scarcity', *Nature Geoscience*. doi: 10.1038/NGEO3047.

van Vliet, M. T. H. and Zwolsman, J. J. G. (2008) 'Impact of summer droughts on the water quality of the Meuse river', *Journal of Hydrology*. doi: 10.1016/j.jhydrol.2008.01.001.

Vogel, E. *et al.* (2019) 'The effects of climate extremes on global agricultural yields', *Environmental Research Letters*. doi: 10.1088/1748-9326/ab154b.

Voldoire, A. *et al.* (2013) 'The CNRM-CM5.1 global climate model: Description and basic evaluation', *Climate Dynamics*. doi: 10.1007/s00382-011-1259-y.

Volpi, E. (2019) 'On return period and probability of failure in hydrology', *Wiley Interdisciplinary Reviews: Water*. doi: 10.1002/WAT2.1340.

Vörösmarty, C. J. *et al.* (2015) 'What scale for water governance', *Science*, 349(6247), pp. 478–479.

van Vuuren, D. P. *et al.* (2011) 'The representative concentration pathways: An overview', *Climatic Change*. doi: 10.1007/s10584-011-0148-z.

Wada, Y., Gleeson, T. and Esnault, L. (2014) 'Wedge approach to water stress', *Nature Geoscience*. doi: 10.1038/ngeo2241.

Wada, Y., Wisser, D. and Bierkens, M. F. P. (2014) 'Global modeling of withdrawal, allocation and consumptive use of surface water and groundwater resources', *Earth System Dynamics*. doi: 10.5194/esd-5-15-2014.

Wahl, T. *et al.* (2015) 'Increasing risk of compound flooding from storm surge and rainfall for major US cities', *Nature Climate Change*. doi: 10.1038/nclimate2736.

Walsh, K. J. E. *et al.* (2016) 'Tropical cyclones and climate change', *Wiley Interdisciplinary Reviews: Climate Change*. doi: 10.1002/wcc.371.

Ward, P. J. *et al.* (2010) 'Partial costs of global climate change adaptation for the supply of raw industrial and municipal water: A methodology and application', *Environmental Research Letters*. doi: 10.1088/1748-9326/5/4/044011.

Ward, P. J. *et al.* (2013) 'Assessing flood risk at the global scale: Model setup, results, and sensitivity', *Environmental Research Letters*. doi: 10.1088/1748-9326/8/4/044019.

Ward, P. J. *et al.* (2014) 'Strong influence of El Niño Southern Oscillation on flood risk around the world', *Proceedings of the National Academy of Sciences of the United States of America*. doi: 10.1073/pnas.1409822111.

Ward, P. J. *et al.* (2017) 'A global framework for future costs and benefits of river-flood protection in urban areas', *Nature Climate Change*. doi: 10.1038/nclimate3350.

Warszawski, L. *et al.* (2014) 'The inter-sectoral impact model intercomparison project (ISI-MIP): Project framework', *Proceedings of the National Academy of Sciences of the United States of America*. doi: 10.1073/pnas.1312330110.

Watts, G. *et al.* (2012) 'Testing the resilience of water supply systems to long droughts', *Journal of Hydrology*. doi: 10.1016/j.jhydrol.2011.10.038.

Whateley, S., Steinschneider, S. and Brown, C. (2014) 'A climate change range-based method for estimating robustness for water resources supply', *Water Resources Research*. doi: 10.1002/2014WR015956.

Wheeler, K. G. *et al.* (2020) 'Understanding and managing new risks on the Nile with the Grand Ethiopian Renaissance Dam', *Nature Communications*. doi: 10.1038/s41467-020-19089-x.

Whitehead, P. G. *et al.* (2009) 'A review of the potential impacts of climate change on surface water quality', *Hydrological Sciences Journal*. doi: 10.1623/hysj.54.1.101.

Whitehead, P. G., Wilson, E. J. and Butterfield, D. (1998) 'A semi-distributed Integrated Nitrogen model for multiple source assessment in Catchments (INCA): Part I - Model structure and process equations', *Science of the Total Environment*. doi: 10.1016/S0048-9697(98)00037-0.

WHO (2014) *Preventing Diarrhoea through Better Water, Sanitation and Hygiene: Exposures and Impacts in Low- and Middle-income Countries, WHO, Water management Current and future challenges andresearch directions*.

Wiberg, D. and Strzepek, K. (2005) *Development of regional economic supply curves for surface water resources and climate change assessments: A case study of China*. Laxenburg.

Wilby, R. L. and Keenan, R. (2012) 'Adapting to flood risk under climate change', *Progress in Physical Geography*. doi: 10.1177/0309133312438908.

Winpenny, J. (2015) *Water: Fit to Finance? Catalyzing National Growth through Investment in Water Security, report of the High Level Panel on Financing Infrastructure for a Water-Secure World*.

Winsemius, H. C. *et al.* (2013) 'A framework for global river flood risk assessments', *Hydrology and Earth System Sciences*. doi: 10.5194/hess-17-1871-2013.

Winsemius, H. C. *et al.* (2018) 'Disaster risk, climate change, and poverty: Assessing the global exposure of poor people to floods and droughts', *Environment and Development Economics*. doi: 10.1017/S1355770X17000444.

World Bank (2012) *Living Standards Measurement Study (LSMS)*. Available at: https://microdata.worldbank.org/index.php/catalog/lsms/?page=1&ps=15&repo=lsms (Accessed: 3 May 2023).

World Bank (2019) *Nature-based Solutions: a Cost-effective Approach for Disaster Risk and Water Resource Management*. Available at: https://www.worldbank.org/en/topic/disasterriskmanagement/brief/nature-based-solutions-cost-effective-approach-for-disaster-risk-and-water-resource-management (Accessed: 4 February 2023).

World Bank (2021) *Population growth (annual %)*. Available at: https://data.worldbank.org/indicator/SP.POP.GROW.

World Bank (2023) *What the Future Has in Store: A New Paradigm for Water Storage*. Washington DC, USA. Available at: https://documents1.worldbank.org/curated/en/099454002022397507/pdf/IDU031e759b40be950485909796045bca5d8e378.pdf.

Wunderlich, S. *et al.* (2021) 'Optimizing household water decisions for managing intermittent water supply in Mexico City', *Environmental Science and Technology*. doi: 10.1021/acs.est.0c08390.

Wutich, A., Beresford, M. and Carvajal, C. (2016) 'Can Informal Water Vendors Deliver on the Promise of A Human Right to Water? Results From Cochabamba, Bolivia', *World Development*. doi: 10.1016/j.worlddev.2015.10.043.

Xiao, D. *et al.* (2019) ‘Streamflow Generation From Catchments of Contrasting Lithologies: The Role of Soil Properties, Topography, and Catchment Size’, *Water Resources Research*. doi: 10.1029/2018WR023736.

Yamazaki, D. *et al.* (2011) ‘A physically based description of floodplain inundation dynamics in a global river routing model’, *Water Resources Research*. doi: 10.1029/2010WR009726.

Yin, J. *et al.* (2023) ‘Future socio-ecosystem productivity threatened by compound drought–heatwave events’, *Nature Sustainability*. doi: 10.1038/s41893-022-01024-1.

Yoon, J. *et al.* (2021) ‘A coupled human-natural system analysis of freshwater security under climate and population change’, *Proceedings of the National Academy of Sciences of the United States of America*. doi: 10.1073/pnas.2020431118.

Yukimoto, S. *et al.* (2019) ‘The meteorological research institute Earth system model version 2.0, MRI-ESM2.0: Description and basic evaluation of the physical component’, *Journal of the Meteorological Society of Japan*. doi: 10.2151/jmsj.2019-051.

Yunzhu, Y. *et al.* (2022) ‘Open AccessArticle Resilience of Critical Infrastructure Systems to Floods: A Coupled Probabilistic Network Flow and LISFLOOD-FP Model’, *Water*, 14(5), p. 683.

Zhang, E. *et al.* (2018) ‘Bottom-up quantification of inter-basin water transfer vulnerability to climate change’, *Ecological Indicators*. doi: 10.1016/j.ecolind.2017.04.019.

Zhang, X. *et al.* (2019) ‘Urban drought challenge to 2030 sustainable development goals’, *Science of the Total Environment*. doi: 10.1016/j.scitotenv.2019.07.342.

Zhao, G. *et al.* (2018) ‘A modeling framework for evaluating the drought resilience of a surface water supply system under non-stationarity’, *Journal of Hydrology*, 563, pp. 22–32.

Ziervogel, G., Shale, M. and Du, M. (2010) ‘Climate change adaptation in a developing country context: The case of urban water supply in Cape Town’, *Climate and Development*. doi: 10.3763/cdev.2010.0036.

Zipper, S. C., Keune, J. and Kollet, S. J. (2019) ‘Land use change impacts on European heat and drought: Remote land-atmosphere feedbacks mitigated locally by shallow groundwater’, *Environmental Research Letters*. doi: 10.1088/1748-9326/ab0db3.

Zorn, C. *et al.* (2020) ‘Evaluating the magnitude and spatial extent of disruptions across

interdependent national infrastructure networks', *ASCE-ASME Journal of Risk and Uncertainty in Engineering Systems, Part B: Mechanical Engineering*. doi: 10.1115/1.4046327.

Zozmann, H. *et al.* (2022) 'Can Tanker Water Services Contribute to Sustainable Access to Water? A Systematic Review of Case Studies in Urban Areas', *Sustainability (Switzerland)*. doi: 10.3390/su141711029.

Zscheischler, J. *et al.* (2018) 'Future climate risk from compound events', *Nature Climate Change*. doi: 10.1038/s41558-018-0156-3.

Zscheischler, J. and Seneviratne, S. I. (2017) 'Dependence of drivers affects risks associated with compound events', *Science Advances*. doi: 10.1126/sciadv.1700263.

10. Appendices

Appendix A: Co-author paper contribution statements 200

Appendix B: Supplement to chapter 4 209

Supplementary Text B-1: Spatial synthetic flood events 209

Supplementary Text B2: Infrastructure criticality 215

Supplementary Text B3: Uncertainty analysis 218

Appendix C: Supplement to chapter 5 224

Supplementary Text C1: Overview of multi-hazard risk analysis (Becher *et al.*, 2023)... 224

Supplementary Text C2: Water balance (Becher *et al.*, 2023). 229

Appendix D: Supplement to chapter 6 232

Supplementary Text D1. Community Water Model. 237

Appendix A: Co-author paper contribution statements

To the Director of Graduate Studies,
School of Geography and the Environment
Environmental Change Institute
South Parks Road, Oxford OX1 3QY

I, Jasper Verschuur, certify that Olivia Becher completed the majority of the work in each of the chapters that form part of his DPhil thesis, as well as any associated articles that have already been published or may be published in the near future.

Signature:



Jasper Verschuur

Date: 21/07/2023

School of Geography and the Environment
Environmental Change Institute
South Parks Road, Oxford OX1 3QY

To the Director of Graduate Studies,
School of Geography and the Environment
Environmental Change Institute
South Parks Road, Oxford OX1 3QY

I, Dr Arpita Mandal, certify that Olivia Becher completed the majority of the work in each of the chapters that form part of his DPhil thesis, as well as any associated articles that have already been published or may be published in the near future.

Signature:

Arpita Mandal

Arpita Mandal

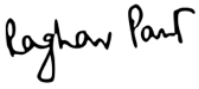
Date: 21/07/2023

Department of Geography and Geology
The University of The West Indies, Mona
Kingston, Jamaica

To the Director of Graduate Studies,
School of Geography and the Environment
Environmental Change Institute
South Parks Road, Oxford OX1 3QY

I, Dr. Raghav Pant, certify that Olivia Becher completed the majority of the work in each of the chapters that form part of his DPhil thesis, as well as any associated articles that have already been published or may be published in the near future.

Signature:



Dr. Raghav Pant

Date: 21/07/2023

School of Geography and the Environment
Environmental Change Institute
South Parks Road, Oxford OX1 3QY

To the Director of Graduate Studies,
School of Geography and the Environment
Environmental Change Institute
South Parks Road, Oxford OX1 3QY

I, Prof Jim Hall, certify that Olivia Becher completed the majority of the work in each of the chapters that form part of his DPhil thesis, as well as any associated articles that have already been published or may be published in the near future.

Signature:



Prof Jim Hall

Date: 24/07/2023

School of Geography and the Environment
Environmental Change Institute
South Parks Road, Oxford OX1 3QY

To the Director of Graduate Studies,
School of Geography and the Environment
Environmental Change Institute
South Parks Road, Oxford OX1 3QY

I, Mark Lawless, certify that Olivia Becher completed the majority of the work in each of the chapters that form part of his DPhil thesis, as well as any associated articles that have already been published or may be published in the near future.

Signature:



Mark Lawless

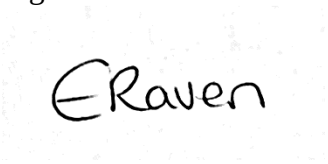
Date: 26/07/2023

JBA Risk Management
Skipton
United Kingdom

To the Director of Graduate Studies,
School of Geography and the Environment
Environmental Change Institute
South Parks Road, Oxford OX1 3QY

I, Emma Raven, certify that Olivia Becher completed the majority of the work in each of the chapters that form part of his DPhil thesis, as well as any associated articles that have already been published or may be published in the near future.

Signature:



Dr. Emma Raven

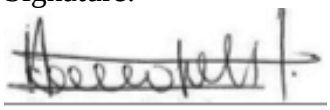
Date: 24/07/2023

JBA Risk Management
Skipton
United Kingdom

To the Director of Graduate Studies,
School of Geography and the Environment
Environmental Change Institute
South Parks Road, Oxford OX1 3QY

I, Homero A. Paltán López, certify that Olivia Becher completed the majority of the work in each of the chapters that form part of his DPhil thesis, as well as any associated articles that have already been published or may be published in the near future.

Signature:



Homero A. Paltán López

Date: 21/07/2023

The World Bank,
Washington DC,
United States



To the Director of Graduate Studies,
School of Geography and the Environment
Environmental Change Institute
South Parks Road, Oxford OX1 3QY

I, Mikhail Smilovic, certify that Olivia Becher completed the majority of the work in each of the chapters that form part of his DPhil thesis, as well as any associated articles that have already been published or may be published in the near future.

Signature:

Mikhail Smilovic

Mikhail Smilovic

Date: 24/07/2023

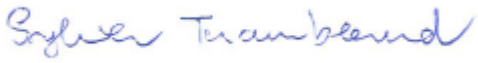
International Institute of Applied Systems Analysis
Schlossplatz 1
A-2361, Laxenburg, Austria



To the Director of Graduate Studies,
School of Geography and the Environment
Environmental Change Institute
South Parks Road, Oxford OX1 3QY

I, Sylvia Tramberend, certify that Olivia Becher completed the majority of the work in each of the chapters that form part of his DPhil thesis, as well as any associated articles that have already been published or may be published in the near future.

Signature:



Sylvia Tramberend

Date: 22/07/2023

International Institute of Applied Systems Analysis
Schlossplatz 1
A-2361, Laxenburg, Austria

Appendix B: Supplement to chapter 4

Supplementary Text B1-2, Figures B1-11, Supplementary Table B1

Supplementary Text B1: Spatial synthetic flood events

26 rainfall-runoff models were established in catchments ranging in area from a minimum of 215 to maximum of 979 km^2 across Jamaica to produce a 10,000-year river flow series, at a daily frequency. This was extracted from a hydrological modelling procedure undertaken at each of approximately 170,000 points of interest covering the globe. Just over 14% of these sites were gauged locations where observed daily or monthly river flow data was available. This was used to calibrate IHACRES-CMD (plus degree-day snowmelt where appropriate) rainfall-runoff models (see Croke and Jakeman, 2004). The input to the calibration process was provided by the global gridded CFSR reanalysis precipitation and temperature data spanning 1980-2010 and aggregated by catchment boundary (Saha et. al., 2010). For ungauged catchments, a nearest neighbour scheme was used whereby the 4 most similar gauged catchments were identified according to a set of catchments attributes including climate type, soil properties, land-use type, elevation, mean annual precipitation and area. The rainfall-runoff models for these 4 donor sites were driven with the input for the ungauged site with the final flow estimate being formed from a weighted mean of the 4 flow outputs based on degree of similarity. Due to the global scale of the procedure, only gauging data available from public databases was used: the streamflow data available from the Jamaican Water Resources Authority was not used to calibrate or validate the rainfall-runoff models, but they were rather run based on catchment attributes from 24 donor catchments globally. That being said, extensive validation of the global hydrological modelling procedure has been conducted. Table S3 below lists types of validation and corroboration data, that have been used to validate the global flood maps where possible.

Supplementary Table B1: Types of data used to validate or corroborate parts of the Global Flood Map development process.

Data	Purpose
Ground control points	Validate elevation values in the DEM
Peak river flows (max in record)	Corroborate peak flow estimates (typically 20 to 100-year return period)

AMAX river flows	Validate peak flow estimates for any return period
Published design flows	Validate peak flow estimates for any return period
Peak rainfall from recent events	Corroborate design rainfall totals (validation not possible due to difficulty in assigning return period)
Published IDF curves	Validate rainfall totals for any return period
Post-flood field survey data (high water marks, measured water depth)	Corroborate hydraulically modelled flood extents and water depths (validation not possible due to difficulty in assigning return period)
Recorded flood outlines (e.g., from satellite imagery or field survey)	Corroborate hydraulically modelled flood extents (validation not possible due to difficulty in assigning return period)
News and media reports and photographs	Corroborate hydraulically modelled flood extents and water depths (validation not possible due to difficulty in assigning return period)
Third-party or other vendor flood maps	Corroborate hydraulically modelled flood extents and water depths (validation not possible due to lack of information about model assumptions or methodology)

Figure B1 shows the observation points and hydrological units (HAZ polygons) used for fluvial and pluvial flood exposure mapping. HAZ polygons are boundaries of hydrological sub-catchments that are capped in size. It is assumed that the return period for an event is approximately homogenous throughout a given HAZ; therefore, they are a convenient way to transfer event severities from rainfall and river discharge Observation Points (OPs) to Exposure Points (EPs) for fluvial flood analysis. Whereas, to transfer events from precipitation gauges to assets for pluvial flooding, assets were assigned to the closest gauge.

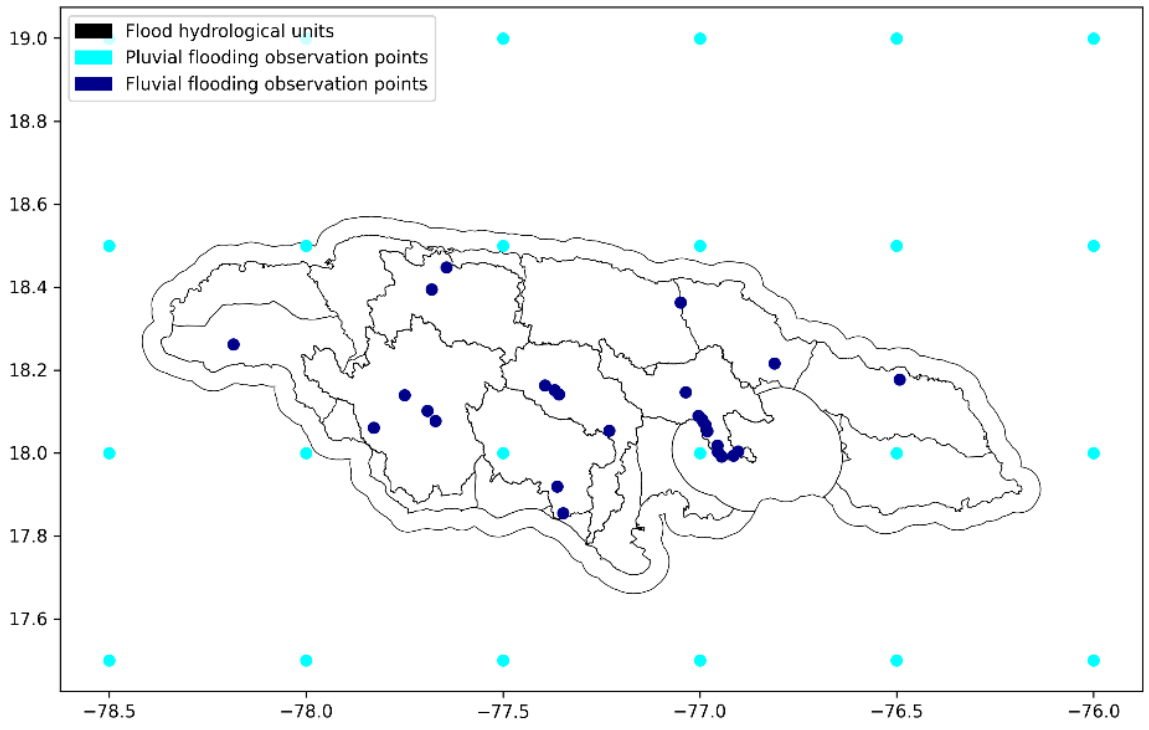
OPs can be linked to Hydrological Accumulation Zones (HAZ), which provide geographic areas for the purpose of transferring observation point event severities to exposure points (EP). Exposure points are attributed with the IDs of the HAZ polygon that contains them and all OPs within the HAZ. HAZ polygons are boundaries of hydrological sub-catchments that are capped in size. It is assumed that the return period for an event is approximately homogenous

throughout a given HAZ; therefore, they are a convenient way to transfer event severities from river and precipitation gauges (OPs) to exposure points.

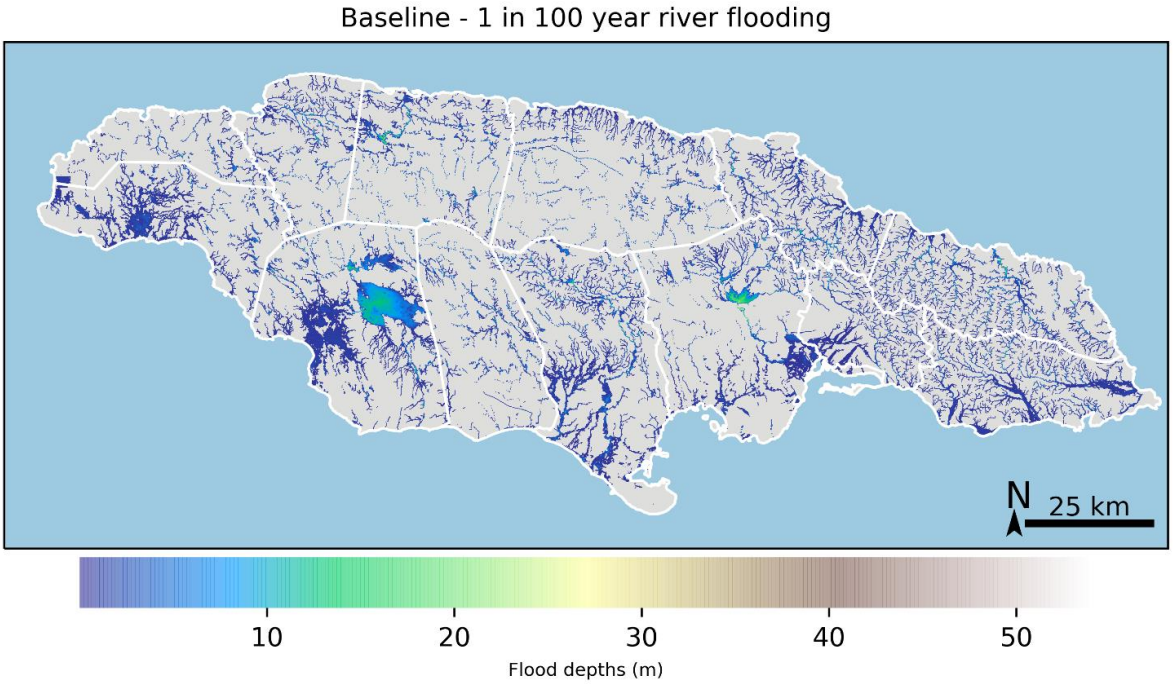
For further information, please refer to the JBA technical report (JBA, 2020b).

There will be several possibilities of OP, EP, HAZ intersections and combinations.

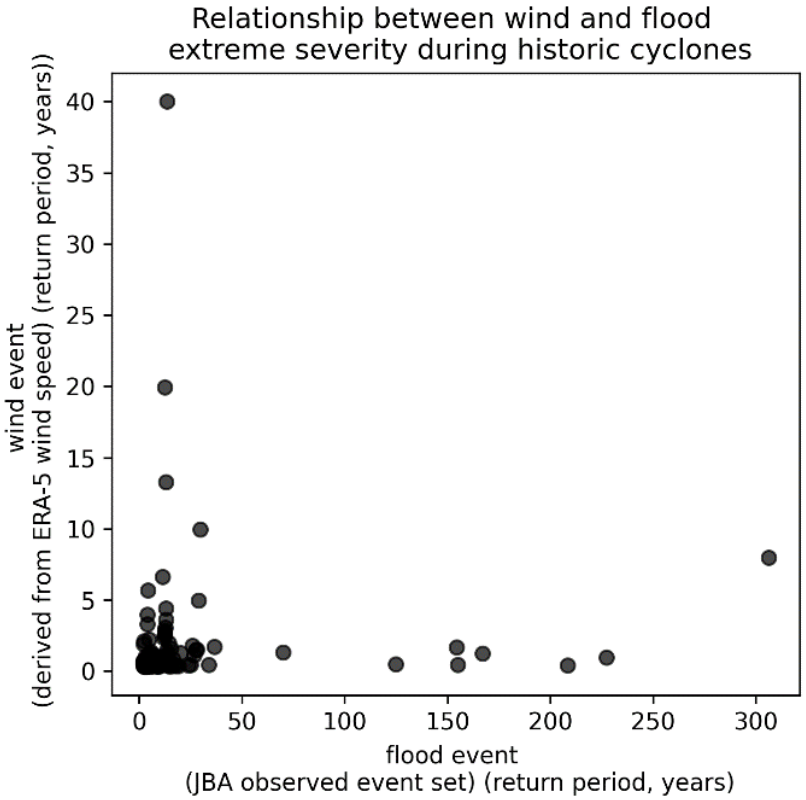
1. If 1 HAZ has only one OP, then all EPs are assigned the return period of the OP.
2. If 1 HAZ has multiple OPs, then the EPs are assigned the geometric mean of all OP return period values.
3. When there is no OP in a HAZ that contains EPs, event severity is taken from the OP closest to the HAZ border within a configurable distance. If no OPs are within this distance for a given flood type, the exposure point is considered not at risk from that flood type.
4. Where an EP is not located within a HAZ geographic area for RP transfer, it is not analysed.



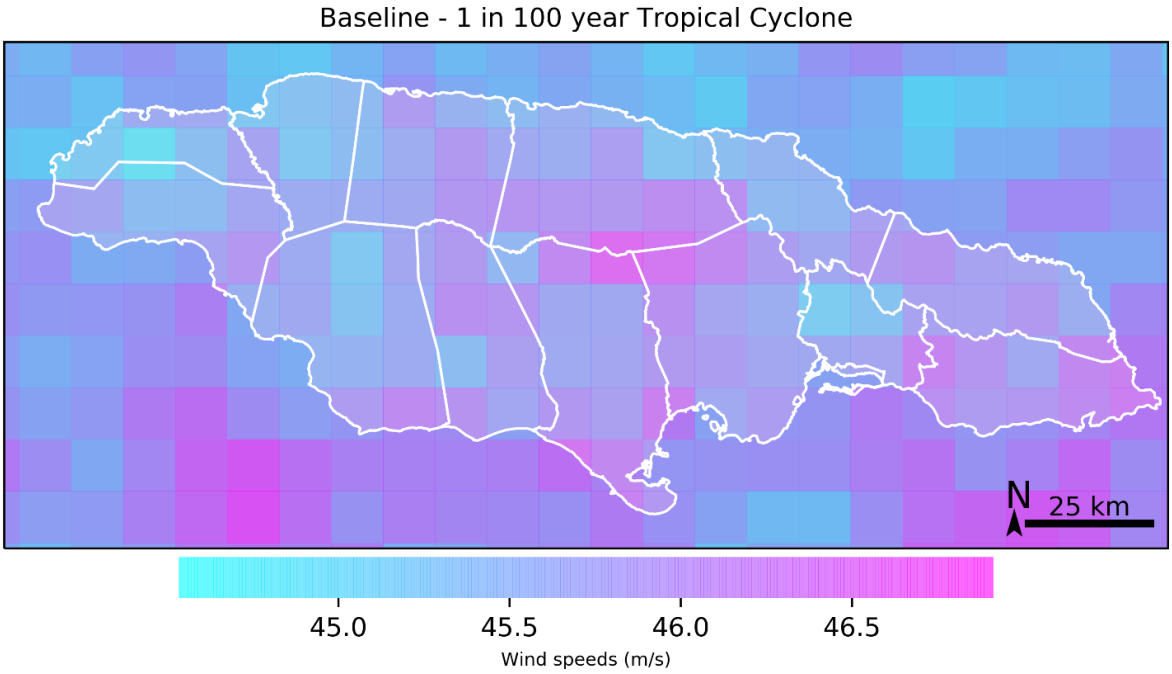
Supplementary Figure B1 Spatial hydrological units for fluvial flooding (black boundaries) and Observation Point locations for fluvial (dark blue) and pluvial (light blue) flooding for the flood event dataset. Datasets were provided by JBA.



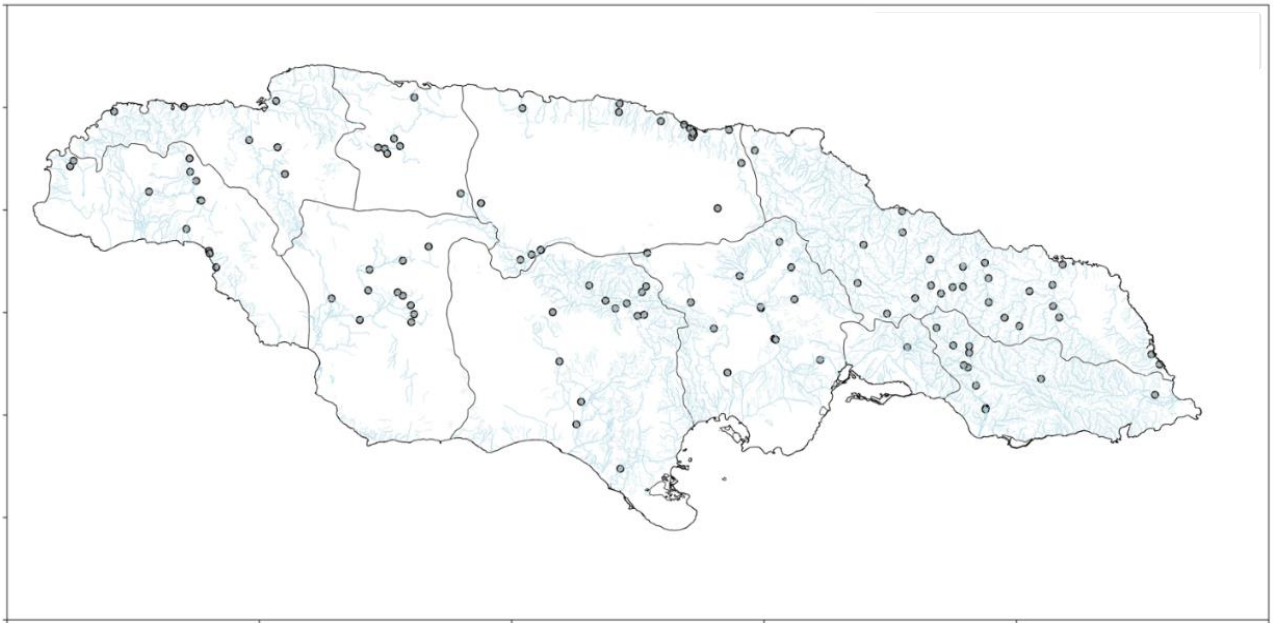
Supplementary Figure B2. 100-year flood inundation (m) under baseline climate conditions (JBA, 2020a).



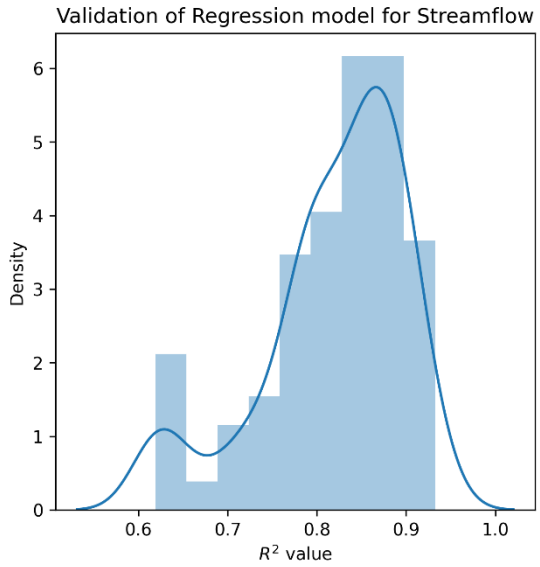
Supplementary Figure B3. Asymptotic relationship between wind and flood severity during historic cyclones.



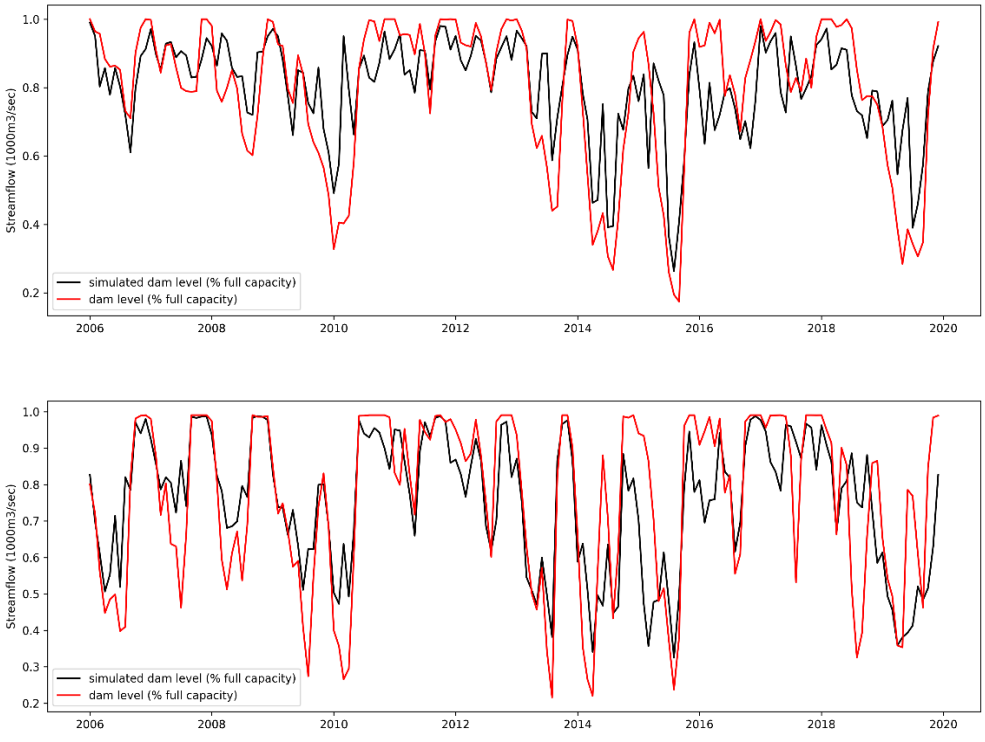
Supplementary Figure B4. 100-year tropical cyclone peak gust speed (m/s) under baseline climate conditions (Bloemendaal et al., 2020).



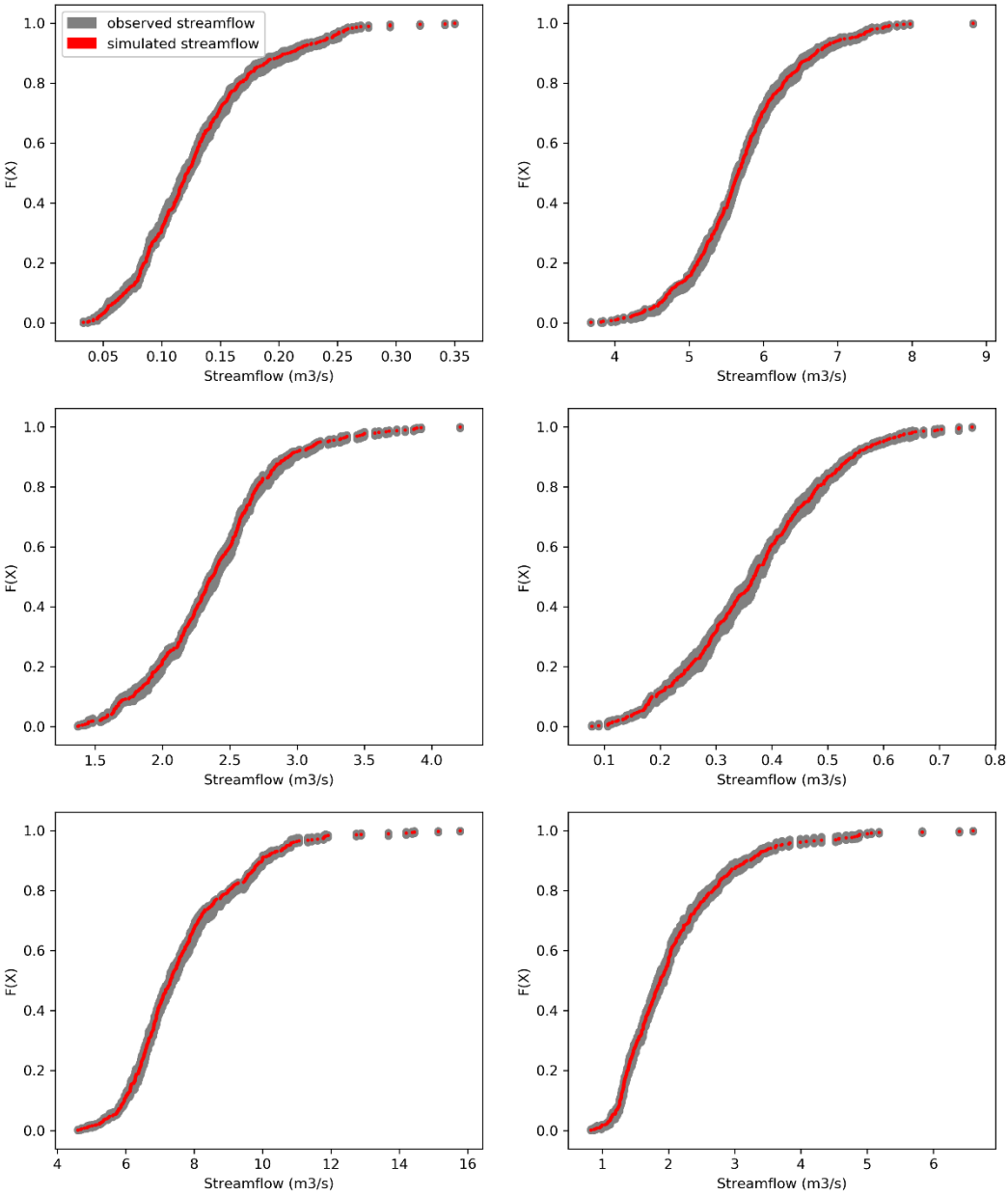
Supplementary Figure B5. Streamflow gauges, rivers and major catchment boundaries. Streamflow gauge data was provided by the Jamaican water resources authority (Jamaica Water Resources Authority, 2020), river and catchment outline were provided by the National Spatial Management Division (National Spatial Data Management Division, 2021).



Supplementary Figure B6. Performance of random forest regression model for statistical derivation of streamflow.



Supplementary Figure B7. Comparison between statically derived reservoir level variations (black) and observed (red) for Mona dam (top panel) and Hermitage dam (bottom panel). Observed dam level variations were provided by the National Water Commission (NWC, 2021b).



Supplementary Figure B8. Comparison between output of synthetic streamflow generator and input data for a selection of 6 streamflow gauges.

Supplementary Text B2: Infrastructure criticality

National water supply infrastructure is made up of multiple systems fed by a given source or set of multiple sources that serve a particular population or demand zone. Each system is, in turn, made up of a set of assets, typically including a source (i.e., a well or river intake), treatment plants, pumping stations, pipelines, storage tanks. To estimate the customers served by individual assets – and thus the number of people to whom water supply would be disrupted if a particular asset was disrupted - it is necessary to understand how different types of water

supply assets interact in a system. Thus, the criticality of the network was established by first determining the configuration of individual assets that make up each system and thus the spatial extent of each system, before allocating the population served by each system to the scale of each individual asset.

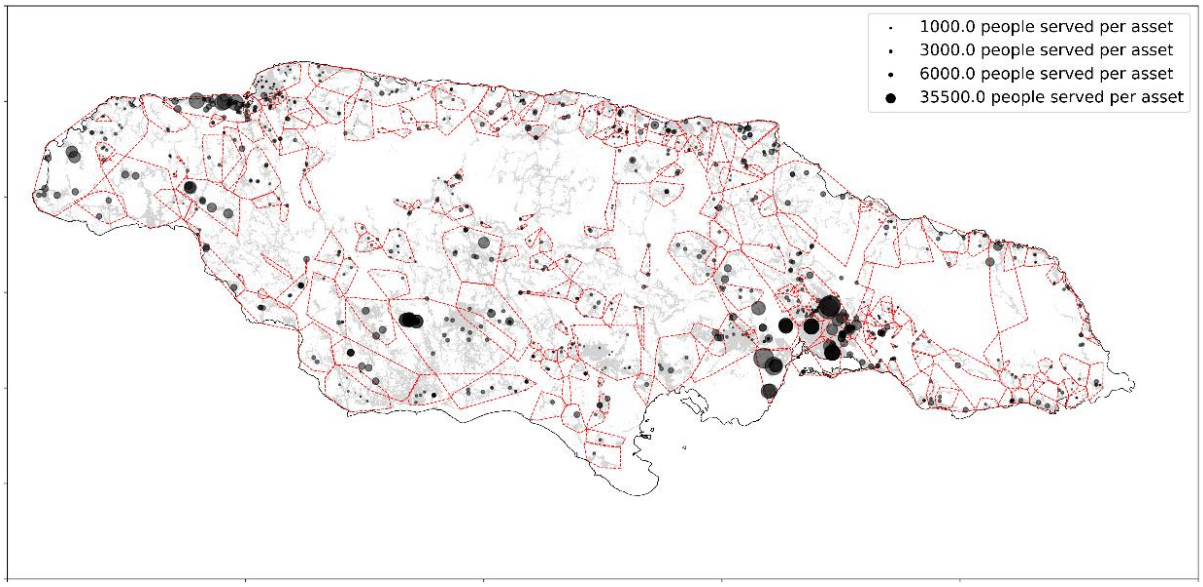
To apportion each potable asset into the respective systems that they serve. The WSPs outline the names of water source (i.e., wells, springs and rivers) supplying each system, such that assets with names matching the name of the system or system sources could be assigned to that system. Similarly, information on the settlements served by each asset/system, as provided by the disruption notices and WSPs, could be used to assign assets named according to the settlement in which they are located to the wider system they are a part of. Assets that were not connected to the piped network, such as catchment tanks, and decommissioned assets were removed from consideration.

The next step is to define the service area of each system, for which Voronoi diagrams were used to delineate the spatial reach of a particular asset. The service area of each system was derived based on the dissolved geometry of each Voronoi polygon belonging to the assets it is made up of. The convex hull of the piped network that fell within each system service area was then used to isolate the service area to the extent of the potable piped network. Thus, each asset was assigned to the system they belong to, which in turn was associated with a population and area served. Note that some systems that were included in the potable asset data but had not been outlined by the WSPs, e.g., no plan was published for water systems in St. Thomas and some systems may also have been constructed after the plans were published. Where this was the case the population within the service area was used based on census data and an average proportion of the total population served by the NWC network. As the plans were published in 2015, the projected population served data for the year 2020 was used.

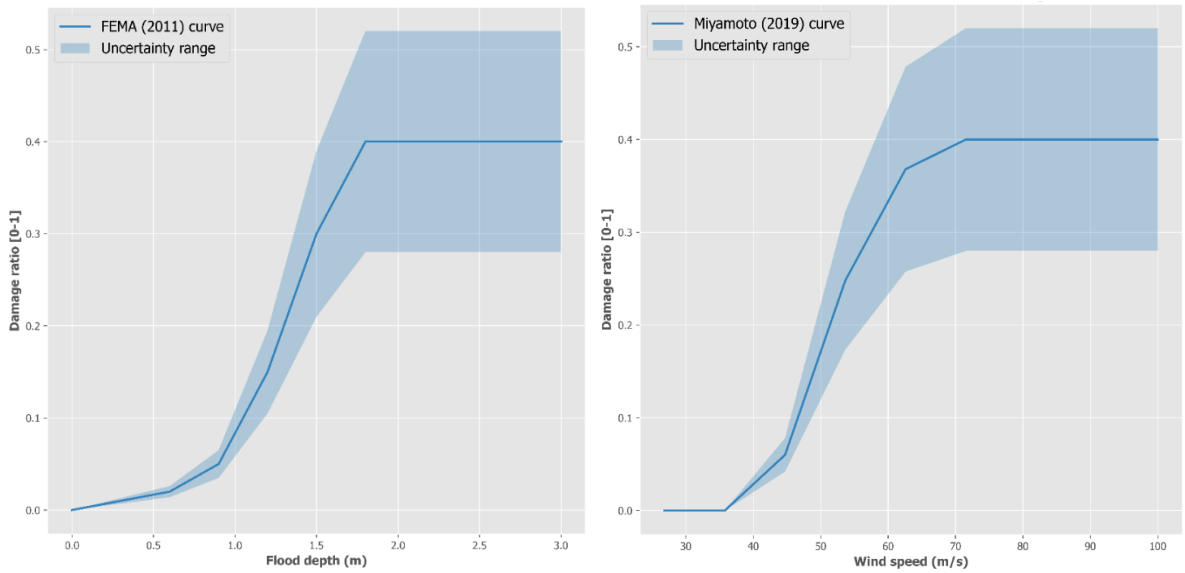
Asset-level criticality in terms of population served can subsequently be estimated based on the nature of the asset in question and the configuration of assets within each system with respect to the piped network. Potable supply systems can be thought of as a hierarchical network with a source feeding a network of pumps which in turn supply storage tanks. Therefore, some types of potable asset can be defined as sources (i.e., spring, wells, reservoirs and treatment plants), junctions that merely convey water across the network but are not consumptive (pumping stations) and sinks that are typically located at the site of served population (storage tank). The population served by each system was apportioned to individual

assets based on the following rules. For sources, where a single source served a given system, it is assigned 100% of the population served by that system. Where multiple sources serve a given system, they are assumed to serve a fraction of the total system population served based on source capacity data, where available. Where source capacity data was not available, the population served by was equally apportioned between sources. For junctions, where the configuration of a system is a linear path from source to sink via a series of junctions or a single junction, the junction was assigned 100% of the population served by that system. Where the configuration of a system includes one or multiple bifurcations, the assets along each bifurcated path were taken to feed a fraction of the total population fed by that system proportionate to the fraction of the piped network within the service area of that path. For sinks, where the configuration of a system is a linear path from source to sink, it is assumed that the full population served per system can be divided equally per storage tank within the system. Where the configuration of a system includes one or multiple bifurcations, the population served by each bifurcated path was divided equally between the storage tanks within the service area of each path.

Disruption notices are published by the NWC on their website and social media platforms to communicate which settlements are either currently subject to a disruption or warn of an incoming disruption, to their water supplies during asset shutdowns or water shortages (National Water Commission, 2020b). This data was used in conjunction with information on the settlements served by each system provided in the Parish WSPs (National Water Commission, 2011) to indicate the level of interaction between systems and the spatial reach of a particular asset or system. It was found that the vast majority of systems do not demonstrate a level of operational interdependency as they do not substitute for another during disruptions, even if they were physically connected via the piped network. This is with the exception of a major inter-system transfer scheme provided by the Ferry/Rio Cobre system which supplements four supply systems in the Kingston and St. Andrew region via two pumping stations, Molynes and Glenmore Road. As these pumping stations are effectively the system sinks for the transferring system and source nodes for the receiving systems, the total population served as outlined in the WSPs was apportioned between these two pumping stations proportional to the relative balance in the receiving systems: Long Mountain, Cavaliers, Constant Spring and Mona. Note that although certain systems share sections of the distribution network, the system population served denoted by the WSPs accounts for this.



Supplementary Figure B9. The criticality of each potable water asset in terms of population served and outlines of the service area of each system. The grey shaded areas indicate the disruption of household across the island. Note that the system service areas indicate the extent of the potable assets belonging to each system, as opposed to the extent of household served by each system.

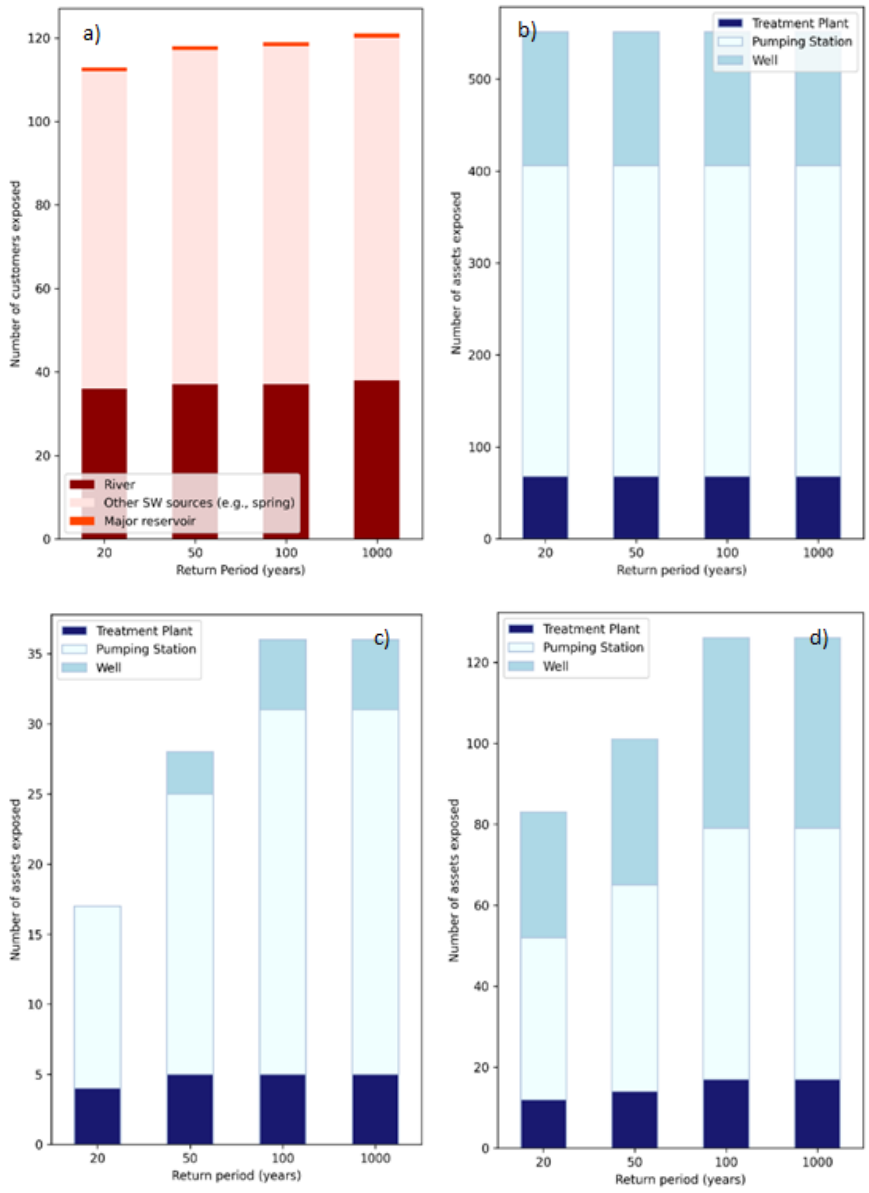


Supplementary Figure B10. Damage curves for water treatment plants for flood depth (left) (Miyamoto, 2019) and wind speed (right) (FEMA, 2020), with an uncertainty range bounded above and below by these curves or we assume a ±30% uncertainty range in the single curve estimates.

Supplementary Text B3: Uncertainty analysis

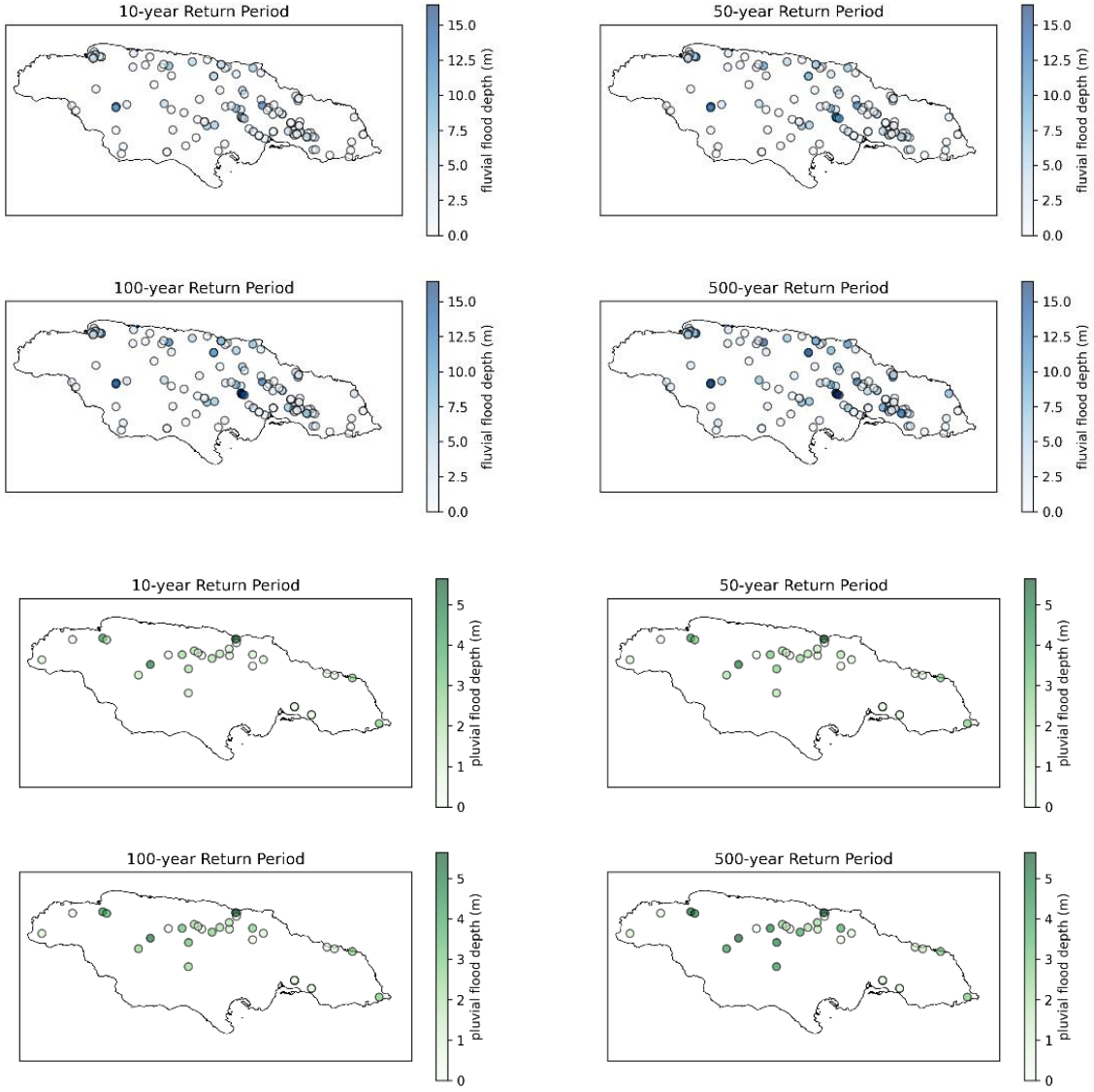
As variations in the vulnerability were found to dominate the effect of proportionate variation in key hazard variables (flood inundation, streamflow and wind speed), the sensitivity of the

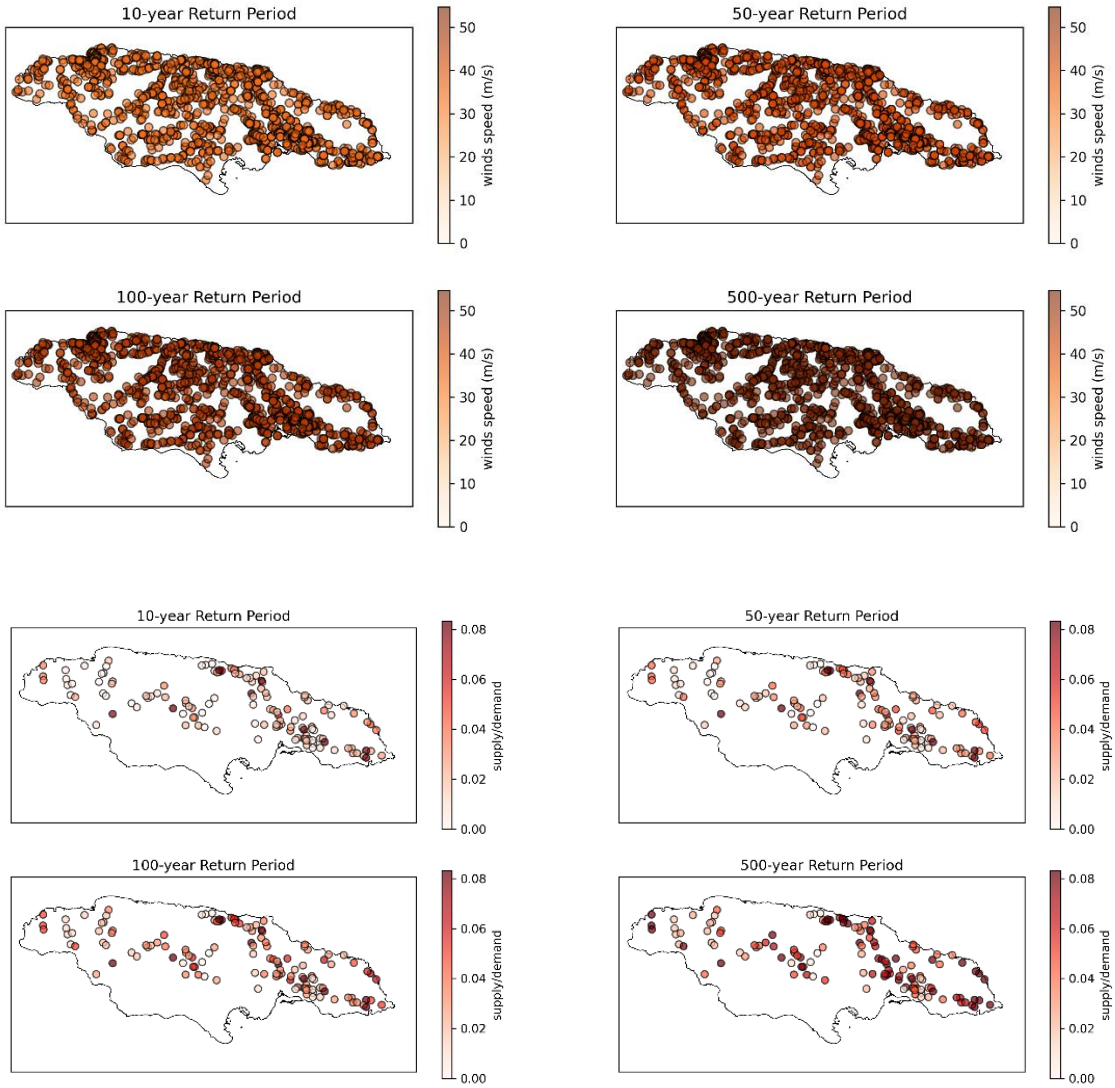
results is investigated by varying key vulnerability parameters. As such, the drought threshold (i.e., the sum of demand, leakage and environment flow volume per system) and the fragility curve (i.e., the amount of damage incurred by a given flood depth or wind speed per asset) was adjusted by a common factor of +/-30% to provide a likely upper and lower bound for the results. Although this approach is simplistic, similar approaches have previously been applied for large scale risk assessments (Koks *et al.*, 2019). However, the analysis could always be with more ground truthed data on the uncertainty associated with the response of assets to hazard severity, and actual uncertainties associated with demand and leakage losses from different systems.



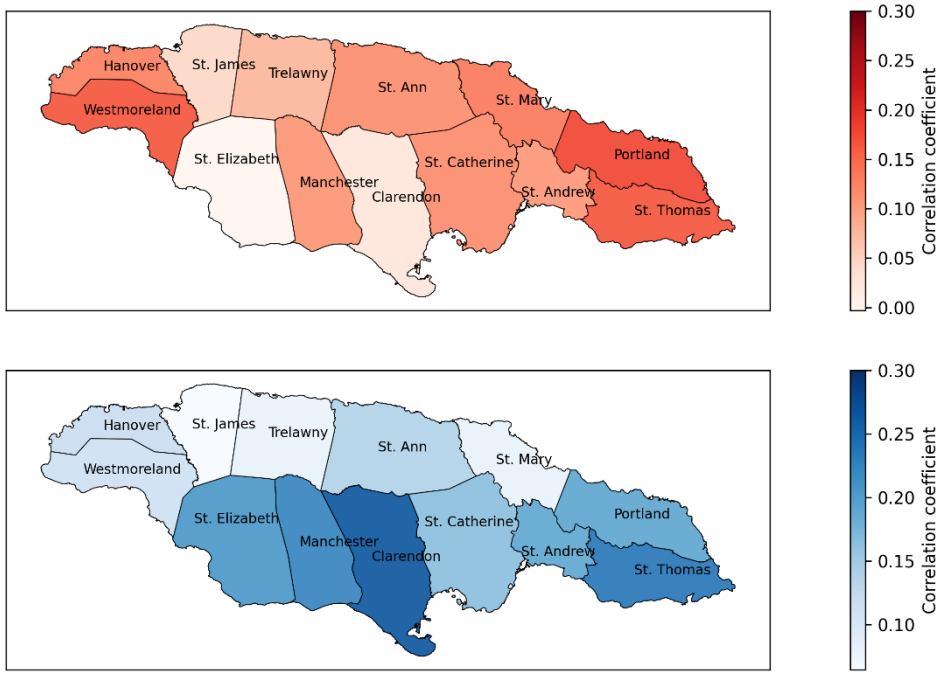
Supplementary Figure B11. Exposure of assets per asset type per hazard: the top left plot (a) is for drought hazard, the top right plot (b) is for extreme wind during hurricanes, bottom left

(c) is for pluvial flood hazard and bottom right (d) is for fluvial flood hazard. Note that here, pumping stations include intakes.

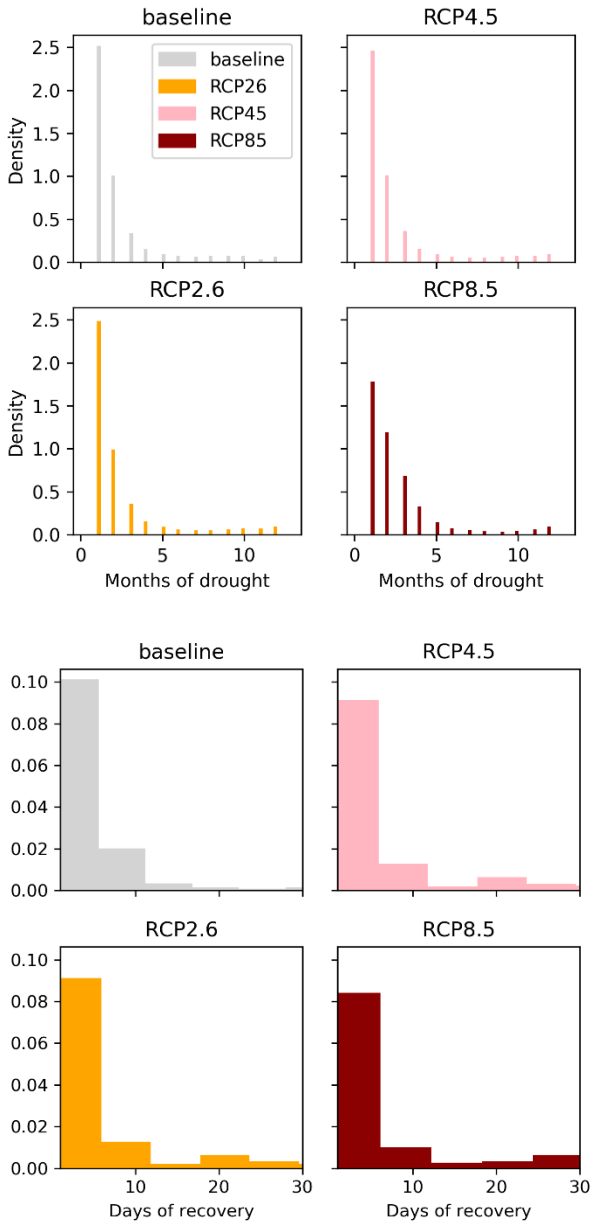




Supplementary Figure B12. Spatial exposure of assets to fluvial flooding, pluvial flooding, extreme wind, and drought deficit for a selection of four return periods. From top to bottom: exposure of assets to fluvial flood inundation, pluvial flood inundation, extreme wind, and drought deficit for a selection of four return periods.



Supplementary Figure B13. Mean correlation coefficient between each parish and all others for water shortage disruptions (left) and asset damage disruptions (right).



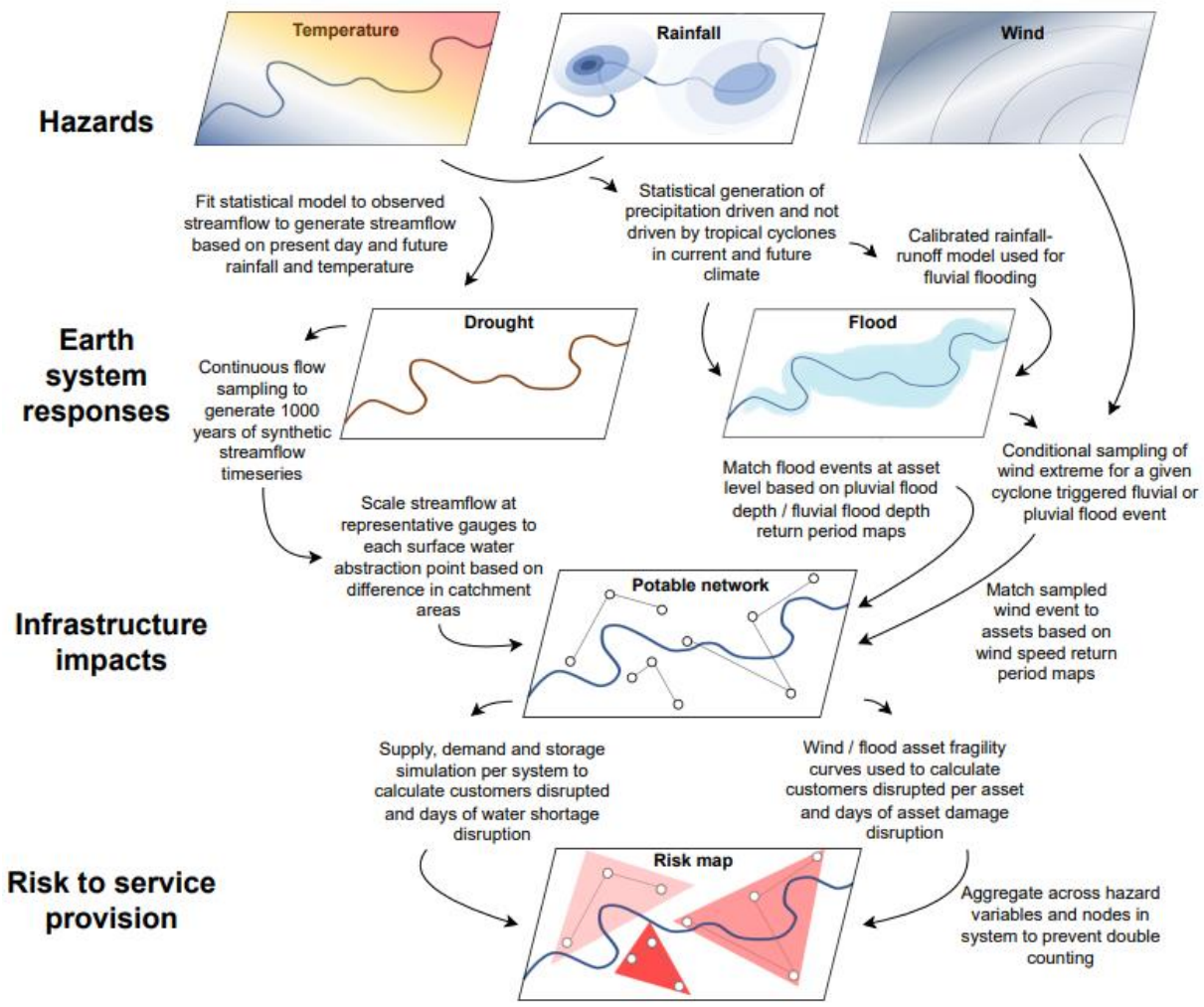
Supplementary Figure B14. *Distribution of water shortage event durations in months (left) and asset damage recovery durations in days (right) per climate scenario.*

Appendix C: Supplement to chapter 5

Supplementary Text C1-2, Tables C1-3, Figure C1-4

Supplementary Text C1: Overview of multi-hazard risk analysis (Becher *et al.*, 2023).

Note that the following section is paraphrasing from Becher *et al.* (2023). Becher *et al.* (2023) developed a multi-hazard risk framework that can be generally applied to quantify the effects of present-day and future climatic shocks on the customers of potable water supply networks. A case study of the national water service provider in Jamaica was adopted to develop the approach. The first step of the framework was to synthetically generate a large set of drought, cyclone and pluvial and fluvial flood events under historic climate conditions and for emissions scenarios RCP2.6, RCP4.5 and RCP8.5, projected to the end of century. The spatial footprint of each event was simulated to capture potential widespread impacts. The second step of the framework involved creating an intermediate complexity model of the national water supply infrastructure network in Jamaica. Assets in the model were assigned their respective damage functions, to characterise their vulnerability to physical damage disruption duration for a variations in wind speed and flood inundation depth. Major storage dynamics were also simulated within the model, along with domestic demand and leakage losses, to simulate the vulnerability of the network to variations in supply. Third, we intersect the event set of multi-hazards with water supply network to estimate quantification of the number of users impacted per day multiplied by the duration of the event (hereafter Customer Disruption Days, or CDD) applies across these different hazard types.



Supplementary Figure C1: Conceptual overview of multi-hazard risk framework (Becher et al., 2023).

Supplementary Table C1: Data sources used for multi-hazard risk (Becher et al., 2023).

Component of analysis	**Sub-component**	**Source**
Flood (fluvial and pluvial)	Return period maps	(JBA, 2020a)
	Event set	(JBA, 2020b)
Drought	Observed streamflow	(Jamaica Water Resources Authority, 2020)
	Synthetic streamflow generator	(Wheeler *et al.*, 2020)
	Meteorological data	(Hersbach *et al.*, 2020)
	Climate change forcing	(Martínez-Castro *et al.*, 2018)
Infrastructure	Potable water assets	(National Spatial Management Branch, 2010)

	Asset damage curves	(FEMA, 2020)
	Parish water supply plans	(National Water Commission, 2011)

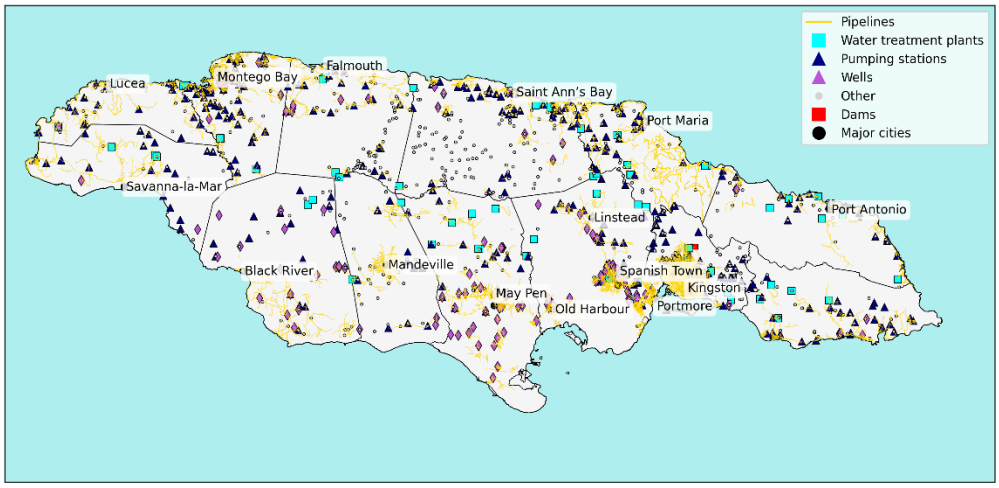
Supplementary Table C2: Asset damage curves (FEMA, 2020).

Asset	**Flood depth (ft)**	**Damage ratio**
water treatment plant	0	0
water treatment plant	1	1
water treatment plant	2	2
water treatment plant	3	5
water treatment plant	4	15
water treatment plant	5	30
water treatment plant	6	40
water treatment plant	7	40
water treatment plant	8	40
water treatment plant	9	40
water treatment plant	10	40
pumping unit	0	0
pumping unit	1	0
pumping unit	2	0
pumping unit	3	0
pumping unit	4	40
pumping unit	5	40
pumping unit	6	40
pumping unit	7	40
pumping unit	8	40
pumping unit	9	40
pumping unit	10	40

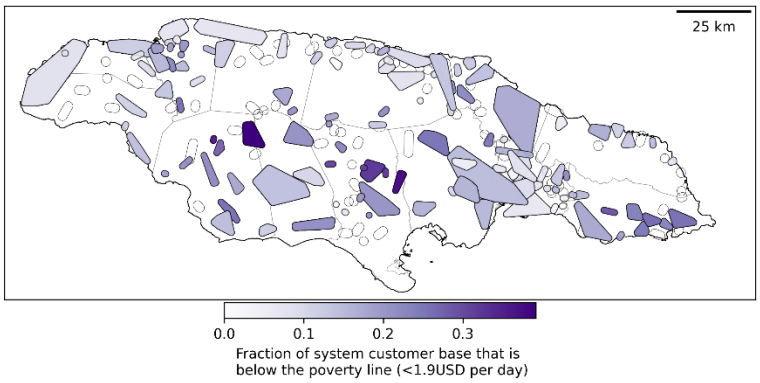
well	0	0
well	1	1
well	2	2
well	3	5
well	4	20
well	5	25
well	6	30
well	7	30
well	8	30
well	9	30
well	10	30

Supplementary Table C3: Water asset cost data (National Water Commission, 2011).

Potable nodes	**Asset count**	**Rehabilitation unit costs**	**Cost unit**
Booster Station	36	5 – 20	J$ million/asset
Catchment	100	N/A*	N/A*
Entombment	14	N/A*	N/A*
Filter Plant	23	200 – 300	J$ million/Mgal*
Intake	14	N/A*	N/A*
Production Well	145	70,000	J$/asset
Pump Station	130	5 – 20	J$ million/asset
Relift Station	126	5 – 20	J$ million/asset
Reservoir	40	N/A*	N/A*
River Source	13	5 – 15	J$ million/asset
Spring	51	5 – 15	J$ million/asset
Storage Tank	477	N/A*	N/A*
Sump	6	5 – 15	J$ million/asset
Treatment Plant	**33**	**200 – 300**	**J$ million/Mgal****



Supplementary Figure C2. Potable water infrastructure database (Becher et al., 2023).



Supplementary Figure C3. Fraction of NWC system customer base that is below the poverty line (1.9USD per day).

Supplementary Table C4. Flood defence cost curve (Mott MacDonald, 2018).

Asset details	**Adaptation option**	**Option cost unit**	**Initial investment**	**Routine cost**	**Routine intervals (years)**
Potable water treatment works	Flood defence around asset	GBP/m	1,000,000	5,000	1
Potable pumping station	Flood defence around asset	GBP/m	50,000	250	1

Supplementary Text C2: Water balance (Becher *et al.*, 2023).

Note that the following section is paraphrasing from Becher *et al.* (2023). Becher *et al.* (2023) considered all surface water abstraction points in Jamaica's National Water Commission's system. Each surface water abstraction point (j) was assigned a drought threshold (DT_j). Each drought threshold for river- or stream-fed systems is equivalent to the sum of the volume of demand that depends on each source (d_j), leakage losses from the network that distributes water from source to end used (l_j) and environmental flow threshold. The flow that is exceeded 90% of the time (i.e., Q10 of the streamflow input), to the abstraction point (NGPRP, 1974) was adopted as the environmental flow threshold (s_j). The drought threshold formulation is provided in Equation C-1. Note that the drought threshold (DT_j) for each reservoir is estimated by matching the timings of disruptions in the receiving potable supply systems with the observed reservoir level variations.

$$DT_j = d_j + l_j + j \qquad \text{(C-1)}$$

Becher *et al.* (2023) ran a monthly simulation of streamflow (for river- / stream-fed systems) or storage variations (for reservoir- / dam-fed systems) on a monthly timestep. If a system experiences a month (m) where supply falls short of the drought threshold, the system faces a drought-related disruption. The magnitude of the disruption is for a given month estimated based on the equivalent positive hydrological deficit (HD), Equation C-2.

$$HD_{y,m,j} = min\left(s_{y,m,j} - DT_j\ (-1), 0\right) \qquad \text{(C-2)}$$

Similarly, for reservoir- or dam-fed systems, a drought event is captured when a drop in reservoir levels $l_{y,m,j}$ occurs relative to the relevant source's DT_j (Equation C-3):

$$HD_{y,m,j} = l_{y,m,j} - DT_j\ (-1) \qquad \text{(C-3)}$$

To translate this monthly hydrological deficit into an equivalent customer disruption, is assumed that the fraction of total demand that is unmet by a given sources is proportionate to a disruption to the fraction of the customers that would be disrupted. To determine the impact of water shortages (ws) on customers in terms of CDD_{ws}, and hence directly compared against asset damage-related disruptions, it is assumed that the disruption would manifest as a reduction in the daily time window (i.e., through zoning) for supply, and hence would be evenly spread across a given month. Equation C-4, C-5 and C-6 provide the drought duration (δ_i),

where d_m denotes the number of days per month, accumulated drought deficit (AD_i) and the water shortage-related Customer Disruption Days CDD_{ws} per event (i), per supply source (j):

$$\delta_i = (m_{max} - m_{min})\, d_m \qquad \text{(C-4)}$$

$$AD_i = \sum_{m_{min}}^{m_{max}} HD_{y,m,j} \qquad \text{(C-5)}$$

$$CDD_{ws}(i,j) = \frac{AD_i}{D_i\, \delta_i} \times C_j \qquad \text{(C-6)}$$

The sample of drought disruption events where, thus, ranked to yield an empirical probability loss distribution. From this, CCD estimates at different return periods (RP) were extracted and the annual average expected $\overline{CDD_j}$ was obtained by integrating $CDD_{i,j}$across the annual exceedance probabilities of events (p), see Equations C-7 and C-8.

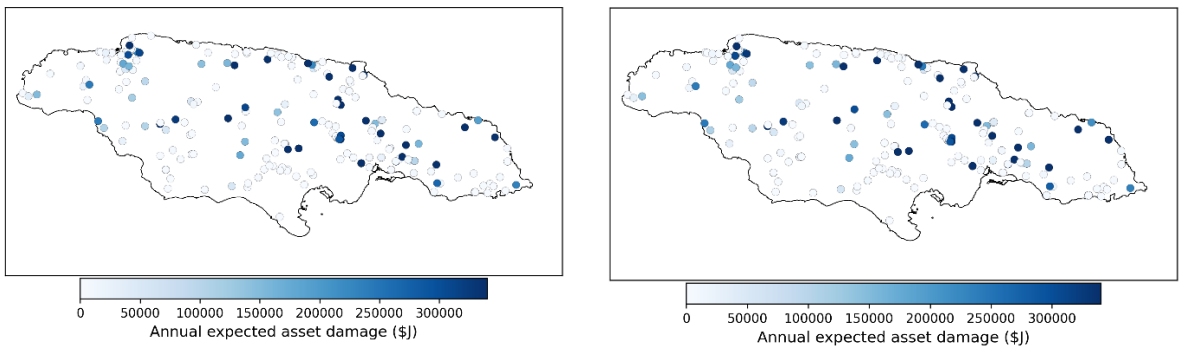
$$p_i = \frac{1}{RP_i} \qquad \text{(C-7)}$$

$$\overline{CDD_j} = \int_0^1 CDD_{i,j}\, dp_i \qquad \text{(C-8)}$$

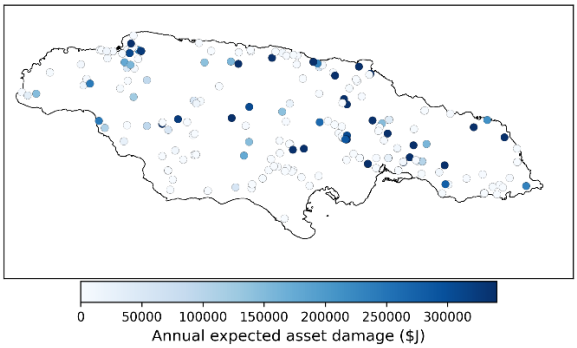
The same procedure was undertaken using synthetic streamflow inputs statistically generated under historic climate conditions and the three future emissions scenarios considered.

Supplementary Figure C4. Annual expected damages all simulated climate change scenarios.

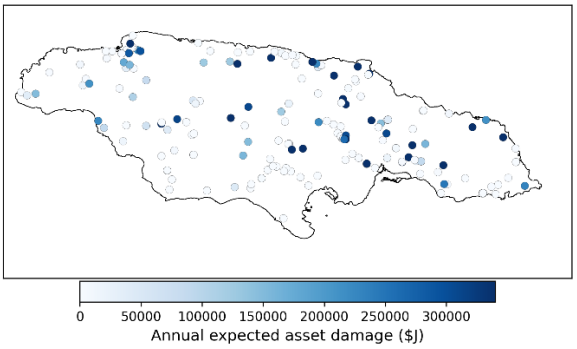
Baseline **RCP2.6**



RCP4.5

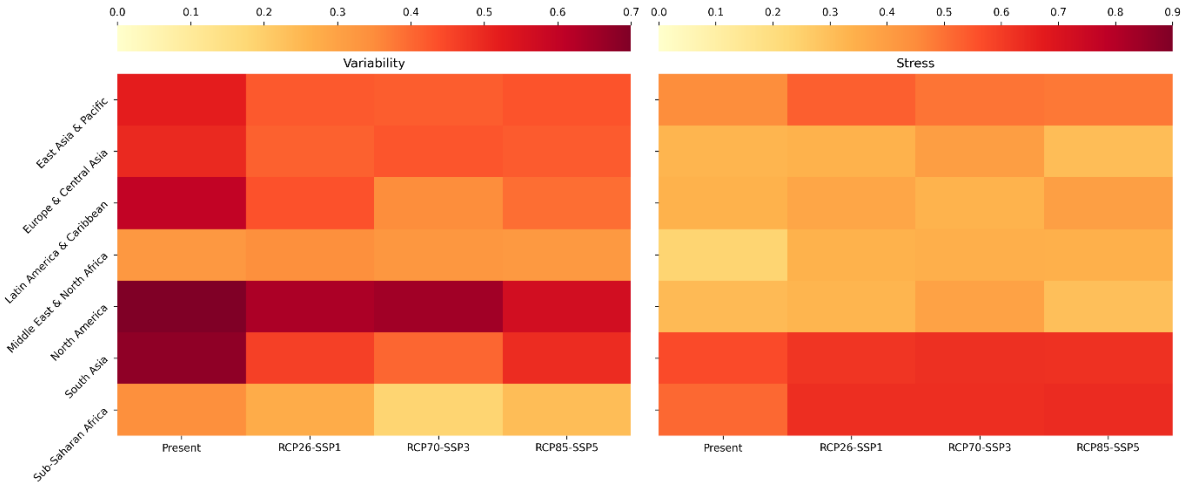

0 50000 100000 150000 200000 250000 300000
Annual expected asset damage ($J)

RCP8.5

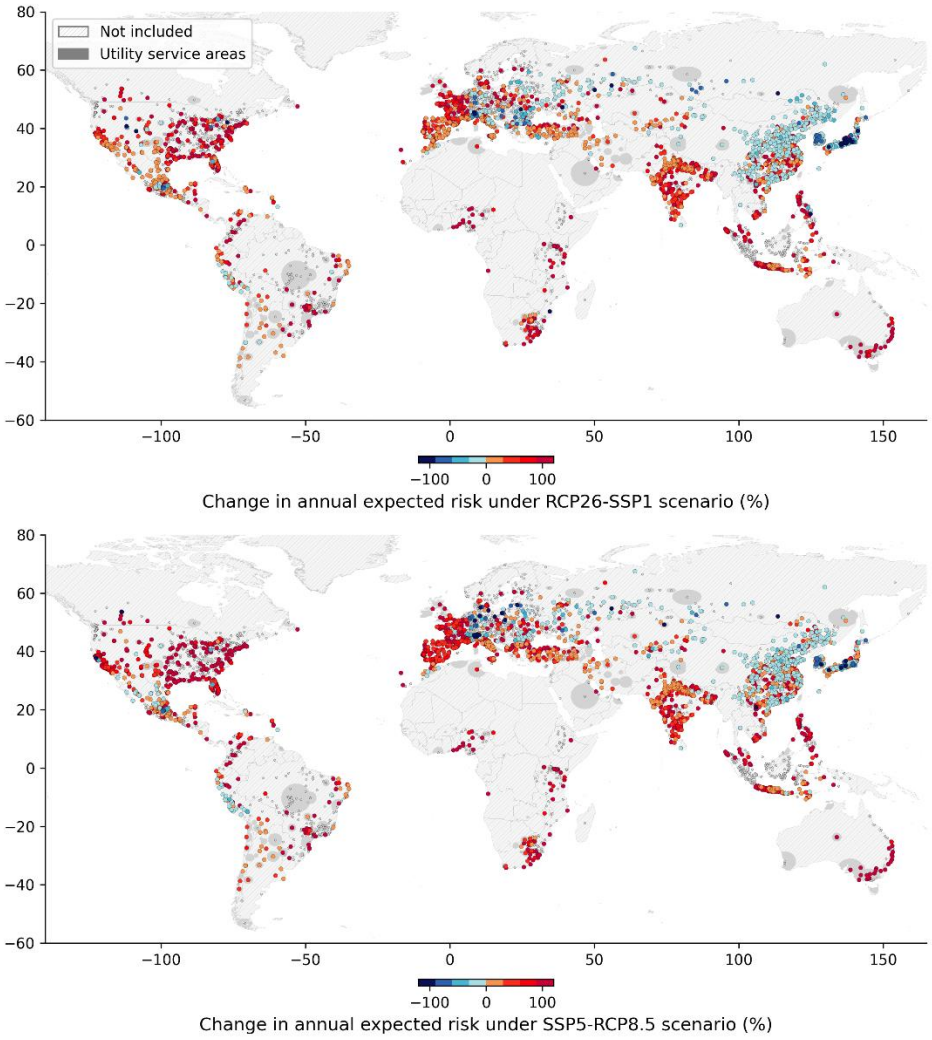

0 50000 100000 150000 200000 250000 300000
Annual expected asset damage ($J)

Appendix D: Supplement to chapter 6

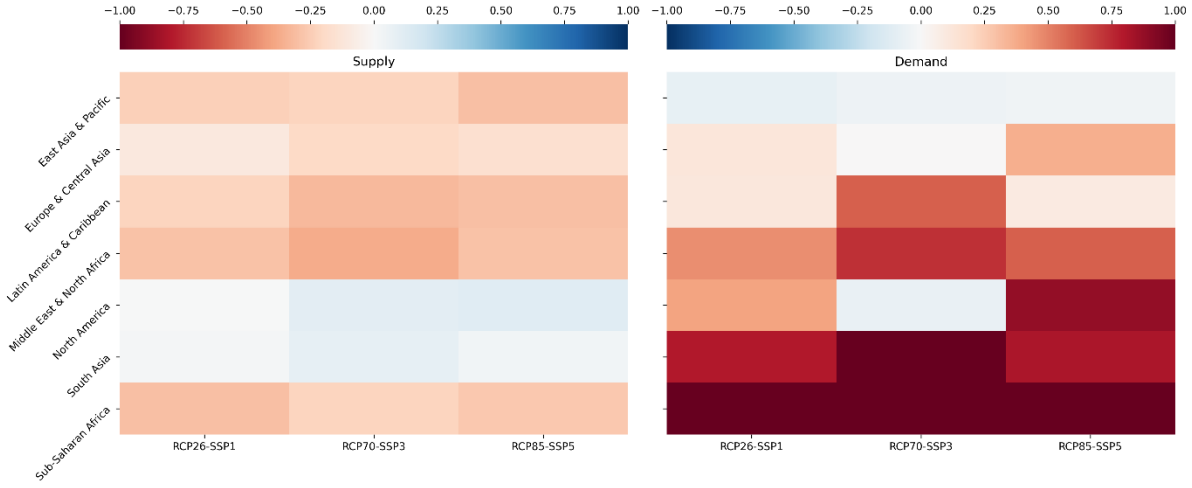
Supplementary Figures 1–9, Tables 1-2 and Text 1.



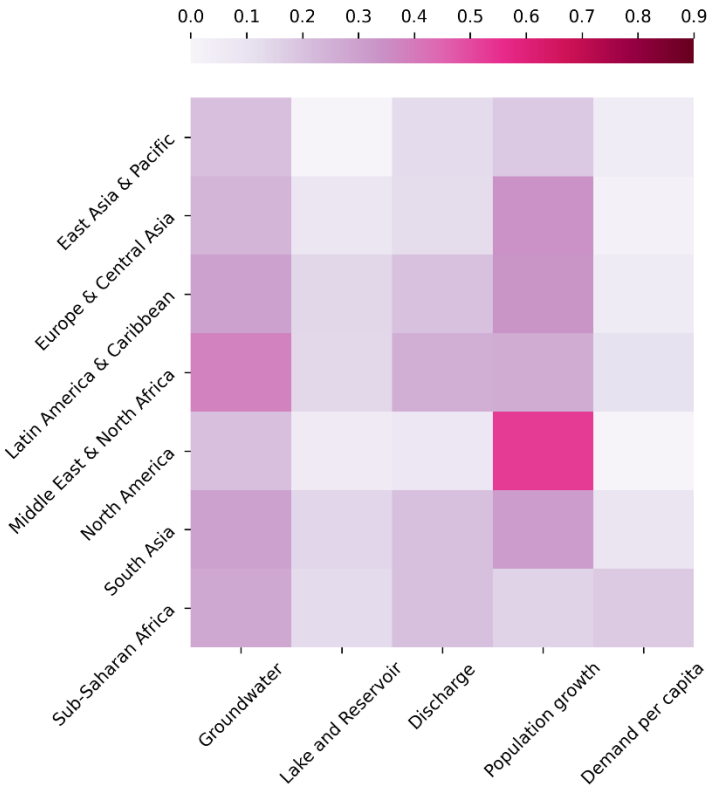
Supplementary Figure D1. Correlation coefficient between hydrological variability and stress on present-day risk per region, per climate scenario.



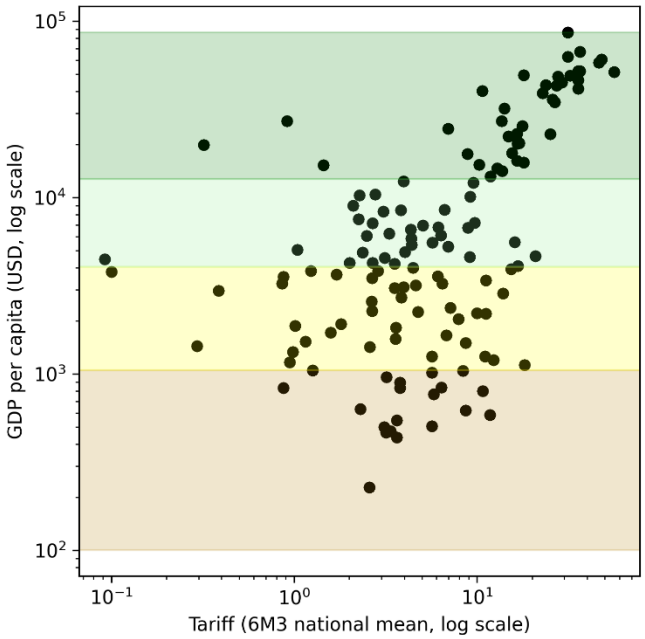
Supplementary Figure D2. Change in future risk projected under optimistic and pessimistic scenarios SSP1-2.6 and SSP5-8.5, respectively. Areas not covered by the utility database used for our analysis are indicated with hatching.



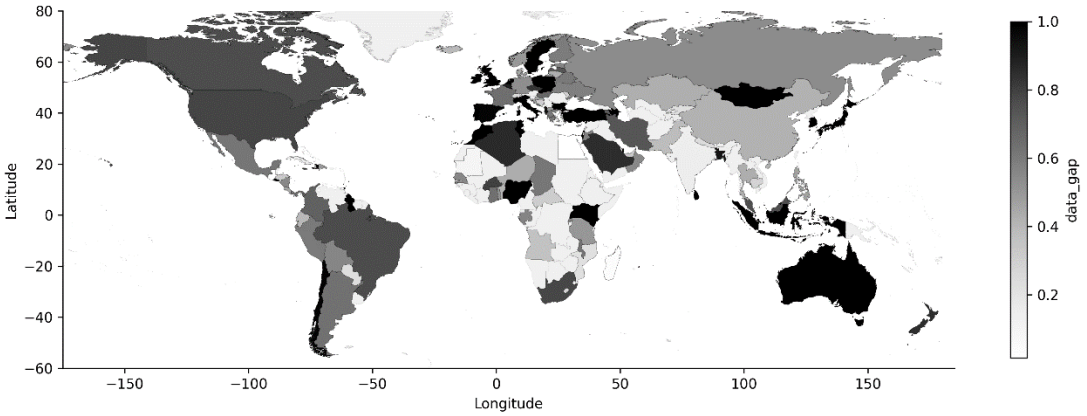
Supplementary Figure D3. Change in supply and demand per considered future scenario from the baseline scenario, per region.



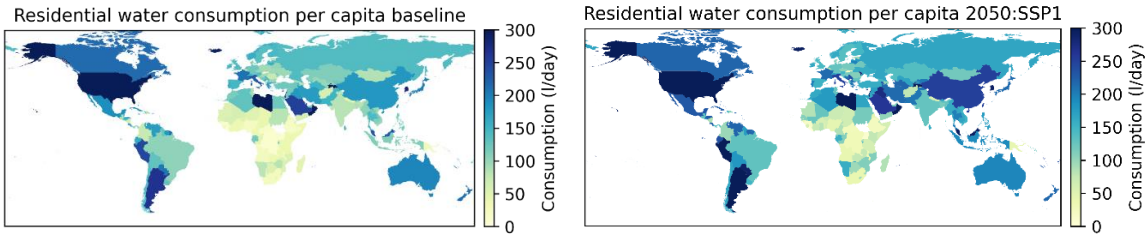
Supplementary Figure D4. Uncertainty range per input, per region. This is calculated as the maximum range across SSP and RCP scenarios considered in this study, divided by the mean.

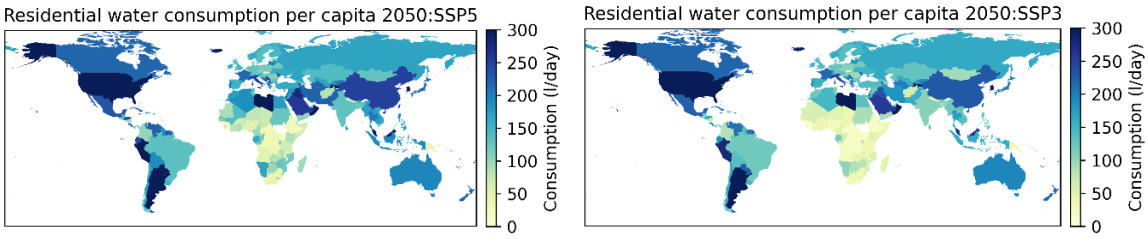


Supplementary Figure D5. Relationship between national mean tariff rate and GDP per capita.

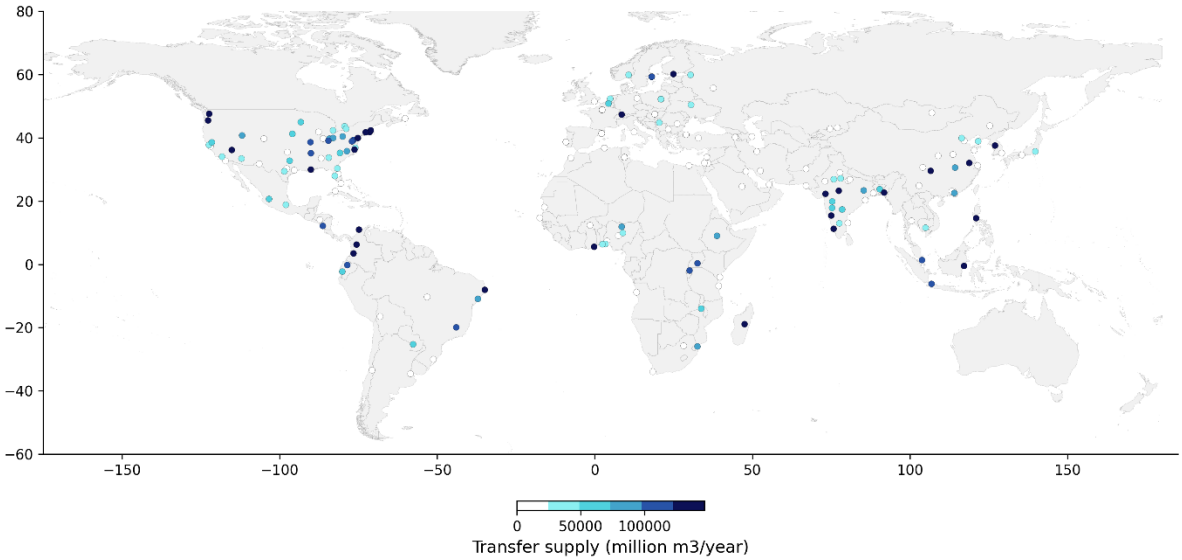


Supplementary Figure D6. Percentage difference between the total number of piped water customers (JMP, 2017) and those included in the utility database (GWI, 2022).

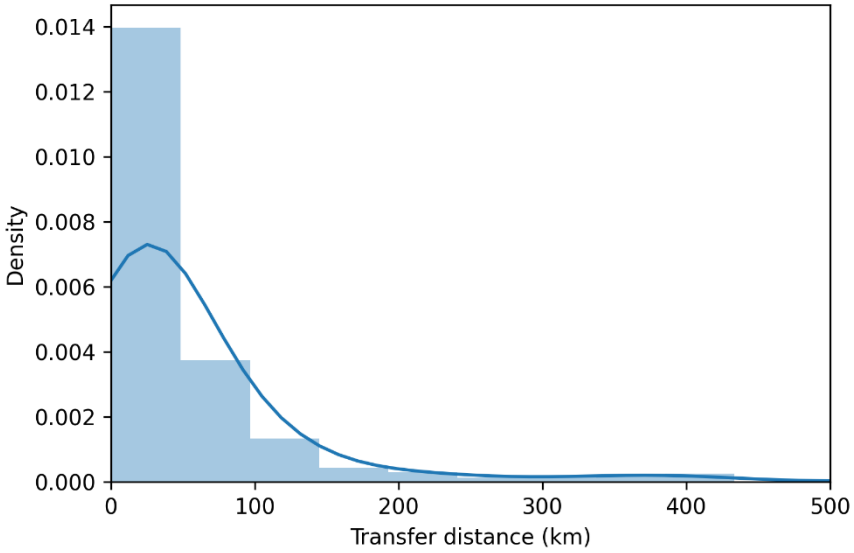




Supplementary Figure D7. Water demand per capita per scenario, with SSP scenarios projected to the year 2050.



Supplementary Figure D8. Spatial distribution utilities with long-distance transfer and their supply volume.



Supplementary Figure D9. Distribution of transfer distance in km (McDonald et al., 2014).

Supplementary Text D1. Community Water Model.

The Community Water Model (CWatM) is a process-based distributed hydrological model (Burek *et al.*, 2020). This paper uses model results at 30 arcminutes at a monthly timestep. Lakes and reservoirs include location, shape, area, and volume derived from the HydroLAKES (Messager *et al.*, 2016). Live storage, inflow, evaporation, abstraction, and outflow are simulated waterbody specific. Reservoir outflow is determined based on three storage limits and three outflow functions (minimum, normal, and non-damaging). Lake outflow is simulated with a weir function from Poleni for rectangular weirs. Groundwater is simulated as a linear reservoir, recharged through soil percolation and preferential flow (renewable), and outflows through capillary rise, baseflow, and pumping. Groundwater demand exceeding availability is facilitated through fossil water (non-renewable).

Supplementary Table D1. Modelling assumptions.

Adaptation option type	Min limits (% of demand)	Max limits (% of demand)
Leakage reduction	10%	Existing rate
Desalination capacity increase	0%	90%
Reservoir storage volume increase	0%	90%

Supplementary Table D2. Option cost inputs (Kahil et al., 2018).

Option	*Sub-component*	*Cost category*	*Value*	*Units*	*Source*
Non-revenue water reduction	Pipe renovation	Capital cost	94-628	Euros/m	(Ahopelto and Vahala, 2020)
Storage	Surface water reservoir	Capital cost	450 (150-2500)	1000$/Mm^3	(Keller, Sakthivadivel and Seckler, 2000; Wiberg and Strzepek, 2005)
		O&M cost	0.05 (0.01 – 0.3)	$/m^3	
Desalination	Plant	Capital cost	1700 (900–2500)	$/m3/day	(Ghaffour, Missimer and Amy, 2013)
		O&M cost	0.25 (0.15-0.35)	$/m3	

	Transfer	Capital cost	57 (34–135)	$/m3 /day	(Albiac *et al.*, 2003; Stillwell, King and Webber, 2010)
		O&M cost	0.01 (0-0.05)	$/m3	
	Pumping station	Capital cost	8.5 (7–15)	$/m3 /day	(Kirshen, 2007)
		O&M cost	0.01 (0-0.05)	$/m3	