College of Engineering, Mathematics and Physical Science

Assessing the Reliability, Resilience and Sustainability of Water Resources Systems in Data-rich and Data-sparse Regions

Submitted by Miguel Headley to the University of Exeter

as a thesis for the degree of

Doctor of Philosophy in Engineering

In February 2018

This thesis is available for Library use on the understanding that it is copyright material and that no quotation from the thesis may be published without proper acknowledgement.

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

Signature: ……………………………………………………………..

Abstract

Uncertainty associated with the potential impact of climate change on supply availability, varied success with demand-side interventions such as water efficiency and changes in priority relating to hydrometric data collection and ownership, have resulted in challenges for water resources system management particularly in data-sparse regions.

Consequently, the aim of this thesis is to assess the reliability, resilience and sustainability of water resources systems in both data-rich and data-sparse regions with an emphasis on robust decision-making in data-sparse regions. To achieve this aim, new resilience indicators that capture water resources system failure duration and extent of failure (i.e. failure magnitude) from a social and environmental perspective were developed. These performance indicators enabled a comprehensive assessment of a number of performance enhancing interventions, which resulted in the identification of a set of intervention strategies that showed potential to improve reliability, resilience and sustainability in the case studies examined. Finally, a multi-criteria decision analysis supported trade-off decision making when the reliability, resilience and sustainability indicators were considered in combination.

Two case studies were considered in this research: Kingston and St. Andrew in Jamaica and Anyplace in the UK. The Kingston and St. Andrew case study represents the main data-sparse case study where many assumptions were introduced to fill data gaps. The intervention strategy that showed great potential to improve reliability, resilience and sustainability identified from Kingston and St. Andrew water resources assessment was the 'Site A-east' desalination scheme. To ameliorate uncertainty and lack of confidence associated with results, a methodology was developed that transformed a key proportion of the Anyplace water resources system from a data-rich environment to a data-sparse environment. The Anyplace water resources system was then assessed in a data-sparse environment and the performance trade-offs of the intervention strategies were analysed using four multi-criteria decision analysis (MCDA) weighting combinations. The MCDA facilitated a

robust comparison of the interventions' performances in the data-rich and data-sparse case studies. Comparisons showed consistency in the performances of the interventions across data-rich and data-sparse hydrological conditions and serve to demonstrate to decision makers a novel approach to addressing uncertainty when many assumptions have been introduced in the water resources management process due to data sparsity.

Acknowledgements

I would like to express my sincere gratitude to my supervisor, Professor David Butler for the invaluable advice and unwavering support that he has given during my PhD research. I also extend my appreciation to my second supervisor, Dr Sarah Ward for her words of guidance and optimism. Thank you both for believing in me and giving me confidence to focus the goal.

Special recognition and thanks goes to the UK Commonwealth Scholarship Commission (CSC) for funding my doctoral research. Additionally, I extend recognition to Oxford Scientific Software and DHI Group for providing me with the software utilised in this thesis. Moreover, I express my sincere gratitude to the Water Resources Authority in Jamaica, as they provided me with invaluable support and assistance during my field visit to Kingston. In addition, I extend my gratitude to my Safe & SuRe research group colleagues for the insightful discussions held these past years.

Lastly, I extend, my sincere gratitude and thanks to my mother Janet, and my family in Barbados and London for keeping me focused and providing support during challenging periods. I would also like to express many thanks to all my friends whom provided me with their moral support.

List of Tables

List of Figures

Notations

Barrel/d	Barrels per day
CO_2	Carbon Dioxide
GL	Gigalitres
kg	Kilograms Carbon Dioxide
KL	Kiloliters
Km	Kilometers
kWh	Kilowatt hour
L/d/household	Litres per day per household
L/hh/d	Litre per household per day
Lpcd	Litres per capita per day
m^3/day	Meters cube per day
m^3/quarter/household	Metres cubes per quarter per household
ML	Megalitres

Abbreviations

AHP	Analytical Hierarchy Process
AWBM	Australian Water Balance Model
CCCCC	Caribbean Community Climate Change Centre
CEFR	Continuous Estimation of River Flows
DM	Demand Management
DO	Deployable Output
EA	Environment Agency
ECHAM5	European Centre Hamburg Model 5
EPSRC	Established Career Research Fellowship Project
ESBD	Economics of Balancing Supply and Demand
FFH	Future Flow Hydrology
GCM	General Circulation Model
HadRCM	Hadley Centre Regional Climate Model
ICWE	International Conference on Water and the Environment
IHACRES	Identification of unit Hydrographs and Component flows from Rainfall, Evaporation and Streamflow data
IRAS	Inter-active River Aquifer System
IGDT	Info-Gap Decision Theory
IWRM	Integrated Water Resources Management
KSA	Kingston and St. Andrew
LTCD	Lower Thames Control Diagram

MCDA	Multi-criteria Decision Analysis
MOEA	Multi-objective evolutionary algorithm
NEPA	National Environment & Planning Agency
NRW	Non-Revenue Water
NWC	National Water Commission
NWI	National Water Initiative
ONS	Office for National Statistics
PET	Potential Evapotranspiration
RCM	Regional Climate Model
RDM	Robust Decision Making
RIBASIM	RIver BAsin SIMulation
RRL	Rainfall-runoff Library
RWSL	Rural Water Supply Limited
SMAR	Soil and Moisture Accounting model
SOP	Standard Operating Policy
SRES	Special Report on Emission Scenarios
TDS	Total Dissolved Solids
TUBs	Temporary use bans
UAE	United Arabs Emirates
UKCIP Programme	United Kingdom Climate Impacts
UKCP09	United Kingdom Climate Change Projections 2009
UNCED	United Nations Conference on Environment and Development
UTR	Upper Thames Reservoir

VBA	Visual Basic Application
WARGI-OPT	Water Resources Graphical User Interface - Optimisation
WARGI-SIM	Water Resources Graphical User Interface - Simulation
WFD	Water Framework Directive
WRA	Water Resources Authority
WRS	Water Resources System WRS
WRZ	Water Resources Zone WRZ
WSDs	Water saving devices (WSDs)
WEAP21	Water Evaluation and Planning 21
WTW	Water Treatment Works WTW

CHAPTER ONE

1. Introduction

1.1. Background and Motivation of Research

Water resources planning deals with planning for and managing of the natural ecosystem such as river basins, and man-made systems such as reservoirs and desalination plants (Palmer and Lundberg, 2004). In the United Kingdom, water companies prepare water resources management plans in consultation with local authorities to demonstrate how they intend to guarantee a high level of supply security over the long-term. In Australia, the extreme hydrological conditions from the millennium drought resulted in the formation of the National Water Initiative (NWI) in 2004, to address future water availability in river and wetland health (Norris *et al*., 2007); and from the beginning of the 21st Century, the majority of the European States integrated the Water Framework Directive (WFD) with their water management policies (Ziolkowska and Ziolkowski, 2016).

In some developing regions, such as Egypt (SSR, 2016), the Ministry of Planning introduced a planning cycle similar to that practiced in the United Kingdom where the country's water resources management plans are prepared in five-year planning cycles (Simonovic *et al*., 1997), and is still the benchmark for water resources planning in Egypt to date (Republic of Egypt, 2005). In contrast, in the Caribbean which is also classified as a developing region (SSR, 2016), a water resources planning structure similar to those practised in the United Kingdom and Egypt, remains a challenge to implement. In the broader context, adequate management in the Caribbean is lacking (Farrell *et al*., 2007), which has been attributed to the lack of investments in instrumentation and inadequate maintenance of existing facilities (Boyce, 2011), and also the negative impacts of tropical hurricanes (Fletcher-Paul *et al*., 2008). These factors have contributed to the level of data sparsity in most islands. In 1995, Hurricane Debbie destroyed several gauging stations in St. Lucia resulting in data gaps for large periods between the recording of streamflow measurements (Fletcher-Paul *et al*., 2008); and more recently, the passage of extreme hurricanes across the Caribbean that resulted in widespread destruction in Dominica and other islands (Phipps, 2017) could have implications for further data-sparse conditions with respect to the recording of streamflow measurements.

Data sparsity exists when limited or no hydrological data is available (Neal *et al*., 2012), and water resources research conducted in data-sparse regions have often utilised stochastic, regionalisation and rainfall-runoff models for estimating missing parameters such as precipitation and streamflow (van der Linden and Woo, 2003; Mirshahi, 2010; Pluntke *et al*., 2014).

1.2. Aims and Objectives

The aim of this work is to develop a water resources system assessment methodology incorporating aspects of data sparsity through its application to a region of Kingston, Jamaica using insights from a data-rich setting in the UK, in order to assess the reliability, resilience and sustainability of interventions proposed for implementation in the data-sparse Jamaican case study.

In order to accomplish this aim, the following objectives are defined:

Objective 1: Review literature on water resources management approaches.

Objective 2: Explore types of water resources models and software applications.

Objective 3: Explore intervention strategies used in water resources management and select a feasible suite of strategies for the management of water resources systems in a data-rich and a data-sparse setting.

Objective 4: Develop and model the implications of plausible future domestic demand and climate change scenarios representative of possible uncertainties which could affect the performance of the data-rich and the data-sparse water resources systems.

Objective 5: Develop a method to evaluate and prioritise the reliability, resilience and sustainability of intervention strategies for the case studies.

Objective 6: Develop a methodology to facilitate the transformation of a water resources system from a data-rich environment to a data-sparse environment in order to demonstrate the validity of strategy selection in a data-sparse setting.

Objective 7: Assess the intervention strategies generated under data-rich circumstances as compared with data-sparse.

1.3. Outline of Thesis

The above objectives have been accomplished across six chapters. The present chapter (Introduction) describes the background of the research project, the main aim and objectives to be achieved, and the thesis structure.

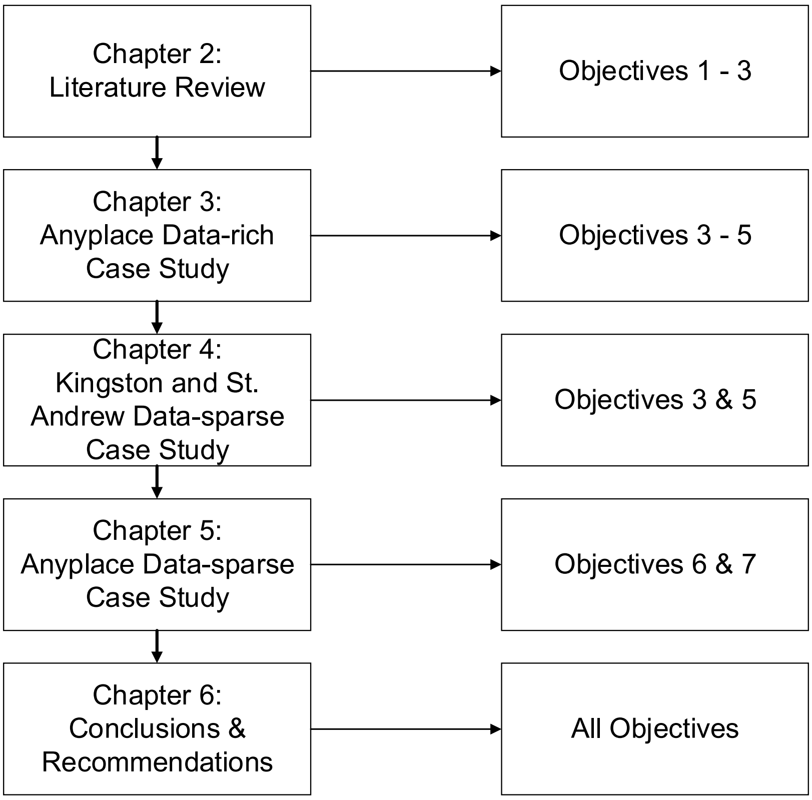


Figure 1.1 Illustration of the thesis structure indicating the chapters and the research objectives explored in each chapter

Chapter 2 presents a review of existing literature on water resources management approaches, water resources models and software applications, and explores types of intervention strategies utilised in water resources management.

Chapter 3 presents the description of the Anyplace data-rich case study and the methods utilised to accomplish the water resources assessment, including the concepts applied in developing future estimates of demand and supply uncertainty, the criteria underpinning the proposed resilience, reliability and sustainability performance indicators, and the prioritisation of the intervention strategies.

Chapter 4 presents the challenges and the justification for classifying the Kingston and St. Andrew case study as a data-sparse region, along with the assumptions and methods utilised to fill data gaps and accomplish the assessment of the water resources system.

Chapter 5, presents and explores a methodology that facilitates the transformation of the Anyplace data-rich water resources system to a data-sparse system. The comparisons of performance trade-offs under data-sparse and data-rich conditions are also examined which is facilitated by the multi-criteria approach developed in Objective 8.

In Chapter 6, summaries of the key findings and conclusions from each chapter are presented, along with discussions relating to the contribution and originality of the work. Recommendations for future research and the practical applicability of the methods utilised in the thesis to the water industry are also presented.

CHAPTER TWO

2. Literature Review

The literature chapter sets the scene by reviewing conventional approaches to water resources planning and management, which are presented in section 2.1, and non-conventional approaches, which are examined in section 2.2. A review of water resources management simulators is presented in section 2.3, and this is followed by a review exploring the types of demand and supply interventions utilized in water resources planning in section 2.4. Finally, in section 2.5, the literature review explores types of qualitative resilience frameworks utilized in water resources planning and management.

2.1. Conventional Approaches to Water Resources Management

Water resources management is characterised as the process of decision-making, allocation, regulation, monitoring and development of water resources (European Commission, 1999). Al Radif, (1999) points out an important aspect pertaining to the history of water management approaches. In the late 19th century towards the early 20th century when "*available water resources were adequate, and in some places in excess of the human needs for domestic, agriculture and industrial uses*" (Al Radif, 1999), the conventional water management approach utilised was supply driven. This approach is simplistic in nature because low levels of stress were placed on available supplies, and safe-provision of water services for domestic, agriculture and industrial uses, was the sole responsibility of institutions. Decisions were made on the short-term with no consideration for the possible consequences resulting from an increase in future demands or changes in the ecosystem. As Naiga *et al*., (2012) points out: "*two areas of concern emerged under the supply-driven approach*" (Naiga *et al*., 2012). These are a) coping with the demands of a rapidly growing society and, b) ensuring sustainability. Regarding the former, one type of intervention employed by decision makers to maximise supplies involved the diversion of watercourses (Al Radif, 1999), however, this practice has adverse consequences and is exampled by what could be considered one of the greatest water mismanagement catastrophes, the degradation of the Aral Sea (Micklin, 1988).

To address growing concerns, the International Conference on Water and the Environment (ICWE) was convened in Dublin in 1992. The ICWE called for new approaches to focus on the development and the assessment of existing resources (ICWE, 1992). The talking points from the ICWE led to the development of reforms at the United Nations Conference on Environment and Development (UNCED) in Rio de Janeiro, 1992, and this resulted in seven programme areas being proposed for water resources management. They are the (United Nations, 1993):

- Integrated water resources development and management;
- Water resources assessment;
- Protection of water resources, water quality, and aquatic ecosystems;
- Drinking water supply and sanitation;
- Water and sustainable urban development;
- Water for sustainable food production and rural development, and,
- Impact of climate change on water resources.

The seven programme areas outlined above have since gained international recognition and could be considered as the pillars on which the Integrated Water Resources Management (IWRM) process has been built. IWRM is considered as "*a process which promotes the coordinated development and management of water, land, and related resources, in order to maximize the resultant economic and social welfare in an equitable manner without compromising the sustainability of vital ecosystems*" (GWP, 2000). The term 'integrated' calls for a coordinated effort at all levels across the water industry (i.e. watershed management, water quality, water supply management, wastewater management, etc.), acknowledging that any improvements or any negative consequences affecting any of the integrated sectors could significantly influence what happens in the other sectors (Matondo, 2002). An

IWRM approach would better facilitate the planning and development of intervention strategies especially in water-stressed developing countries such as Jamaica (WRI, 2013).

Several academics have questioned the IWRM approach including Jønch-Clausen, (2004) who refers to it as a "*one size fits all*" approach, and argues that successful policies and strategies introduced in one region may not necessarily guarantee success in another region. Funke, *et al*., (2007) also agrees with Jønch-Clausen, (2004) and added that while it may be easier to achieve the principles outlined in the IWRM guidance in European states, in lesser developed regions such as Africa the situation tends to be more complicated due to the "*lack of familiarity with technical terminology, poverty and poor democratic governance*". Biswas, (2004) also questioned the governance policies of IWRM, advocating for more clarity regarding the concept's practical applicability. Biswas, (2004) also calls for an adaptable framework to accommodate different situations in developing countries. Simonovic *et al*., (1997) suggests that the criteria set out for water resources planning in developing countries should be different to those utilized in developed countries. Moreover, Simonovic *et al*., (1997) recommends that the revisited criteria should be developed to address prevailing constraints on the physical, financial and human resources in order to efficiently allocate sparse or limited resources.

Despite the concerns underpinning IWRM, the need to consider water resources management in the context of a broader landscape is evident. Referring to the UN WATER, (2012), evidenced studies show a successful uptake of the seven programme areas in 80% of 130 countries. Nevertheless, to avoid a repetition of past consequences which led to the ICWE and the UNCED conferences, there is a clear need to update existing guidelines and develop new innovative strategies to cope with the dynamics of diverse modern societies and the changing climate. In response, recent studies in the academic arena have captured the water resource planning and management processes aided by water resources simulation model applications using non-conventional approaches.

2.2. Non-conventional Approaches to Water Resources Planning and Management

Crow-Miller *et al*., (2017) argues against traditional demand management approaches which consist of supply augmentation projects such as the construction of pipelines, desalination, and reservoir capacity increase projects. Welsh and Endter-Wada, (2017) argue that new supply augmentation strategies could result in communities enduring the environmental and social risks associated with those types of water infrastructure projects. They also argue that new projects could result in higher water costs for water infrastructure that could potentially result in being unreliable, as water resources are being threatened by multiple unknowns resulting from climate change, an increased frequency of droughts, and population growth. Lach *et al*., (2004) also advocate against built infrastructure projects and highlight that "*water managers work to avoid criticism by routinizing the uncertain through the construction of infrastructure, agreements, and other organizational processes*".

Some researchers have performed water resources assessments of built infrastructure to evaluate how they would respond to multiple unknowns resulting from climate change, an increased frequency of droughts, and population growth. In the United Kingdom, Matrosov, Woods and Harou, (2013) employed Robust Decision Making (RDM) and Info-Gap Decision Theory (IGDT) as approaches to planning intervention strategies under uncertainty for the Thames Basin. RDM characterizes the vulnerabilities of selected intervention strategies using a process known as 'scenario discovery' explored in detailed in the literature (Bryant and Lempert, 2010). Scenario discovery "*rigorously identifies sets of future conditions, or 'scenarios', where the system under the preferred plan would be under most stress*" (Matrosov *et al*., 2013). In IGDT, the robustness of a strategy is maximised given a set of performance requirements. An example of IGDT can be found in a study conducted on the Thames Basin by Matrosov *et al*., (2013), where 7 plausible supply options from the Revised Water Resources Management Plan (Thames Water, 2010) were selected, and plausible demands for the study were obtained from the Thames Water Utility Limited's stochastic water demand forecasting tool (Thames Water, 2010). The IGDT produced charts of system performance with robustness and opportuneness plots, where the system performance was illustrated for the combination of supply options tested and plausible demand. The RDM identified through 'scenario discovery' the combinations of supply options resulting in the most system vulnerabilities.

Moreover, Matrosov et al., (2013) evaluated two approaches for testing a suite of supply and demand management interventions for the Thames water resources zone. The first approach is the Economics of Balancing Supply and Demand (ESBD) which is "*typically implemented using optimisation models that minimise total economic costs of meeting future water demands given portfolios of supply and demand management (DM) options*" (Padula *et al*., 2010) as cited in Matrosov *et al*., (2013). The second approach considers RDM discussed above and identifies a 'robust' plan (not the optimal strategy) of strategies that perform well under the scenarios tested. Matrosov *et al*., (2013) revealed that both of the approaches (i.e. RDM and ESBD) resulted in the selection of different strategies and this is primarily due to the ESBD returning the optimal result and the RDM returning a plan of strategies. Later, for the same case study, Huskova *et al*., (2016) employed a multi-objective evolutionary algorithm (MOEA) to demonstrate the testing of a suite of supply and demand management interventions. "*MOEAs are heuristic global search algorithms that simulate the process of natural evolution*" (Huskova *et al*., 2016) and return a set of solutions that meet the criteria. The interventions were first evaluated using historical parameters and then utilizing the parameters of the Future Flow Hydrology. The comparison of the results revealed that 60% of strategies that were considered sufficient when evaluated for historical parameters, failed when evaluated for the Future Flows Hydrology parameters. The implications of the study further reinforce Milly and Julio, (2008)'s argument that historical conditions are insufficient for planning for unknowns in the future.

Hall *et al*., (2012) proposed a risked-based approach to water resources planning that incorporates the use of the UKCP09 climate change projections.

In their research, risk is defined in the context of the probability of failing to meet a level of service requirement, which is defined in terms of "*the target probabilities of water shortages that a water company seeks to ensure will not be exceeded*" (Hall *et al*., 2012). The UKCP09 climate projections are a set of "*perturbed physics ensemble (PPE) of simulations, downscaled using the regional climate model (RCM) HadRM3 at a resolution of 25km*" (Walsh and Blenkinsop, 2015). Projections from 12 other climate models are also incorporated to allow the sampling of structural modelling errors from a multi-model ensemble (Walsh and Blenkinsop, 2015). These ensembles are combined within a Bayesian statistical framework to produce the UKCP09 probabilistic projections. They argued that updating the current ESBD approach utilized by water companies to a water resources planning risked-based approach would facilitate a more rigorous approach to planning for uncertainty and allow for better resource allocation. Turner and Blackwell, (2014) demonstrated the risked-based approach on the West Cumbria water resources zone. They defined the level of service in terms of the risk of the probability of drawdown occurring below the threshold of interest (i.e. reservoir drawdown to the level 0.74m, which is defined as a Trigger 3 Apply for Statutory Drought Measures) (Turner and Blackwell, 2014). The authors conclude that the risked-based method could be useful to inform decision makers as to how plausible futures could impact on a water resources system, therefore allowing criteria such as the RDM to generate the most robust planning strategies.

2.3. Simulation Models used in Water Resources Management

Simulation models used in water resources management are those that "*quantitatively represent the response of a water resource system*" (Fawthrop, 1994) when evaluated for a set of hydrological parameters. The parameters introduced in the simulator could be of a historic nature or synthetic parameters generated for future scenario analysis. According to Loucks *et al*., (2005), key factors to consider in the selection of water resources models for research purposes include the research question, the characteristics of the water resources system, time and finances. Matrosov *et al*., (2011) characterizes simulation models commonly used in assessing water resources systems into two groups. They are ‘rule-based' or ‘optimisation-driven'. Rule based-models utilize computing programming languages such as Visual Basic, C++ or FORTRAN to define the operating rules for key components of the water resources system. Examples of rule-based models are Aquator (Oxford Scientific software, 2015), IRAS (Wurbs, 1997), RIBASIM (Omar, 2013; Deltares, 2015), and WARGI-SIM (Swayne *et al*, 2010).

Optimisation driven models seek to find the optimal solution or a set of optimal solutions that comply with an objective function(s) and model constraints. In the context of water resources management, this could relate to finding the optimal operational conditions or interventions strategies. An advantage of these types of simulation models is that all possible combinations of operating rules and intervention strategies can be tested in a relatively short space of time, whereas in the case of the rule-based models, the programmer is burdened with developing programming code for each possible scenario. In contrast, Matrosov *et al*., (2011) argue that the complexity of some optimisation models can be regarded as a pitfall. This, however, depends on the nature of the water resources system, as some operating "*rules may be difficult to represent using optimization and model results may not be easy to replicate in practice*" (Matrosov *et al*., 2011). Examples of optimisation-based models include AQUATOOL (Sulis and Sechi, 2013), MIKE BASIN (DHI, 2003), WEAP (WEAP, 2017), MODSIM (Swayne *et al*., 2010) Aquator (Oxford Scientific software, 2014b). Aquator is also categorized as an optimization-based model because the programmer has the option not to utilize programming scripts to introduce operating rules, and optimisation is possible in Aquator using the Aquasolver optimiser (This is a tool in Aquator consisting of a suite of optimisation algorithms). In the subsection 2.3.1, the model features of both ruled-based and optimisation driven models are examined in addition to the criteria which led to the selection of Aquator for research purposes in this thesis.

2.3.1. Model Features

Inter-active River Aquifer System (IRAS)

IRAS is a rule-based water resources management simulator used for analysing surface and groundwater resources (Wurbs, 1994). It was designed for catchment and groundwater simulations, and water quality assessments. Its structure is based on the principles of the mass balance, and key problems addressed are flow routing, seepage, evaporation, and information or design changes related to catchment or groundwater components. It is characterized as rule-based in the context that programming code is used to input/add operational rules or constraints to the water resources model. Analyses are performed for daily time-steps, and water resources components (demand regions, storage, catchments etc.) are represented in the model as nodes. According to French and Taylor, (1996), the IRAS model software is known for its flexibility for allowing programmers to incorporate model modifications and configure multiple threshold constraints. IRAS is used by a large volume of university researchers and in the water industry in the United Kingdom. Most recently, Matrosov *et al*., (2011) and Matrosov *et al*., (2013) utilized IRAS to test a suite of interventions for the Thames River basin study, and Borgomeo *et al*., (2014) utilized IRAS to demonstrate the practical applicability of the Risk-based approach to water resources management explored in subsection 2.2.

RIBASIM

RIBASIM (RIver BAsin SIMulation) is used in river basin management to assess the response of river basins for a wide range of hydrological conditions. It is rule-based in nature, as the programmer can define operating/planning scenarios where each scenario is characterized by a particular operating rule and/or water supply projection (Swayne *et al*., 2010). Simulations are conducted for daily time-steps, and hydrologic methods such as the "*Manning formula, Flow-level relation, 2-layered multi-segmented Muskingum formula, Puls method and Laurenson non-linear lag and route method*" (Deltares, 2015) are key features of RIBASIM. Johnston and

Smakhtin, (2014) utilized RIBASIM to fill gaps not reported in previous studies related to the hydrological modelling of the Nile River.

WARGI-SIM

WARGI-SIM is a rule-based simulation software that analyses the interrelationships between demands and supply (Swayne *et al*., 2010). According to Sulis and Sechi, (2013), this simulator is simplistic in nature as non-experts are able to understand the complexities of water resources system problems simulated in WARGI-SIM. Water resources components are represented schematically as nodes, and users can "*define preferences for each combination of possible transfers between the resource and demand node*" (Sechi and Sulis, 2009). A key characteristic of this model is that simulations can be conducted for a variety of time-steps ranging from hourly to seasonal. In the literature, WARGI-SIM has been used in the drought study found in Sechi and Sulis, (2009) to assess a water resources system's capability to meet demands during water shortage periods in the Water Basin-Plan case study in Sardinia Italy. In that study, mitigation strategies are optimised by linking the WARGI-SIM simulator with the deterministic optimisation module (WARGI-OPT). This suggests that if optimisation is preferred, users would have to link the WARGI-SIM to an optimizer module.

Water Evaluation and Planning 21 (WEAP 21)

WEAP21 is a user-friendly simulator that takes an integrated approach to water resources planning (WEAP, 2017). Recall the IWRM concept explored above which calls for a coordinated effort at all levels across the water industry. The WEAP model has been designed to facilitate a large proportion of the dimensions of the IWRM process including stakeholder participation and policy scenarios. It also considers plausible uncertainties associated with climate change as illustrated in Table 2.1. WEAP consists of a linear programming allocation algorithm that allows researchers and water managers to prioritize demand and supply preferences (Yates *et al*., 2005). Lastly, WEAP 21 can be linked with other software for comprehensive modelling of water quality as highlighted in Table 2.1.

Table 2.1 Summary of the key features of the WEAP21 as cited in WEAP, (2017)

Integrated Approach	Unique approach for conducting integrated water resources planning assessments
Stakeholder Process	Transparent structure facilitates engagement of diverse stakeholders in an open process
Water Balance	A database maintains water demand and supply information to drive mass balance model on a link-node architecture
Simulation Based	Calculates water demand, supply, runoff, infiltration, crop requirements, flows, and storage, and pollution generation, treatment, discharge and instream water quality under varying hydrologic and policy scenarios
Policy Scenarios	Evaluates a full range of water development and management options, and takes account of multiple and competing uses of water systems
User-friendly Interface	Graphical drag-and-drop GIS-based interface with flexible model output as maps, charts and tables
Model Integration	Dynamic links to other models and software, such as QUAL2K, MODFLOW, MODPATH, PEST, Excel and GAMS

MODSIM

MODSIM is a simulation model primarily used in river basin management. It is considered as rule-based because users can configure operating rules and constraints. For example, a user can "*assign relative priorities for meeting diversion, instream flow, and storage targets, as well as lower and upper bounds on flows and storage*" (U.S. Army Corps of Engineers, 1998). Additionally, the Lagrangian relaxation algorithm RELAX-IV found in (Bertsekas and Tseng, 1994) can be employed to find optimal operating rules for procedures such as reservoir regulation and demand allocation (U.S. Army Corps of Engineers, 1998).

MIKE BASIN DHI

MIKE BASIN is a water resources simulator developed by DHI Software that uses GIS information to facilitate a water resources assessment (Ireson *et al*., 2006). In MIKE BASIN, river systems are schematically represented as branches and the nodes represent point locations which could be a diversion, confluences, and abstractions. According to Ireson et al., (2006) calibration is not possible in MIKE BASIN, however, calibration is conducted utilizing other modules from the MIKE DHI library. For instance, river flows can be calibrated in the NAM Rainfall-Runoff model explored in Chapter 3. Other types of water resources assessments conducted in MIKE BASIN include water quality assessments.

AQUATOOL

AQUATOOL is a decision support system (DSS) designed to assist water managers and engineers with the decision-making process in the early stage of water resources projects (Andreu *et al*., 1996). It consists of a library of modules which includes SimWin that simulates the movement of water in the water resources system, SimRisk which is used to conduct an assessment of drought conditions, OptiWin which allows users to utilize the optimisation, and AquiVal to simulate groundwater (Sulis and Sechi, 2013). Similar to the rule-based models described above, researchers can also utilize C + +, Visual Basic and FORTRAN for modifying operating rules and in scenario testing.

Aquator

Aquator is a commercial water resources management software that allows for both simple and complex water resources systems to be modelled (Oxford Scientific software, 2014). Water resources infrastructure such as hydropower facilities, catchments, reservoirs and service reservoirs, water treatment facilities, gauging stations, pumping stations, river channels and boreholes (groundwater) can be modelled in Aquator. In addition, key actions such as diversion of flows, the confluence of streams and abstraction licenses can also be modelled. When compared to the other water resources models explored in this subsection, Aquator presents the advantage of being flexible because it allows for water managers to address a broad range of dimensions in water resources management. Aquator uses daily time-steps to simulate the operation of a water resources system. A key characteristic of Aquator is that it can be categorized as both a 'rule-based' model and as an optimisation-based model. In the context of rule-based, Visual Basic Application (VBA) scripting is accessible in a Microsoft Windows Environment, and this allows for users to test a wide range of operational protocols, modify abstraction licenses, or facilitate the testing of scenarios. Moreover, users could utilize the Aquasolver Algorithm. Depending on the nature of the problem, the objective functions of the Aquasolver algorithm can be configured to find the optimum solution/route that allows for supplies to meet demands, while enforcing all operating rules and constraints, in addition to minimising operational costs and the impact on water resources when the water availability is scarce (Oxford Scientific software, 2014a).

Aquator is used extensively in the water industry and in the research domain in the United Kingdom. Water companies such as Thames Water (Thames Water, 2014) and (Severn Trent, 2014) have used Aquator to assist in the preparation of Water Resources Management Plans, and by academics (Arena *et al*., 2014; Arena *et al*., 2015) to determine the best low cost water resources alternatives for the region of Apulia, Italy; and Turner and Blackwell, (2014) demonstrated the application of the Risk-based management approach in the Ennerdale case study. Aquator is utilized in this thesis to facilitate the testing of the intervention strategies formulated in Chapters 3 and 4, and the performance outputs from the model are used to demonstrate the applicability of the Social and Environmental resilience indicators, which are based on the Safe and SuRe resilience definition discussed in Chapter 3. Based on the review of water resources models explored above, the criteria underpinning the selection of Aquator for use in the thesis is as follows:

a) Aquator allows for modelling on a catchment scale, the groundwater scale, and the river basin scale.
b) A broad range of water resources problems/scenarios can be addressed through the modelling of different types of water resources infrastructures and protocols (i.e. abstraction licenses).
c) In the absence of GIS information, water resources systems can be built and modelled in Aquator.
d) The Microsoft Environment facilitating the use of VBA is easily accessible in Aquator’s User Interface, and users do not have to resort to the use of external software for scripting purposes.

Table 2.2. Summary of simulation models used in water resources management

Model	Features
Inter-active River Aquifer System (IRAS)	A rule-based water resources management simulator. Designed for catchment and groundwater simulations, and water quality assessments. Flexibility - water resources models can be modified to incorporate multiple threshold constraints using programming code.
RIBASIM	A rule-based simulator. Primarily used in river basin management to assess the response of river basins for a wide range of hydrological conditions. Analyses performed at daily time-steps. Flexibility - planning scenarios can be tested using programming scripts where each planning scenario is characterized by a particular operating rule and/or water supply projection.
WARGI-SIM	A rule-based simulator. Primarily utilised to analyse the interrelationships between demands and supply. Analyses can be performed at, hourly time-steps, daily time-steps, monthly time-steps and seasonal time-steps.
Water Evaluation and Planning 21 (WEAP 21)	Adopts an integrated approach to water resources planning. The WEAP model has been designed to facilitate a large proportion of the dimensions of the IWRM process including stakeholder participation and policy scenarios. WEAP consists of a linear programming allocation algorithm that allows researchers and water managers to prioritize demand and supply preferences. Can be linked with software applications that focus on water quality modelling.
MODSIM	A rule-based simulator. Primarily used in river basin management. The Lagrangian relaxation algorithm RELAX-IV can be employed to find optimal operating rules for procedures such as reservoir regulation and demand allocation.

Model	Features
MIKE BASIN DHI	A water resources simulator that uses GIS information to facilitate a water resources assessment. For model calibration, the water resources simulator is linked with other modules designed for calibration found in the MIKE DHI library.
AQUATOOL	AQUATOOL is a decision support system designed to assist water managers and engineers with the decision-making process. It is linked with other modules such as SimWin that simulates the movement of water in the water resources system. Programming languages such as C + +, Visual Basic and FORTRAN for modifying operating rules, and for scenario testing.
Aquator	A commercial water resources management software that allows for both simple and complex water resources systems to be modelled. Various water resources infrastructure can be modelled such as hydropower facilities, catchments, reservoirs and service reservoirs, water treatment facilities, gauging stations, pumping stations, river channels and boreholes (groundwater). Current and proposed abstraction licenses can be modelled and evaluated. Daily time-steps are utilised to simulate the operation of a water resources system. Aquator can be categorized as both a ‘rule-based’ model and as an optimisation-based model. The programming language Visual Basic Application can be utilised to test a wide range of operational protocols, modify abstraction licenses, or facilitate the testing of scenarios. The optimisation algorithm, the Aquasolver algorithm can also be used to find the optimum solution/route that allows for supplies to meet demands, while enforcing all operating rules and constraints, in addition to minimising operational costs.

2.4. Intervention Strategies Utilised in Water Resources Planning and Management

Climate change extremes are becoming more common (Parmesan *et al*., 2003), and planning for uncertainty poses a great challenge for decision makers. Milly and Julio, (2008) argue that a pitfall of utilizing conventional water resources management approaches relates to the planning of intervention strategies (in response to the changing climatic conditions and population changes) using historical parameters and assuming that those previously experienced events are adequate enough for making decisions regarding planning strategies. An important topic captured in this research relates to intervention strategies, where emphasis is placed on utilizing a water management approach that allows for the implementation of best management practices now and in the future. In the context of water resources management, best practices could be considered as an action or a set of actions that aim to produce positive results or improve upon existing results when striving to meet special goals (e.g. reduction in the water demand; compliance with operational standards). According to Bazza, (2002), an effective water management plan should consider best management practices that capture long-term and short-term goals.

In the World Commission on Dams, (2000) report, the negative environmental and social consequences of dams and reservoirs are outlined. This report points out that despite the benefits gained from these types of interventions in the past (in terms of increasing supply availability), in China and India approximately 4 million persons have been displaced annually between 1986 and 1993 as a result of dams and reservoir construction (World Commission on Dams, 2000). Crow-Miller *et al*., (2017) suggests that in recent years, there has been a resurgence in the construction of built infrastructure which includes dams, and desalination plants in Asia, Africa, Europe and South America. Moreover, Perry and Praskievicz, (2017) identifies a trend where the uptake of built infrastructure is reoccurring in both developing and developed countries, and in some cases, decommissioned infrastructures and existing dams are being reused to increase supply storage (e.g. dam capacity augmentation), as opposed to also acknowledging that the rising water demand must be ultimately constrained (Crow-Miller *et al*., 2017). Recent investments by the World Bank in developing Asia have focused on dams, as they are regarded as ‘climate buffers' (Crow-Miller *et al*., 2017); and, according to (Bosshard, 2013) the World Bank "*still finds it easier to spend billions of dollars on mega-projects than to support the small, decentralised projects that are most effective at expanding energy access in rural areas*". Crow-Miller *et al*., (2017) suggest that the backward shift towards 20th-century interventions are not sufficient to cope with the emerging challenges of human growth and climate uncertainty, and recommends that supply-side and demand-side interventions should be used together.

2.4.1. Demand-side Intervention Strategies

Smart Water Meters

According to Sønderlund *et al*., (2014), smart water meters are electronic devices that record consumption in real-time, or near real-time, and communicates this information to the service provider. This strategy is advantageous over the traditional water meter (readings are taken monthly/quarterly) as the service provider can identify irregularities in customer usages and communicate this information to the customers to determine if leakages are occurring. Additionally, consumption information is more accurate, and this leads to more accurate billing. Research conducted on the contribution of smart water meters can be found in Davies *et al*., (2014) where it was reported that the average monthly savings of 1.82 kL/month were obtained for 82 households in a pilot study in Sydney; in the United Kingdom, savings of 8.1 L/d/household was observed in a United Utilities pilot study (Waterwise, 2011b) and in Canada, a study by Abbotsford Mission Water & Sewer Services, (2012) revealed savings of $49m^3$ per year from the households in that region.

Financial Incentive Programs

According to Colorado Water Conservation Board, (2010), retrofitting of toilets, washing machines, showerheads, and faucets in single family residences has resulted in the reduction of the water demand by approximately 30 percent. In Seattle, the local utility company offers rebates (up to $100 USD) to residents who install water-efficient devices (or water saving devices) (Saving Water Partnership, 2017). Tsai *et al*., (2011) assessed the effectiveness of rebate programs in reducing the water demand and reported significant positive water savings averaging between 3.94 and 5.38 m^3/quarter/household for a water resources zone in Ipswich, Massachusetts. Moreover, financial incentives implemented with tariff blocks resulted in a 27% water consumption observed over a 12-year period (1996 – 2008) in Zaragoza, Spain (Kayaga and Smout, 2014).

Information and Educational Programmes

Government and privately-operated water utilities have embarked on social media marketing campaigns, schools, and general public outreach programmes to promote a greater awareness of the need to be more water efficient and adopt conservation practices. However, a large number of authors argue that the effectiveness of these programmes is often short-term. Syme *et al*., (2000) points out that most often, campaigns are launched during dry periods and consumption patterns are likely to return to normal levels after the dry period. Moreover, Roibás *et al*., (2007) highlight that awareness campaigns can be most effective when introduced with other interventions. March, Hernández, and Saurí, (2015) suggest more stringent awareness campaigns should be adopted, and makes several recommendations which include the promotion of financial incentives and rebate programmes to continue to achieve further reductions in the period following the dry event.

Regulatory Actions: Temporary Use Bans

The Temporary Use Bans Act 2010 (UK Government, 2010) has been introduced as a water conservation strategy. It has been utilized by Affinity Water and Thames Water (Affinity Water, 2012; AP, 2012) in response to low groundwater levels preceding two dry winters. According to the Temporary Use Bans Act 2010, restrictions are imposed on actions such as water gardening, car washing, and filling pools. The full list of restricted actions can be accessed in the literature Temporary Use Bans Act 2010 (UK Government, 2010). Moreover, an educational awareness campaign by Scottish Water highlighted that up to 5% savings can be gained from the implementation of this measure (Scottish Water, 2005). Thames Water, (2016) considers *Temporary Use Bans* as a restriction to implement if the storage volume falls below a certain level, and they characterized this as a Level 3 restriction to implement for conserving storage.

2.4.2. Supply Augmentation Interventions

Pipelines and National Grid Interventions

There has been considerable debate regarding the merits of built infrastructure as water resources intervention strategies. Crow-Miller *et al.*, (2017); Welsh and Endter-Wada, (2017) have argued that a trend is on the rise associated with the return of built infrastructural projects as solutions to water challenges; and they have also criticised organisations such as The World Bank for investing in such projects and not in more demand management projects.

In the United Kingdom, the idea of a national grid connecting water resources have been discussed (Water UK, 2012; Waterwise, 2017). According to Desai *et al.*, (2005), the advantages of such projects include: a) minimising the water imbalance across water resources zones, b) better flood control, c) facilitating better management of water resources during droughts, d) reduce stress placed on the groundwater table and, e) improvements in the management of the domestic supplies. Moreover, the disadvantages extend to the: *i*) adverse environmental impacts, *ii*) challenges associated with land acquisition, *iii*) high levels of evaporation loss if canals are utilized and, *iv*) high maintenance costs. In 1984, in Libya, the largest water transfer project 'The Great Man-Made River' was conceived to transport approximately 5-6 million m³/day to the northern cities. Water was abstracted from 500 wells approximately 500 m deep and transported along 4000 km of pipelines (Abdelrhem *et al*., 2008; FAO, 2016). In Libya, there are no permanent rivers in Libya, only ephemeral rivers or wadis (FAO, 2016), and Libya's population could conceivably double in the next 25 years (Abdudayem and Scott, 2014). These types of challenges have resorted to the reliance on this type of intervention strategy which incurs high costs associated with pumping water from great distances and great depths, and also in the degradation of the quality of water (Abdudayem and Scott, 2014). On a smaller scale in the United Kingdom, water companies perform water trading and inter-company transfers. According to Deloitte LLP, (2015), environmental and social benefits can be gained from water trading. In the context of the environmental benefits, the movement of water from a region where it is abundant to a region where supplies are low would relieve the stress place of the environment, and in the social context, it would provide the possibility for water companies to be better able to meet customer demands.

Rainwater Harvesting

Rainwater harvesting systems capture runoff from rooftops, and they aim to reduce the stress placed on potable water supplies by replacing its usage with rainwater for toilet flushing and outdoor purposes (Tsai *et al*., 2011). An example of their value was demonstrated in response to the negative effects of the millennium drought (1997 – 2010) as rainwater harvesting was embraced by a large number of residents, industries, and schools in Melbourne. According to Low *et al*., (2015), in January 2003 the Victorian Government introduced a Water Smart Rebate Scheme, where rebates between $800 - $1500 AUD were issued for rainwater tanks, dual flush toilets, permanent greywater systems, hot water recirculators, and efficient showerheads. These strategies resulted in the reduction of the potable demand by 0.35GL/year over an eight-year period (Low *et al*., 2015), and in particular, rainwater harvesting contributed to water savings of up to 15% per household (Low *et al*., 2015). Overall by 2010, approximately a quarter (26%) of households in Australia had installed rainwater tanks, and this is a significant increase from the 2007 figure of 15% (Beatty and McLindin, 2012). Furthermore, Talebpour *et al*., (2015) provided a valuable contribution in studying the energy consumed by the pumps of rainwater harvesting systems (as they delivered water to the intended end-use) in nineteen households in South-east Queensland, Australia. One important aspect the study revealed was the link between pump energy and flow rate behaviour. When the results were analysed on a household scale (i.e. sets of individual household results and not collectively), they found that the overall efficiency of rainwater harvesting system comprising of single speed pumps was poor in comparison to other types of pumps used. This extends to larger quantities of energy expended by those pumps in delivering water to the intended end-use.

A comparative study conducted by (Amos *et al*., 2017) in a developed country (Australia) and a developing country (Kenya) revealed that Kenyan authorities do not provide rebates as an incentive to encourage rainwater harvesting uptake and rely on customer perspective to ensure the security of supplies. Moreover, they argued that similar economic benefits achieved in Australia could be achieved in developing countries such as Kenya where the uptake of rainwater harvesting is primarily mainstream in the slums; and, this "*demonstrates…social acceptability to some extent*" (Amos *et al*., 2017).

In contrast, in the developing Caribbean islands for example in Jamaica and Barbados which have been categorized as water stressed (WRI, 2013), rainwater harvesting is not mainstream. The island of Barbados boasts of approximately 2081 mm/year (based on historical data 1901 – 2015), and Jamaica receives 2117 mm/year (based on historical data 1901 – 2015) (The World Bank, 2017a). Small progress has been made in Barbados, as it has become mandatory that residents construct a water storage tank (concrete and sub-terrain) if the area of their roofs exceeds 3000 sq feet (OAS, 2005). However, no policies or incentives have been proposed to address other types of residential households. More specifically in Kingston, only 0.1% of the total annual rainfall was reported as harvested in 1990 (UWA, 1990). The poor uptake in Kingston could primarily be attributed to the lack of widespread knowledge associated with the benefits of such technology. Surprisingly, a walking survey of 220 houses conducted by the researcher in the Mona Heights community revealed that 43% of the households contained at least a single water tank (Figures B1 to B4 provided in the Appendix B as supplementary information). Additionally, all of the residents surveyed revealed that the tanks are used to store potable water from the mains for use during periods of water shortages. Furthermore, informal interviews conducted with Kingston residents by the researcher revealed that residents prefer for example the construction of a new reservoir as opposed to the uptake of rainwater harvesting (refer to Table B1 in Appendix B).

Greywater Recycling

As mentioned above, the Victorian Government introduced a Water Smart Rebate Scheme which included rebates for the uptake of greywater technology primarily for gardening use (Low *et al*., 2015). Greywater constitutes "*low polluted wastewater from bathtubs, showers, hand washing basins and washing machines*" (Sadashiva *et al*., 2016) as cited in Kraume *et al*., (2010). Jamrah *et al*., (2008) conducted a water resources assessment to investigate the potential for greywater use in Oman and its acceptability. The results revealed that approximately 150 *Lpcd* could be generated from greywater use (80 to 83% of the total freshwater consumption), and the acceptance for this technology was 76% for gardening, 53% for car washing and 66% for toilet flushing. Several issues of concern emerging from that study relate to the lack of consideration in addressing health concerns associated with this type of intervention, and the sample size (five university staff residences) can be argued to be relatively small. Referring to Table 2.3, the advantages and disadvantages of greywater usage are presented, and one of the key disadvantages refers to the use of greywater on plants. In the studies Ryan *et al*., (2009) and Al-Mashaqbeh *et al*., (2012), the use of greywater for watering/irrigation of gardens are encouraged, however, the potential health risks associated with this were not explored in those studies. Domínguez *et al*., (2017) acknowledges that greywater poses a risk to public health especially on a neighbourhood scale system, and excludes the use of greywater sourced from the kitchen basin for irrigation.

Table 2.3 Advantages and disadvantages of greywater reuse adapted from (Sadashiva et al., 2016)

Advantages
Reduction of overall water demand
Reduction of Organic and hydraulic loadings on the municipal wastewater system
Reduction in water bills
Replenishment of groundwater which contributes to a healthy water cycle
Protection of aquatic ecosystems due to decreased diversions of freshwater
Disadvantages
Cannot be stored for more than 24 hrs (since nutrients break down and cause bad odour)
Biodegradable soaps and detergents can also present a problem over a period of time when greywater is used for irrigation
Health standards of the water and quality concerns
Contains fats, oils, grease, hair, lint, soaps, cleansers, fabric softeners, and other chemicals that are harmful to plants

Yu *et al*., (2015) evaluates the potential costs (installation and operational) associated with greywaters systems in Los Angeles and revealed that a small system designed for a single family ranged between $6000 - $13000 USD.

Moreover, in Australia where mandatory maintenance is undertaken by a qualified professional, annual costs are between $200 - $900 USD. Yu *et al*., (2015) suggest that if Los Angeles were to adopt a similar policy, the cost of greywater systems would likely be driven up unless rebates programs are implemented. The cost of the system could be considered a key disadvantage of this technology and Yu *et al*., (2015) considers it a factor contributing to the technology's low uptake. Some researchers have performed analysis on the economic benefits of greywater systems, and in Munoz *et al*., (2016) it was revealed that an average payback period of 10 years is likely for a system valued at $4000 USD for a single family in California. However, Munoz *et al*., (2016) also point outs that water rates and the lifestyle of the occupants are factors that make it difficult to quantify a payback period. On the other hand, Mandal *et al*., (2010) assessed the economic feasibility of a greywater system installed in a residential block for a family of five in Nagpur, India, and the study revealed a payback period of 1.6 years and savings of up to 48% on potable water. The comparison of the results from the case studies suggests that payback could also be influenced by the current state of the country's development. García-Montoya *et al*., (2015) employed the use of a multi-objective optimisation algorithm to assess the potential of greywater and rainwater harvesting as interventions in Morelia, Mexico. A scenario testing approach was utilized (i.e. greywater use only; rainwater use only; both interventions together) and the results revealed that individually, greywater systems resulted in lower operating costs than that of the rainwater harvesting system. However, unsurprisingly for the scenario where both systems (i.e. rainwater harvesting and greywater systems) were tested together, the results showed that significant reductions can be obtained in the total fresh water consumption and fresh water pumping costs.

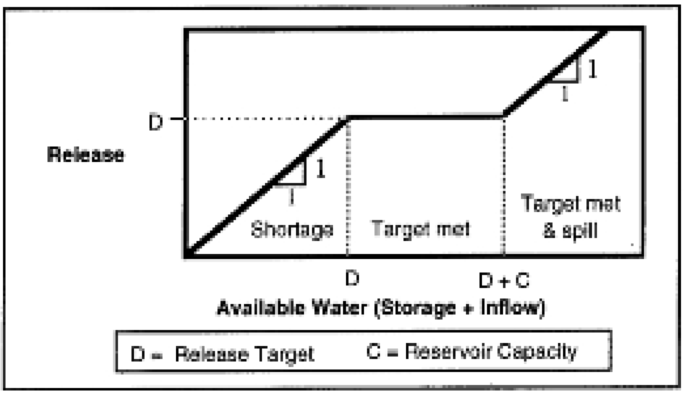
Desalination

In the Middle Eastern region, there is a shortage of potable water, and the Gulf States make up the leading region in the world that has resorted to the uptake of desalination technology (Shatat and Riffat, 2014). The Kingdom of Saudi Arabia is highly reliant on desalination technology and produces approximately 18% of the global desalinated water (Shatat and Riffat, 2014), and the United Arabs Emirates accounts for 22% (Murad *et al*., 2007). Ouda, (2014) performed an assessment of the potential of desalination technology to satisfy the demands forecasted in the year 2040 for three demand scenarios. "*The country is currently using about 1.5 million barrel/d for desalinated water production"* (Ouda, 2014). The results revealed that for the optimistic (lowest) demand scenario, double the current amount of crude oil would be required for the production of desalinated water. A clear implication of such a scenario is that this would place significant stress on the country's major source of revenue, which is crude oil. (Liu *et al*., 2015) studied the carbon footprint of the different types of desalination technology utilized in the UAE (i.e. multi-stage flash, multiple effect distillation, and reverse osmosis). The results revealed that the multi-stage flash method produces the largest footprint with 2.988 kg of CO_2 and the multiple effect distillation produced the lowest footprint 1.280 kg of CO_2. Heck *et al*., (2016) conducted research on public's awareness focusing on the impacts of desalination on the ecosystem. The indicators investigated: "*1) public awareness of the plant and brine discharge; (2) self-assessed level of knowledge about impacts from the plant; and (3) factual knowledge about brine discharge, including composition of the brine, location of discharge, and movement of brine in the ocean*" (Heck *et al*., 2016). The results revealed that approximately 94.1% were knowledgeable about desalination plants; however, 52% of the respondents were not aware that desalination plants discharge brine, and only 17% of those were aware that brine is discharged into the sea. (Ouda, 2014) describes public awareness on water resources related issues as 'sub-optimal' and calls for the implementation of intensive public awareness campaigns in Saudi Arabia.

Reservoir Hedging

Several researchers have employed the use of inflow forecast ensembles to inform decision makers on how to improve decision making in the management of reservoir systems. Yao and Georgakakos, (2001) utilized historical inflow records (1965 -1993) and two ensembles of Global Circulation Models (1993 – 2050): 1) assuming no CO_2 increase and 2) assuming a 1% CO_2 increase, the results revealed that planned strategies were sufficient to manage the conditions resulting from the climate change forecasts. Other studies which have utilized inflow forecast ensembles in reservoir management include Faber and Stedinger, (2001); Hamlet *et al*., (2002); Kim *et al*., (2007) and Georgakakos *et al*., (2012). Recently, Walsh and Blenkinsop, (2015) employed the use of the UKCP09 climate projections to obtain weather variables (i.e. precipitation and potential evapotranspiration) and utilised a rainfall transformation process to generate discharge for the 2050's period, to test the implementation of a new reservoir system of capacity 100 million m^3 for the Thames Basin in the United Kingdom. The results revealed that the implementation of this new reservoir would avoid resorting to the introduction of the level 4 restriction (i.e. severe water rationing). Moreover, Huskova *et al*., (2016) utilized the Future Flow Hydrology explored in greater detail in Subsection 3.2.3 to assess the performance of the Upper Thames Reservoir (UTR) in the Thames Basin. The results show that the UTR would improve the system design's robustness to the tested plausible future conditions.

In research studies where the option to build a new reservoir system is not possible, hedging policies can be utilized to modify the reservoir operating rules to cope with the conditions of the surrounding environment. The aim of reservoir hedging is to conserve the storage volume during dry periods or droughts (Zhao *et al*., 2011). The Standard Operating Policy (SOP) is illustrated in Figure 2.1, and according to Felfelani *et al*., (2013), when demand is high and the reservoir storage volume (in Figure 2.1 - existing storage + inflow) is low, under the SOP all of the reservoir's storage is utilised in an attempt to meet domestic demand (D). In contrast, if the available storage volume is more than sufficient to meet domestic demand (D), then spillage occurs. According to Rittima, (2009) this approach is insufficient for water resources planning as it does not guarantee (assuming no demand management measures are implemented) that water will be stored for usage in the future, and it also does not control reservoir spillage.



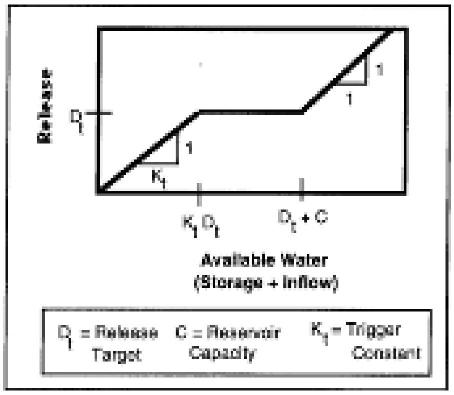


Figure 2.1 Illustration of the Standard Operating Policy (left) and One-Point Hedging (right) (Lund, 1996)

One-point hedging or the Trigger Hedging Value is also illustrated in Figure 2.1 and is an extension of the SOP. Graphically, a trigger value (K_t) is introduced along the SOP curve at a point on the target level of release. The value of K_t is less than 1 ($K_t < 1$), and according to Shih and ReVelle, (1994) the smaller the value of K_t, the greater the reduction in releases.

Similar to One-point hedging, the Two-point hedging connects to the SOP line at two points, P_1 and P_2. The trigger value defining the slope of the line from the origin to P_1, is less than 1 ($K_t < 1$), and the point P_2 intersects the target release line of the SOP from the period where hedging is no longer required (Lund, 1996). Beyond the point P_2, the SOP is reintroduced (Lund, 1996) as seen in Figure 2.2. Critique of this policy is expressed in Felfelani *et al.*, (2013) highlighting that releases are triggered more abruptly, despite allowing for the storage volume to be conserved over a longer duration. The following pseudo-code is used to mathematically express two-point hedging:

If ($St + I_t$) < P_1 then $R_t = S_t + I_t$
else if (S_t+I_t) < P_2 then
$R_t=[(D - P_1)/(P_2 - P_1)]*[S_{t-1}) + I_t]+P_1 - P_1*((D - P_1)/(P_2 - P_1))$
else if (S_t-1+ I_t) < (C+D) then $R_t = D_t$
else $R_t = S_t - 1 + I_t - C$

where, S_t is the storage at beginning of period t, I_t *is the* expected inflow for period t, C is the reservoir capacity, D_t is the demand for period t, and R_t is the release in the period t (Lund, 1996).

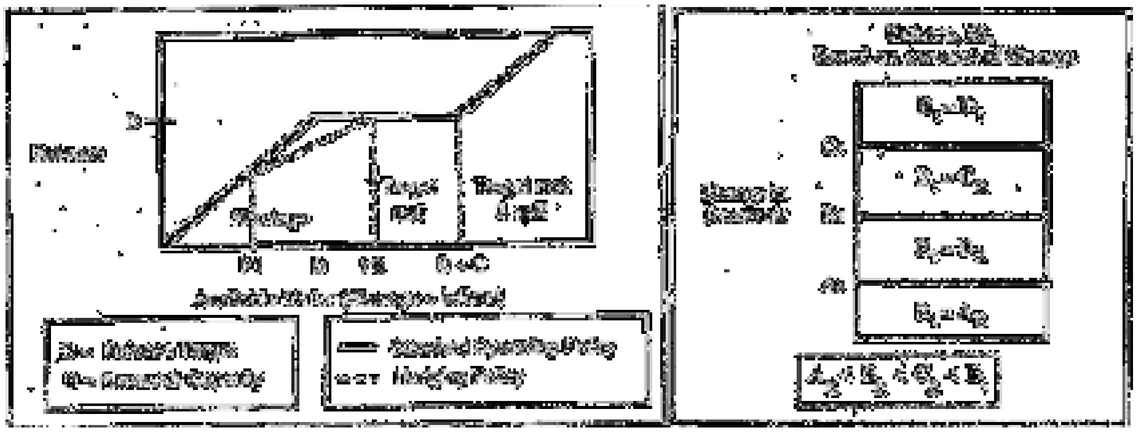


Figure 2.1 Illustration of Two-point Hedging (left) and Zonal Hedging (right)

Moreover, three-point hedging is an extension of the two-point hedging and under this concept, a third point between P_1 and P_2 is placed if the extent of the releases is considered too abrupt. This method addresses the critique discussed above for the two-point hedging. Zonal hedging is illustrated in Figure 2.2, and as seen in the illustration, the reservoir storage volume is segmented into different levels or zones. There are no set criteria associated with how the zones are fixed, however a typical zone could be segmented to facilitate flood control, emergency storage and dead storage (i.e. also considered as permanent storage) as exampled in Lund, (1996), and ideally, different release targets are set for each of the zones (Felfelani *et al.*, 2013).

Several researchers have extended the SOP concept to develop reservoir operating rules. Vudhivanich and Rittima, (2003) developed reservoir operating rules which are based on four levels of risk (0.05, 0.10, 0.20, 0.30) for the probability that water shortages and water spillages will occur. They compared the results to the SOP simulated for the same hydrological conditions. The results revealed that the probability-based policies were more effective in operation than the SOP in the context of quantifying spillage. For the water shortages, the probability-based policies resulted in a higher frequency of water shortages. Moreover, Felfelani *et al*., (2013) proposed the '*Goal-seeking Hedge*' to improve on the disadvantage of the one-point hedging where a single value (i.e. the Trigger value K_t), is utilized as the reduction coefficient for the duration of the period (e.g. a drought period). The *Goal-seeking Hedge* is explored in greater detail in the subsection 3.2.6, as it was tested as one of the intervention strategies in this thesis.

Thames Water, (2016) developed an operating policy better known as the Lower Thames Control Diagram (LTCD). This operating policy was developed by calculating the deployable output of historical records from 1920 – present, and this allowed for the Lower Thames to be segmented into four control levels (i.e. this is similar to the zonal hedging concept). The control levels inform water managers and allow them to make decisions on the types of restrictions to implement. Additionally, these control levels also prescribed the volume of target flow that is allowed to flow downstream towards the Teddington Weir (a man-made structure designed to control the river flow rate downstream of the weir).

2.5. Qualitative Resilience Frameworks Utilised in Water Resources Planning and Management

This section of the literature review explores conceptual framework developed to characterize resilience. Cabinet Office, (2011) proposed a conceptual framework for quantifying resilience. They express resilience as "*the ability of assets, networks, and systems to anticipate, absorb, adapt to and/or rapidly recover from a disruptive event*" (Cabinet Office, 2011). The framework illustrates four properties fundamental for resilience: resistance, reliability, redundancy and response and recovery. Resistance is conceptualized as the ability to provide protection and prevent damage or disruption. It can be argued that a critique of this property (resistance) is that strategies designed from assessments using historical records may not be sufficient to resist potential unexampled extremes which could affect the system in the future as a result of climate change.

Kinzig *et al*., (2010) proposed a generic framework for evaluating the resilience of social-ecological systems. It is composed of five steps: *i*) describing the system; *ii*) understanding the system dynamics, *iii*) system interactions, *iv*) system governance and, *v*) acting on the assessment.

Referring to step (*i*) above, this can be conceptualised as defining the 'resilience of what to what?'; step (*ii*) characterises the transitions the system undergoes as it moves from one state to the next, and sets the scene for establishing a critical threshold; *iii*) in this step, any interactions between ecological components with social components are characterised, *iv*) in this step, "*adaptive governance approaches recognize cross-scale interactions and promote interactions across organizational levels*" is explored (Kinzig *et al*., 2010) and, *v*) the information gathered in the previous steps is use to populate the two templates that underpin the resilience framework, and these are found in the literature (Kinzig *et al*., 2010).

Ainuddin and Routray, (2012) proposed a community resilience framework for improving community preparedness and increasing earthquake awareness. In developing the framework, surveys were administered to 200 residents selected at random in the community in Baluchistan, Pakistan. The framework was formulated from the information gathered from the surveys. The framework is underpinned by the notion that the community is considered as the first responder post-disaster, and is segmented into the following categories: (*i*) identify hazard/disaster characteristics, (*ii*) determine individual/community vulnerability, (*iii*) risk reception and awareness preparedness, and (*iv*) improve social, economic, and physical resources.

Ifejika *et al*., (2014) present a conceptual and analytical framework for characterizing livelihood resilience to capture an understanding of people's adaptive capacities. They describe a livelihood approach as the "*resources that people have and the strategies they adopt to make a living*" (Ifejika *et al*., 2014). The authors express livelihood resilience as comprising of three key properties, a) 'buffer capacity', b) 'self-organization' and c) the 'capacity for learning'. The buffer capacity refers to the amount of change a person or a system can undergo whilst attempting to retain the same structure, function, or identity. Moreover, self-organization refers to the opportunity to self-organize or reliance on one's own resources; and the capacity for learning characterizes a system or a person's ability to adapt.

Shirali *et al*., (2012) proposed a resilience engineering framework to investigate deficiencies in the resilience of the plant operations. Surveys were administered to the plant's employees and management to gather information associated with seven-safety indicators (schedule delays, safety committees, meeting effectiveness, safety education, worker's involvement, competence, safety training) and four managerial indicators (centralization or decentralization control systems, management of change, risk management and accident analysis, management commitment to safety and resilience). The authors revealed that deficiencies were found to be present in all the indicators assessed, especially in the areas of safety training, management of change, and risk management and accident analysis.

Vugrin *et al*., (2011) developed a semi-quantitative assessment framework for assessing the resilience of infrastructure and economic systems. In the context of infrastructure systems, resilience is underpinned by the three dimensions illustrated in Figure 2.3. An economic resilience index is utilized to measure the costs incurred during failure and the cost incurred during recovery.

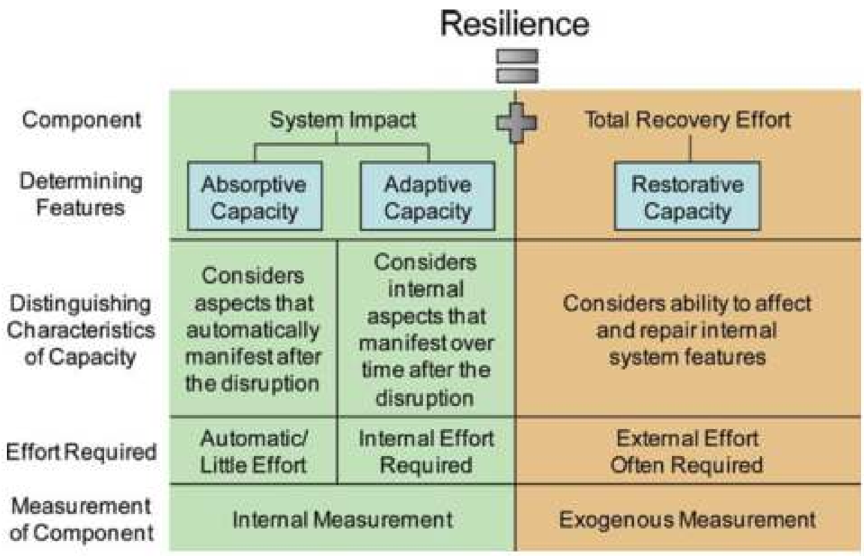


Figure 2.3 Illustration of resilience framework proposed by Vugrin *et al*., (2011)

Sterbenz *et al*., (2011) proposed a framework for assessing the resilience of communication network systems. In their work, resilience is expressed "*as the ability of the network to provide desired service even when challenged by attacks, large-scale disasters, and other failures*" (Sterbenz *et al*., 2011), and is comprised of the following six properties: defend, detect, diagnose, remediate, refine, and recover. The property 'defend', captures reducing the probability that a fault could result in the failure of the network. In their work, this is exampled by performing a model analysis (e.g. firewall analysis). If the analysis reveals that network defences have been breached, then the next step involves detecting network failures. Moreover, 'remediate' refers to actions implemented to maintain network functionality, such as rerouting protocols around the detected failures. Depending on the nature of the fault, remediation actions could result in the network possibly recovering to its 'sub-optimum' state or baseline. Post-recovery diagnosis addresses prevention of similar events in the future either by removing the fault or by introducing redundancy measures. Lastly, 'refine', captures the ability to learn from past system failures.

A weakness of this framework extends to the lack of consideration for addressing unknown unknowns, and this is transparent in the 'remediate' phase of the framework discussed above.

Bruneau *et al*., (2003) proposed a semi-quantitative resilience assessment framework for examining the resilience of earthquake engineering structures, based on the definition "*the ability of the system to reduce the chances of a shock, to absorb a shock if it occurs (abrupt reduction of performance) and to recover quickly after a shock (re-establish normal performance)*" (Bruneau *et al*., 2003). This definition is underpinned by the following properties of resilience: *i*) robustness, *ii*) redundancy, *iii*) resourcefulness and, *iv*) rapidity, and considers the ideal resilient system to be one that demonstrates: a) reduced failure probabilities, b) reduced consequences from failure and, c) reduced time to recovery (Bruneau *et al*., 2003).

The research approaches explored above are underpinned by the assumption that an 'event' resulting in the degradation of the system performance is known. For example, the concepts underpinning the framework found in Bruneau *et al*., (2003) suggests that in order to characterize the system resilience, the nature of the disruption is known. In environments where there is a high level of unpredictability and uncertainty regarding an event, it could be difficult to apply this research framework (i.e. when the event is characterized as an 'unknown unknown'). To further underscore the shortcomings of that research framework, two of the properties considered as fundamental to achieving resilience, which are redundancy and resourcefulness would be difficult to implement under a high level of unpredictable or uncertainty. It would be challenging to characterize the level of redundancy required by interventions actions and/or the amount of resources that would be required to minimise degradation in the system performance. The redundancy property underpinning the Cabinet Office, (2011) research framework and the remediate property underpinning the research framework in Sterbenz *et al*., (2011) both inherit these shortcomings.

The Safe & SuRe research framework developed by Professor David Butler as a part of the EPSRC Established Career Research Fellowship Project, addresses the shortcomings explored above. The framework has been developed to address existing and emerging concerns in the water industry; however, the concepts underpinning the structure of the research framework can be applied beyond the water industry. The framework provides key definitions for reliability, resilience, and sustainability performance indicators, and establishes the relationship between each of them as illustrated in Figure 2.4.

Figure 2.4 Illustration of the relationships underpinning the Safe and SuRe research framework (Butler *et al*., 2014)

According to Butler *et al*., (2014), a Safe and SuRe system is one which is "*reliable, built upon by resilience and topped off with sustainability*". Referring to Figure 2.4, Safe essentially means reliability which is considered as the bedrock of the Safe and SuRe research framework. This is concerned with the (water resources) system's functionality with respect to maintaining established or an expected level of service (Mugume, 2015). The level of service criteria varies depending on the type of system being evaluated.

Ideally, systems are normally designed to be reliable and not fail, however, in reality these systems are being challenged by emerging threats such as climate change, demand increases and compliance with new policy regulations. Therefore, depending on the type of system, preventing failure (especially in older systems) is not always possible. To address these emerging concerns, the framework progresses to the middle phase of the Safe and SuRe pyramid which captures resilience, where the aim is to minimise both how long failure occurs, and the extent or the magnitude of that failure. Lastly, the summit of the pyramid addresses sustainability with emphasis placed on the design life of the system. Sustainability therefore seeks to ensure that any strategy implemented to enhance both reliability and resilience is able to achieve that over the entire design life of the system. The middle phase and the summit of the pyramid which addresses both resilience and sustainability respectively, essentially is characterized as – SuRe. Table 2.4 summarises the definitions of reliability, resilience and sustainability expressed under the Safe and SuRe framework.

Table 2.4 Summary of the Safe and SuRe performance indicators

Author	Resilience
Butler *et al.,* (2014)	Resilience is expressed in the Safe and SuRe framework as "*the degree to which the system minimises the level of service failure magnitude and duration over its design life when subject to exceptional conditions*". **Reliability** “Reliability is expressed in the Safe and SuRe framework as “*the degree to which the system minimises level of service failure frequency over its design life when subject to standard loading*”. **Sustainability** Sustainability is expressed in the Safe and SuRe framework as “*the degree to which the system maintains levels of service in the long-term whilst maximising social, economic and environmental goals*”.

The relationships discussed above have been used to facilitate the developments of four methods for assessing the reliability, resilience, and sustainability of systems. They are illustrated in Figure 2.5 and are summarised below (Butler *et al*., 2017):

1) Top-down approach;
2) Bottom-up approach;
3) Middle-based approach and the;

4) Circular approach

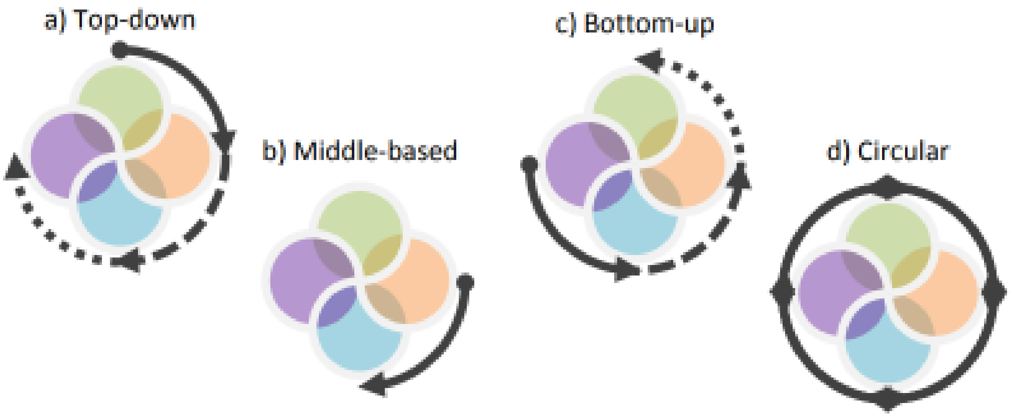


Figure 2.5 Illustration of the Safe and SuRe analysis methods.

In the figure above, the green circle characterises ‘Threats’, and these are events resulting in the degradation of the system performance. The orange circle characterises the ‘System’, and the blue circle characterises ‘Impact’, which addresses compliance with the level of service. The purple circle characterises ‘Consequences’ which captures the socio-economic and environmental effects of non-compliance with the level of service (Butler *et al.*, 2017).

In the Top-down approach, the conventional approach to water resources management is captured (Butler *et al.*, 2017), and the assessment of systems is guided in a clockwise direction from threat to impact or threat to consequence. In the Middle-based approach, the emphasis is placed on identifying and quantifying how (water resources) systems respond to extreme conditions. In this method the system is stress tested (e.g. reducing inflows to a reservoir by 40%) and the response of the system under these conditions reveal different ways the system could be impacted. The bottom-up approach progresses anti-clockwise from Consequences to Impact, or progresses anti-clockwise from the System phase, and presents a key advantage associated with its applicability. The Bottom-up method is consequence focused (Butler *et al.*, 2017) and can be implemented in environments where detailed information regarding likely threats and impacts is limited. This is accomplished by assessing how society, the economy and the environment would be affected by the removal of "*a critical system or service*" (Butler *et al*., 2017). Lastly, the Circular approach considers all the elements of the Safe and SuRe research framework from Threats to Consequences (moving in a clockwise direction), with the emphasis placed on Learning, which is characterized at the intersection of Consequences and Threat. This allows for strategies and policies to be continually updated as key information is gathered on the system performance, and the social, economic and environmental consequences. The concepts underpinning the analysis in the Safe and SuRe framework captures uncertainty from a system and consequences perspective.

2.6. Conclusion

In this chapter, reviews on water resources planning, water resources simulators, interventions strategies, resilience and research frameworks are presented. The chapter begins by exploring the shortcomings associated with conventional water resources planning, and highlights arguments calling for a transition from conventional approaches to water resources management, to an integrated approach. Moreover, the literature review explores various types of intervention strategies utilized in water resources planning and management, and highlights studies that call for the widespread implementation of supply and demand type interventions as an integral part of the water resources planning process. However, a review of recent literature suggests that in several regions within Asia and Africa, a backward shift towards the use of conventional approaches and conventional strategies (i.e. built infrastructure) are re-emerging due to some investors regarding those types of interventions as 'climate change buffers'.

The chapter progresses by examining the strengths and weaknesses of the different types of water resources management simulators commonly used in the planning process, leading to the justification and selection of Aquator which the simulator utilised in this thesis.

Following from the above, a critical review of research frameworks from different research and professional disciplines was conducted, exploring how resilience as a performance indicator is quantified. A common shortcoming highlighted in these frameworks extends to their lack of consideration in addressing uncertainty. More specifically, in the research frameworks established in the engineering arena, an emphasis is placed on redundancy and resourcefulness, which have been characterized in some studies as properties fundamental to achieving resilience; however, they fail to capture how uncertainty and extreme events would be addressed. Lastly, the literature review focuses on the Safe and SuRe framework which improves on the shortcomings identified from the reviewed frameworks by proposing four analysis approaches. The concepts underpinning the Middle-base and

Bottom-up approaches provide implications for addressing the shortcomings identified in the other research frameworks.

CHAPTER THREE

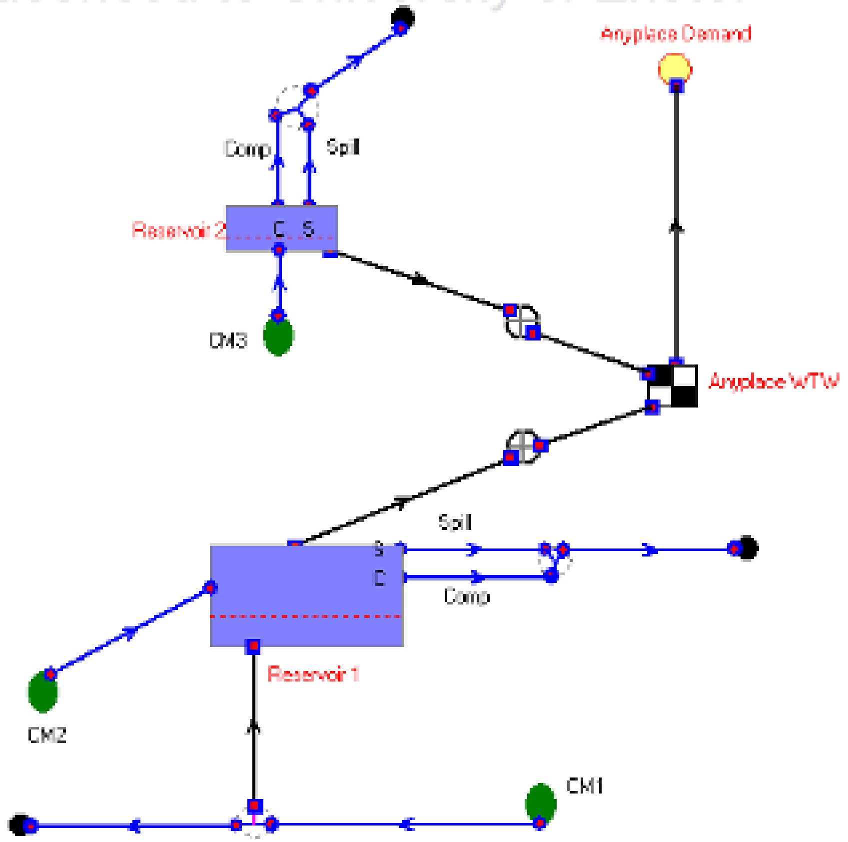
3. The Anyplace Case Study

This chapter provides a detailed description of the Anyplace case study, and the materials and methods utilized to accomplish research objectives 4, 5, 6, 7 and 9. In section 3.1, a description of the water resources system is presented. In section 3.2 demand and supply scenarios and the interventions are explored; and in section 3.3, the concepts underpinning the Safe and SuRe Circular approach are applied in this chapter. The social and environmental performance indicators are presented in section 3.4, and this is followed by the modelling of the water resources system in section 3.6. The results are discussed in section 3.7, and in section 3.8, a method is developed to assess the performance trade-offs of the intervention strategies.

3.1. Description of the Anyplace water resources system

The Anyplace case study region is a real-life water resources system located in Northern United Kingdom. The exact location of the Anyplace water resources system is not revealed throughout study as permission was not granted by the water supply undertaker governing the area due to data sensitivity associated with the performance of their assets. The water resources system serves approximately 260,500 inhabitants and comprises of two reservoirs systems RV1 and RV2 of capacity 7303 Megalitres (ML) and 2939 ML respectively. They both supply water to a treatment works facility of design capacity 500 ML/d. Referring to the schematic illustration of the Anyplace Aquator model (see section 3.1.2) in Figure 3.1, the larger of the two reservoir systems (RV1) is supplied with abstracted water from catchment CM1. Abstraction costs are £15 per ML, and the daily abstraction permit limit is 50 ML/d (i.e. real-life abstraction costs for the Anyplace WRS). In addition, water from the catchment CM2 also supplies RV1. The smaller of the two reservoirs RV2 obtains it supplies from CM3 as illustrated in Figure 3.1.

Licenced to University of Exeter



An Abstraction allows water to be taken from a river network to a supply storage component (i.e. reservoir, water treatment works).

Catchment marks the beginning of one branch of a river network, and adds water daily to the river network at that point.

A Reservoir provides storage for the system. Storage can either be in the river network or the supply system.

Water treatment works is located in the supply system and processes water to be sent into distribution.

The Demand centre represents the source of the demand (i.e. a city, village etc).

This component is the river reach and conveys flows from catchments into the supply and storage components.

A link represents connections between components in the supply distribution network. Examples of link are pipelines, aqueducts etc.

A termination is requirement in the Aquator water resources model and represents the last component downstream of a river reach. This accounts for water leaving the system (e.g. water not abstracted from a river).

Figure 3.1 Illustration and description of the infrastructural components in the Anyplace water resources system

As previously mentioned, Anyplace is located in the United Kingdom, and according to Sanderson *et al*. (2012) the impacts of climate change, along with increases in demand are expected to exert a 'high level of stress'[1] on the water supplies in the United Kingdom. In 2010, the Anyplace region experienced an extended dry period resulting in low levels of rainfall, and a deficit of 85% to 95% of the 30-year average 1981 – 2010 was recorded (Met Office, 2017). This placed high levels of stress on existing supplies. Moreover, if the population in that region increases, this could result in a possible increase in demands, and this is likely to exert additional stress on supply availability.

To assess this possibility, uncertainty associated with population increase is explored by applying the principles underpinning the Office for National Statistics (ONS, 2016) population projections for the United Kingdom. In this case study, the population is estimated for the period 2020 - 2049. This is explored in greater detail in section 3.2.2 and allows for a comprehensive assessment of the response of the Future Flow Hydrology to demand uncertainty. The Future Flows Hydrology explored in detail in subsection 3.2.3, represent a plausible set of future hydrological flow conditions.

3.1.2. Brief description of the Anyplace Aquator Model

Aquator is a water resources management software package developed by Oxford Scientific software, (2014a). The main components of a water resources system such as reservoirs, pumping stations, water treatment works, abstractions, catchments and water transfers are represented within an Aquator model (refer to Figure 3.1 for illustration of an Aquator model schematic). The model is driven by a demand value which represents domestic demands, and supplies are defined by time series of river flow or groundwater sources. The movement of water from the supply side to the demand side is governed by control rules that are typically representative of the real-life system operation. A vital characteristic and key advantage of

[1] In the context of this study, this could be expressed as the number of instances or periods where hydrological conditions result in low streamflow. The consequences of this could result in less inflows to reservoirs or extend to over abstraction from groundwater sources to compensate for a deficit in available supply.

Aquator is its flexibility. This allows for custom alterations to be made to key model components, and operational rules can be scripted as a series of code using Visual Basic Application (VBA). In this study, VBA is used to operationalise the suite of intervention strategies listed and described in subsection 3.2.4. The initial parameters used as input during modelling are presented in Table 3.1.

Table 3.1 Anyplace configuration settings

Component	Input	Type	Value
Demand Centre	High Demand Scenario	Parameter value	52 ML/d
Reservoir RV1	Capacity	Parameter value	7303 ML
Reservoir RV1	Compensation Release	Parameter value	4.546 ML/d
Reservoir RV2	Capacity	Parameter value	3909 ML
CM1; CM2; CM3	FFH	Time series	-
Abstraction from CM1	Fixed Cost	Parameter value	£15/ML
Abstraction from CM1	Permit limit	Parameter value	50 ML/d
Anyplace WTW	Design Capacity	Parameter value	500 ML/d
VBA	Miscellaneous	-	-

3.2. Anyplace case study methodology

The purpose of this section is to describe the main concepts, methods and frameworks applied in this thesis to achieve the research aims and objectives. The methods presented in Figure 3.2 were utilized to achieve the research aim.

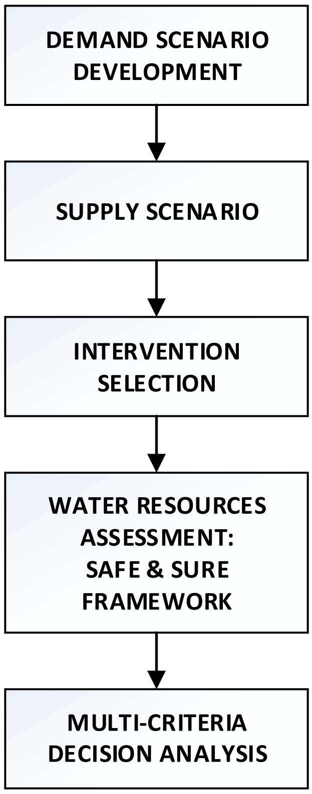


Figure 3.2 Illustration of methods explored in this section

3.2.1. Anyplace case study scenario development

Socio-economic scenario development is an important method for exploring uncertainty in the future. The Foresight Programme is widely known for developing scenarios for examining social, political, economic and environmental implications for the future (DTI, 2002). These scenarios cover a broad spectrum which extends to economic development, planning, and the built environment, agriculture, water, biodiversity and coastal zone management. In addition, the UK Climate Impacts Programme (UKCIP) also developed a suite of socio-economic scenarios to use for climate change impact assessments, and these are built on the early work conducted by the Foresight Programme. In the context of water resources, the Environment Agency EA, (2001) developed a suite of four plausible scenarios for water demand. Makropoulos *et al*. (2008) developed a suite of scenarios to explore the realm of uncertainty associated with drinking water, wastewater and stormwater. Further to this, Casal-Campos *et al*. (2015) developed scenarios to explore the robustness associated with green and grey strategies for the horizon of 2050.

The scenarios developed in this chapter consider possible changes that could influence the Anyplace water demand for the evaluation period 2020 to 2049. In the literature, this (30-year period) is the minimum period considered adequate for the long-term planning of intervention strategies, and is exampled in recent the studies (Hall and Borgomeo, 2013; Forsythe *et al*., 2014; Walsh and Blenkinsop, 2015). The scenarios explore the uncertainty associated with future water demands for the Anyplace region, and these demands are used in conjunction with the supply uncertainty associated with the Future Flows (subsection 3.2.3).

The combination of demand and supply scenarios enhances the water resources assessment, and allows for a greater level of uncertainty planning to be employed when selecting interventions. To develop demands for the evaluation period, a suite of socio-economic drivers of demand are considered. These include policy changes, technological development and demographic changes and are explored in subsection 3.2.2.

3.2.2. Drivers of Demand

To investigate the influence that future variations in population could have on the Anyplace's water resources, the Office for National Statistics (ONS) population projections scenarios were adopted as a starting point.

The ONS developed 10 variants quantifying the possible changes in the UK population, which are illustrated in Figure 3.3. They are based on drivers of change such as future fertility, mortality, and migration; and previously observed deviations in population trends.

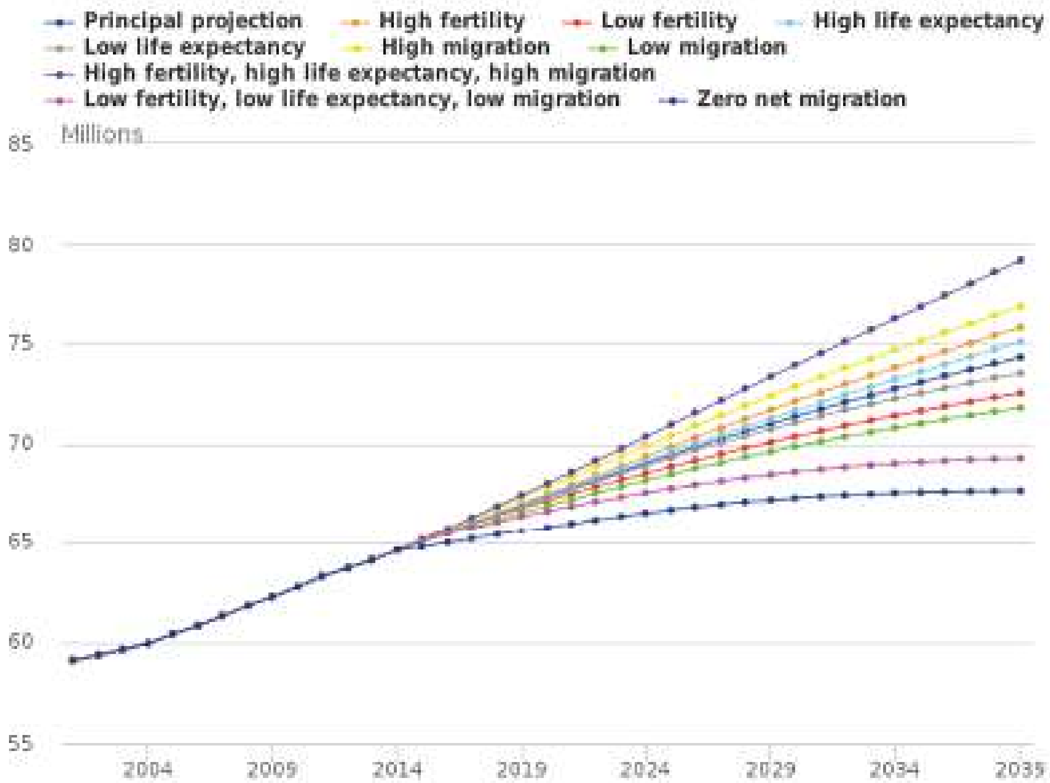


Figure 3.3 Office for National Statistics Illustration of Population Projections Variants for the UK (ONS, 2016)

Referring to Figure 3.3, the Low Migration Variant envisions a future where migration to the UK is very low. This underlying concept is used as the starting point for developing the Low Demand Scenario. In this scenario, it is assumed that the UK population could be approximately 71.8 million by the year 2039. Applying the incremental increase method for population projection (Equation 3.1), an estimated population value for the United Kingdom projected for the year 2049 was determined in Microsoft Excel, from the Low Migration Variant. The results are summarised in Table 3.2. Assuming the Anyplace population closely follows the trends in growth and decrease observed in Table 3.2 for the Low Migration variant, the estimated population projection for Anyplace for 2049 was determined and summarised in Table 3.3. The incremental increase equation is presented below:

$$P_n = P + X * n + \left(\frac{n(n+1)}{2}\right) * Y \qquad (3.1)$$

Where, P_n = population after n^{th} decade, X is the average increase and Y is the incremental increase.

Referring to Table 3.2 for the Low Migration Population statistics, the value in the 3rd column represents the change in the population per decade, and allows for the average value to be determined (i.e. [(4.2 +3.1+7.2)/3]). Moreover, in the 4th column, the increment in the Y parameter was determined by subtracting the difference in the increments obtained for each decade (i.e. 3.10 – 4.20 = 1.10). Lastly, P_n was determined by substituting the values in Equation 3.1. This process was repeated for the UK Principal and the High Fertility, High Life Expectancy and High Migration variants (explored below) to estimate the population for the year 2049.

For the Mid-range demand scenario, the UK Principal projection is used as the starting point. This variant is based on the underlying assumptions that past observed trends (i.e. migration, fertility and life expectancy) will remain the same in the future, and estimate that the UK population could reach approximately 74.3 million by the year 2039. The approach described above to develop the Low Migration Variant was applied here, and the trends underpinning the UK Principal variant were used to extend this estimate to the year 2049. Similarly, the assumption was made that the Anyplace population closely follows trends observed in the UK Principal Variant, and the estimated population for 2049 was determined by applying the incremental increase method as shown in Table 3.3.

Lastly, a High demand scenario was developed for Anyplace region based on the High Fertility, High Life Expectancy, and High Migration Variant. In this variant, it is assumed that a sharp rise in the drivers of change fertility, life expectancy and migration occur. Under this variant, it is estimated that the UK population could reach 79.1 million by 2039. The approach and assumptions described to develop the Low and the Mid-range demand scenarios were applied in this case to estimate the Anyplace population for 2049 (Table 3.3).

The estimated residential demand is determined for each of the three scenarios by first assuming the average daily consumption in the UK remains at approximately 150l/p/d (Waterwise, 2012), and then multiplying this value by the estimated population for each of the three population projections scenarios as seen in Table 3.4. In this estimation, it is also assumed that the average daily consumption patterns in the UK would remain the same in the future.

Table 3.2 Estimated UK Population for the Year 2049 using the incremental increase method based on the ONS variants

Year	Low Migration Population (million)	Increment (X)	Increment (Y)	UK Principal Population (million)	Increment (X)	Increment (Y)	High Fertility, High Life Expectancy, High Migration Population (million)	Increment (X)	Increment (Y)
2009	62.30	-	-	62.30	-	-	62.30	-	-
2019	66.50	4.20	-	66.90	4.60	-	67.40	5.10	-
2029	69.60	3.10	-1.10	71.00	4.10	-0.50	73.30	5.90	0.80
2039	76.80	7.20	4.10	74.30	3.30	-0.80	79.10	5.80	-0.10
		14.50	3.00		12.00	-1.30		16.80	0.70
		4.80	1.50		4.00	-0.65		5.60	0.35
	Total			**Total**			**Total**		
	Average			**Average**			**Average**		
2049	**83.10**			**77.70**			**85.05**		

Table 3.3 Estimated determined for Anyplace Region for the Year 2049 using the incremental increase method

Year	Low Migration Population (million)	Increment (X)	Increment (Y)	UK Principal Population (million)	Increment (X)	Increment (Y)	High Fertility, High Life Expectancy, High Migration Population (million)	Increment (X)	Increment (Y)
2009	260500	-	-	260500	-	-	**260500**	-	-
2019	276130	15630	-	278735	18235	-	**281340**	20840	-
2029	287175	11045	-4585	295459	16724	-1511	**303847**	22507	1667
2039		-287175	-298220	307277	11818	-4906	**325116**	21269	-0.1238
		-260500	-302805		46777	-6417		64616	429
		-86833.3	-151402.5		15592	-3208.5		21539	214.5
	Total			**Total**			**Total**		
	Average			**Average**			**Average**		
2049	**300579**			319661			**346869**		

Table 3.4 Summary of the estimated demands for each of the future scenarios

Scenario	Estimated Population 2049	Average daily consumption UK (l/d/p)	Average daily estimated demand Anyplace 2049 (ML/d)
Low Demand	300579	150	45.09
Mid-range Demand	319661		47.95
High Demand	346869		52.03

3.2.3. Supply side scenario development

The Environment Agency (EA) 'Climate Change approaches in water resources planning – overview of new methods report' (EA, 2013), recommends that for all water resources assessments, the impacts of climate change on the supply availability should be assessed. In the report, four methods are recommended by the EA, one of which consists of the use of Future Flows Hydrology (FFH). The FFH was chosen in this research because they can be applied to studies where it is not possible for a water company or other researchers to conduct their own rainfall-runoff modelling (EA, 2013). This particularly applies to the Anyplace water resources system because of the high level of anonymity associated with the Anyplace location.

According to Prudhomme *et al*. (2013) the FFH database comprises of a "*set of river flow and groundwater level projections for 281 river sites and 24 boreholes across Great Britain to enable the investigation of the role of climate variability on river flow and groundwater levels nationally and how this may change in the future*" as illustrated in Figure 3.4. They have been developed using eleven regional climate change models by the UK Met Office Hadley Climate Centre (11 HadRCMs) (EA, 2013), and they represent hydrological scenarios that reflect the future climate change impacts of the 281 river sites.

They can be accessed from the National River Flow Archive (NRFA) online portal.

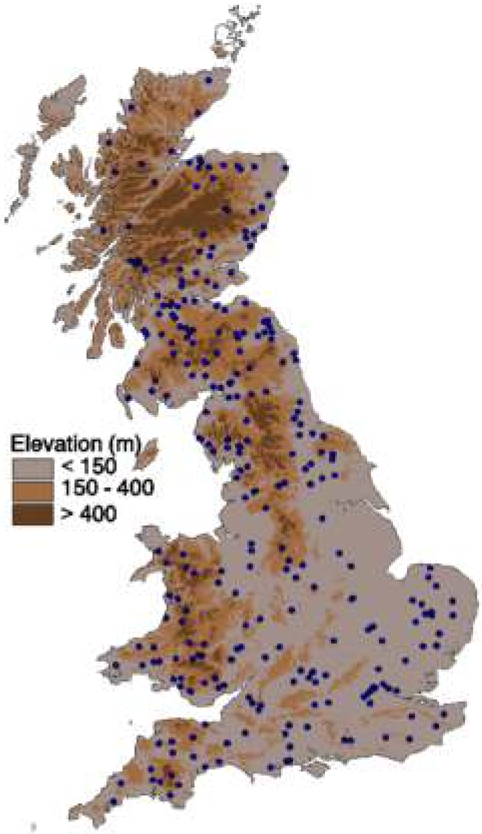


Figure 3.4 The location of 281 catchments in the UK where Future Flows Hydrology parameters have been generated (Prudhomme, *et al*., 2013).

The 11 HadRCM ensembles cover the 1950 and 2098-time horizon (Prudhomme *et al*., 2013). The ensemble provides time series of precipitation and temperature at 25-km grid spatial resolution. Potential evapotranspiration is also generated at a 5-km resolution using the HadRM3-PPE-UK experiment, and this is based on the FAO-56 Penman-Monteith method described in Allen *et al*. (1998). Each of the 11 ensembles is driven by the same historical and SRES A1B inputs, however, one of the eleven members remain unperturbed with the remaining members perturbed to different atmospheric parametrizations (Murphy *et al*., 2009). This allows for comparisons to be made between past conditions and the perturbed conditions. The SRES A1B scenario envisions a future where a balance between continuing use of fossil-intensive and non-fossil energy sources are utilized (IPCC, 2000).

The FFH flow time-series are generated using three models: regionalized, catchment and hybrid. The time-series chosen for the Anyplace was generated by a Continuous Estimation of River Flows (CEFR) regionalized rainfall-runoff model. The key feature of the CERF model is that the model structure and parameters do not vary for different catchments located in the same region. This is possible in the CERF model because all the catchments are assumed to "*behave in a hydrological similar manner*" (Griffiths *et al*., 2006). As stated earlier, the 30-year period 2020 - 2049 is used in this assessment, and therefore is also used to assess the impact of climate change on future supply during this period.

3.2.4. Proposed demand side and supply intervention strategies

In this research, a key focus is on the application and testing of intervention strategies associated with the management of the supply-demand balance of the case studies' water resources. These strategies were compiled following the comprehensive review in Chapter 2 and are summarised in Figure 3.5.

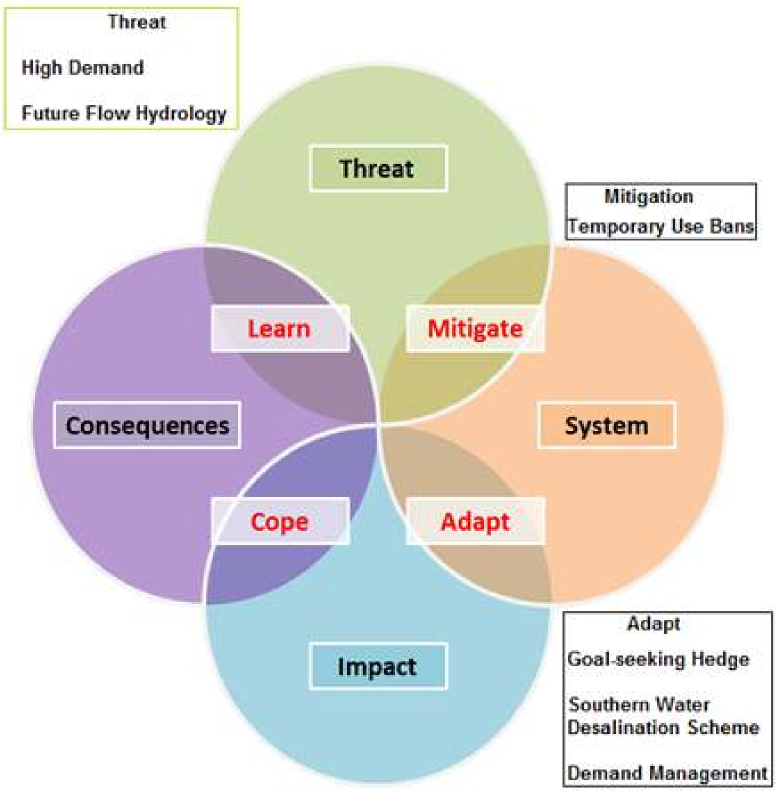


Figure 3.5 Illustration of the threats and the intervention strategies mapped on the Safe and SuRe framework.

The strategies focus on both the demand and the supply side and have been selected using the Evidence-Based Intervention Decision Making Matrix developed by (Andrews and Buettner, 2002). In Figure 3.6, the interventions labelled increase reservoir capacity, water transfers, and groundwater abstractions are categorized as 'untested and unavailable'. This is due to lack of information available in the public domain, and the high level of anonymity associated with the case study region (recall the Anyplace location is confidential). The intervention strategies rainwater harvesting, and desalination were categorized as 'evidence-based and available' because a large volume of studies detailing the potential desirable and undesirable benefits of rainwater harvesting and desalination are available in the academic domain. The desalination scheme was adopted from the Southern Water water resources management plan because its design capacity has been proposed to service a water resources region of similar size to that of the Anyplace region.

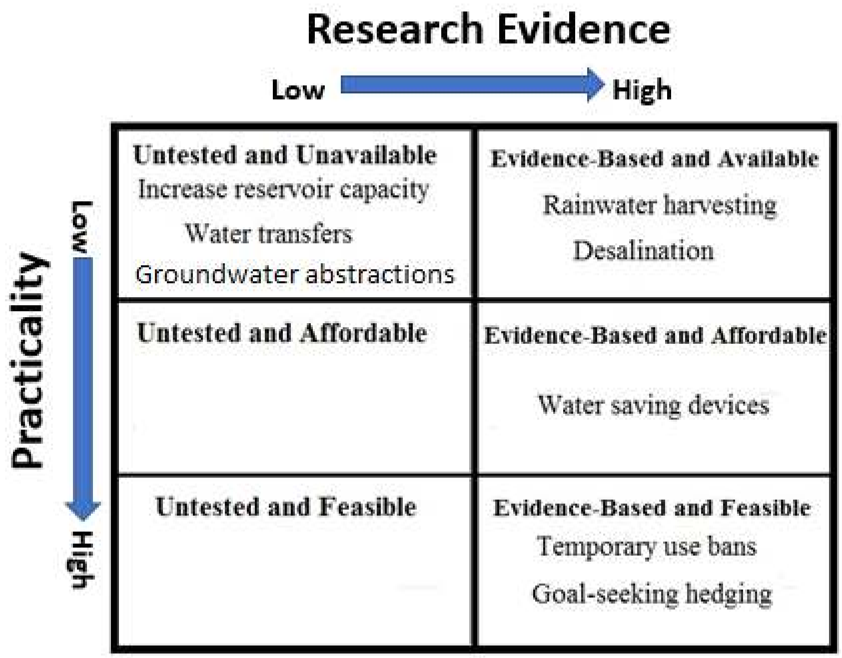


Figure 3.6 Evidence-based intervention making matrix used to prioritise the intervention strategies (Andrews and Buettner 2002)

The *Goal-seeking Hedge* is a new policy proposed in Felfelani *et al*. (2013), and it can be argued that it should be categorized as 'untested and feasible'. This stems from this policy being a relatively new concept to the reservoir management discipline and has not been implemented in practice on a real-life reservoir system at the time of writing. However, the application of the intervention was exampled on a small reservoir system in the research of Felfelani *et al*., (2013), and further testing in this research would enhance the feasibility of that strategy. Therefore, it has been categorized as 'evidence-based and feasible'. Additionally, *Temporary Use Bans* are policy driven and are considered a feasible strategy to implement during drought conditions or in the period following two dry winters.

Finally, water saving devices have been categorized as 'evidenced based and affordable' because of the growing literature demonstrating the water conservation benefits. They are considered affordable because it is assumed that more financially secure individuals can afford the costs associated (e.g. installation), and some utility companies such as South West Water provide the basic water saving devices that do not require installation, free of charge (SWW, 2017). More detailed descriptions of the proposed strategies are described in the subsections below along with the criteria supporting their implementation.

3.2.5. Demand Management Strategies

Water saving devices

Water saving devices (WSDs) are proposed as a water demand management strategy to explore the influence they could have on water resources management under the future scenarios. The term water saving devices represents devices such as low flow showerheads, dual flush toilets and save a-flush-bags. In Waterwise (2011) a study carried out by United Utilities revealed that consumption was reduced by 28.7 L/property/day when these three devices were installed. This intervention was tested assuming that for the evaluation period, these three WSDs would be installed in all the properties in the Anyplace region. Additionally, it was assumed that the average occupancy of 3 persons per household remains the same, and this allows for an estimate of the number of properties to be determined (i.e. 346,869/3 = 115,623 households). The total average daily reduction in consumption was determined to be 3.32 ML/d (i.e. 115,623 *28.7 L/hh/d), and this saving is modelled in Aquator using a VBA script.

Rainwater Harvesting – household scale

The second type of water demand intervention strategy explored in this case study is rainwater harvesting. Harvested rainwater was assumed to be used for outdoor and WC purposes. Literature estimates found in (Chao *et al*., 2015), stated that rainwater usage contributes to approximately 6% to 10% of the total water use in summer and 26% in winter. The implications of this trend are tested in the case study using these estimates. These seasonal contributions are modelled in Aquator using a VBA script. The time-step in the evaluation period which sees the introduction of the demand management strategies are summarised in Table 3.5.

Temporary Use Bans

The influence of *Temporary Use Bans* (TUBs) is evaluated in the Anyplace study, and they are introduced in a period immediately following a 2nd dry winter. According to a Scottish Water, (2005) water conservation report, outdoor water use such as gardening and washing cars contributes to 5% of the consumption from mains. The TUBs are applied in this case study with the assumption that restrictions are placed on potable water use for outdoor usage (i.e. gardening and car washing). A VBA script was written to test this intervention in Aquator.

3.2.6 Proposed Supply-side Intervention Strategies

Goal-seeking Hedge

Goal-seeking Hedge was tested in this research as a supply-side intervention strategy. This intervention was developed by Felfelani *et al*. (2013) and is applied to the reservoir systems as a result of dry conditions or when the reservoir system has been operating at below the emergency storage level for 14 days. A VBA script was utilized to modify, when necessary, the reservoir operating rules in Aquator. The *Goal-seeking Hedge* coefficient is defined in (Felfelani *et al*., 2013) by Equation 3.2 as follows:

$$\propto = \left(1 - \frac{1}{i}\sum_{j=1}^{i} \frac{D_j - v_j}{D_j}\right)$$

(3.2)

where, α is the goal seeking coefficient, v_j is the deficit volume, D_j is the total demand volume in each period, and i is the total number of time steps.

The aim of the '*Goal-seeking Hedge* policy' is to reduce the quantity of water released from the reservoirs system when drawdown surpasses the emergency storage volume level (Felfelani *et al*., 2013). The key characteristic of the '*Goal-seeking Hedge*' when compared to the more conventional hedging policies is that the coefficient is dynamic in nature and does not remain at a fixed value (Felfelani *et al*., 2013). As the storage volume changes with each time-step, the value of the 'goal seeking' coefficient also changes. This contrasts with the conventional 'One-point hedging' coefficient that introduces a fixed reduction value or 'trigger value' from the point of its inception until the system can be returned to normal operation (Lund, 1996). The goal-seeking coefficient is calculated in monthly time-steps in this study.

Desalination technology

A 15 ML/d desalination scheme is tested in this case study. It is one of the preferred options from the list of new supply resources published in the

Southern Water water resources management plan (Southern Water, 2014). As mentioned in subsection 3.2.4, its selection was based on its potential to service a region of similar size to Anyplace.

Table 3.5 Intervention scheduling for the Anyplace case study

Intervention	Comments
Desalination	Introduced from the time-step representing the beginning of the largest degradation in the system performance
Temporary Use Bans	Introduced from the time-step/period representing the period following a second dry winter
Demand Management schemes	Introduced from the time-step representing the largest consecutive duration where domestic demands are not met.
Goal-Seeking Hedge	Introduced at the time-steps when the reservoir system performance is at or below the emergency storage level for one month

The next section 3.3 provides a description of the performance framework utilized in this thesis, followed by the conceptualization of the proposed performance indicators.

3.3. The Safe and SuRe Framework

To assess the performance of the water resources systems considered in this thesis, performance indicators must be established. This section explores the Safe and SuRe research framework's Circular approach to establish a relationship between threats, the system, impacts, consequences, and the intervention strategies. In this thesis, the circular approach progresses in a clockwise direction beginning the analysis from the threat phase and terminating at the consequences phase (Figure 3.7).

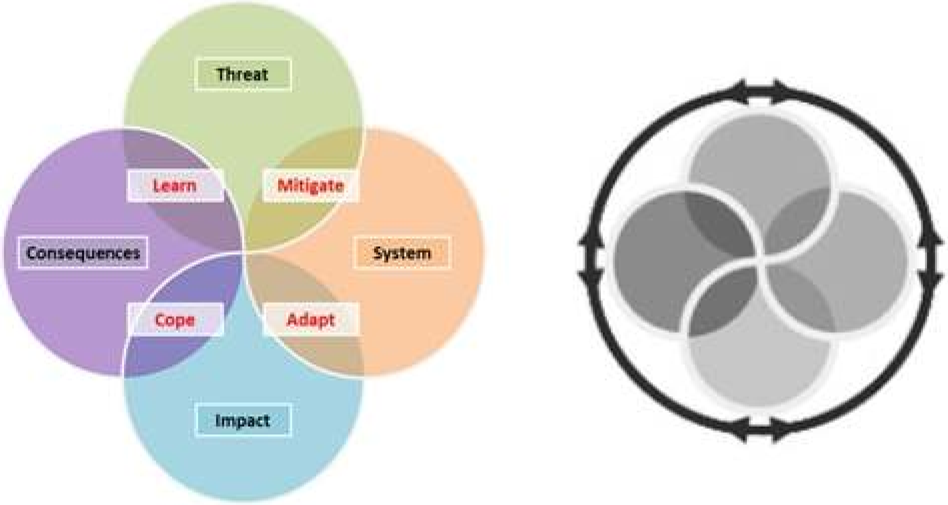


Figure 3.7 Illustration of the Safe and SuRe Framework and the Safe and SuRe Circular Approach (Butler *et al.*, 2017)

In the Anyplace case study, the threats to the system are demand increase, which has been linked to plausible changes in the Anyplace population, and the effects of climate change on supply availability, which has been considered in the Future Flow scenarios. In this thesis, the circular approach moves in a clockwise direction from Threats to System, and System to Impacts as seen in Figure 3.7. In this case study, the undesirable impact which could affect the system is categorized in terms of degradation of the reservoir's system performance below the emergency storage level, and that has been referred to in this work as 70% of the total storage volume. The impacts on the system performance may also have implications on the compliance with domestic demands and environmental requirements, which could lead to possible undesirable and non-sustainable outcomes. These are defined in the socio-economic segment of the framework as consequences, and underpin how the three pillars of sustainability are affected. Referring to Figure 3.8, emergency storage has been selected as 70% because during past historical droughts (e.g. 1974-76; 1984 and 1995-96) in the United

Kingdom, the reservoir systems (RV1 and RV2) have been drawn down below the storage capacity of 70% for long durations.

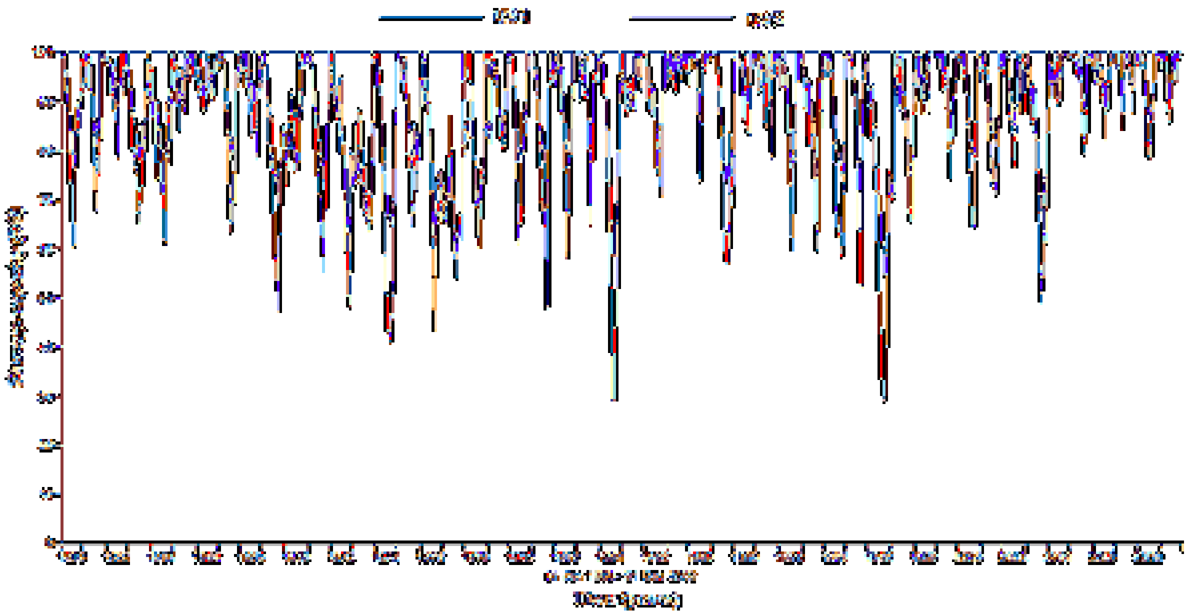


Figure 3.8 Illustration of the storage performance based on historical parameters of streamflow, and historical demand 32.1 ML/d for the period 1960 to 2009 (RV1 – Reservoir 1; RV2 = Reservoir 2).

Moreover, the intersecting segments of the framework represent intervention strategies introduced to minimise and better manage the undesirable impacts and the consequences. These segments are denoted by mitigation, adaptation, coping and learning strategies. Mitigation strategies are defined in Butler *et al*., (2017) as long-term actions, and these may be physical or non-physical to reduce the extent of the impact on the system, and in the long term, the effects on the consequences. For example, referring to the threat population growth, this could possibly result in an increase in the demand. Long-term actions to implement could be public education and awareness campaigns as measures to ensure the security of supplies.

Adaptation strategies are both more effective in the short and long-term and, are typically employed when threats cannot be mitigated (Butler *et al*., 2017). These typically include actions or modifications to the existing system; however, their successful implementation also depends on the availability of sufficient investment. For example, a targeted short-term action could be the introduction of a new hedging policy in response to drier conditions. Moreover, for a water resources system affected by high levels of non-revenue water, the replacement of ageing water mains could be a lengthy process due to slow investments, or little knowledge pertaining to the locations of some of the old water mains is an example of long-term actions. With time (i.e. post-completion) this type of asset management will contribute to the reduction of non-revenue water. This could result in favourable implications such as improvements in the conveyance and distribution of supply, and could also lead to more consistent compliance with customer demands. In addition, the introduction of a new source such as desalination is also an example of a long-term action.

Coping is defined in Butler *et al*., (2017) as "*any preparation or action taken to reduce the frequency, magnitude or duration of the effects of an impact on a recipient*". A Coping strategy could be employed when planned mitigation or adaptation strategies have not been introduced or are considered insufficient. For example, in the context of an extended dry period, customers could respond to temporary water shortages by purchasing water from private water tankers or bottled water. In both cases, there are financial implications for the customers.

The final intersection of the Safe & SuRe framework in Figure 3.7 analyses the effectiveness of the interventions strategies to determine what can be learned. This extends to best practices and is critical if continued improvements are to be incorporated. Learning is not explored in this research and therefore the circular approach is utilized up to the Consequences phase. The performance indicators utilized in the case studies are explored in the next section.3.4.

3.4. Performance Indicators

The performance indicators selected in the study to assess the performance of the Anyplace water resources system are reliability, resilience, and sustainability. Resilience is expressed in this thesis from a social and environmental consequence viewpoint, using the Safe and SuRe definition of resilience. Referring to Butler *et al*., (2014), resilience is expressed as "*the degree to which the system minimises the level of service failure magnitude and duration over its design life when subject to exceptional conditions*". In characterizing resilience from social and environmental perspectives, the aim is to minimise the number of consecutive days that residential demand is not met (failure duration), and the summation of the total deficit not delivered during that period (failure magnitude). This is conceptualized in Figure 3.9. The blue point represents the baseline performance. The baseline performance in the figure represents the consecutive failure duration in days and the sum of the demand deficit (social) or compensation releases deficit (environmental) for the water resources system without testing any intervention strategies. The black points represent the performance of the water resources system after an intervention or a suite of strategies have been tested; and the red point conceptually represents the target resilience performance. The *x* and *y*-axis have been rescaled to the range 0 to 1 to avoid confusion when interpreting results. For instance, a performance result for the blue point could return a consecutive failure duration of 200 days and magnitude of 2000 ML, and the red point could return a performance result of 10 consecutive failure days and a magnitude of 15 ML. Graphically, results of this nature would be difficult to illustrate, and therefore rescaling the results to the range 0 to 1 is preferred. These performance indicators may be of interest to water engineers and water management stakeholders, as they could support decision making in terms of planning the period of implementation of intervention strategies (e.g. the implementation of *Temporary Use Bans* during extended dry periods) as demonstrated in this work (refer to Table 3.5).

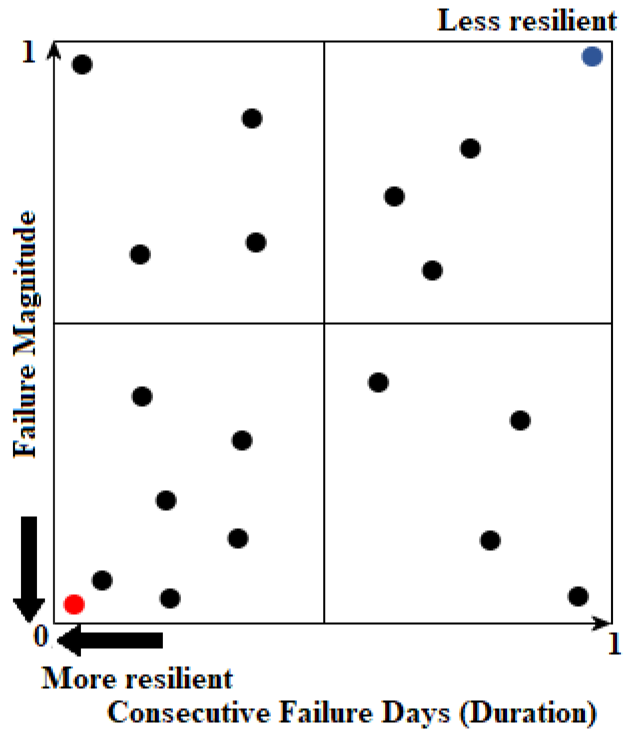


Figure 3.9 Conceptual illustration quantifying the Social and Environmental resilience concepts

The reliability criterion utilized in this study is based on the proportion of time-steps in the time horizon that the reservoir performance does not fall below the emergency storage (i.e. defined in this study as 70% of the total storage volume), and this is conceptualized in the Figure 3.10. This criterion may be of interest to water engineers and stakeholders in the planning and development of reservoir hedging policies.

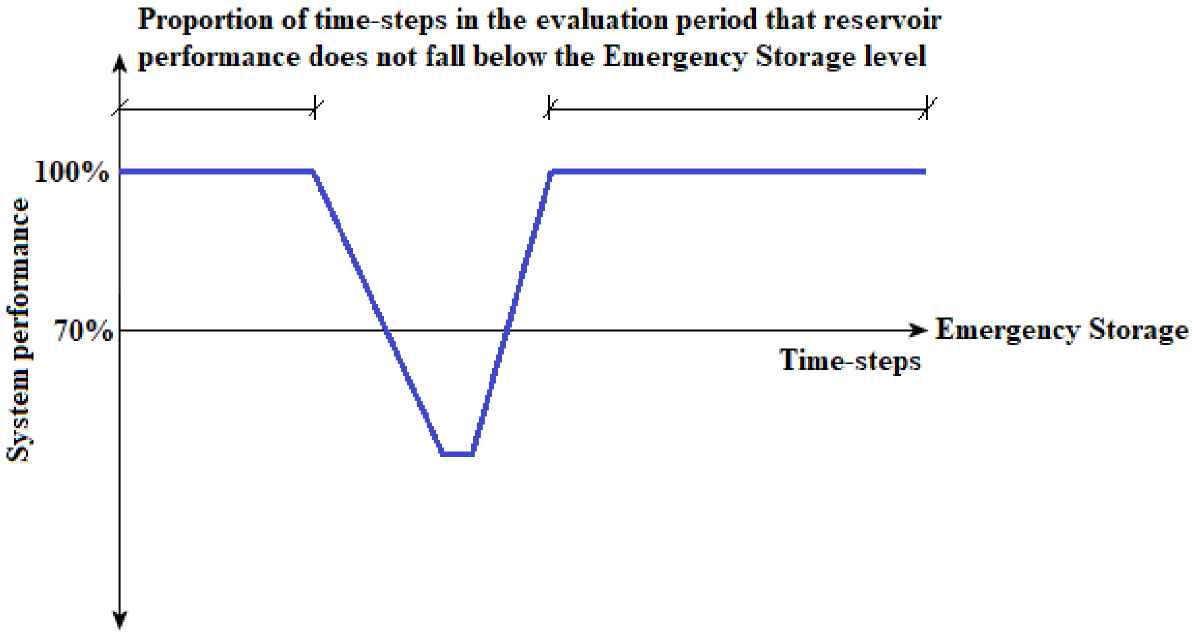


Figure 3.10 Conceptual illustration quantifying the Reliability concept

Lastly, sustainability is conceptualized in this case study in terms of the consequences to the environment in context of minimising abstraction and operating costs (utilised in the Anyplace case studies), and minimising the energy footprint (utilized in the Kingston and St. Andrew case study explored in Chapter 4). However, in the broader sense, the performance indicator could be explored in detail to include the construction costs of a desalination plant and the energy utilized to transport materials to the construction site, and as documented in the research by Lattemann and Höpner, (2008), chemical agents (i.e. coagulants, anti-scalants, anti-foaming agents, etc.) utilised in desalination processes such as reverse osmosis and thermal plants could also be taken under consideration. This, however, depends on the availability of data. A summary of the performance indicators used to assess the performance of the data-rich Anyplace (this chapter), the Anyplace data-sparse (Chapter 5), and the data-sparse Kingston and St. Andrew (Chapter 4) water resources systems are presented in Table 3.6.

Table 3.6 Summary of the performance indicators used in this thesis to define the system impacts and consequences in all of the case studies

Requirement	**Resilience**
Domestic Demand	Minimise largest consecutive failure period (days) Summation of the demand deficit during that period (ML)
Compensation Releases	Minimise largest consecutive failure period (days) Summation of the compensation flow not delivered during that period (ML)
Impact	**Reliability**
System Failure	The proportion of time-steps in the evaluation period that reservoir performance does not fall below the Emergency and dead storage levels
Objective	**Sustainability**
Cost	Abstraction and Operational costs (£) (Anyplace)
Energy usage	Energy used during production; (kWH) (Kingston)

3.5. Implementation

The methodology illustrated below is designed to guide readers through the assessment procedure of the Anyplace water resource system.

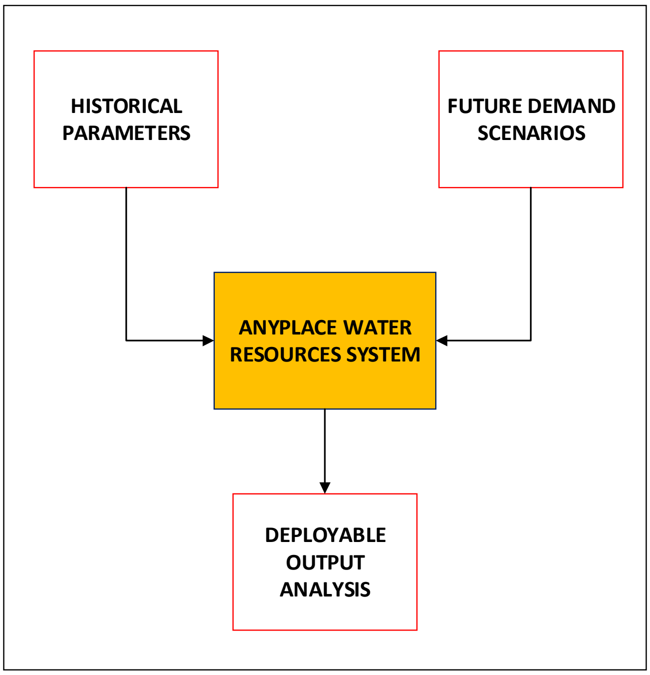


Figure 3.11 Illustration of the parameters used in the deployable output analysis.

The first phase of the methodology in Figure 3.11, explores how the Anyplace water resources system performs when evaluated for historical conditions, and for the plausible future conditions associated with the Future Flows Hydrology. This phase of the methodology allows the researcher to obtain a more precise understanding of how the water resources model performed historically. This is of great importance because the Anyplace Aquator model had been previously calibrated in another (confidential) study. The historical and future flow parameters summarised in Table 3.7 were utilized to perform a deployable output analysis for each of the three levels of demands developed in subsection 3.2.2 (N.B. the Anyplace model was already calibrated and changing any inputs would have required further calibration).

According to Turner *et al*., (2014), "*DO is a measure of the greatest level of compliance with customer demand a water resources system can deliver without incurring failure during the model simulation*". The DO was initially evaluated at five-year time intervals, and additional simulations were conducted to capture the performance for the historical drought periods and dry spells during 1976, 1977, 1984, 1990, 1996 and 2003 (Met Office, 2017).

Table 3.7 Parameters used to evaluate the Deployable Output of Anyplace Water Resources System

DO Analysis	**Demand (ML/d)**	**Parameters (Daily time-steps)**
Low Demand Scenario	45	Precipitation (1960 - 2009); Potential evapotranspiration (1960 - 2009); River flow (1960 - 2009)
Mid-range Demand Scenario	48	Precipitation (1960 - 2009); Potential evapotranspiration (1960 - 2009); River flow (1960 - 2009)
High Demand Scenario	52	Precipitation (1960 - 2009); Potential evapotranspiration (1960 - 2009); River flow (1960 - 2009)

The results of the DO analysis are presented in Table 3.8 and illustrated in Figure 3.12. In Table 3.8, the supply-demand balance for each demand scenario was calculated. In each of the scenarios, there are deficits for most of the periods especially in each of the historic drought years and dry spells. The results for historical demand (32.1 ML/d) were not summarised in Table 3.8 because a supply equals demand result (i.e. a supply-demand balance was achieved) was returned for each period evaluated as seen in the Figure 3.12. Moreover, the year 1996 returned the least desirable results across all three scenarios, and this is an indication that the water resources system in its current state would be unable to handle past experienced events if the demands were greater. This is transparent in Figure 3.13 where the storage volumes for the reservoir systems RV1 (red) and RV2 (blue) are shown. In the case of an extreme demand (i.e. 52 ML/d) the reservoirs were utilizing the emergency storage and the dead storage levels for long periods. It is acknowledged that reservoir operating policies such as hedging would be introduced to limit the severe strain placed on the systems, or on the demand side (i.e. *Temporary Use Bans*) to reduce consumption. From a visual examination of the reservoir system performances, it can be concluded that as the demand increases so does the level of unreliability. Based on the results obtained from the DO analysis, the water resources system will be evaluated for the most extreme demand scenario (i.e. high demand) to determine how the system responds during intervention testing.

Table 3.8 Summary of the deployable output results when evaluated for the three demand scenarios (Ml/d)

Average Annual Estimated Demand for the Low Demand Scenario (45 ML/d)

Year	**1960**	**1965**	**1970**	**1975**	**1976**	**1977**	**1980**	**1984**	**1985**	**1990**	**1995**	**1996**	**2000**	**2003**	**2005**	**2009**
DO	45.00	44.14	39.80	43.55	31.77	32.83	40.29	41.75	44.62	44.97	44.28	30.35	45.00	40.65	45.00	45.00
Demand	45.00	45.00	45.00	45.00	45.00	45.00	45.00	45.00	45.00	45.00	45.00	45.00	45.00	45.00	45.00	45.00
Balance	0.00	-0.86	-5.20	-1.45	-13.23	-12.17	-4.71	-3.25	-0.38	-0.03	-0.72	-14.65	0.00	-4.35	0.00	0.00

Average Annual Estimated Demand for the Mid-Range Demand Scenario (48 ML/d)

Year	**1960**	**1965**	**1970**	**1975**	**1976**	**1977**	**1980**	**1984**	**1985**	**1990**	**1995**	**1996**	**2000**	**2003**	**2005**	**2009**
DO	48.00	45.98	40.23	43.34	32.30	32.80	41.41	42.29	47.10	47.94	46.93	27.64	48.00	37.02	48.00	48.00
Demand	48.00	48.00	48.00	48.00	48.00	48.00	48.00	48.00	48.00	48.00	48.00	48.00	48.00	48.00	48.00	48.00
Balance	0.00	-2.02	-7.77	-4.66	-15.70	-15.20	-6.59	-5.71	-0.90	-0.06	-1.07	-20.36	0.00	-10.98	0.00	0.00

Average Annual Estimated Demand for the High Demand Scenario (52 ML/d)

Year	**1960**	**1965**	**1970**	**1975**	**1976**	**1977**	**1980**	**1984**	**1985**	**1990**	**1995**	**1996**	**2000**	**2003**	**2005**	**2009**
DO	52.00	46.64	40.87	43.28	32.96	33.25	42.61	40.57	49.27	51.82	47.94	25.96	51.94	34.27	52.00	52.00
Demand	52.00	52.00	52.00	52.00	52.00	52.00	52.00	52.00	52.00	52.00	52.00	52.00	52.00	52.00	52.00	52.00
Balance	0.00	-5.36	-11.13	-8.72	-19.04	-18.75	-9.39	-11.43	-2.73	-0.18	-4.06	-26.04	-0.06	-17.73	0.00	0.00

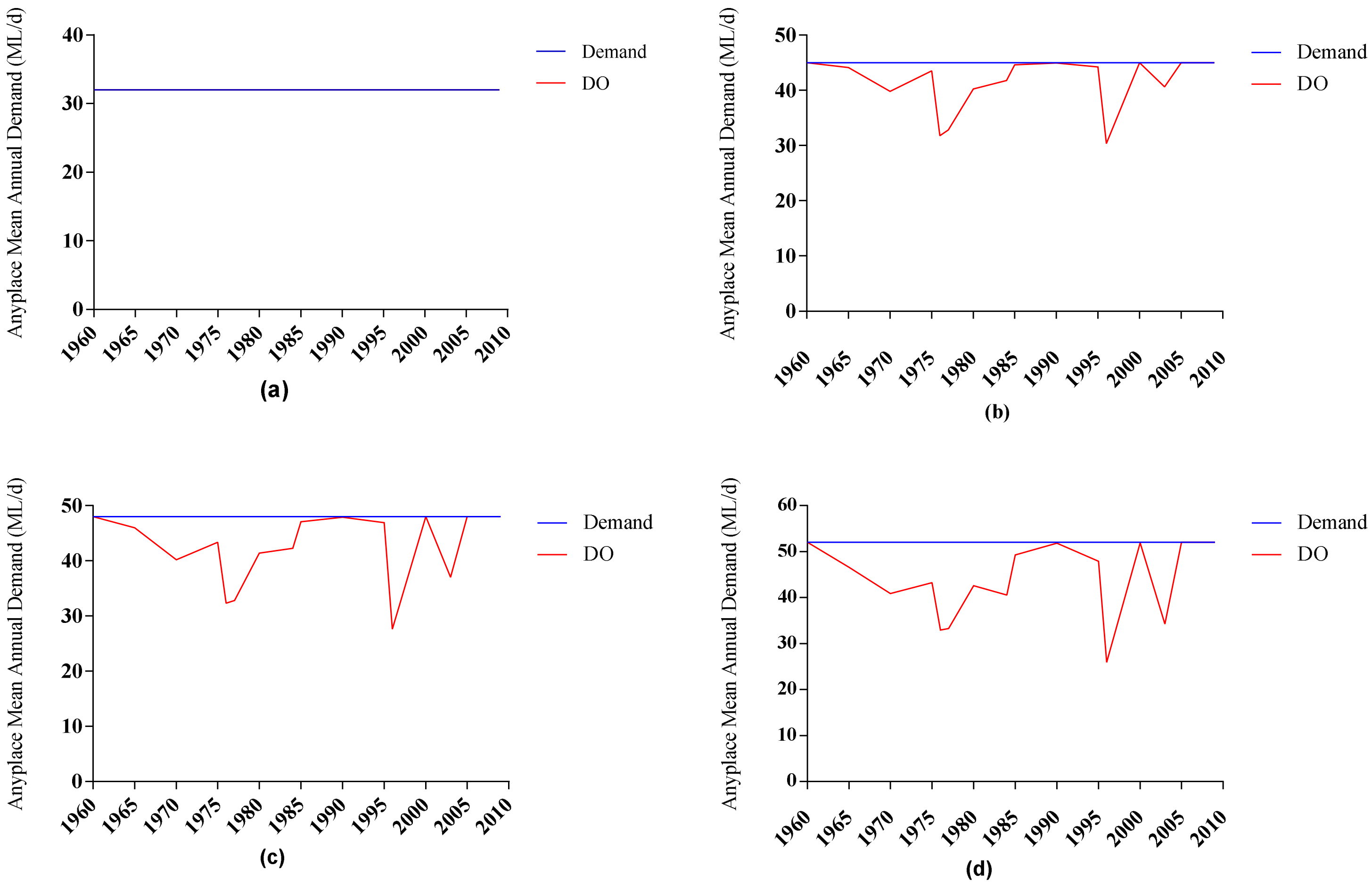


Figure 3.12 Illustration of the Deployable Output (DO) Analysis results for the historical demand (a); low demand (b); the Mid-range demand (c) and the high demand scenarios (d)

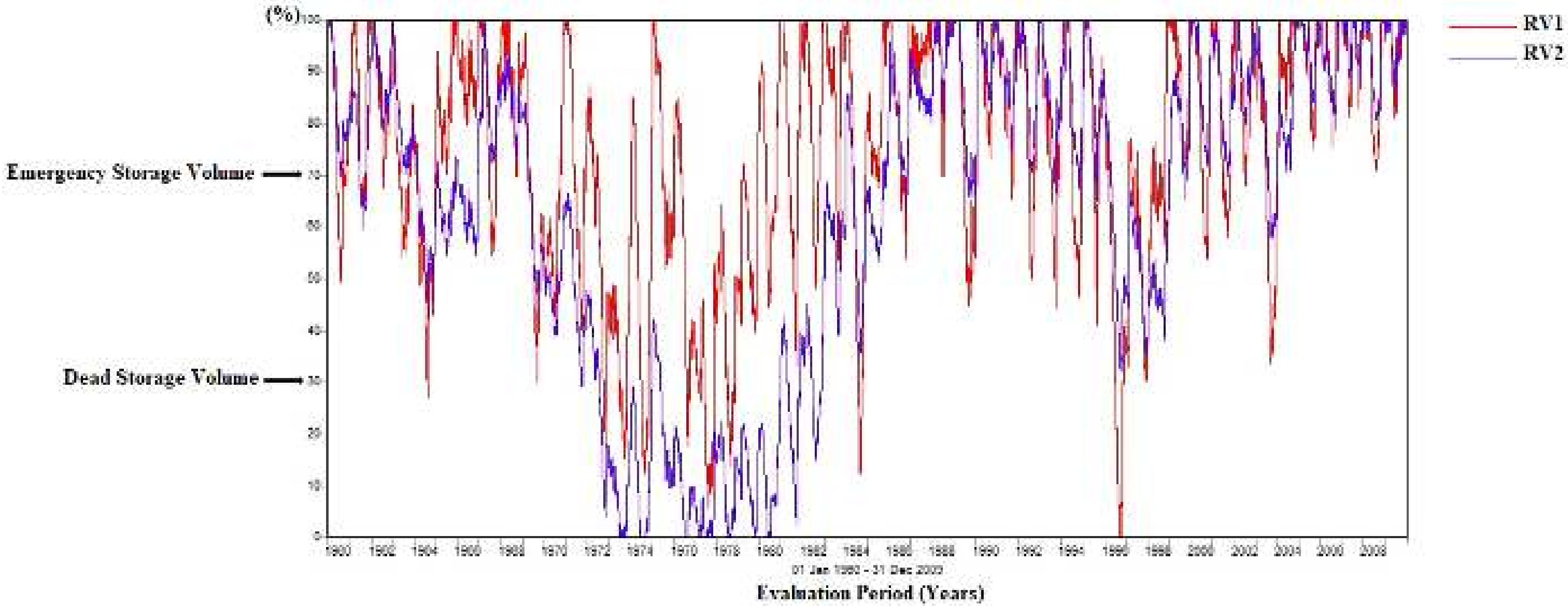


Figure 3.13 Illustration of reservoir storage performance when the deployable output (DO) was evaluated using historical datasets for the high demand scenario (RV1 has a capacity of 7303 ML and RV2 has a capacity of 3909 ML)

The second phase of the methodology is illustrated in the Figure 3.14 and adopts the guidance set out in the Safe and SuRe circular approach explored above in section 3.3.

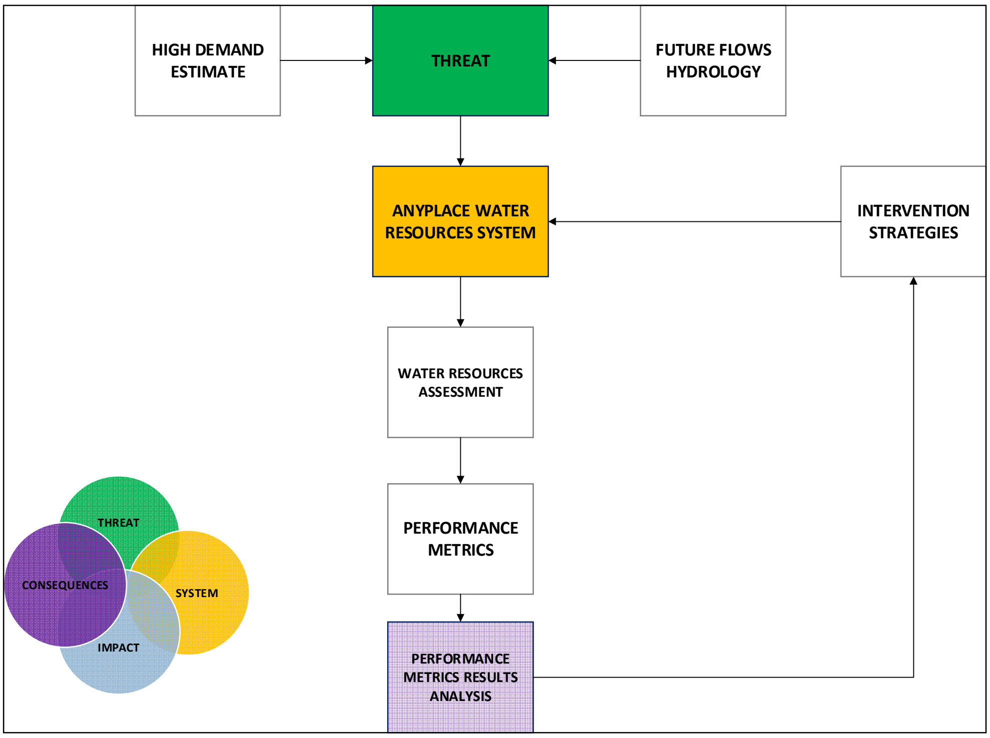


Figure 3.14 Illustration of the Anyplace water resources assessment procedure

The colour coding allows for the link between the Safe and SuRe framework and the steps utilized in the water resources assessment to be established. First, the high demand value and one of the 11 Future Flows is introduced as inputs to the Aquator model. These are represented by the green rectangle in Figure 3.14. Subsequently, an initial simulation is performed for the period 2020 to 2049 without any interventions to establish a baseline of how the water resources system performs under those uncertain conditions. This process is repeated a further 10 times at the same demand, and for each repetition, a different FFH ensemble is utilized as the supply input. Therefore, in total 11 sets of baselines results are returned.

For each of the 11 plausible future baselines, the performance indicators (section 3.4) discussed above are utilized to determine to what extent the system is affected and to quantify the severity of the consequences. This identifies at which time-steps the intervention strategies are likely to be introduced. Guidance on the scheduling of the interventions is summarised in Table 3.5, and the water resources assessment is repeated by testing each of the proposed strategies. If the results from strategy testing show no improvement, then the system is reassessed considering different combinations of intervention strategies. The results of the intervention testing are explored in the following section.

3.6. Results and Discussion for the Social and Environmental Resilience Performance Indicators

3.6.1. No Interventions

This section presents the results of the Anyplace water resources assessment for the High demand scenario with no interventions introduced. In Figure 3.15 a two-dimensional scatterplot of the total failure magnitude (deficit in ML) vs the failure duration (consecutive failure days) is shown. Each of the points represents the longest consecutive period that the residential demand has not been met, and this is represented on the *x*-axis, with the *y*-axis representing the total accumulated demand deficit for the failure duration.

As seen in Figure 3.15, all the FFH points result in high consecutive failure days and high failure magnitudes. The data on the x-axis has been rescaled in Microsoft Excel to the range [0,1]. Moreover, the points at the extreme ends of the axis are the FFH points Q6 and FFH point Q9, represent values of 0.47 (44 consecutive failure days) and 1.0 (94 consecutive failure days), respectively. The corresponding demand deficits for these points are 1748.73ML and 3906.12 ML respectively. The results of the intervention assessments are summarised in Table 3.9, Table 3.10 and Table 3.11 in this chapter, and Table A.1, Table A.2, and Table A.3 in Appendix A. These tables quantify the social and environmental resilience performance indicators in the context of the resilience definition explored in Section 3.4.

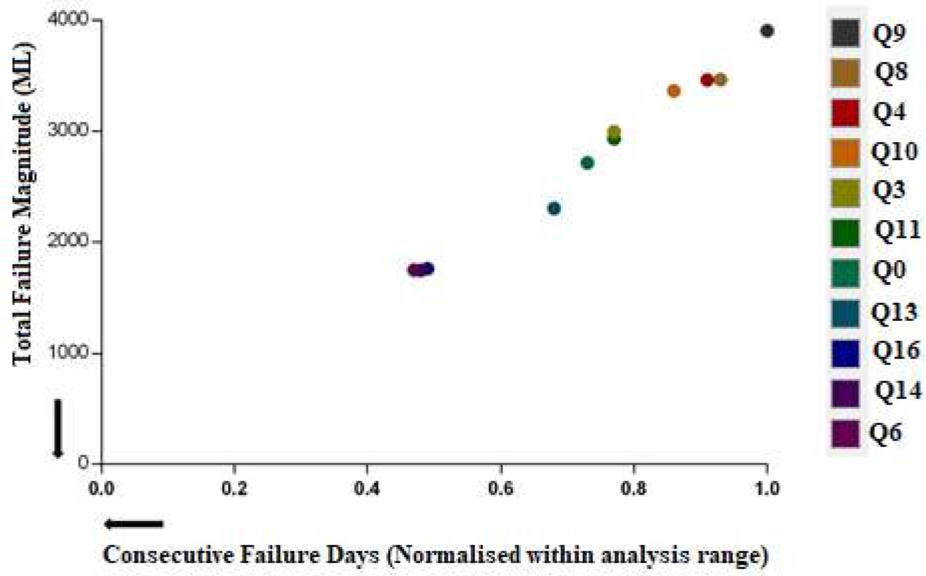


Figure 3.15 Scatter of the results for the social resilience performance indicator illustrating the longest consecutive days during which domestic demands have not been met, as well as the total deficit for the 30-year evaluation period for the *No Interventions* strategy (Q represents a Future Flow Hydrology Ensemble)

Table 3.9 Summary of the Social resilience and the Environmental resilience performance results for the No Interventions strategy assessed for the high demand and the 11 FFH[2].

		No Interventions				
		Social Resilience		**Environmental Resilience**		
FFH ID	**Consecutive Failure Days**	**Magnitude (ML)**	**Duration change**	**Consecutive Failure Days**	**Magnitude (ML)**	**Duration change**
Q0	69	2716.39	-	55	250.03	-
Q16	46	1760.85	-	**18**	**81.83**	-
Q14	45	1745.19	-	27	122.74	-
Q13	64	2303.55	-	37	168.2	-
Q11	72	2932.36	-	69	313.67	-
Q10	81	3364.90	-	80	363.68	-
Q3	72	2995.20	-	72	327.31	-
Q9	*94*	*3906.12*	-	*93*	*422.78*	-
Q4	86	3462.78	-	76	345.50	-
Q6	**44**	**1748.73**	-	29	131.83	-
Q8	87	3464.88	-	60	259.12	-

Table 3.10 Summary of the Social resilience and the Environmental resilience performance results for the Desalination strategy assessed for the high demand and the 11 FFH[2].

			Desalination			
	Social Resilience			**Environmental Resilience**		
FFH ID	**Consecutive Failure Days**	**Magnitude (ML)**	**Duration change**	**Consecutive Failure Days**	**Magnitude (ML)**	**Duration change**
Q0	*63*	*2537.04*	-6	*55*	*250.03*	0
Q16	46	1760.85	0	18	81.83	0
Q14	38	1438.81	-7	27	122.74	0
Q13	39	1611.33	-25	28	127.29	-9
Q11	21	531.60	-51	13	56.18	-56
Q10	38	1455.83	-43	31	139.82	-49
Q3	56	1614.79	-16	46	209.12	-26
Q9	39	1191.38	-55	11	47.00	-82
Q4	**10**	**289.17**	-76	**1**	**4.55**	-75
Q6	33	1116.87	-11	29	131.83	0
Q8	14	344.38	-73	6	27.25	-54

[2] Values in bold show great potential to improve social and environmental resilience, values in italics so the least potential to improve social and environmental resilience.

Table 3.11 Summary of the Social resilience and the Environmental resilience performance results for Demand Management, Temporary Use Bans and Desalination strategy assessed for the high demand and the 11 FFH[2].

	Demand Management, Temporary Use Bans and Desalination					
	Social Resilience			**Environmental Resilience**		
FFH ID	**Consecutive Failure Days**	**Magnitude (ML)**	**Duration change**	**Consecutive Failure Days**	**Magnitude (ML)**	**Duration change**
Q0	*51*	*231.23*	-18	*51*	*231.23*	-4
Q16	25	798.32	-21	12	50.24	-6
Q14	33	1208.70	-12	27	122.74	0
Q13	39	1471.11	-25	29	128.49	-8
Q11	22	445.60	-50	13	56.18	-56
Q10	41	1457.77	-40	31	139.82	-49
Q3	19	615.93	-53	11	46.30	-61
Q9	34	877.98	-60	5	17.32	-88
Q4	6	157.87	-80	1	4.55	-75
Q6	29	1106.64	-15	29	128.49	0
Q8	**2**	**22.58**	-85	**0**	**0**	-60

Referring to the Environmental resilience column in Table 3.9 for the *No Interventions* strategy, the FFH scenario resulting in the lowest number of consecutive failure days is the point Q16 with a normalized (rescaled) value of 0.2 (18 consecutive failure days) as seen in Figure 3.16. The point FFH Q9 represents the worst result returned with 93 consecutive failure days. Coincidently, this FFH also performs the worst with 94 consecutive failure days in the social resilience category (see columns 2 and 5 in Table 3.9).

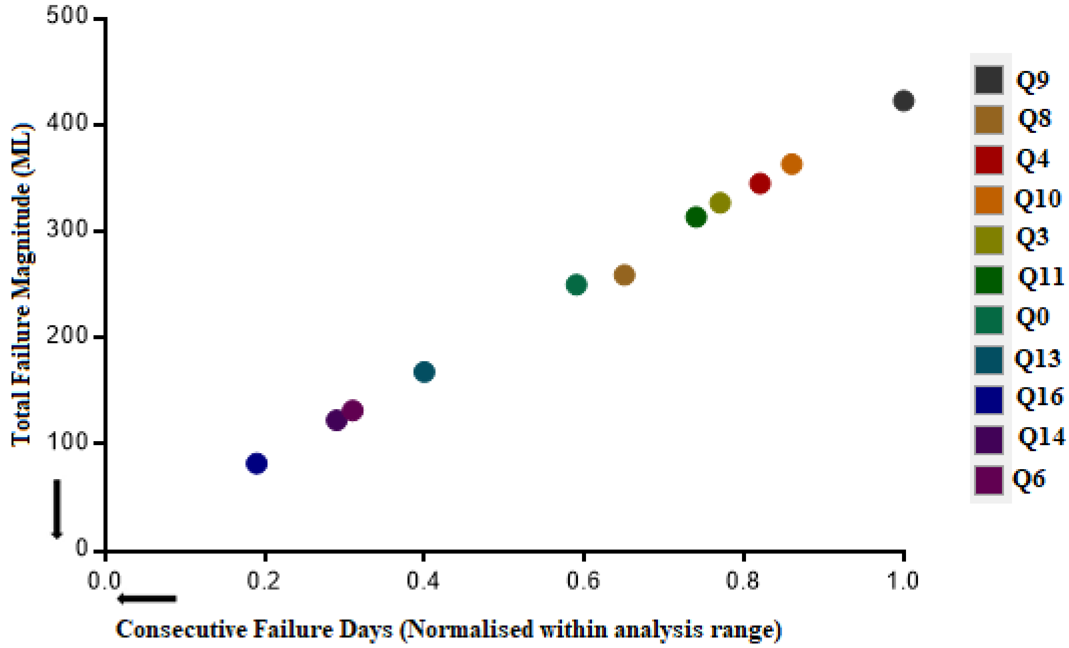


Figure 3.16 Scatter plot of the results for the Environmental performance indicator illustrating the longest consecutive days during which compensation releases are not delivered downstream, and the total deficit for the 30-year evaluation period for the *No Interventions* strategy

3.6.2. Goal-seeking Hedge

A reservoir hedging policy is tested in this case study, and it is categorized as an adaptation strategy in the Safe and SuRe framework as seen in Figure 3.5, to address the system failures. The *Goal-seeking Hedge* is an adjustment made to the standard operating protocol, which is introduced to minimise the extent of a failure affecting the operational capabilities at the reservoir RV1. The results returned for the 11 FFH are summarised in Table A1 in Appendix A, and have been expressed based on the indicators capturing social and environmental resilience. When compared to the *No interventions* results, the introduction of a hedging policy resulted in an increase in the consecutive days of failure and significant increases in the demand deficit, indicating decreased social resilience. The FFH most affected by this intervention is the FFH Q4 owing to some 1573 days increase in consecutive failure days. In contrast, the environmental resilience has shown an improvement under the hedging policy as the number of consecutive failure days returned for each of the FFH points has been reduced. The best result (in terms of minimising the failure magnitude and duration) was returned for the FFH Q8 as full compliance with the required compensation releases was achieved. These results are attributed to the water resources model configuration which has been set to prioritize compensation releases even during the drier periods.

3.6.3. Desalination Intervention Strategy

Regarding the contribution of the desalination to the demand side, a comparison is made between the *No Interventions* (*a*), and the *Desalination* (*b*), for the social resilience performance indicator. This is illustrated in Figure 3.17; and from a visual assessment, the FFH Q4, Q8, Q11 returned the best responses in terms of reducing the number of consecutive failure days and the demand deficit. The remainder of the FFH showed improvements, but a large demand deficit still exists (i.e. in excess of 1000ML). The scheduling criteria in Table 3.5 states that desalination is introduced from the time-step representing the period of the largest degradation in the reservoir (i.e. RV1 because this system returned the larger failure magnitude for all 11 FFH when compared to RV2) system performance. The implication of this is that there are other periods before the introduction of desalination where severe degradation occurs but for shorter durations, and because of this the reliability scores returned for some of the evaluations are low. This is discussed in section 3.7 where reliability is addressed.

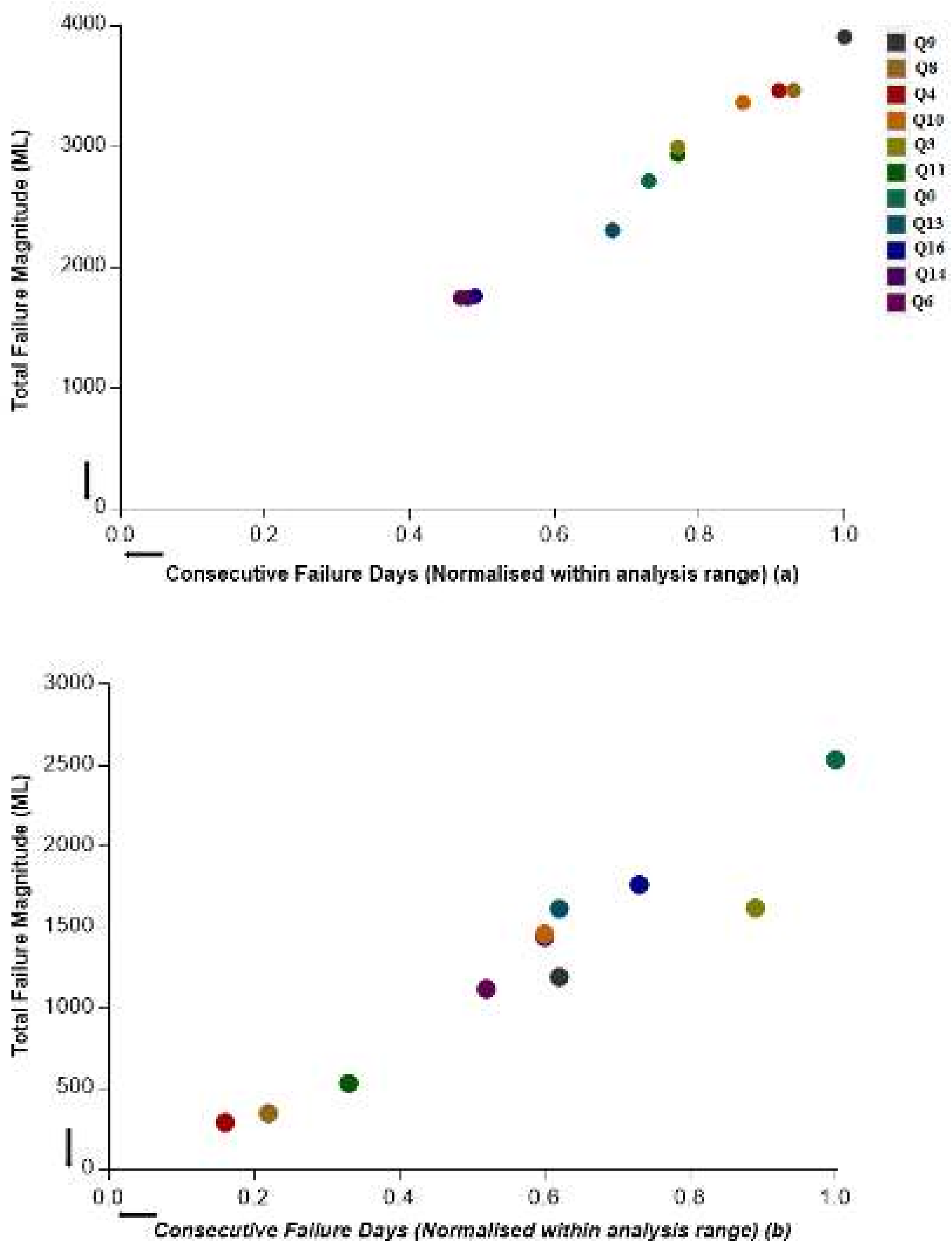


Figure 3.17 Two-dimensional illustration of results of the intervention strategies No Interventions (a) and Desalination (b). Each FFH point represents the failure magnitude and duration for the Social Resilience performance indicator.

A set of positive results were obtained regarding the compliance of compensation releases when desalination was introduced. The introduction of this strategy allowed for more RV1 storage to comfortably meet the compensation releases requirement. The most notable improvements occurred in the FFH Q4 and Q9. These FFH recorded reductions in the number of days where compensation releases were not met by 82 and 75 days respectively, as seen in Table 3.10.

3.6.4. Temporary Use Bans

The *Temporary Use Bans* intervention strategy was introduced in a period preceding two consecutive dry winters. The overall results of this strategy quantifying Social and Environmental resilience are summarised in the Table A2 in Appendix A. When this strategy was tested the most notable contribution was returned for the FFH Q13. A reduction of 18 consecutive days was obtained for the social resilience performance indicator. The remainder of the scenarios resulted in very small reductions in the failure days, but in most of the results, no improvements were achieved in comparison to the *No interventions* strategy. In relation to the compensations flows, the most notable improvement was returned for the FFH Q4 where 12 consecutive days reduction in the failure duration was achieved. In the other scenarios (FFH Q0, Q10, Q11, Q14, Q9 and Q3), this intervention resulted in no improvement.

3.6.5. Demand Management Interventions

The demand management interventions tested in this case study comprised of rainwater harvesting and water saving devices. These addressed the consequences of a demand deficit affecting customers, and are mapped on the Safe and SuRe framework as adaptation strategies (Figure 3.5). These strategies were introduced from the time-step representing the largest consecutive duration where domestic demands were not met. Referring to Table A3 in Appendix A, the most significant improvement obtained from this strategy occurred in the FFH Q8 for the social resilience performance indicator. In this scenario, the consecutive failure duration was reduced by 14 days. The other notable improvements occurred in the FFH scenarios Q6 and Q13, as they both achieved a reduction by 5 days. For the environmental resilience performance indicator, the most improved result when compared to the *No interventions* strategy was obtained for the FFH scenario Q13 as seen in Table A3. The FFH Q13 achieved the largest reduction in the non-compliance with compensation releases by 9 days (i.e. 9 days that compensation releases were not met), however, the FFH Q16 resulted in the lowest consecutive days of non-compliance with compensation releases (12 days).

For all the interventions tested above, desalination returned the best results in terms of minimising the failure magnitude and duration in the context of the social and the environmental performance indicators criteria. The next section seeks to find the best combination of interventions strategies that minimise the impacts and reduces the societal and environmental consequences.

3.6.6. Combination of Interventions

This section presents the results obtained when the interventions described above were tested for a suite of intervention combinations. The six schemes below were modelled following the schedule guidelines set out in Table 3.5, and the results obtained are illustrated in Figures 3.18 and 3.19:

a) *Demand Management and Goal-seeking Hedge*;
b) *Demand Management, Goal-seeking Hedge, and Desalination*;
c) *Demand Management, Goal-seeking Hedge, and Temporary Use Bans*,
d) *Demand Management and Temporary Use Bans*,
e) *Demand Management, Temporary Use Bans, and Desalination*,
f) *Demand Management and Desalination*.

In Figure 3.18, the combined strategies Demand *Management and Goal-seeking Hedge* denoted by (*a*), the *Demand Management, Goal-seeking Hedge and Desalination* strategy denoted by (*b*), and the *Demand*

Management, Goal-seeking Hedge and Temporary Use Bans denoted by (*c*) returned the most undesirable results for the social resilience performance indicator. This is unsurprising because the *Goal-seeking Hedge* has demonstrated in subsection 3.6.2, the tendency to increase the number of consecutive days that the demand is not met. In the strategy *b*, the FFH Q4 (shown in brown) resulted in the lowest total failure magnitude of 91.26 ML over a consecutive failure duration of 206 days. Comparing this result with that obtained in Table 3.9, the Q4 scenario also showed the most improvement for the desalination intervention, but was one of the worst performers for the demand management and the hedging interventions (Table 3.9). This demonstrates the importance of desalination in the FFH Q4 scenario.

Overall the strategy *Demand management, Temporary Use Bans and Desalination* denoted by (*e*) in Figure 3.18 returned the best performance in the context of achieving compliance with the social (resilience) performance indicator criteria. For the strategy *e*, the best performing FFH was Q8 with a total failure magnitude of 22.58 ML over a consecutive duration period of 2 days, and the least favourable results were returned for Q0, Q10 and Q13 which resulted in total failure magnitudes of 231.23 ML, 1457.77ML and 1471.11 ML over consecutive duration periods of 52, 41 and 39 days respectively.

In relation to the environmental (resilience) performance indicator, the *Demand Management, Temporary Use Bans and Desalination* combination strategy denoted by (*e*) in the Figure 3.19 resulted in all of the FFH scenarios achieving the lowest consecutive days of non-compliance with compensation releases when compared to the results in the other strategies (i.e. *a, b, c, d, f*); and in that strategy, the FFH Q8 was the only scenario where full compensation releases were met, and this is followed by FFH Q4 where compensation releases were not met for a total duration of one day. For the *Demand Management and Desalination* scheme denoted by *f* in Figure 3.19, the scenario FFH Q8 was the only scenario to result in full compliance with compensation releases Overall, *Demand Management and Desalination* showed the greatest potential to improve environmental compliance across the 11 FFH scenarios.

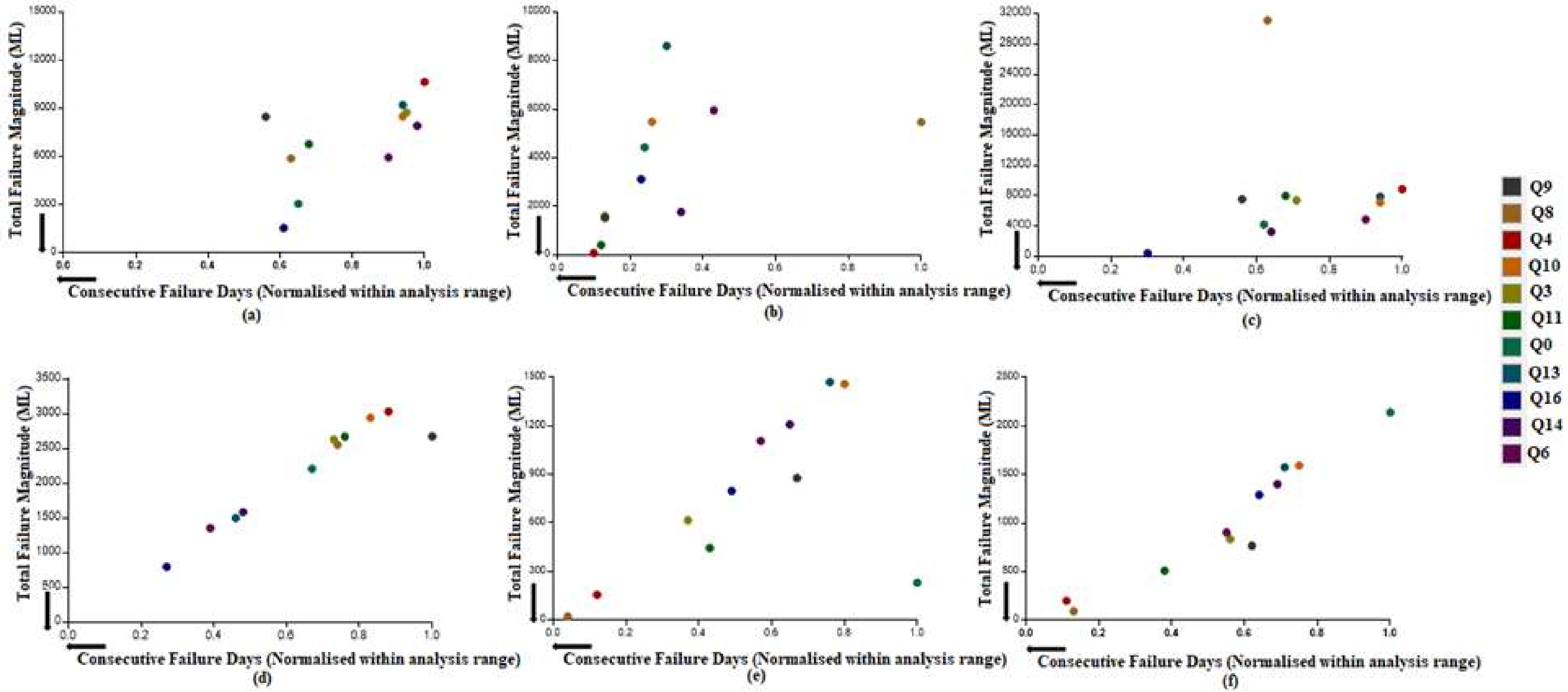


Figure 3.18 Plot illustrating the results of the social resilience performance indicator for the combined suite of intervention strategies a-f[3].

[3] a- Demand Management and Goal-seeking Hedge; b- Demand Management, Goal-seeking Hedge, and Desalination; c- Demand Management, Goal-seeking Hedge, and Temporary Use Bans; d- Demand Management and Temporary Use Bans; e- Demand Management, Temporary Use Bans, and Desalination; f - Demand Management and Desalination.

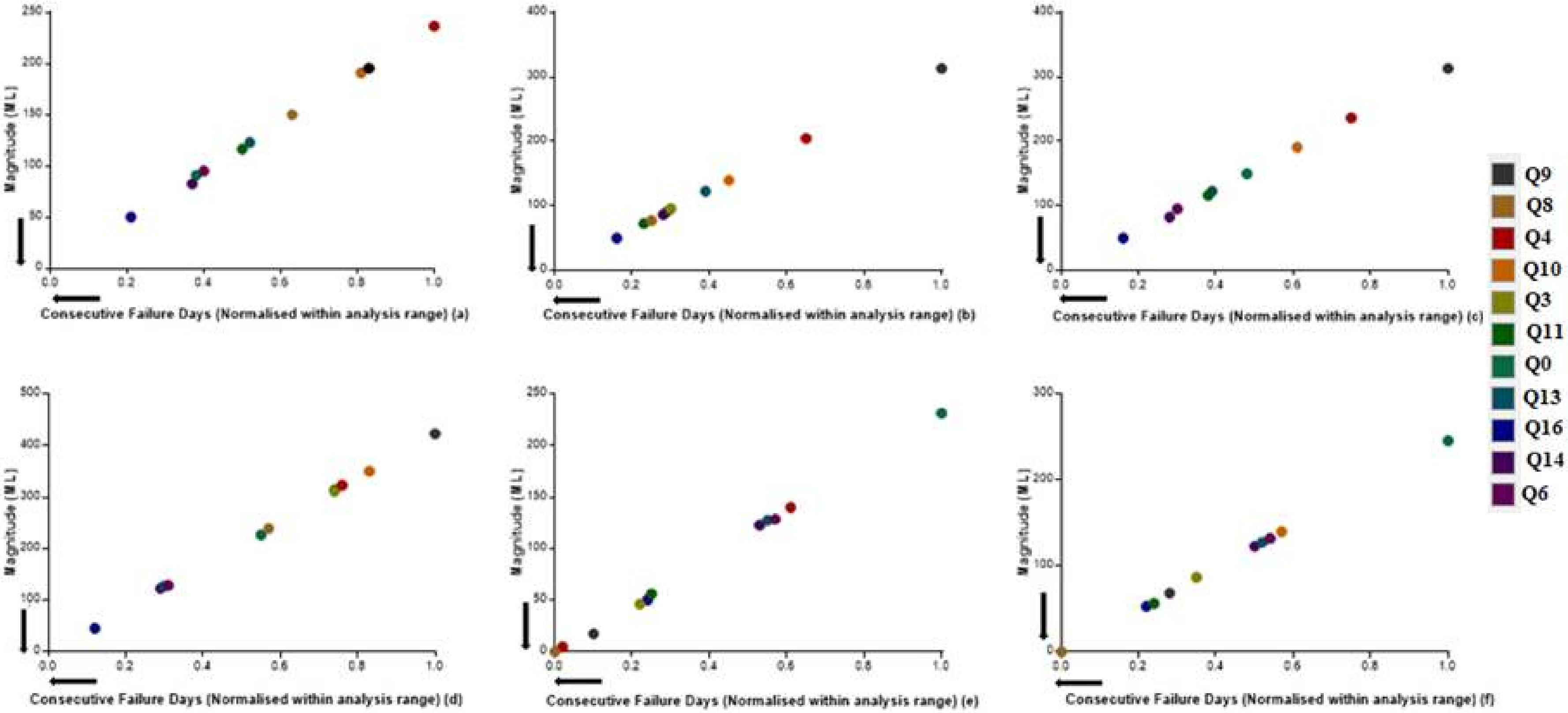


Figure 3.19 Plot illustrating the results of the environmental resilience performance indicator for the combined suite of intervention strategies (a-f)[3].

3.7. Results and discussion for the reliability performance indicator

3.7.1. No Interventions

The reliability of the reservoir systems RV1 and RV2 are graphically illustrated in Figure 3.20. As discussed in Section 3.4, the reliability is expressed as the proportion of time-steps during the evaluation period that the reservoir storage performance is not below the emergency storage level (i.e. defined in this study as 70% of the total storage). In all the FFH, the reservoir system RV2 operates at a high level of reliability. These results are underpinned by the configuration which allows RV2 to contribute 20% of the total supply that is delivered to the WTW to be treated before being distributed to customers; and because of this, RV1 is constantly operating under a high level of stress.

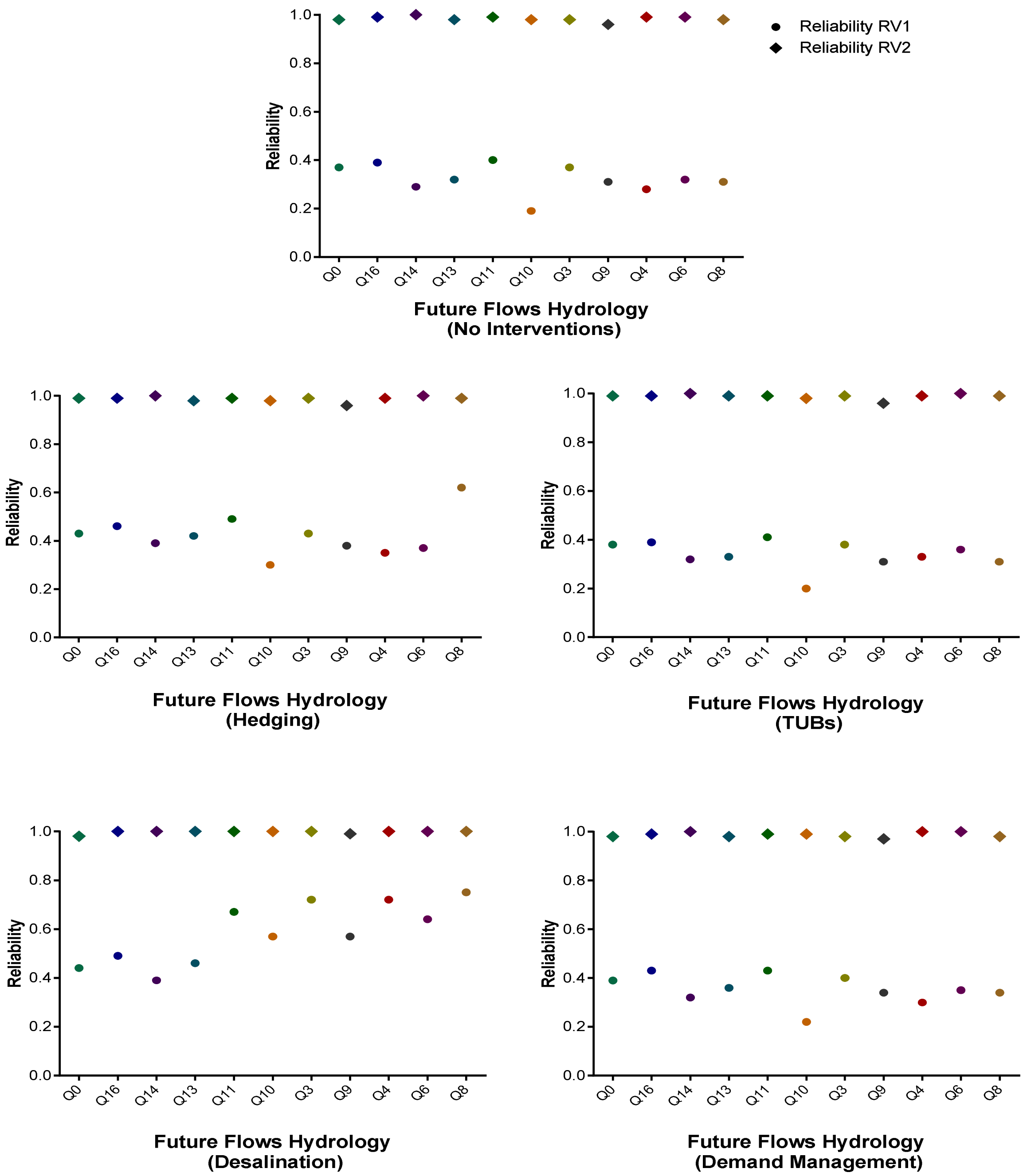


Figure 3.20 Scatter plots illustrating the reliability for the interventions strategies tested for the high demand scenario and the 11 FFH.

In relation to the reliability of RV1, the reliability scores returned for the *No interventions* strategy range from 0.19 to 0.40. This indicates that for most of the time steps of the evaluation horizon, the reservoir system RV1 is in operation below the emergency storage level. Another contributing element to consider is the nature of the hydrological conditions present in each of the FFH scenarios. For example, further analysis of the time series input for the FFH Q10 (this returned the lowest reliability score) in Figure 3.21 shows that there are many instances where the river flows are at low levels for long durations. Referring to Figure 3.21, examples of these low values of flow can be seen at around the time-steps 500, 2000; 3400, 4200, 5100, 6300, 7500 and 10000. The introduction of the hedging policy in the next sub-section demonstrates improved reliability in the presence of these conditions.

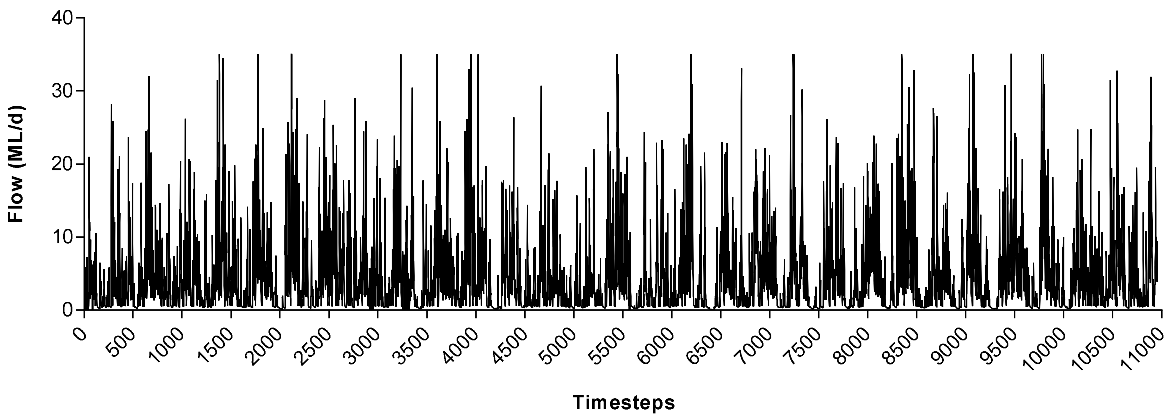


Figure 3.21 Example illustration of the FFH Q10 ensemble timeseries.

3.7.2. Goal-seeking Hedge

The aim of reservoir hedging (*Goal-seeking Hedge*) is to reduce the quantity of water released from the reservoirs systems when affected by extended dry conditions. A clear benefit of the hedging policy is that it allows storage to be conserved in the reservoir for the duration of the dry conditions. This contributes to an improvement in the reliability of the RV1 performance for all the FFH scenarios as seen in Figure 3.20. The FFH resulting in the best (in terms of compliance with the reservoir's defined level of service) reliability score is FFH Q8, for a value of 0.59. This indicates that the reservoir operates above the emergency storage level for most of the time-steps of the evaluation horizon. Despite an improvement in the reliability scores, the FFH Q10 remained at a low level of reliability with a score of 0.30.

3.7.3. Desalination

The *Desalination* intervention was introduced to minimise the longest period the reservoir RV1 is in operation below the emergency storage. Analysis of the system performance results for RV1 reveals that the longest period RV1 is in operation below the emergency storage is different for each of the 11 FFH scenarios. Therefore, for each of the evaluations, desalination is modelled from a different time step in the evaluation period. This strategy has been introduced with a focus on minimising the strain placed on existing supplies of RV1. The contribution of this strategy towards improving the system reliability is summarised in Figure 3.20. All the reliability scores showed small improvements when this intervention was introduced, however, the reliability scores for the FFH Q0, Q14, Q13 and Q16 remained below 0.5, and this means that for those scenarios, the reservoir RV1 is operating below the emergency storage level for a large proportion of the evaluation period.

3.7.4. Temporary Use Bans

The impact of the *Temporary Use Bans* intervention on improving the reliability of RV1 is not significant for across the 11 FFH. The biggest improvement in reliability (when compared to the *No interventions* strategy) occurred for FFH Q4, resulting in an increase in reliability of 0.05 (FFH Q4 reliability = 0.33). In FFH Q8, Q9 and Q16, no improvements were shown. Recall in Table 3.5, where the intervention scheduling was defined, *Temporary Use Bans* is only implemented in the period following two dry winters. Therefore, the impact of this intervention on improving the reliability was not significant as it was not implemented for long durations or extended periods during the evaluation.

3.7.5. Demand Management

The Demand management interventions water saving devices and rainwater harvesting achieved small improvements in reliability for all of the FFH. A factor underpinning this result is that rainwater harvesting has been modelled on a seasonal schedule. The schedule detailing its implementation can be found in Table 3.5. The biggest improvement in reliability recorded was 0.04 (in relation to the *No interventions* strategy) and this occurred for the FFH Q16, which also returned the highest reliability score of 0.49. This means that for approximately half of the evaluation period, the storage volume for RV1 operates below the emergency storage. Overall, the lowest reliability score of 0.22 was returned for FFH Q10 as shown in Figure 3.20. Given that none of the reliability scores has resulted in a significant improvement when compared to the reliability scores for the *No interventions* (Figure 3.20), collectively these are insufficient to relieve the stress exerted on the FFH supplies. As mentioned in Chapter 2, Crow-Miller *et al*., (2017) recommend the implementation of both demand and supply interventions, the implications of which are examined in the following section.

3.7.6. Combination of Interventions

The reliability results returned when the combination of intervention strategies (described above in the social and environmental resilience subsection) was tested and are illustrated in Figure 3.22. Of the 11 plausible FFH tested, the *Demand Management, Temporary Use Bans, and Desalination* intervention strategy returned the showed the greatest potential to improve reliability for the majority of the FFH except Q13 and Q14, where the *Demand Management and Desalination* intervention resulted in higher reliability values. In the *Demand Management, Temporary Use Bans, and Desalination* intervention strategy, the FFH Q4 and Q8 both resulted in the highest reliability scores which were 0.79 for RV1 and 1.0 for RV2, and in the FFH Q14, the lowest reliability score of 0.40 was returned. Despite the low score, the FFH Q14 achieved an improved reliability by 0.11 when compared to the reliability value for the *No interventions* strategy. Outside of the above mentioned combined strategies, the *Desalination* intervention strategy produced the next set of promising reliability values across the 11 FFH. Moreover, the *Demand*

Management and Temporary Use Bans intervention strategy contributed the least towards improving the reliability when tested under the conditions presented in the 11 FFH scenarios. This strategy inherits the low contributions to improving the reliability that is described in the individual *Demand Management* and the *Temporary Use Bans* intervention strategies.

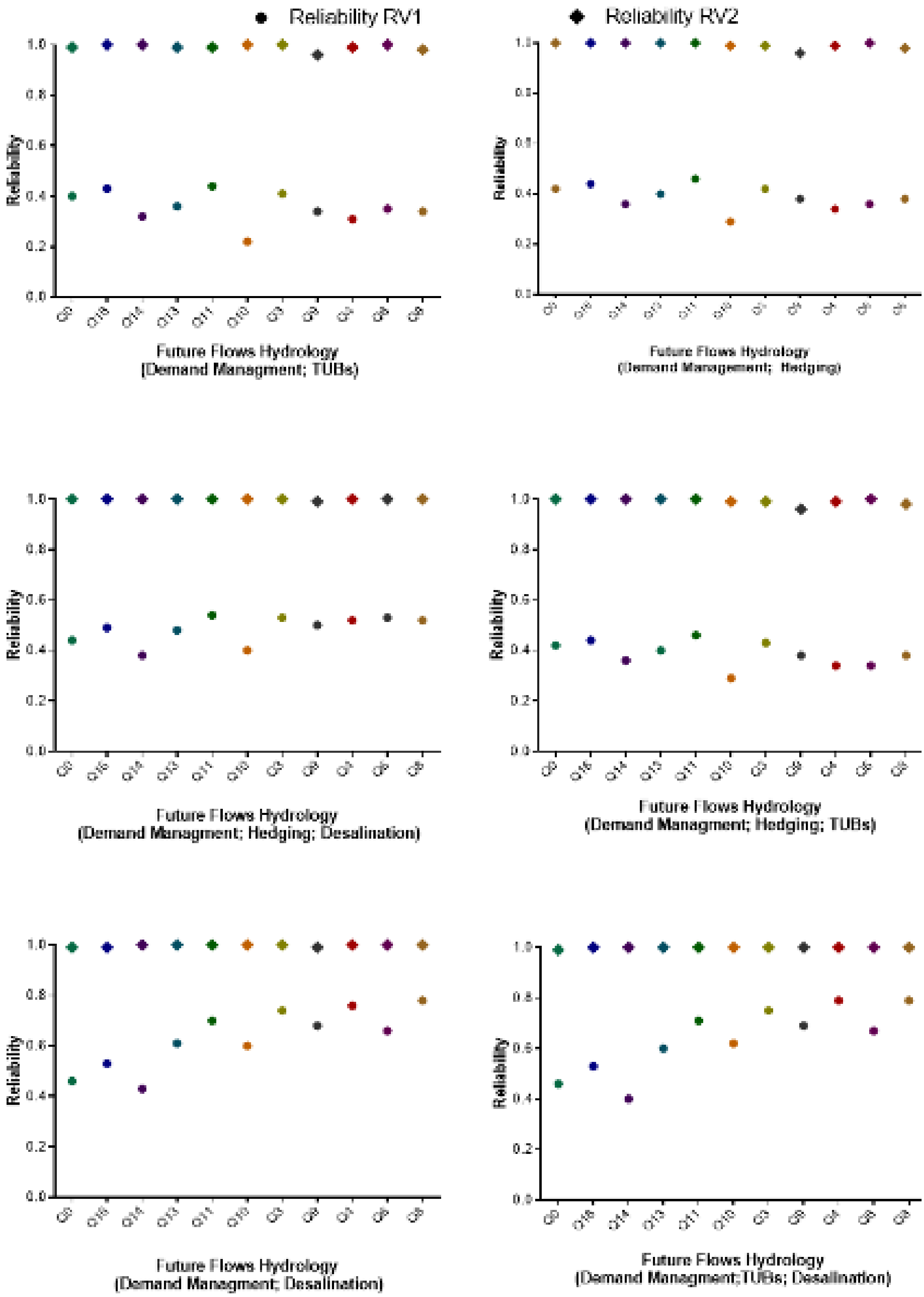


Figure 3.22 Scatter plots illustrating the reliability in each FFH for interventions strategies grouped together to form a suite of combinations that were evaluated for the high demand scenario.

3.8. Results and Discussion of the Sustainability Performance Indicator

In Table 3.12, abstraction and desalination operating (i.e. desalination, and combined interventions with desalination) costs are summarised for all intervention strategies evaluated in the water resources assessment. The values highlighted in bold represent the minimum abstraction and operating costs incurred over the evaluation period, and the values in the italics represent the minimum abstraction costs incurred by interventions not combined with desalination. From a revision of Table 3.12, it is difficult to indicate which of the strategies combined with desalination showed greater potential to improve sustainability, as the strategies *Demand Management*, *Temporary Use Bans*, and *Desalination*, and *Demand management and Desalination* both resulted in low costs. Possible factors contributing to the mixed results include the nature of the hydrological conditions encountered in each of the FFH, and the scheduled implementation of desalination as outlined in the guidelines presented in Table 3.5. Similarly, it was also difficult to indicate which of the strategies without desalination provided improved sustainability as the strategy resulting in the lowest abstraction costs varied in each of the FFH evaluated.

Table 3.12 Illustration of operating costs[4] for prioritised intervention strategies and the combined intervention strategies[5]

FFH	Intervention Strategies Costs £										
	No Schemes	**Hedging**	**TUBs**	**Demand Management**	**DM; Hedging**	**DM; Hedging; TUBs**	**DM TUBs**	**Desalination[6]**	**DM; Hedging; Desalination**	**DM; TUBs; Desalination**	**DM; Desalination**
Q0	701,770	*693,629*[7]	700,464	695,123	693,665	693,665	694,729	15,443,885	15,455,613	15,439,409	**15,434,782**[8]
Q16	746,991	742,826	746,190	743,006	744,303	744,303	*742,660*	13,870,439	13,889,176	**13,862,557**	13,863,754
Q14	772,957	*763,789*	769,120	767,262	765,720	765,720	767,053	13,909,092	13,913,738	13,904,358	**13,899,616**
Q13	694,135	*684,612*	694,205	686,335	687,195	687,195	686,189	59,768,851	59,828,009	59,756,416	**59,755,831**
Q11	657,629	648,201	656,791	648,545	650,544	650,544	*647,605*	59,751,909	59,734,738	**59,734,738**	59,735,478
Q10	702,025	*698,787*	701,986	700,559	699,098	699,098	700,485	69,646,183	69,715,634	**69,617,321**	69,630,222
Q3	747,225	729,583	734,414	730,956	729,945	*729,583*	730,974	82,795,058	82,864,924	**82,759,455**	82,764,976
Q9	747,595	741,723	747,002	742,849	741,009	*684,257*	743,074	72,957,833	73,036,696	**72,934,642**	72,941,893
Q4	688,420	684,339	*682,340*	683,610	684,257	741,009	683,550	72,923,384	72,993,121	**72,897,448**	72,909,708
Q6	696,016	689,339	689,616	688,504	689,808	689,808	*688,094*	53,191,898	53,254,125	**53,172,968**	53,175,167
Q8	665,438	*624,791*	665,331	661,656	662,905	662,905	661,589	82,738,595	13,788,238	13,690,206	**13,688,106**

[4] The operating cost for each time-step of the 30-year evaluation period was calculated by the water resources simulator Aquator. The operating costs in Table 3.12 represents the total accumulated operating costs for the evaluation period.

[5] Operating costs include abstraction cost (£15/ML) provided by the utility company, and treatment cost for desalinated water.

[6] Treatment costs (£1.50/ML) utilised in this thesis are based on the current market price estimate for the treatment of desalinated water In Texas, USA by the Guadalupe-Blanco River Authority.

[7] The values in the italics represent the minimum abstraction costs incurred by interventions not combined with desalination.

[8] Values highlighted in bold represent the minimum abstraction and operating costs incurred over the 30-year evaluation period.

3.9. Multi-criteria Decision Analysis (MCDA)

MCDA is a decision-making methodology where numerical values are employed to compare objectives and alternatives (Li, 2007). This type of analysis is implemented in this thesis as a tool to assist with comparing and validating the results obtained under data-sparse and data-rich conditions, and also to establish a stable platform that allows the interventions to be prioritized.

Decision makers can typically be from the same organizations with the same values or from different institutions working collaboratively on the same project. Lunduka *et al*., (2013) segments stakeholders into three categories: private, public and environmental; which underpins the call for an integrated approach to water management explored in Chapter 2. In this thesis, it is assumed that stakeholders are representative of these categories. A potential hurdle to integrated decision making highlighted in Vogler *et al*., (2017) relates to making decisions when multiple objectives (performance indicators) are to be considered. To prioritise intervention strategies that capture different stakeholder performance preferences, four MCDA weighting combinations are examined in this section as a part of the MCDA methodology illustrated in Figure 3.23. This methodology is based on the Analytical Hierarchy Process (AHP) found in Saaty, (2008) and demonstrates how intervention strategies are selected based on stakeholder preference regardless of the hydrological environment. The AHP considers a set of evaluation criteria (in this thesis referred to as performance indicators), and a set of options (interventions), and the most suitable trade-offs are examined against the different criteria (the proposed weighting combinations).

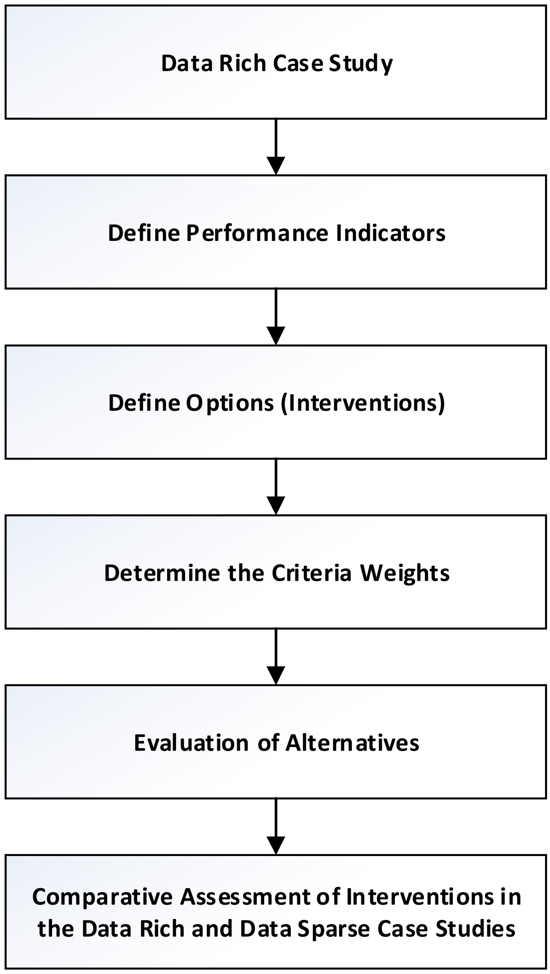


Figure 3.23 Illustration of the steps in the MCDA methodology

Performance Indicators

In the second phase of the methodology, the performance indicators were defined. These were explored in Section 3.4 and are the social and environmental performance resilience indicators.

Recall in section 3.6, the results for the social and the environmental performance indicators were expressed in consecutive failure days (duration), and deficit (failure magnitude). To enable a direct comparison between these performance indicators, and reliability, which is expressed using the scaled range of [0,1] and sustainability, which is expressed in £, the social and environmental and the sustainability performance indicators were scaled within the range of [0,1]. For example, consider the FFH Q0 for *No Interventions* (Table 3.6) which resulted in the longest consecutive failure period of 69 days. The total demand deficit for that period was 2719.39 ML, and the total demand for that period was 3588ML. Therefore, the value of the demand met was determined as follows: ((3588-2719.39)/3588) resulting in a value of 0.24. This approach was also applied in determining the scaled values for the environmental performance indicator. Moreover, the scaled values for sustainability were determined by dividing the abstraction costs returned during the evaluation period by the total abstraction costs the water resources system would incur if the maximum allowable daily volume permitted for abstraction (N.B. recall the abstraction permit limit of 50 ML/d)) was withdrawn each day during the evaluation period.

Intervention Strategies

A simple weighting criteria was applied to evaluate the relative importance of each of the performance indicators that captures different stakeholder perspectives and thinking. In assigning weights to different performance indicators, the sum of all weights should equal to one (i.e. for n = 3 performance indicators, $n_1 + n_2 + n_3 = 1$) (Saaty, 2008).

Under ideal conditions, reliability, resilience, and sustainability should be considered as equally important to the decision-making process. However, considering an integrated approach to water resources management involving stakeholders from different environments, views and opinions on how they prioritize performance indicators could vary. In this thesis, four weighting combinations were examined to provide a simplistic approach for capturing a broad range of stakeholder views and to allow for intervention strategies to be prioritised by ranking them from the highest (1st) to lowest (11th) performing.

In the first weighting combination, it was assumed that stakeholders consider each of the performance indicators to be equally important in the decision-making process. Therefore, the resilience, reliability and sustainability indicators are assigned equal weights (i.e. resilience: 0.33 reliability: 0.33; sustainability: 0.33). The second weighting combination considers sustainability and reliability to be equally weighted (i.e. resilience: 0; reliability: 0.5; sustainability: 0.5), and in the third combination, sustainability and resilience are equally weighted (i.e. resilience: 0.5; reliability: 0; sustainability: 0.5). Lastly, in the fourth combination, resilience and reliability are equally weighted (i.e. resilience: 0.5; reliability: 0.5; sustainability: 0).

The combinations scenarios explored above were utilised to demonstrate trade-offs among the intervention strategies and were implemented using payoff tables (Levine, 2008). In Table 3.13, an example application of weighted combinations is demonstrated. In this example, the fourth weighting combination described above is illustrated, with trade-off decision making demonstrated among the interventions: *Demand Management*, *Desalination*, and *No Interventions*. In this example, the intervention strategies when evaluated for each of the performance indicators returned the results highlighted in italics in Table 3.13. For instance, the weighted score of the *Demand Management* strategy is determined by multiplying the results obtained when assessed for a performance indicator (i.e. the assessment results in section 3.6 to 3.8 for each performance indicator) by the weighted score assigned to that performance indicator (i.e. 0.30 *x* 0.50; 0.35 *x* 0.5; 0.8 *x* 0). The total weighted score is determined as the summation of the weighted intervention values, and this process is repeated for each intervention strategy. The intervention strategy with the lowest total weighted score is the intervention strategy resulting in the highest rank (i.e. 1st). This is due to the implications of sustainability where the emphasis is low operating and abstraction costs.

Table 3.13 Example application of the weighted scoring method

Performance Indicators	Performance Results		
	Demand Management	Desalination	Goal-Seeking Hedge
Reliability	*0.30*	*0.90*	*0.95*
Resilience	*0.35*	*0.65*	*0.40*
Sustainability	*0.80*	*0.60*	*0.50*

Performance Indicators	Weights	Intervention Strategies		
		Demand Management	Desalination	Goal-Seeking Hedge
Reliability	0.5	0.15	0.45	0.48
Resilience	0.5	0.18	0.33	0.20
Sustainability	0	0.00	0.00	0.00
Total Weighted Scores	1	**0.33**	0.78	0.68

3.9.1. Anyplace Data-rich MCDA results

In sections 3.6 to 3.8, the results of the data-rich Anyplace water resources assessment were presented. The results revealed the combined intervention strategy *Demand Management, Temporary Use Bans, and Desalination* showed greater potential to improve reliability, resilience and sustainability. This strategy allowed the WRS performance to achieve low consecutive failure day values, and also low demand and compensation deficits (for the social and environmental performance indicators); and these results are presented in Tables 3.9 to 3.11, Figures 3.18 to 3.19 and in Appendix A (Tables A1 to A3). Moreover, reliability was improved as reservoir levels were maintained. However, mixed results were obtained for the sustainability indicator as other strategies returned lower costs in some of the FFH.

To assist with the selection and prioritization of interventions, the MCDA described above was utilized to analyse the results. Referring to Figure 3.24, the Anyplace data-rich MCDA results are presented. The y-axis indicates the ranks of the intervention strategies from highest (1st) to lowest (11th), and the x-axis lists the names of the intervention strategies.

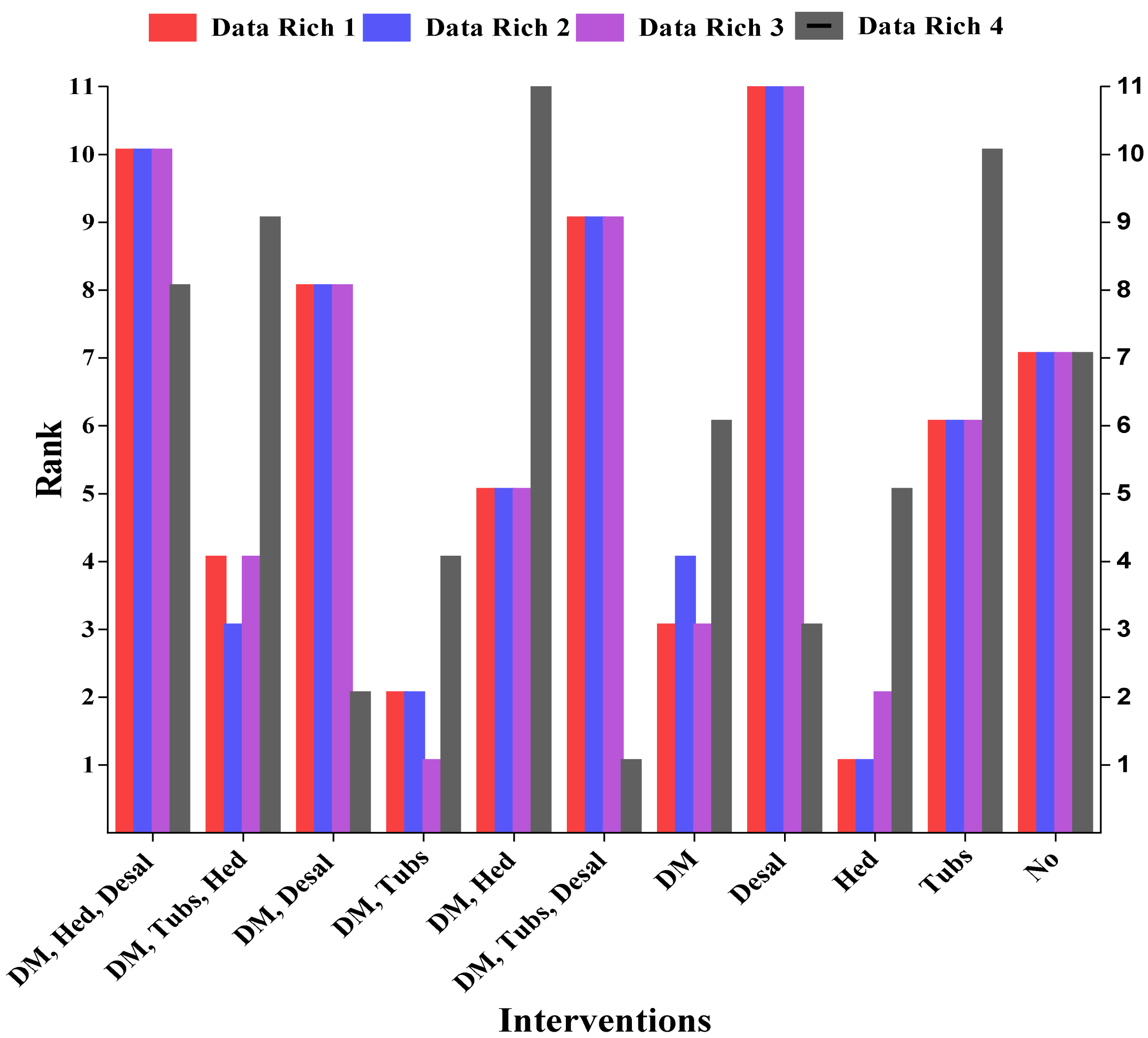


Figure 3.24 Illustration of the MCDA Anyplace data-rich results examining the implications of the reliability, social resilience and sustainability performance indicators[9]

[9] Data Rich 1 – The Anyplace case study evaluated for the MCDA weighting combination 1; Data Rich 2 – The Anyplace case study evaluated for the MCDA weighting combination 2; Data Rich 3 – The Anyplace case study evaluated for the MCDA weighting combination 3; Data Rich 4 – The Anyplace case study evaluated for the MCDA weighting combination 4.

The *Goal-seeking Hedge* strategy resulted in the highest ranked strategy (i.e. 1st) when tested across three of the four weighting combinations, except in the MCDA weighting combination 4, where sustainability was weighted as zero. This occurred as a result of the *Goal-seeking Hedge* strategy allowing for greater volumes of the reservoir's storage to be conserved, and by doing so, this resulted in lower quantities of water abstracted directly from the catchment CM1 (refer to the Aquator model illustration), that led to lower abstraction costs. In contrast, when sustainability was not considered during decision making (i.e. MCDA weighting combination 4), the strategy *Demand Management, Temporary Use Bans and Desalination* was ranked first, and this strategy's result complements those obtained above in sections 3.6 to 3.8 as that strategy was the best from a social and environmental perspective; and also from a reliability perspective. The results for the MCDA weighting combination 4 also showed that when *Desalination* was implemented alone, and combined with other strategies as a part of the planning process, those types of strategies were some of the highest ranked (i.e. 1st, 2nd, 3rd, and 8th in MCDA weighting combination 4). This set of results highlight the importance of sustainability (i.e. in this thesis, sustainability captures abstraction and desalination production costs) in the decision-making process, which is further underpinned by the poor results for interventions comprising of desalination technology across the MCDA weighting combinations 1, 2 and 3.

In the case of the MCDA weighting combination 3 (i.e. when reliability was weighted as zero), the *Demand Management and Temporary Use Bans* strategy showed the greatest potential to improve resilience and sustainability. Based on that MCDA weighting, the implications of this result are as follows:

1) Desalination costs are non-contributory (i.e. the higher the costs, the lower the sustainability);
2) Demand management strategies indirectly result in lower abstraction costs (i.e. assuming customers are more water aware and more water is conserved allowing the WRS to achieve a higher level of compliance with domestic demands). Additionally, this indirectly results in a positive

impact on the supply side, as lower volumes of abstractions occur, resulting in lower costs.

3) The introduction of *Temporary Use Bans* (e.g. hose pipe bans) directly reduces the stress placed on available supplies, by lowering the demand and allowing for an improved level of compliance with domestic demand. The cost implications discussed above in 2 also extend to this implication.

For the MCDA weighting combination 2 where resilience is weighted as zero, the *Goal-seeking Hedge* strategy was also the most promising. The results in sections 3.6 and 3.7 revealed that the *Goal-seeking Hedge* contributed very little towards improving social resilience but significantly towards improving the WRS reliability and sustainability as discussed above. Unsurprisingly, for the MCDA weighting combination 1, the hedging strategy was the most promising, and this is due to the implications of reliability and sustainability explored above.

A consistent result was returned for the *No Interventions* (i.e. do nothing) strategy which was ranked 7 across all of the MCDA weighting combinations. This result implies that from a reliability, resilience and sustainability perspective, doing nothing is not a feasible strategy given the uncertain nature of the FFH and the high demand scenario.

In the context of comparing trade-offs for the environmental performance indicator, and the reliability and sustainability indicators (Figure 3.25), the *Goal-seeking Hedge* strategy was the highest ranked result across all the MCDA weighting combinations except for the MCDA weighting combination 4. This result compliments the implications explored above, and also the valuable contribution of sustainability in the decision-making process (i.e. when sustainability was assigned a zero weighting, the hedging policy showed less potential to improve reliability and resilience as seen in Figure 3.25). Additional implications of this result are:

1) The hedging results in section 3.6 are further underpinned as the results obtained here also allowed for greater compliance with environmental

downstream flows (environmental resilience) which was prioritised during configuration of the Aquator model;

2) Through greater conservation of reservoir releases, hedging enabled an improved level of compliance with the reliability performance criterion to be achieved as the reservoir RV1 remained in operation above the set level of service for longer periods;

3) Desalination type strategies are among the lowest ranked, as treated desalinated water is supplied directly to the Anyplace demand centre, and therefore results in little or no impact towards improving the compliance with compensation release from the reservoir.

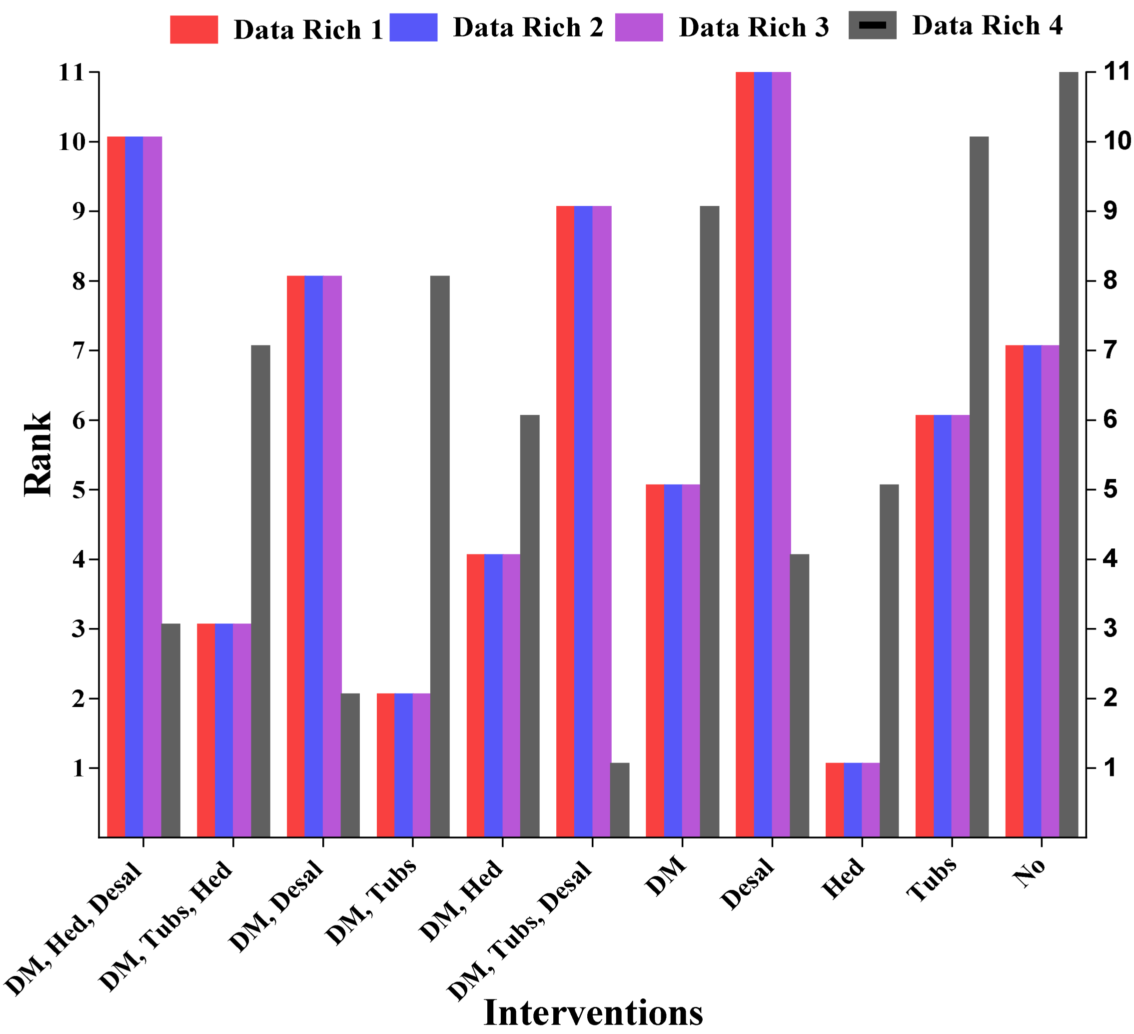


Figure 3.25 Illustration of the MCDA Anyplace data-rich results examining the implications of the reliability, environmental resilience and sustainability performance indicators[9].

3.10. Conclusions

This chapter explored the assessment of the Anyplace water resources system. On the supply side, 11 plausible FFH ensembles of daily river flows were used as inputs for the Anyplace catchments, and these represent multiple possibilities of how supplies in the Anyplace region could be affected by climate change in the future. Moreover, on the demand side, uncertainty associated with how the Anyplace water resources system would respond to a large increase in demand is tested in a High demand scenario. These two extremes are tested together and resulted in a comprehensive water resources assessment.

The assessment results summarised in Tables 3.9 to 3.11 in Appendix B, enable decision makers to envision the undesirable impacts and consequences which could occur in possible futures (i.e. any of the 11 FFH) where '*No Interventions*' are implemented, and also when a range of strategies are tested.

Summarising the '*No Interventions*' results for the social resilience indicator, it was revealed that a large proportion of the Anyplace residents could be severely affected for periods spanning up to 94 consecutive days in the FFH Q9, establishing the conditions associated with this future as the worst-case scenario. Moreover, in the Anyplace region where the downstream waterbody environment is of high importance, the environmental resilience indicator presents decision makers with an indication of how severe conditions downstream of the reservoir could be in the future. This was highlighted in the FFH Q9 which resulted in 93 consecutive days without compliance with the required compensation releases. A valuable contribution of the social and environmental performance indicators is that they provide decision makers with a simple interpretation of the performance results, whereas with some of the resilience indicators explored in Chapter 2, results are communicated as a value between 0 and 1 and not necessarily with a contextual explanation.

Moreover, a decision-making methodology was implemented to achieve Objective 8, that demonstrated a comprehensive approach to assist with the decision making when different performance objectives are of interest to stakeholders. The results of the MCDA showed how strategy selection can be influenced by different decision making perspective, especially in the context of sustainability, which was the most influenced performance indicator.

In the next chapter, the data-sparse Kingston, and St. Andrew case study is explored and evaluated using the Safe and SuRe research framework. This is followed by exploration of the results of an MCDA comparing the Anyplace data-rich case study with the results of the MCDA Anyplace data-sparse study in Chapter 5.

3.11 Limitations

The following limitations were encountered in the Anyplace case study:

- Time – Initially, this research aim and objectives were designed to capture aspects of data sparsity in the islands of Barbados, Grenada and Jamaica, and develop a methodology to assess the reliability, resilience and sustainability of interventions proposed for the implementation in each of those islands. Unfortunately, the researcher was unable to obtain information (and hydrological parameters) that has previously been agreed with the Barbados Water Authority and the National Water and Sewerage Authority in Grenada. Moreover, a considerable amount of time had been invested in gathering information from grey literature and available sources which if successful, would have allowed the researcher to build and characterise water resources systems representative of existing conditions in those islands in Aquator. However, the information gaps in the Barbados and the Grenada cases studies remained challenging to fill, and the decision was made approximately midway through the second year of the PhD course, that these case studies were not comprehensively characterised with evidenced and historical data to facilitate model calibration, and the introduction of scenario representative of plausible conditions for those Caribbean case studies regions. Subsequently, this led to the research on the Anyplace case study which began from approximately midway through the second year of the PhD course.
- Data sensitivity – Another research limitation encountered in the Anyplace case study is associated with the anonymity (i.e. not being able to disclose the exact location) of the study region. This extends to permission not being granted by the water supply undertaker to reveal the names of the facilities (e.g. RV1 – reservoir 1) as that would have placed sensitive information in the public domain (in relation to the performance of their water infrastructure facilities).

CHAPTER FOUR

4. Kingston & St. Andrew Data-sparse Case Study

This chapter provides a detailed description of the Kingston and St. Andrew case study and the materials and methods utilised. The chapter begins by introducing the background and the geophysical characteristics of the case study regions in sections 4.1 to 4.3. In the sections 4.4 to 4.5, the structure of the water resources system is presented. In the subsections 4.6, the demand and supply uncertainties are explored in addition to the intervention strategies. The methods used to frame the case study are also explored in these subsections. In section 4.7, the results and discussions are presented.

4.1. Background

The Caribbean and the Latin American region are one of the most urbanised in the world with 80% of its population currently residing in cities (UNWater, 2017), and according to UNDESA, (2004) urbanisation is expected to increase to approximately 86% by 2050. The island of Jamaica identified by the geo-referencing in Figure 4.1 is in the north-western region of the Caribbean Sea, and is centred on latitude 18°15'N and longitude 77°20'W. It is located 850 km to the south of Cuba, and approximately 1000 km to the North of Panama. It is the largest English speaking island in the Caribbean.

Figure 4.1 Location of the island of Jamaica in the Caribbean (Google Maps, 2017)

The area of Jamaica is approximately 10,990 km^2 and is comprised of 14 parishes as seen in Figure 4.2. Each of the parishes is governed by a local council that make up the administrative government of Jamaica. One of the island's main topographical features is the Blue Mountain range which is a series of mountains located along the West-North-Western to the East-South-Eastern (Hanover to Saint Thomas). The parish of Kingston (and the Parish of Saint Andrew) which is the location of the case study region contains the capital city. It is located to the South-East of the Blue Mountains which is the origin for many of the rivers and springs flowing on the South-Eastern part of the island. According to the National Library of Jamaica NLJ, (2007), there are 28 major rivers across the island. The Rio Minho River in Clarendon is the longest surface water resource with an approximate length of 92.6 Km; and in the case study region, the Hope River which has an approximate length of 19.6 Km, is the longest river.

Figure 4.2 Map of Jamaica illustrating the location of Kingston and suburban St. Andrew (Holness, 2017)

In Jamaica, the city of Kingston and the suburban communities of the parish of St. Andrew (known collectively as Kingston and St. Andrew or KSA) can be considered the megacity of the English-speaking Caribbean. This is attributed to the KSA region being the most densely populated city when compared to other capital cities in the English, Dutch and French-speaking islands. According to the 2010 census, the population of KSA was approximately 25% of the island's 2.7 million inhabitants (Statinja, 2011), a marginal increase of 1.64% from the 2000 census. Socio-economic factors have contributed to the increase in migration from the rural communities to Kingston, and according to Tindigarukayo, (2014) these are "*the underdevelopment of the rural sector in comparison with the urban; lack of a strong peasantry system in Jamaica, capable of keeping small growers wedded to the land; and prevalence of poverty within rural areas, arising mainly from unequal distribution of land in disfavour of small growers*".

4.2. Water Governance in Jamaica

The structure of the governance of water management in Jamaica is illustrated in Figure 4.3. The Ministry of Water, Land, Environment and Climate Change

(MWLECC) is responsible for overseeing the development and evaluation of water resources strategies by the water agencies in Jamaica. Within the portfolio of this ministry, several agencies were created to govern water resources in Jamaica. They are the National Water Commission (NWC), the Water Resources Authority (WRA), National Environment & Planning Agency (NEPA), and the Rural Water Supply Limited (RWSL).

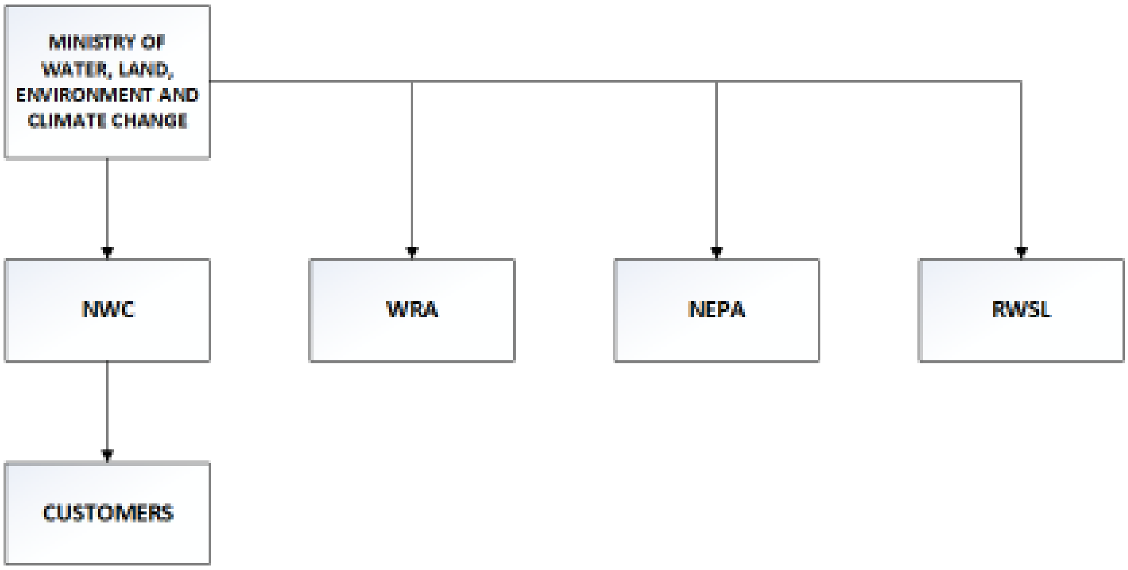


Figure 4.3 Illustration of the structure of Jamaica's water resources sector (Cole, 2013)

The NWC is the agency responsible for the provision of water supply and sewerage services in Jamaica and supplies approximately 16.8 million imperial gallons of water to residents in the KSA WRZ annually (NWC, 2011). The WRA oversees the collection of streamflow measurements for the entire island, which can be accessed from their online database which is referred to as the WebMap (WRA, 2012). NEPA coordinates the management and planning strategies for the island's watersheds, and the RWSL is the agency responsible for the development of water supply infrastructure to the standards set by the NWC and the NEPA.

4.3. Geophysical Characteristics

The geology of the island of Jamaica is made up of three types of rock formations as illustrated in Figure 4.4. They are the quaternary alluviums, tertiary limestones and cretaceous volcaniclastics (Tarr, 1900; WRA, 2016).

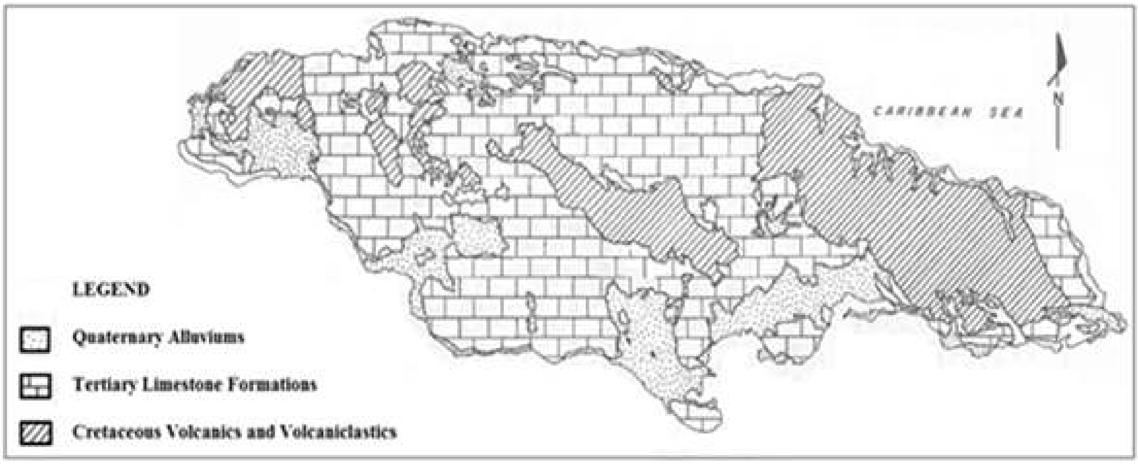


Figure 4.4 Distribution of the types of rock formations across Jamaica (UWA, 1990)

In Table 4.1, the types of rock formations are listed followed by descriptions of their characteristics. These formations are classified into two hydrostratigraphic units, called aquifers and aquicludes (UWA, 1990). Aquifers are defined as rock formations with sufficient permeability and underground storage, meanwhile, aquicludes are defined as rock formations with low permeability that do not support underground storage. The distribution of the island's hydrostratigraphic units are shown in Figure 4.5; and as is illustrated in the figure, the parish of Kingston is predominantly characterised by Alluvium Aquifer/Aquiclude and Basement Aquiclude. This signifies that the water sources in the region are comprised of both groundwater and surface water sources.

Table 4.1 Description of the types of rock formation found in Jamaica. Adapted from (UWA, 1990; WRA, 2016)

Jamaica's Rock Formations	
Quaternary Alluviums	Generally moderate permeability, which occupies about 15 percent of the land area - mainly in the coastal plains and in the floors of interior valleys.
Tertiary limestones	Variably developed karstification and moderate to high permeabilities, which occupies about 60 percent of the land area.
Cretaceous volcaniclastics	Low permeability, occupy about 25 percent of the land area - mainly within inliers along the upland axis.

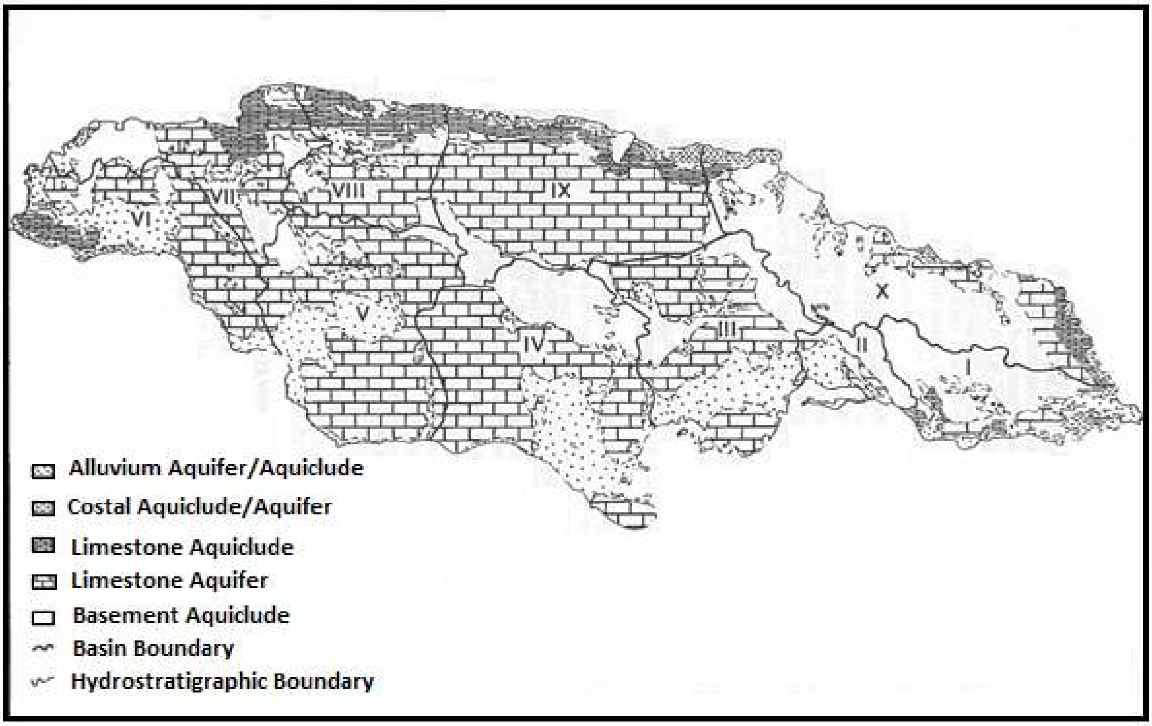


Figure 4.5 Hydrostratigraphic units for Jamaica (UWA, 1990)

4.4. Jamaica and Kingston and St. Andrew Water Resources

Jamaica's water resources are defined by ten hydrological basins or water resources zones (WRZ) as illustrated in Figure 4.6. The hydrological basin which represents the Kingston and St. Andrew parishes is called the Kingston basin and has an area of approximately 25km^2 (WRA, 2012). To provide a sense of clarity and avoid confusion between the parishes and the hydrological basins, the Kingston basin from here onwards will be referred to as the Kingston and St. Andrew Water Resources Zone (KSA WRZ).

Figure 4.6 Illustration of Jamaica’s ten hydrological basins (WRA, 2012)

As mentioned in the subsection 4.3, the water resources in the KSA WRZ comprises of surface water and groundwater sources. In the NWC publication, (1993), it was reported that the Kingston and St. Andrew WRZ accounted for approximately 26% of the total of the island's water usage (Table 4.2). Moreover, the KSA WRZ is primarily dependant on surface water sources, and approximately 75% of its surface water is obtained from the Hope, Wag, Ginger, and Moresham rivers (UWA, 1990); with a significant proportion supplied by the Yallahs pipeline which runs from the Blue Mountain South WRZ to the KSA WRZ (refer to Figure 4.7). In extended dry periods, water is transferred from the Rio Cobre River located in the Rio Cobre WRZ to supplement the deficit affecting the KSA WRZ.

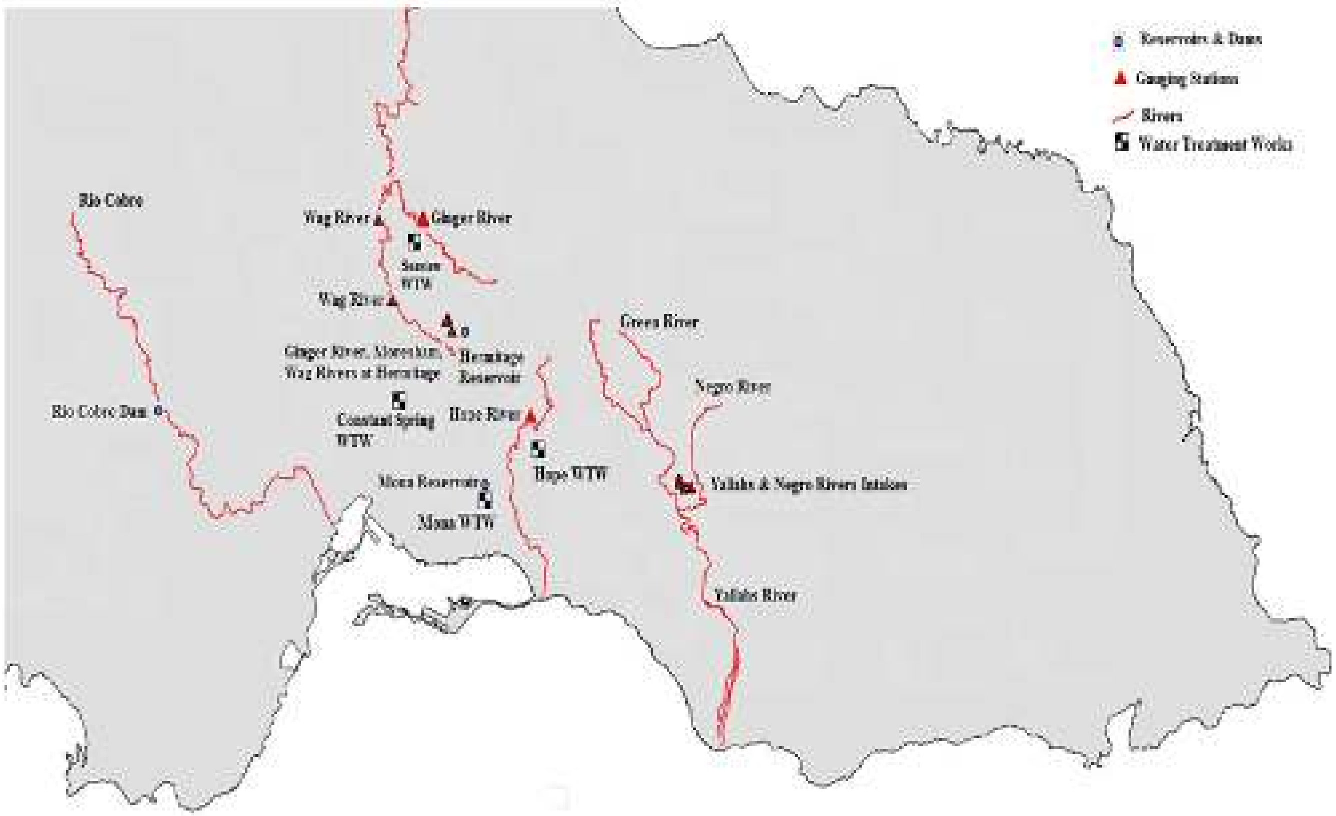


Figure 4.7 Illustration of the surface water sources and major facilities in the KSA WRZ, the Rio Cobre and the Blue Mountain South WRZ (WRA, 2012)

Table 4.2 Breakdown of Jamaica's water resources used in each of the water resources zones (NWC, 1993)

Water resources zone	Water resources usage (%)
KSA	26
Westmoreland	1
Saint Mary	5
Saint Catherine	20
Hanover	1
Portland	3
Clarendon	10
Saint James	4
Saint Thomas	4
Manchester	2
Trelawny	9
Saint Elizabeth	9
Saint Ann	6

The potable water supply in the KSA WRZ is plagued by a high level of non-revenue water (NRW). This is water produced by treatment facilities that is not billed or accounted for, and is attributed to factors such as leakage, illegal connections, unmetered connections, etc. According to Observer, (2017), the current level of non-revenue water is approximately 59%, owing to an increase of 15% from the 2010 figures reported in (NWC, 2011). The cause of this high level of NRW is attributed to ageing infrastructure, lack of investments and poor management of existing resources (NWC, 2011). Additionally, the lack of widespread use of water efficient devices (water savings devices) among the Kingston and St. Andrew residents (surveyed during a fieldwork investigation conducted in January 2016) also contribute to the stresses placed on the potable supply. The approach utilised, limitations and results of the fieldwork survey capturing the residents' attitudes towards water saving devices are all summarised in Appendix B as supplementary information.

In addition to high levels of NRW, Jamaica was joint 1st in a publication by the World Water Institute that lists the top 36 water-stressed countries in the world (WRI, 2013). Despite the high ranking, Jamaica receives an average annual rainfall of approximately 2117 mm which is almost double the amount received in the UK, which is approximated as 1184mm/yr. This is based on rainfall parameters for 1901 – 2015 provided by the World Bank database (The World Bank, 2017b). The island's historic rainfall distribution is illustrated in Figure 4.8, and the northern slopes of the Blue Mountains Range receive the most annual rainfall average for that region which is between the range 3000 to 5000 mm/yr. In contrast to the southern side of the island, the island's annual rainfall varies between 1000 to 2000 mm/yr.

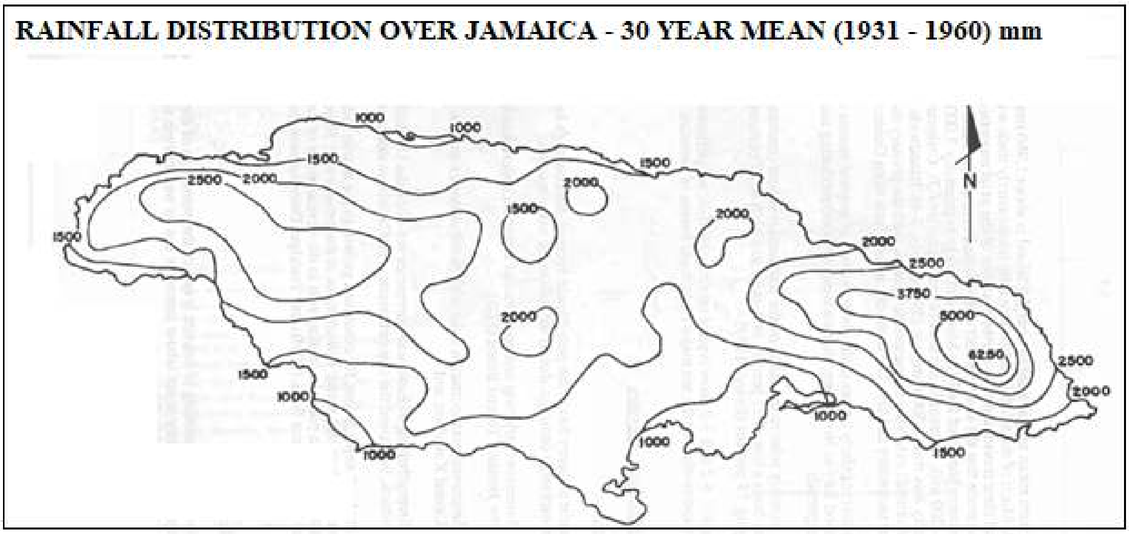


Figure 4.8 Distribution of the rainfall across Jamaica (Evans and Vickers, 1973)

Groundwater abstractions account for the remaining 25% of potable supply in the KSA WRZ. However, owing to high levels of abstractions at depths of up to 8 meters below the sea level between the period 1930 and 1973, fresh and salt water have interfaced, which has seen a rise in the chloride levels up to 200mg/l (Cashman, 2014). The Ferry Springs which are located on the border of the Rio Cobre and the KSA WRZs are the most contaminated groundwater source in the region with a saline water content of 600 – 1200 ppm CL (UWA, 1990).

4.5. Description of the Kingston and St. Andrew Water Resources System

As mentioned in Section 4.4, the KSAWRZ is supplied with water predominantly from surface water sources due to high levels of saline intrusion affecting the groundwater sources. Referring to Figure 4.7, water is supplied from the Yallahs and the Negro Rivers in the South-eastern part of the island by the Yallahs pipeline. The Yallahs pipeline was initially constructed in 1986 to provide an average annual yield of 87 megalitres per day (ML/d). However, its capacity is now limited to 75 ML/d due to the ageing of the pipeline (NWC, 2016), and recent historic inflows from the pipeline are estimated at approximately 64 ML/d (NWC, 2016). Inflows from the Yallahs pipeline are conveyed 30 km directly from its intakes which are located in the Blue Mountain South WRZ (Figure 4.6), to the Mona water treatment works (WTW) (capacity 56 Ml/d) (NWC, 2016), and it also has a by-pass valve which allows for water to be conveyed to the Mona Reservoir.

The Mona reservoir is the largest reservoir on the island with a capacity of 3062 ML. In addition to occasionally being supplied with water from the Yallahs pipeline, the reservoir is predominantly filled with supplies from the Hope River by the Hope Aqueduct, which has a capacity of 212 ML/d. The Hope River which is the largest river in the KSA WRZ also supplies the Hope WTW with supplies (design capacity 30 ML/d). No information was available on the WebMap portal WRA, (2012) regarding the upper abstraction limit for water taken from the Hope River in order to maintain a sustainable ecological health.

In the northern region of the KSA WRS, the Seaview WTW (capacity 8 ML/d) is fed directly with raw water from the Ginger River, which also supplies the Hermitage reservoir (capacity 1437 ML located in rural St. Andrew) by the Ginger River Pipeline. The Hermitage reservoir was constructed at the confluence of the Moresham and Wag Rivers. Releases from the Hermitage reservoir are conveyed directly to the Constant Spring WTW facility (capacity 75ML/d) in the northwest of Kingston. To the west of the KSA WRZ, the

Ferry/Rio Cobre pipeline which intakes raw water from the Rio Cobre River in the Rio Cobre WRZ assists the KSA WRS with meeting water requirements in extended or drought conditions by transferring 15ML/d to that WRZ (NWC, 2011). An Aquator schematic representation of KSA WRS can be found in Figure 4.9.

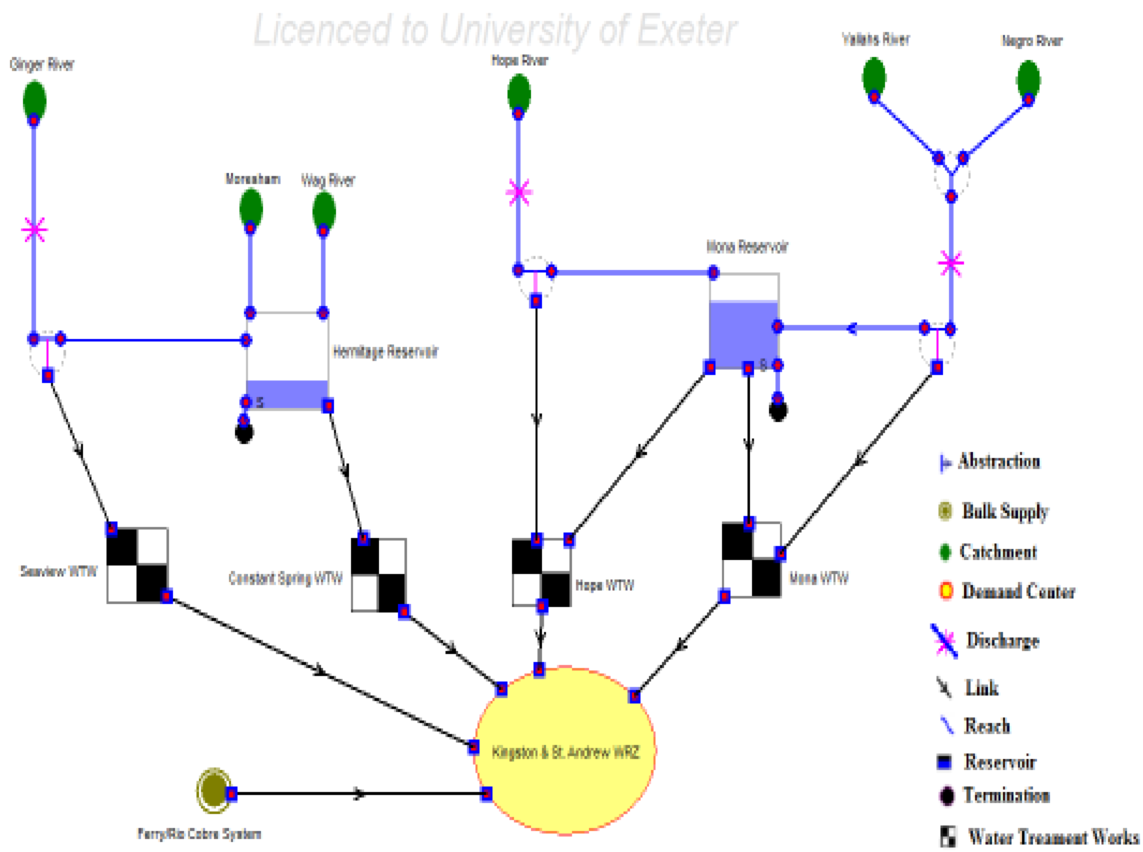


Figure 4.9 Schematic representation of the full Kingston & St. Andrew Water Resources System components in Aquator

4.6. Framing the Kingston and St. Andrew Case Study Problem

4.6.1. Classification of the datasets: The Webmap Database

The WRA's Webmap portal is an online database that contains archives of historical hydrological parameters for the island of Jamaica. In Figure 4.10, an illustration of the Webmap platform is shown. The Webmap database is intended to provide users with a comprehensive characterisation of hydrological parameters for the entirety of Jamaica. The types of parameters accessible from the database range from historical measurements of rainfall, streamflow and groundwater abstractions, to the location and historical storage performance of dams and reservoirs.

A major obstacle encountered from working with the Webmap database is that a large proportion of the hydrological parameters have not yet been assigned any datasets. More specifically, in the KSA WRZ, the Hope River is the only surface water source that comprises of relatively consistent historical records (i.e. from May 1955 to December 2016, a total of 67 daily streamflow measurements are missing). The remainder of the rivers that have archived data are burdened with large gaps between the daily and/or monthly measurements. Moreover, there are no records characterising the historical performance of the reservoir systems. Hence the KSA WRS is characterised as data-sparse due to the inaccessibility and inconsistency of data through the recognised authorities and in the public domain.

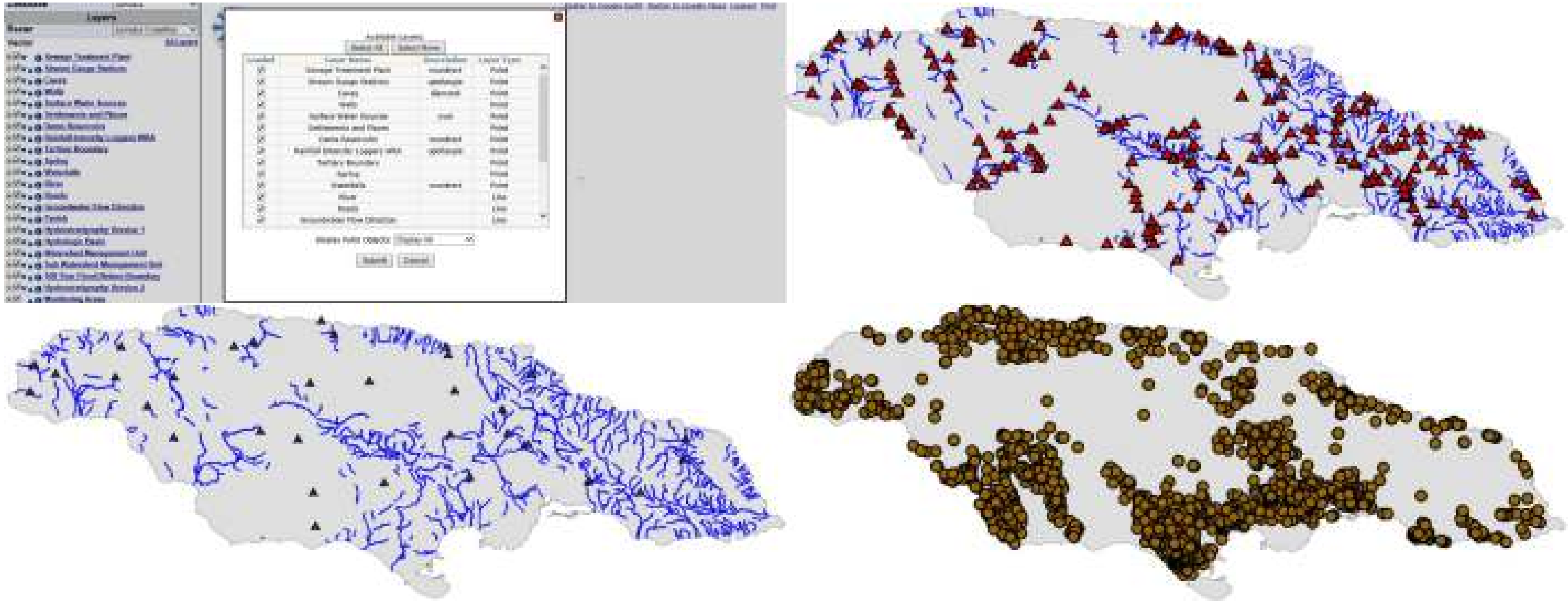


Figure 4.10 Illustration of the WRA Webmap database characterised by the types of datasets available (top left), river networks and gauging stations (top right), river network and rainfall intensity loggers (bottom left) and water wells (bottom right) (WRA, 2012)

To overcome this data sparsity and fill in the data gaps encountered in this project, the researcher pursued three approaches. In the first approach, the researcher travelled to Kingston to conduct fieldwork in January 2016. However, the fieldwork was unsuccessful which can be attributed to a lack of cooperation by the NWC[10].

In the second approach, a process called Parameter Regionalisation was attempted to fill the gaps. Parameter regionalisation is explored extensively in the following studies (Croke and Norton, 2004; Götzinger and Bárdossy, 2007; Visessri and McIntyre, 2016) however, limited success was achieved from that process as regionalisation is more suitable for implementation for catchments within the same local geographical region (i.e. for example, catchments in the south-east of England are more suitable to regionalise catchments in the surrounding areas). In the final approach, climate projections of meteorological data were obtained for the KSA WRZ and discharge for the Hope River was generated through a rainfall transformation process explored in section 4.6.8. The final approach was used in combination with the Aquator model to assess the KSA WRS. As a part of this process, demand and supply uncertainty were examined, and these are explored in the subsections 4.6.2 and 4.6.3 respectively.

4.6.2. Demand Uncertainty

This section describes how the demand value which is used as the demand input was derived. In this study, half of the KSA WRS is modelled and assessed due to the lack of consistent parameter sets that are required to characterise the entire water resources system. According to (Observer, 2017), the Mona and the Hope production and distribution facilities supply approximately 50% of the domestic demand to customers in the KSA WRZ. Coincidentally, the Hope region and parts of the Mona region are the better-

[10] The fieldwork to Jamaica was planned on short notice (i.e. within approximately eleven days), and the WRA was the only local organisation to acknowledge the researchers visit to the island. Officials at the WRA attempted to organise meetings with water professionals and also site visits to infrastructure (e.g. the Hermitage Reservoir and several water treatment facilities) owned by the NWC through email and telephone correspondence to the NWC head office. The emails sent to the NWC officials were continually forwarded to persons whom the researcher was informed was the official that would be able to grant permission to visit NWC sites, and fill in the missing information gaps. Further details can be found in Section 4.9.

characterised regions found in the Webmap database. For these reasons, the Mona and the Hope regions of the KSA WRS will be assessed at this level, and this system is hereafter referred to as KSA50.

The value used to represent the demand for these two regions is taken from a study conducted by the NWC to estimate the future demand in the KSA region, and this is found in the Kingston and St. Andrew Water Supply Plans document (NWC, 2011). The projected demand for the KSA water resources system in the year 2030 was estimated to be 140 ML/d, therefore the demand used for Mona and the Hope region (referred to as KSA50) was 70 ML/d.

4.6.3. Supply Uncertainty

This section describes the supplied layout of the KSA50 WRS and the parameters required to characterise the supply side of the WRS.

Parameters required to characterise individual components of the KSA WRS were not available, and therefore the rainfall-runoff transformation process explored in the subsections 4.6.4 to 4.6.6 was used to generate discharge to characterise the WRS. Meteorological parameters are utilised during the rainfall transformation process, and these are obtained from the climate projections for Kingston, which are accessible from the Caribbean Community Climate Change Centre (CCCCC) database. The type of parameter generated for the case study was discharge for the Hope River catchment (i.e. the main river in KSA50). Therefore, the uncertainty associated with how the effects of climate change could impact on river flows is considered in the water resources assessment.

4.6.4. Rainfall-runoff Transformation Process

Due to the data-sparse nature of the KSA50 region, there was significant supply uncertainty and data gaps. A rainfall-runoff transformation was used as a process to close these gaps as far as possible to enable the system performance to be assessed using Aquator. Rainfall transformation involves the simulation of the mechanics of the natural water cycle, where the output of interest derived from this process is discharge (Mackay *et al*., 2014). Rainfall-runoff models are categorised into five types: i) simple empirical models, ii) large-scale energy-water balance equations, iii) conceptual ‘lumped' iv) landscape daily hydrological models and, v) fully distributed physically based models (Vaze *et al*., 2011). Empirical models capture the transformation process through the use of empirical equations to establish regression relationships. In the large-scale energy-water balance equations model, the Budyko concept is applied to estimate the catchment's evaporation as a function of the aridity index (Gerrits *et al*., 2009). Moreover, the landscape daily hydrological model conceptualises the characteristics of a catchment such as its geology, soil types and land use, etc, by often employing the use of digital elevation models to capture all of these processes (Vaze *et al*., 2011). These processes are modelled using the equations found in the literature (Liang *et al*., 1994). Physically based models are the most complex types of rainfall-runoff models and a large number of parameters typically ranging from 10 to 1000 have to be quantified in order for the model to be run Vaze *et al*., (2011); and this type of model is predominantly utilised in flood forecasting studies, as exampled in (Liu, Martina and Todini, (2005) and Sharif, Al-Zahrani and Hassan, (2017). They are based on the "*understanding of the physics of the hydrological processes which control the catchment response*" (Vaze *et al*., 2011), and use complex physically based equations found in (Beven, 1989) to define the transformation process. Lastly, conceptual rainfall-runoff transformation models, also referred to as lumped conceptual models, employ spatially mathematical equations which represent the hydrological process at the input and the output of the conceptual stores, as runoff is generated within the catchment (Fawthrop, 1994). The concept lumped indicates that the properties of the catchment remain the same throughout the transformation process (Fawthrop, 1994), and according to Vaze *et al*., (2011) its resulting performance has been classified as variable when these types of models are utilised for hydrological conditions different to those encountered during model calibration.

Despite the shortcomings of conceptual models mentioned above, these (conceptual rainfall-runoff) models have widely been utilised in the literature to conduct a rainfall transformation. According to Crawford and Burges, (2004) the Stanford Watershed Model was the first mathematical model in the 1960's. The model structure represents a catchment using four stores (surface, soil, groundwater and river routing). A minimum set of inputs are required to define this model which are precipitation and evapotranspiration. However, the assumption is made that the hydrological inputs are uniform for the entire catchment, and therefore average values are used to define the model equations (Dawdy and O'Donnell, 1965). A literature search revealed that this model has not been utilised in recent hydrological studies, however, an example of its application can be found in (Egbuniwe and Todd, 1976) where it was used to test the correlation between gauged and ungauged catchments. Another rainfall-runoff model used to test the correlation between gauged and ungauged is IHACRES. It is used extensively for the procedure known as parameter regionalisation which has been applied to the studies found in Hansen *et al*., (1996), Anderson and Goodall, (2006), and Lalozaee, Pahlavanravi and Bahreini, (2013).

The Rainfall-runoff Library (RRL) was developed by (eWater, 2004) and generates time series of daily catchment runoff. The inputs required are time series of precipitation and evapotranspiration, and historical time series of discharge which is used for a goodness of fit compared with the simulated time series of flow produced during transformation. The RRL is comprised of a database of lumped rainfall-runoff models through which calibration of parameters is performed manually or by calibration optimisers. These models are the: 1) Australian Water Balance Model (AWBM), 2) Sacramento model, 3) Simhyd model 4) Soil and Moisture Accounting model (SMAR) and, 5) the Tank Model. Referring to the data sparsity challenge described above in this chapter, a significant factor that prevented the use of any of the fives models above in this research relates the lack of data that is required to define site-specific characteristics of the KSA catchment (e.g. baseflow coefficient, infiltration coefficient, etc.). For instance, in the Sacramento model, seventeen parameters are required to define the transformation process, and the modeller must also set an upper and lower value for each of the parameter values. In the case of AWBM, eleven parameters (upper and lower limit values) must be defined, and in the Simhyd and the SMAR, nine parameter values (upper and lower limits values) must be defined. However, in the case of SMAR, detailed knowledge of soil characteristics is also required. In relation to the Tank model, sixteen parameter values (upper and lower limits values) are required to define the transformation process.

Table 4.3 Summary of database of lumped rainfall-runoff models in the RRL

	Australian Water Balance Model (AWBM)	**Sacramento model**	**Simhyd Model**	**Soil and Moisture Accounting model (SMAR)**	**Tank Model**
No. of parameters	11	17	9	9	16
Time-steps	Daily and hourly	Daily	Daily	Daily	Daily
Stores	3	5	3	3	4
Inputs	Daily rainfall and evapotranspiration	Daily rainfall and evaporation	Daily rainfall and areal potential evapotranspiration	Daily rainfall and evaporation	Daily rainfall and evaporation
Outputs	Surface run-off	Surface run-off	Surface run-off	Surface run-off, groundwater discharge, evapotranspiration and leakage from the soil profile	Surface run-off

After a review of rainfall transformation models, it was decided that the NAM rainfall-runoff model a product within MIKE DHI database will be utilised. The NAM model requires that upper and the lower limits for nine parameter values be established. However, a key characteristic of the NAM model is that the upper and lower limits have already been defined for each of the parameter values as illustrated in Table 4.4. Therefore, in the absence of detailed information in relation to site-specific characteristics of the KSA catchment, it is possible to achieve a goodness of fit between the historical time series of flow and the simulated time series by utilising automatic and manual calibration which is discussed in the subsection 4.6.6. In the following subsection, the NAM rainfall-runoff model is explored in detail.

4.6.5. MIKE DHI and the NAM Rainfall-Runoff Model Structure

The MIKE product suite is a water environment management software developed by the Danish Hydrological Institute (DHI). It is comprised of several modelling modules which are categorised in terms of water resources management, coastal and sea management, urban cities management and groundwater and porous media analysis. The NAM (Nedbor Afstrommings Model) rainfall-runoff model, which is part of the water resources management module, was used in this study to simulate the inflow discharge into the Hope River at a daily time step for the 2020 to 2036 horizon.

The NAM rainfall-runoff model is a deterministic, lumped and conceptual model that continuously accounts for the water content in four interrelated storages (Nielsen and Hansen, 1973). These storages are illustrated in Figure 4.11 and are snow storage, surface storage, root zone storage and groundwater storage.

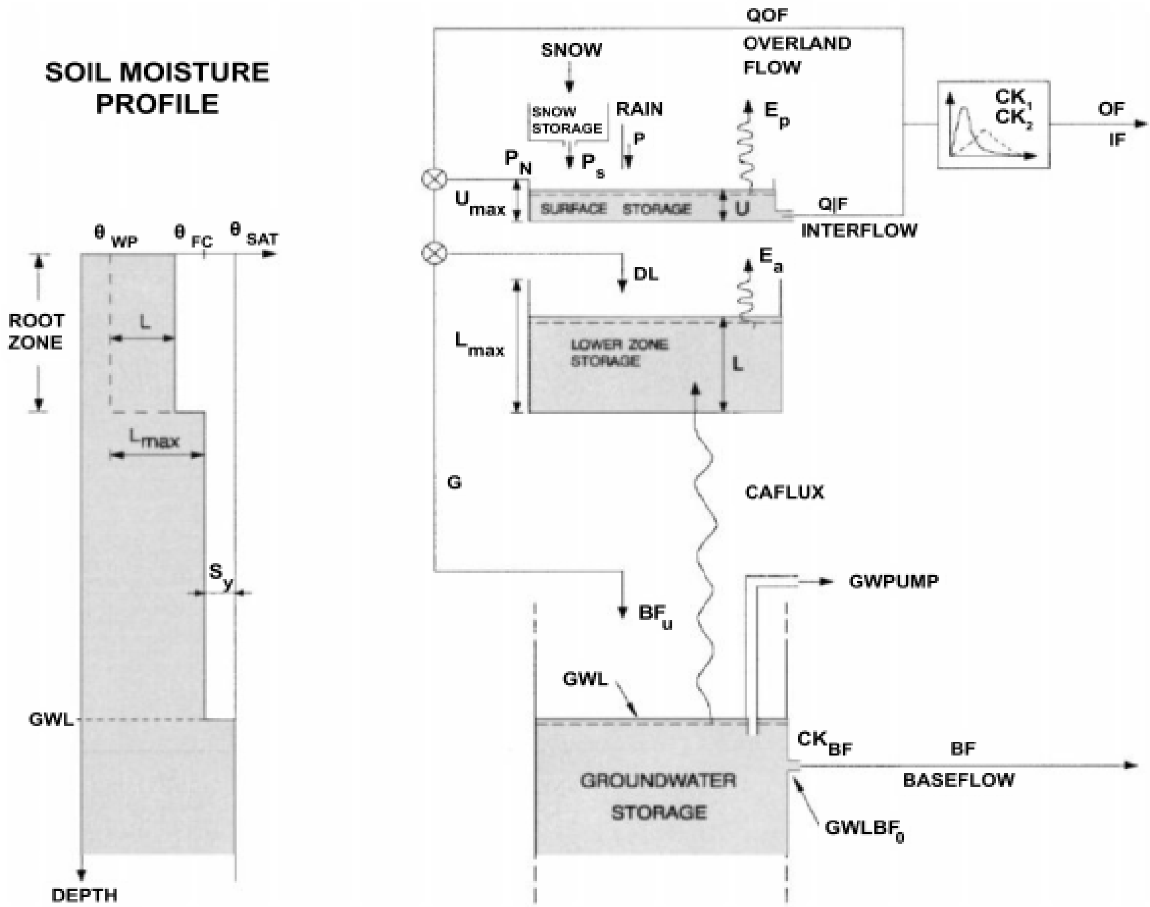


Figure 4.11 NAM rainfall runoff-model structure (Agrawal and Desmukh, 2015)

As mentioned above, the NAM rainfall-runoff model is characterised by 9 parameters that represent the physical elements of a catchment (Table 4.4). The snow storage is not considered in this study because the climate of Jamaica is tropical for the entirety of the year. The parameter U_{max} denotes the maximum water content in the surface storage. When the parameter *U* is more than U_{max}, the excess water P_N produces overland flow *QOF* and infiltration *DL* as seen in Figure 4.11. The interflow is determined from two linear reservoirs CK_1 and CK_2 with the same time constants (DHI, 2003). The parameter L_{max} denotes the maximum water content in the root zone storage. The value of L_{max} is dependent on site-specific soil characteristics of the root zone, and the value of *L* is dependent on the quantity of water that the vegetation uses during the transportation process. The amount of water recharging in the groundwater storage is denoted by the parameter *G* which is dependent on the nature of the soil composition in the root zone, and the amount of water available in that storage. The amount of baseflow (*BF*) discharged from the groundwater storage is dependent on the parameter *G*, and is discharged in relation to a time constant *CKBF*. A description of the parameters and their threshold values are presented in Table 4.4.

Table 4.4 Parameter description and boundary conditions for the NAM rainfall-runoff model (DHI Software, 2003)

Parameter	Unit	Description	Storage	Lower Bound	Upper Bound
Umax	mm	Maximum water content in surface storage	Surface	10	20
Lmax	mm	Maximum water content in root zone storage	Root zone	100	300
CQOF		Overland flow coefficient	Surface	0	1
CKIF	Hours	Interflow flow coefficient	Surface	200	1000
CK1,2	Hours	Time constants for routing overland flow	Surface	1	50
TOF		Root zone threshold value for overland flow	Surface	0	0.99
TIF		Root zone threshold value for interflow	Surface	0	0.99
CKBF	Hours	Time constant for routing baseflow	Groundwater	1000	4000
Tg		Root zone threshold value for groundwater recharge	Root zone	0	0.99

4.6.6. NAM Model Calibration Datasets

The input parameters required to calibrate the NAM model are historical time series of precipitation, potential evapotranspiration, and discharge. The precipitation time series used in model calibration was obtained from Webmap for the Wakefield weather station (some distance from KSA50) and is representative of the four-year period 2009 to 2012. The potential evapotranspiration parameters were obtained from the Norman Manley weather station and are representative of a 30-year monthly average for 1981 to 2010. The time series of discharge for the Hope River was obtained for the Webmap database for the period 2009 to 2012. These were the only input parameters available for the island of Jamaica, and were sourced the Webmap database (WRA, 2012), and The State of Jamaica Climate report (UWI, 2012). that In Figure 4.12, the parameter locations with respect to the KSA WRZ are illustrated and Table 4.5 shows the height above sea level for each of the parameter locations. This has implications that in each of the parameter locations, different climatic conditions could be present.

Figure 4.12 Illustration of the site locations for each of the parameters used in the model calibration (Google Earth, 2017)

Table 4.5 Height above sea level for historical parameters used as inputs for the NAM calibration

Parameter	Location	Height Above Sea Level (m)
Discharge (Hope River)	Gordon Town KSA	608
Precipitation	Wakefield	120
Potential Evapotranspiration	Norman Manley Airport	5

4.6.7. NAM Model Climate Change Datasets

Daily time series of future climate change precipitation and evapotranspiration projections for Kingston, Jamaica for the period 2020 to 2036 were used as inputs into the NAM rainfall-runoff model. The precipitation and evapotranspiration datasets are available from the Caribbean Community Climate Change Centre (CCCCC) portal. These datasets were produced using the ECHAM5 General Circulation Model (GCM). The ECHAM5 GCM model "*simulates local scale climate features such as orographic precipitation, extreme climate events and regional scale climate anomalies at high spatial (between 12 – 50 km) and temporal (daily time step) resolutions*" (Mugume *et al*., 2013). The spatial resolution available for the Caribbean region is 25 km as illustrated in Figure 4.13.

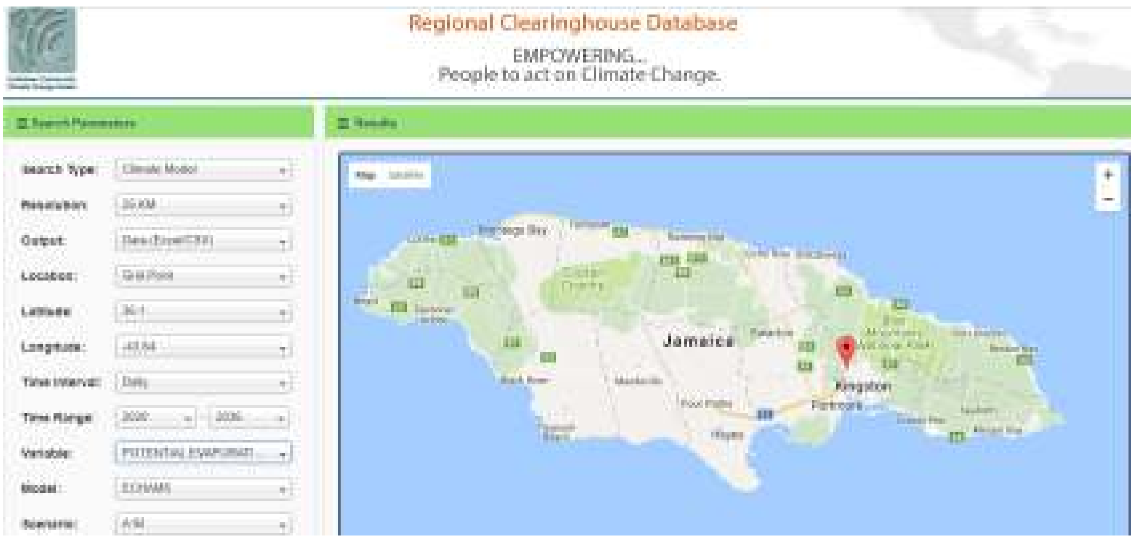


Figure 4.13 Illustration of the CCCCC regional clearinghouse database (CCCCC, 2015)

4.6.8. Proposed Intervention Strategies

Ferry Springs Brackish Desalination Scheme

The Ferry Springs brackish desalination scheme outlined in the Water Resources Development Master Plan (UWA, 1990) was identified as a potential intervention to enhance the resilience of the KSA50 WRS. Ferry Springs is located on the border of the Rio Cobre and the KSA WRZs and has been contaminated with high levels of saline water due to the exploitation of groundwater resources. The Ferry Springs schemes propose the construction of a brackish desalination facility at two locations near the springs. The type of desalination process is not explored in detail in the UWA Water Resources Development Master Plan report (UWA, 1990), however, in this research it is assumed that the desalination process is reverse osmosis. The site-specific details are summarised in Table 4.6.

Table 4.6 Summary of the Ferry Springs Brackish Desalination Schemes (UWA, 1990)

Item	Site A-East	Site A-East + Site B
Net Pond Area (ha)	52	252
Daily production (ML/d)	12.5	48
Unit cost of Water (£)	0.34	0.78
Capacities of canals m^3/h	2520	2520
Conveyance System Length (Km)	0.25	13.1

Asset Management

In the year 2010, NRW losses were estimated to be 44%, and this value increased to 59% in 2017 (Observer, 2017). Asset management as an intervention refers to the replacement of ageing infrastructure to reduce NRW by 30% as outlined in (Observer, 2017). It is assumed that asset management works will be conducted along the Yallahs pipeline enabling it to operate at its maximum capacity when required, as well as implementing a maintenance strategy at the Mona reservoir.

4.6.9. Performance Criteria

As mentioned in section 3.5, the performance indicators selected to assess the performance of the water resources system in this case study are reliability, resilience and sustainability. Resilience is expressed in this thesis from a social and environmental viewpoint. For the Social resilience concept, the aim is to minimise the number of consecutive days that the residential demand has not been met (failure duration), and the summation of the total deficit not delivered during that period (failure magnitude). This is conceptualised in Figure 3.9. Moreover, for the Environmental resilience criterion, the aim is to minimise the number of consecutive days that environmental releases (failure duration) have not been delivered downstream, and the summation of the total volume of releases not delivered during that failure period (failure magnitude), which is also conceptualised in Figure 3.9. As mentioned in section 3.5, reliability is based on the fraction of time-steps in the time horizon that reservoir performance does not fall below the dead storage (i.e. defined for the KSA50 case study as 30% of the total storage). Sustainability is conceptualised in terms of the energy used by an intervention during the production of water (e.g. desalination).

4.6.10. KSA case study assessment methods

The methodologies applied in the data-sparse KSA case study are illustrated in Figures 4.14 and 4.15. The first phase of the methodology presented in Figure 4.14 defines the calibration of the NAM model. Calibration attempts to yield the best possible solution where a set of parameters are chosen to demonstrate the ability of the model to minimise the differences between the historical hydrological conditions and the simulated hydrological conditions.

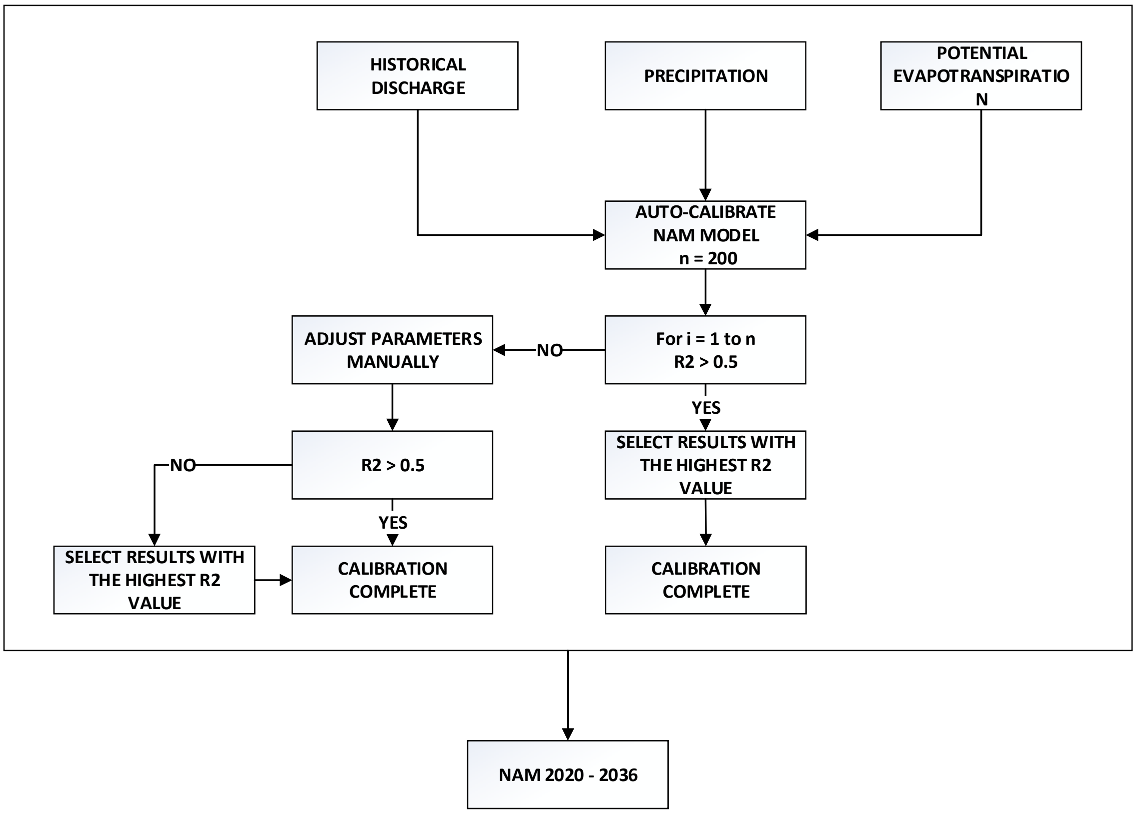


Figure 4.14 Illustration of the outline of the NAM rainfall-runoff calibration methodology

The historical parameters of discharge, precipitation and potential evapotranspiration were introduced as the inputs to the NAM model. Calibration of the NAM model can be performed automatically, by trial and error, or utilising both approaches. The methodology presented in Figure 4.14 is designed to incorporate both the auto-calibration and the trial and error approaches. The NAM model conducts an automated search (auto-calibration) in the parameter space producing a set of 200 possible parameter combinations (e.g. $i = 200$). In this case study, the value $i = 200$ was selected due to time limitations (associated with the manual calibration process described below) and therefore the methodology is not confined to this limit. Each of the 200 parameter combinations was evaluated by trial and error (manual calibration) to determine which of them yielded the best possible hydrological conditions (i.e. the closest possible goodness of fit when compared to the Hope River historical discharge). Manual calibration involved replacing the default values for the 9 parameters listed in Table 4.4 with those returned from the auto-calibration, and then re-running the NAM model (this was process was conducted 200 times until the best fit was obtained). The performance criteria utilised was the coefficient of determination, where according to Moriasi and Arnold, (2007), an R^2 >= 0.5 is considered satisfactory (in their research, a comprehensive literature review of reported ranges of values and corresponding performance ratings for R^2 was conducted). If none of the combinations yielded a satisfactory result, then the parameters were manually adjusted until an R^2 > 0.5 is returned. In this case, study, if after manual adjustment of the parameters, the results yielded an R^2 < 0.5, then the highest possible R^2 achieved is selected.

In the second phase of the methodology, the calibrated NAM model is used to generate future time series of discharge for the Hope River for the period 2020 to 2036. The Hope River is the main watercourse in the KSA50 region. Referring to Figure 4.15, climate data for Kingston Jamaica was utilised as the input into the calibrated NAM model. The model undertakes the process explained in subsection 4.6.6 and generates discharge for the Hope River based on the site-specific parameters obtained from the calibration.

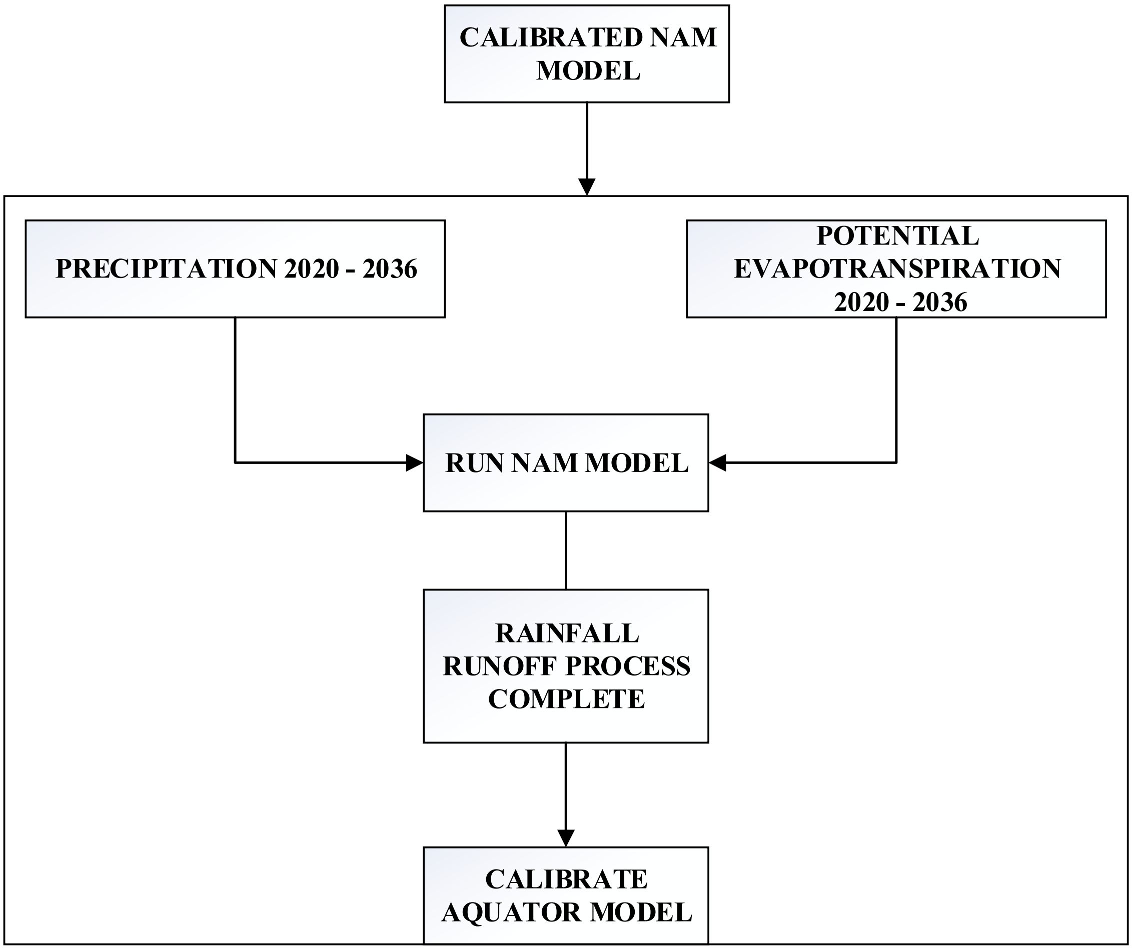


Figure 4.15 Illustration of the NAM rainfall-runoff calibration procedure

In the final phase of the methodology presented in Figure 4.16, the historical inflows from the Yallahs pipeline and the future time series of discharge (for the Hope River) were introduced into the KSA Aquator model, and a water resources assessment conducted to evaluate KSA50 to the hydrological conditions. The water resources assessment is explored in greater detail in the section 4.6.11.

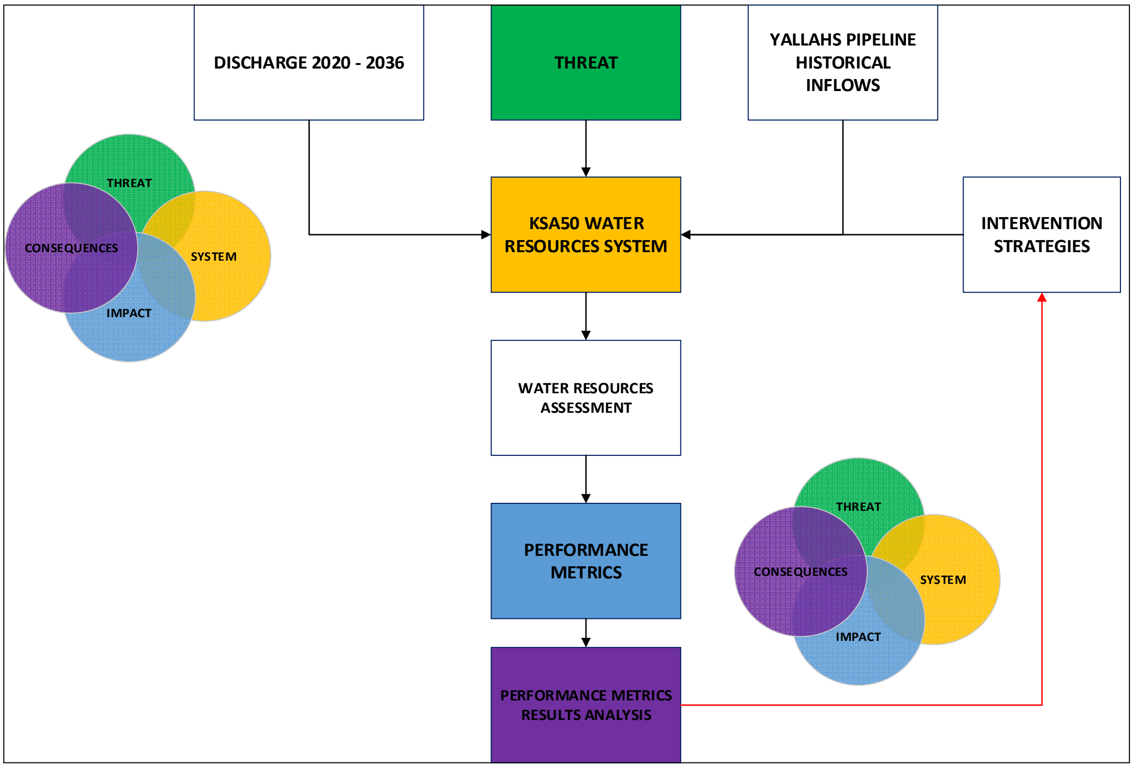


Figure 4.16 Illustration of KSA50 water resources assessment methodology

4.6.11. KSA50 model configuration

Before testing the intervention strategies, the KSA50 model was configured to reproduce the operating conditions outlined in the (NWC, 2011) report. This was done to establish a baseline state for the model.

In the NWC, (2011) report, the demand for the KSA region in the year 2030 was estimated to be approximately 31 million imperial gallons per day (140 ML/d). Therefore, the estimated demand value for KSA50 utilised in this assessment was 70 ML/d. The value utilised as inflows from the Yallahs pipeline was the average historical inflows value (64 ML/d), and at the Hope River, the discharge generated from the NAM rainfall runoff procedure was utilised as input. The model for KSA50 is illustrated in Figure 4.17.

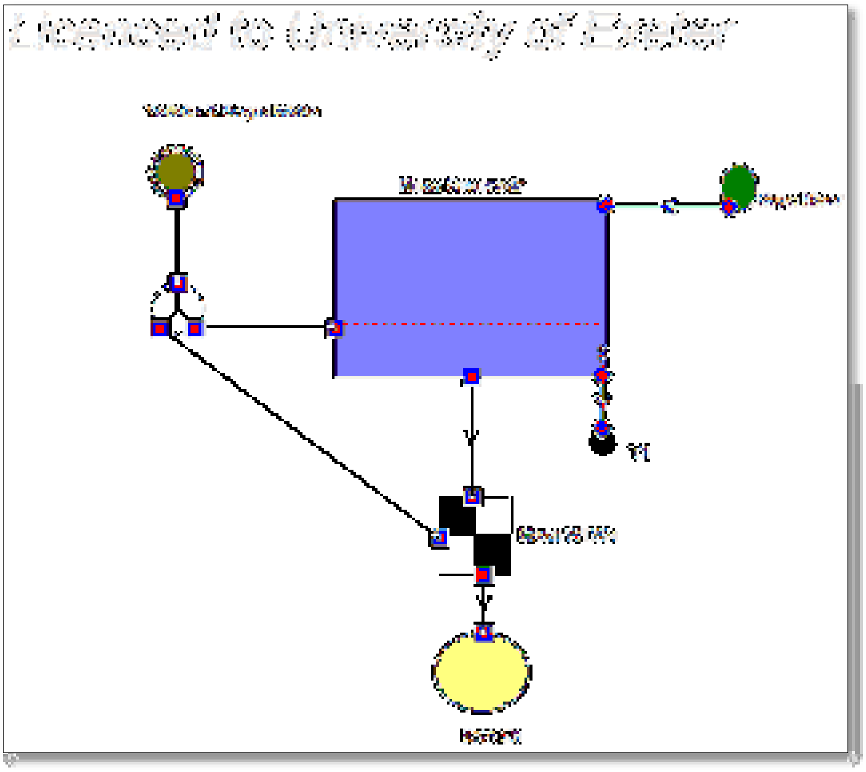


Figure 4.17 Aquator model representation of the KSA50 WRS

Moreover, the KSA50 model was configured to reproduce half of the physical losses reported in the (NWC, 2011) report for the supply side. Recall that the Mona and the Hope sections of the WRS have the potential to meet 50% of the demands for the KSA WRS (Observer, 2017). Therefore, it is assumed that on the supply side the losses are distributed evenly throughout the WRS, therefore the KSA50 model is configured to operate under these conditions. According to NWC, (2011), 50% of the physical amount equated to approximately 7.7 million imperial gallons per day in the year 2010 (35 ML/d), and this quantity of losses highlighted in the NWC report was assumed to remain at the same level in the 2030s (NWC, 2011). In addition to the assumptions above, the following additional assumptions were made in this thesis to justify where those losses are occurring:

1) the losses are occurring along the 30.6 km length of the Yallahs pipeline and;
2) losses resulting from seepage occurring at the Mona reservoir.

The losses along the Yallahs pipeline are modelled in Aquator by utilising the Leakage Loss Rate parameter, and at the Mona reservoir, they are modelled utilising the Seepage-Percent Storage parameter. The range of values tested for the Leakage Loss Rate parameter and the Seepage-Percent Storage parameter are presented in Table 4.7. The model returned a physical loss value of **191,790** ML (Table 4.7) during configuration (i.e. calibration of the KSA50 model) when a value of 0.01% was tested as the leakage loss parameter value, and 8.5% was tested as the seepage loss parameter value. The configured value obtained was found to be approximately 478 ML (less than 1%) below half of the target supply side losses for the 2030s reported in (NWC, 2011). Recall the assumption above by the NWC that losses between 2010 and 2030 remained at the same level.

Table 4.7 Summary of parameter values tested for the model calibration of KSA50

Component		**Parameters**		**16-year Physical Losses Simulated (ML)**	**16-year Physical Losses Required (ML)**
Mona Reservoir Capacity (ML)	**Yallahs Pipeline Length (Km)**	**Leakage Loss Rate (ML/Km/d)**	**Seepage Percent Storage (%)**		
3028	30.6	0	0	0	**192,268**
		0.0001	1	0	
		0.0001	2	0	
		0.0001	3	0	
		0.0001	4	3,832	
		0.0001	5	23,333	
		0.0001	6	63,167	
		0.0001	7	113,934	
		0.0001	8	166,981	
		0.0001	9	208,890	
		0.0001	8.5	189,981	
		0.001	8.5	190,092	
		0.01	8.25	180,725	
		0.01	**8.5**	**191,790**	

4.6.12. KSA50 Water resources assessment

The Safe and SuRe framework and the Safe and SuRe circular approach explored in Section 3.3, is utilised in this case study to assess the performance of the KSA50 WRS. Referring to Figure 4.16, the threats to the system are demand increase and climate change. As mentioned in subsection 4.6.6, the discharge obtained from the NAM rainfall-runoff model was used as the Hope River input. Moreover, Milly and Julio, (2008), argue that changes in the Earth's climate have resulted in variability in the "*means and extremes of precipitation, evapotranspiration, and rates of discharge of rivers*" (Milly and Julio, 2008); and therefore, utilising the historical average inflow from the Yallahs pipeline for the water resources assessment would not account for extreme changes that could potentially occur. To account for these extremes, an assessment was conducted on the configured model to consider possible increases and decreases in the inflow from the Yallahs pipeline. The average inflow value was adjusted in 5% increments up to a value that is 20% (the water resources system experienced failure for every time-step when evaluated at this upper bound) more than the average inflow. This process was repeated by considering possible declines in the inflows, by decreasing the average inflow value in decrements of 5% until a value that is -20% of the average inflow was obtained. In other words, the threat and the intervention strategy remained the same while a series of model runs are conducted considering the different levels of uncertainty regarding flows for the Yallahs pipeline.

4.7. Results and Discussion

4.7.1. NAM Model Calibration Results

The NAM model was calibrated for the four-year period 2009 to 2012. The model parameters returned from the calibration represent possible site-specific conditions for the Hope River, and these were found to be within the NAM model upper and the lower boundary limits. The results for the calibrated parameters are summarised in Table 4.8.

Table 4.8 Summary of calibrated NAM Parameters

Parameter	Unit	Calibrated Result	Parameter Range
Umax	mm	18.50	10 – 20
Lmax	mm	219	100 – 300
CQOF	-	0.89	0.1 – 1
CKIF	Hours	206.50	200 – 1000
CK1,2	Hours	24.70	1 – 50
TOF	-	0.67	0 – 0.99
TIF	-	0.74	0 – 0.99
CKBF	Hours	2898	1000 – 4000
Tg	-	0.84	0 – 0.99

The model calibration resulted in a coefficient of determination (R^2) of 0.31 and is based on the guidelines summarised in Moriasi and Arnold, (2007), the value 0.31 indicates that a satisfactory relationship does not exists between the observed discharge and the simulated discharge values.

Visual analyses of the calibration results are illustrated in Figures 4.18 and 4.19; and from these analyses, it was concluded that the calibrated parameters in Table 4.8 have shown the tendency to overestimate low flows and to underestimate peak flows. In Figure 4.19 the difference between the simulated accumulative discharge and the observed accumulative discharge values appears to be small up to the time-steps dated 29th September 2010 to 30th September 2010 where an event resulted in a large volume of discharge. This large event was not reproduced at the same intensity as in the simulated output. A likely implication underpinning this result could be the possibility that different climatic conditions are present in each of the parameter locations (Figure 4.12), as the heights above sea level are different in each location. In Figure 4.12 and Table 4.5, the location and the height above sea level for each of the parameters used as inputs during calibration are indicated. In each of the three locations, there are differences in the heights of the weather stations above the sea level. At Wakefield station, the height above the sea level is 120m and this is the location of the rainfall station; and at the Norman Manley station, the height above sea level is 5m, and this is the location of the PET parameters, and at the Gordon Town gauging station at Hope River, the height above sea level is 608m. This variation in altitude and proximity is a potential explanation for the differences between the observed and simulated outputs.

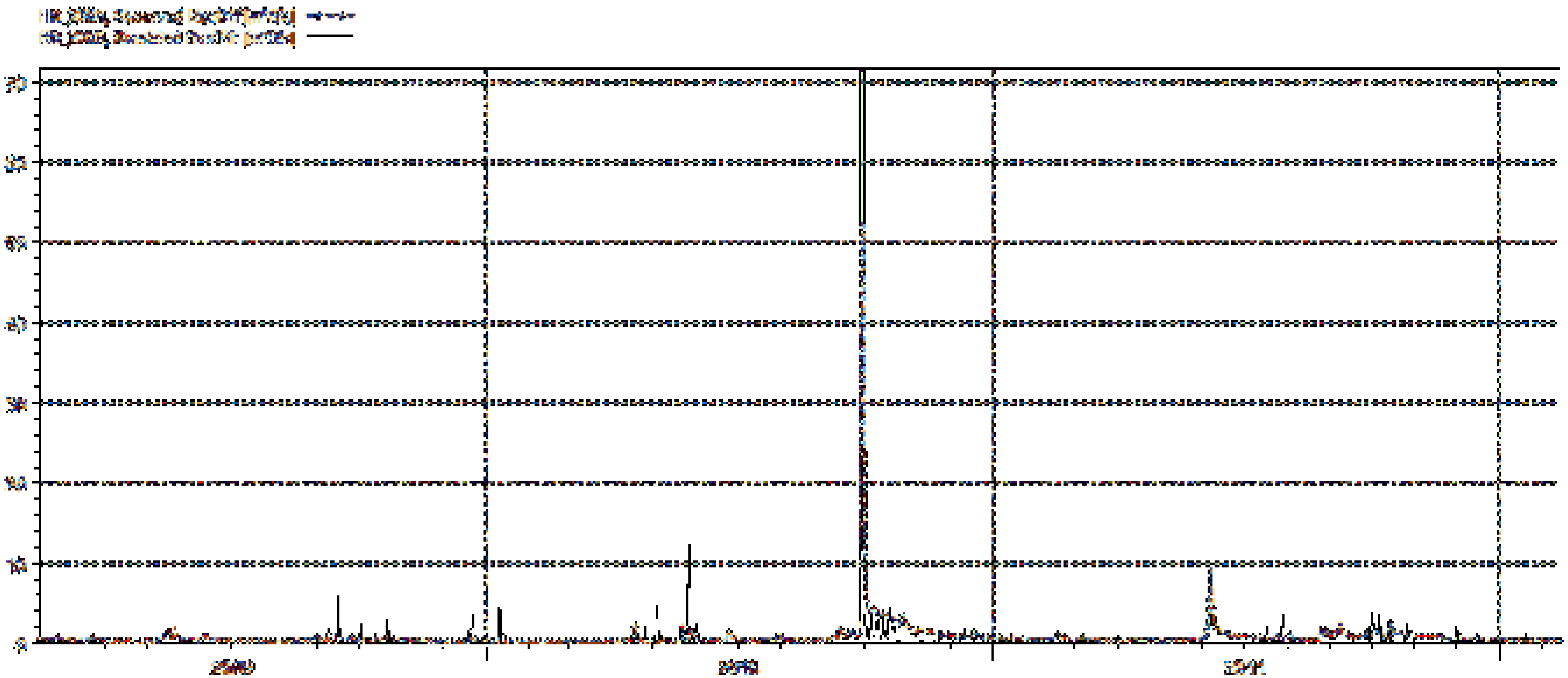


Figure 4.18 Illustration of the degree of agreement between the observed runoff and the simulated runoff for an R2 value of 0.31.

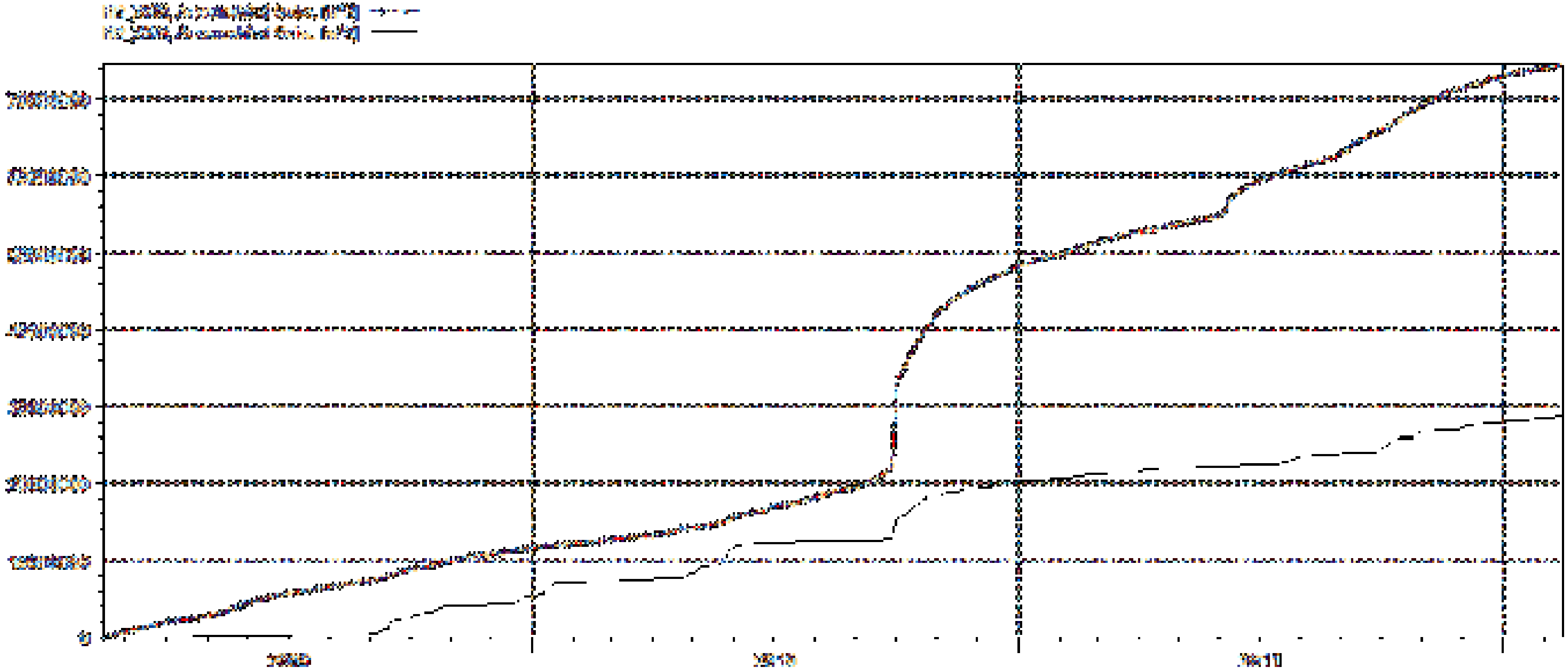


Figure 4.19 Illustration of the accumulative observed runoff and simulated observed runoff for an R2 value of 0.31

4.7.2. KSA50 Water Resources Assessment Results

4.7.2.1. No Interventions and Coping

Social Assessment and sustainability results

The threats considered in this case study are demand increase and climate change. In addition, supply uncertainty was explored by varying the inflows from the Yallahs pipeline. In Figure 4.20, the threats and intervention strategies utilised during the KSA50 water resources assessment are illustrated. Initially, no interventions are modelled in Aquator for this assessment, however, a coping scenario was tested after the model simulation to demonstrate the economic consequences to customers, and this is summarised in Table 4.9.

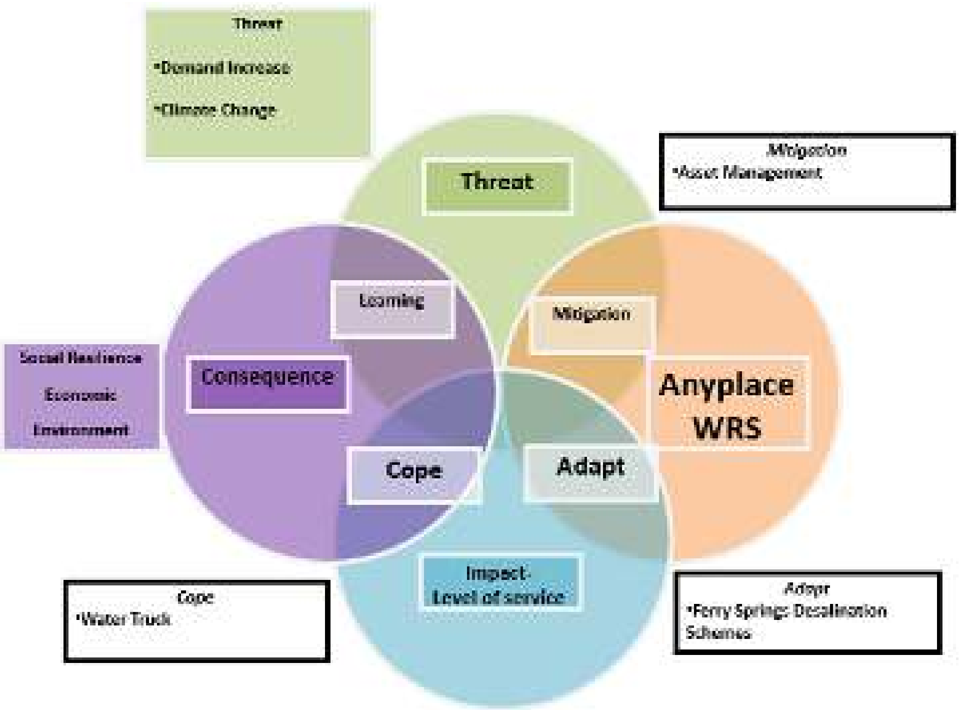


Figure 4.20 Illustration of the threats and the intervention strategies in the Safe and SuRe framework

In Table 4.9, the results of assessing social resilience are expressed in terms of consecutive failure days and total accumulated deficit during the dry season. The dry season in the Caribbean begins in December and runs until

April (Cashman *et al*., 2009), and the period representing this in this thesis is **December 2025 to April 2026.** This period was chosen because it represents the most severe dry period in the evaluation horizon.

Table 4.9 Summary of the accumulated demand deficit for the 16-year period, and the costs associated with the coping intervention concept results returned for the demand increase and climate change considering varying levels of uncertainty at the Yallahs pipeline

Level of Uncertainty at the Yallahs Pipeline	Consecutive Failure Days	Failure Magnitude (L)	Unit cost per L (£)	Total Cost (£)
-20%	151	5624000000	0.01	56,240,000
-15%	151	5157000000		51,570,000
-10%	151	4680000000		46,800,000
-5%	151	4192000000		41,920,000
0%	151	3649000000		36,490,000
5%	96	2726000000		27,260,000
10%	90	2344000000		23,440,000
15%	86	1955000000		19,550,000
20%	82	1577000000		15,770,000

A 20% reduction in the inflows from the Yallahs pipeline, would result in customers enduring 151 consecutive days of domestic demands non-compliance. This result is indicative of a worse case supply uncertainty scenario. Moreover, when the inflow from the Yallahs pipeline remains at normal conditions (i.e. scenario 0%), the non-compliance duration is the same as the worst-case scenario (i.e. -20%), but only differs in the failure magnitude that has accumulated for the time-steps during that period. Significant improvements in the results were obtained when the Yallahs pipeline inflow capacity was increased. The greatest improvement was observed when the inflows were varied/increased by 20%. At this level, customers endured 82 consecutive days of demand deficits. This evidence demonstrates the potential benefits to be gained from upgrading the Yallahs infrastructure (e.g. Asset management intervention).

The coping intervention is characterised in Table 4.9 by conceptualising that the deficit can be minimised by purchasing water from water tankers. Information on the financial well-being of Kingston residents is not available and therefore the total cost in the 5th column is representative of potential costs that would occur if this were the only intervention strategy implemented. This is calculated by multiplying the unit cost per litre in the 4th column (WaterAid, 2016) by the total demand deficit accumulated from December 2025 to April 2026 dry period, which is summarised column three. The unit cost per litre of water (£0.01) represents the UK price equivalent of water from water tankers in less developed countries, and this this found in The State of the World's Water report 2016 (WaterAid, 2016). Unsurprisingly, the cost is lower when the inflows from the Yallahs are greater. In addition, it is acknowledged that in the event of a drought or prolonged dry spells, the inflows from the Yallahs may be reduced, therefore the varied range of reduced Yallahs inflows tested (i.e. -5% to -20%) can be considered as reference points for decision makers to understand the socio-economic consequence to the Kingston residents if no action is taken.

Reliability Assessment Results

On the system side, the impacts of threats are evaluated by employing the reliability concept discussed in section 3.5 and are summarised in Table 4.10. For each of the increases or decreases in the Yallahs inflows, the impact on the Mona reservoir system performance results in a poor level of reliability (i.e. operating below the emergency storage level), and this is an indication that emergency supplies are consistently being utilised. At the Dead storage level (i.e. 30% of the total storage volume), the results varied between 0.42 and 1.00. For example, for the -20% scenario, the reliability score of 0.42 indicates that the Mona reservoir operates below the Dead storage level for most of the simulation daily time-steps, and the value 1.00 returned for the 15% and 20% scenarios indicates that system performance does not fall below the dead storage. Overall, the poor results obtained by quantifying the system performance at the Emergency storage level (i.e. 70% of the total storage volume). At the level, a reliability score of 0 was returned for each level of uncertainty tested (Table 4.10) and are reflective of the threats and uncertainty affecting the system. To improve the system performance, intervention strategies are required and is represented in this case study by desalination. The impacts on the system are explored below.

Table 4.10 Summary of the reliability scores characterising the performance of the Mona reservoir

Level of uncertainty at the Yallahs pipeline	Reliability at Dead Storage performance
-20%	0.42
-15%	0.57
-10%	0.71
-5%	0.78
0%	0.86
5%	0.93
10%	0.98
15%	1.00
20%	1.00

4.7.2.2. Ferry Springs brackish desalination scheme

Social assessment

To minimise the social, economic and environmental consequences and the impact on the system performance, the Ferry Springs brackish desalination schemes outlined in the Water Resources Development Master Plan (UWA, 1990) were tested. Two construction sites are proposed under the scheme plans and the site-specific details are summarised in Table 4.6.

The site A-East feasibility is evaluated for the threats outlined in the Safe and SuRe circular approach in Figure 4.20, and the social resilience results are summarised in Table 4.11. The results show that the A-East option improves on the failure magnitudes and durations summarised in Table 4.9, although customers would still experience high levels of demand deficits during the dry season. However, the implementation of the 'A- East + Site B' desalination scheme results in full compliance with demands for each level of uncertainty during the most severe dry season of the evaluation period. In relation to the environmental consequence, the use of desalination has implications as it contributes large amounts of energy utilised during operation.

Table 4.11 Comparison of the results for the social resilience performance indicator for the Site A East Ferry Springs desalination scheme

Intervention	Level of uncertainty at the Yallahs pipeline	Consecutive Failure Days	Failure Magnitude (ML)
Ferry Springs Site A East - 12.5 ML/d	-20%	132	3628
	-15%	97	2757
	-10%	97	2511
	-5%	91	2221
	0%	90	1916
	5%	82	1574
	10%	80	1247
	15%	38	566
	20%	37	405

In Table 4.12, the energy use for both desalination options is summarised. Column two represents the total desalinated water utilised for each level of uncertainty, and the 3rd column, is representative of the total energy (660 kWh/ML) that is expended during the production of 1 megalitre of desalinated water, which was adopted from an American research study by (Gleick and Cooley, 2009). Therefore, for example, the energy footprint for desalination is determined by multiplying 1888 ML * 660 kWh/ML and converting the units to MWh, and desalination costs are determined by multiplying the values in the 2nd column by the unit cost of 1 ML of desalinated water given in Table 4.5.

Table 4.12 Comparison of the total desalinated water produced and the total energy usage during operation for the Ferry Springs desalination schemes

Intervention	Level of uncertainty at the Yallahs pipeline	Total[11] Desalinated Water Utilised (ML)	Energy Footprint Desalination (MWh)	Total Cost of Desalinated Water[12] (£) (thousand)
Site A East – 12 ML/d	-20%	1888	1246	642
	-15%	1888	1246	642
	-10%	1888	1246	642
	-5%	1888	1246	642
	0%	1888	1246	642
	5%	1888	1246	642
	10%	1888	1246	642
	15%	1888	1246	642
	20%	1888	1246	642

Intervention	Level of uncertainty at the Yallahs pipeline	Total Desalinated Water Utilised (ML)	Energy Footprint Desalination (MWh)	Total Cost of Desalinated Water (£) (thousand)
Site A East + Site B – 48 ML/d	-20%	7044	4649	5494
	-15%	6968	4598	5435
	-10%	6873	4536	5360
	-5%	6773	4470	5282
	0%	6656	4392	5191
	5%	6525	4306	5089
	10%	6390	4217	4984
	15%	6251	4125	4875
	20%	6113	4034	4768

The Site A-East desalination option produces a constant quantity of water during the dry period because of the severity of the period and small capacity

[11] The dry season in the Caribbean begins in December and runs until April (Cashman *et al.*, 2009), and the values for the total desalinated water in the third column is representative of the future period December 2025 to April 2026.

[12] Table B3 in Appendix B demonstrates how the production costs are calculated. The cost to produce 1 ML of desalinated water was taken from Table 4.6. It should be noted that these values represent the cost to produce 1 ML of desalinated water in Jamaica in the year 1991. These values were sourced from the Underground Water Authority, Water Resources Development Master Plan report (UWA,1990).

of the system. However, the larger capacity desalination system is not operating at full design capacity when tested across the different levels of uncertainty corresponding to the variable operation of the Yallahs pipeline.

4.7.2.3. Asset Management and Desalination (Mitigation and Adaptation)

Social performance indicators assessment

From a review of the environmental consequences, in subsection 4.7.2.2, best practice guidance can be informed through the application of a mitigation measure to improve the performance of the WRS on the supply side. The mitigation intervention termed asset management was tested along with the Ferry Springs desalination scheme (adaptation). Asset management refers to the replacement of ageing infrastructure to achieve a reduction in the NRW losses target which has been set at 30%. This objective can be achieved through significant upgrades to the Yallahs pipeline and implementation of a maintenance schedule for the Mona reservoir.

Referring to Table 4.13, the results returned for the combined migration and adaptation strategies are summarised. When compared with the results in Table 4.11, the desalination option A-East and the asset management intervention contributed to a significant improvement in reducing the consecutive days that the required demand has not been met. Unsurprisingly, the lowest number of consecutive days and lowest demand deficit was obtained when the Yallahs flows were tested at a 20% increase. The Site A – East + Site B desalination scheme resulted in full compliance across all levels of uncertainty.

Table 4.13 Comparison of the results returned for the social resilience indicator for the Site A East Ferry Springs desalination schemes combined with asset management.

Intervention	Level of uncertainty at the Yallahs pipeline	Consecutive Failure Days	Failure Magnitude (ML)
Ferry Springs Site A East - 12.5 ML/d	-20%	91	2470
	-15%	90	2133
	-10%	75	1683
	-5%	72	1332
	0%	68	1035
	5%	37	467
	10%	22	257
	15%	19	165
	20%	17	87

Sustainability

The contribution of asset management towards reducing the quantity of desalination production required was analysed by comparing Table 4.12 and Table 4.14. For the A-East scheme, asset management reduced the quantity of water required for production at the baseline level of uncertainty (i.e. 0%), and for each of the other increasing levels of uncertainty. The energy expended in the production of desalination water was also less. In the remainder of the levels of uncertainty, asset management does not contribute to reductions in the quantities of desalination water produced or the energy footprint of desalination.

Table 4.14 Comparison of the total desalinated water produced and the total energy usage during operation for both Ferry Springs desalination options combined with asset management

Intervention	Level of uncertainty at the Yallahs pipeline	Total Desalinated Water Utilised (ML)	Energy Footprint Desalination (MWh)	Desalinated Water Cost (£) (thousand)
Ferry Springs Site A East - 12.5 ML/d	-20%	1888	1246	642
	-15%	1888	1246	642
	-10%	1888	1246	642
	-5%	1888	1246	642
	0%	1126	743	382
	5%	730	481	248
	10%	432	285	146
	15%	218	143	74
	20%	79	52	26
Intervention	**Scenarios**	**Total Desalinated Water Utilised (ML)**	**Energy Footprint Desalination (MWh)**	**Desalinated Water Cost (£) (thousand)**
Ferry Springs Site A- East + Site B - 48 ML/d	-20%	6657	4393	5192
	-15%	5828	3846	4545
	-10%	6385	4214	4980
	-5%	6241	4119	4867
	0%	6096	4023	4754
	5%	5959	3932	4648
	10%	5820	3841	4539
	15%	5678	3747	4428
	20%	5517	3641	4303

For the ‘A-East + Site B scheme', the total desalinated water produced, and the energy footprint of the desalination plant was reduced when all levels of uncertainty were tested.

Reliability

Referring to the reliability of the system, the reliability scores in Table 4.15 for both desalination options combined with asset management are presented. The results show an improvement in reliability in the Mona reservoir’s dead storage performance when compared to the performance scores for the *No Interventions* strategy in Table 4.10. However, despite improvements in the performance at the dead storage level, no improvement in reliability was achieved for the emergency storage performance indicator. Implications of the poor performance for emergency storage indicator are underpinned by the following:

1) The introduction of desalination and asset management reduced the effect of the social and economic consequences to the customers and the environment, however on the system side, the remainder of the losses affecting the system side is at a high level; and despite intervention actions, the reservoir operates below the emergency storage level. This implies that additional interventions measure would be required.
2) To improve on the above implication, consideration could be given to the implementation of additional measures such as capacity augmentation or desilting.

Table 4.15 Comparison of the reliability scores for both Ferry Springs desalination options combined with the reduce losses intervention

Intervention	Level of uncertainty at the Yallahs pipeline	Reliability at Dead Storage performance
Ferry Springs Site A East - 12.5 ML/d	-20%	0.81
	-15%	0.89
	-10%	0.95
	-5%	0.99
	0%	1.00
	5%	1.00
	10%	1.00
	15%	1.00
	20%	1.00

Intervention	Level of uncertainty at the Yallahs pipeline	Reliability at Dead Storage performance
Ferry Springs Site A-East + Site B - 48 ML/d	-20%	0.82
	-15%	0.90
	-10%	0.96
	-5%	0.99
	0%	1.00
	5%	1.00
	10%	1.00
	15%	1.00
	20%	1.00

On a positive note, both combinations of interventions resulted in the reservoir operating above the dead storage level for most of the time-steps of the period analysed (i.e. the most severe dry season). This can be seen in the 3rd column in Table 4.15 for the levels of uncertainty -5% to -20%. For the other levels of uncertainty, the reservoir operates between the dead storage level and the emergency storage level.

4.8. Conclusions

In this chapter, a water resources assessment was performed on the data-sparse Kingston and St. Andrew case study. The major challenge encountered in the case study is related to the inconsistencies of the data sets to characterise key components of the WRS, and as a result, the research focus was shifted from assessing the entire KSA WRS to assessing the potential of the Mona and Hope segments of the KSA WRS to service 50% of the KSA's demand (KSA50). A synthetic time series of discharge was generated from climate change projections (2020 to 2036) using the NAM rainfall-runoff model for the Hope River, which supplies untreated water to both the Mona and Hope zones. The first phase involved generating time series of discharge required for the calibration of the NAM model. The most consistent parameters were used for the model calibration which resulted in a coefficient of determination value of R^2=0.31. As mentioned in subsection 4.6.7, it is believed that the different micro-climates where the parameter sets were sourced played a key role in underpinning the R^2 score. The model performed poorly as it failed to reproduce a critical peak discharge value.

The discharge time series generated from the NAM was used as an input for the Hope River component in the Aquator model. The historical inflow from the Yallahs pipeline was used as an input for the other supply source in the model. A baseline was established for the water resources model by configuring it to reproduce the supply losses reported in the NWC 2011 water supply plans. As part of the configuration, key assumptions were taken into consideration in relation to the NRW losses occurring on the supply side of the WRS, explored in detailed in subsection 4.6.10. The configuration that resulted in the best fit in reproducing the reported losses occurred when 8.5% of the Mona reservoir total storage volume was lost to seepage, and a leakage rate of 0.01 ML/km occurred along the Yallahs pipeline. Moreover, a high level of uncertainty remained in relation to how the performance of the Yallahs pipeline will vary under climate change influences. This was explored using a sensitivity analysis to test different levels representing increases and decreases in the inflows. To explore uncertainty in detail, the model was simulated considering the threat and interventions for each level of uncertainty.

The Safe and SuRe circular approach was used as the tool guiding the water resources assessment. Intervention strategies outlined in Jamaica's water resources management plans were tested. This included two brackish desalination schemes and an asset management scheme in the form of undertaking infrastructure upgrades. The assessment was first conducted without considering these mitigation and adaptation interventions, and the most severe dry season period (December 2025 to April 2026) was analysed. The social resilience measure proposed revealed severe consequences to customers resulting from high levels of demand deficits for each level of uncertainty tested. A coping strategy, provision of water tankers, was also employed to conceptualise the likely costs to meet demands. The results revealed that it would cost between £15 million to £56 million to meet demands during the most severe dry conditions if no other interventions were introduced.

The brackish desalination scheme 'Site A- East + Site B' showed the greater potential to improve resilience. When tested alone, this intervention enabled full compliance with demands met during the most severe dry period as seen in Figure 4.11. The contribution of asset management was tested in combination with the two desalination schemes. When tested with the desalination scheme A-East, reductions in the number of failure days and demand deficits were achieved only if the inflows from the Yallahs pipeline were increased as seen in Table 4.14. Regarding the 'Site A- East + Site B' scheme, asset management resulted in the reduction in the energy expended during desalination, and also the cost that would be incurred in the production of desalinated water. Depending on the type of investment available in the case study region, assets management when combined with desalination could be a strategy that contributes to improved reliability and resilience.

The most unsatisfactory results were obtained for the reliability of the Mona reservoir. For each of the intervention strategies tested, the reservoir remained in operation below the emergency storage level. The addition of desalination and the combination of interventions marginally improved the reliability when compared to the *No Interventions* strategy.

4.9 Limitations

The following limitations were encountered in the Kingston and St. Andrew case study:

- Time and financial resources– The first and the second year of the PhD research course was spent trying to network and establish a research/professional relationship with key decision makers at the NWC (the NWC is the sole water supply undertaker in Jamaica and the owner of all the water facilities in the country). Networking with key figures at the NWC was important in the early stages of the research as the researcher is not a national of Jamaica. Therefore, in order to obtain the information required to build and characterise an Aquator model representative of the Kingston and St. Andrew water resources system, a significant proportion of time was dedicated to networking. In January 2016, the researcher was travelled to Kingston meet with a senior hydrologist at the Water Resources Authority (the island's water regulator). The hydrologist attempted to arrange meetings with key figures at the NWC. The visit was unsuccessful, and this could be attributed to financial resources made available to the researcher at the time of the visit. The funding provided by the Commonwealth Scholarship Commission was only sufficient to finance an 18 day stay in Kingston, Jamaica.
- Lack of professionalism– Several attempts were made by a key figure at the WRA on behalf of the researcher to arrange site visits to NWC facilities and also interviews with key persons within the NWC organisation (through email correspondence). Unfortunately, for the entirety of the visit to Kingston and for the remainder of the PhD course, the researcher did not receive any email correspondence in acknowledgement of the communications (emails sent on behalf of the researcher) sent.
- Access to data and timing of study – The researcher was provided with access to the island's WebMap database by the Water Resources

Authority. WebMap is an online archive containing historical hydrological parameters (e.g. stream flow records, reservoir levels, etc). However, at the time of writing of the thesis, data characterising key facilities and river networks in Kingston had not been uploaded to the online archive.

CHAPTER FIVE

5. Anyplace Data-Sparse Case Study

5.1. Introduction

In section 3.9, an MCDA methodology was developed to recommend intervention strategies to decision makers. However, it was not possible to implement this methodology in the KSA50 case study due to a large number of assumptions introduced to fill data gaps encountered in that case study. This also implies that conditions are unfavourable to propose the recommendation of any intervention strategies to Kingston decision makers. This chapter builds on the early work conducted in section 3.9 by presenting the case for the selection of strategies in a data sparse environment. A methodology is proposed that can set the scene for decision makers to validate intervention strategies which have been planned for implementation in a data-sparse environment. To achieve this, the methodology presented in section 5.2 transforms the Anyplace data-rich case study to an (Anyplace) data-sparse case study, allowing for Objective 6 to be achieved.

5.2. Implementation of data-sparse conditions

To examine the Anyplace water resources system under data-sparse conditions, data gaps were introduced in the 11 FFH time series at catchment CM1 which was chosen using random sampling in MATLAB. In Figure 5.1 an illustration of the data gaps that are present in the most recent time series profile for the Yallahs river pipeline is shown. This time series was obtained from the Water Resources Authority WebMap database for the 15-year duration of 2001 – 2015 (WRA, 2012). To achieve the transformation from data-rich to data-sparse conditions, some of the time series values found in the CEH 11 FFH time series were substituted with zero values following the guidance of the pattern illustrated in Figure 5.1 (e.g. in the first year representing data-sparse conditions, all of the time series values for the months of May and June were substituted with zero values). The FFH time series represents 11 plausible supply scenarios over a 30-year period, and to determine the starting period of data-sparse conditions was a challenging task. To overcome this, Monte Carlo sampling was employed to select a starting period between 2020 and 2034. Selecting a starting period after 2034 would mean that some of the 15-year data-sparse conditions would not be included in the evaluation. For example, in the FFH Q10 time series, the period representing data-sparse conditions commenced from the year 2027 to 2041; and for FFH Q14, data-sparse conditions were experienced from 2030 to 2044, etc. The summary of the data-sparse periods for each of the FFH are summarised in Table 5.1.

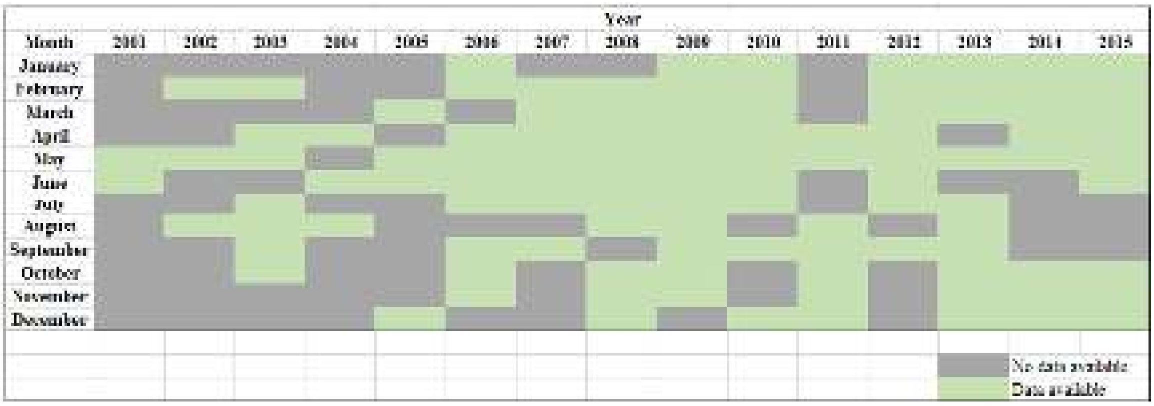


Figure 5.1 Illustration of the data gaps present in the Yallahs River pipeline time series for the period 2001 - 2015 (WRA, 2012)

Table 5.1 Summary of data-sparse periods tested at the catchment CM1 during the transformation of Anyplace data-rich to Anyplace data-sparse

FFH	15- year Data-sparse Period
Q0	2020 - 2034
Q10	2027 - 2041
Q3	2022 - 2036
Q6	2025 - 2039
Q4	2032 - 2046
Q13	2028 - 2042
Q9	2025 - 2039
Q8	2020 - 2034
Q14	2030 - 2044
Q16	2031 - 2045
Q11	2029 - 2043

Following the guidance of the methodology explored in Chapter 3, Aquator was used to perform simulations of the Anyplace WRS under data-sparse conditions. The results of those simulations are explored below.

5.3. Results and discussion

As mentioned in Section 5.1, it is challenging to identify and prioritise the best strategy in the KSA50 case study, due to the wealth of challenges and uncertainty associated with the case study's data-sparse nature. To overcome this, and to assess the performance of promising intervention strategies in a data sparse environment, the methodology explored in section 5.2 was developed. This sets the scene for observing how well promising strategies obtained in a data rich environment perform in a data-sparse setting.

MCDA Weighting Criteria Combination 1

Referring to MCDA weighting combination 1 in Figure 5.2, the performance results for Anyplace evaluated under data-sparse and data-rich conditions are illustrated when the performance indicators are weighted equally (i.e. reliability

= 0.33; resilience = 0.33; sustainability = 0.33). The x-axis indicates the names of the interventions evaluated, and the y-axis ranks the strategies from the high (1st) to the low (11th). The results indicate that under equal weighting, no differences were observed in the performances of the strategies when ranked from high to low under both sets of environments (data-sparse and data-rich). Interestingly, under both environments, the *Goal-seeking Hedge* intervention showed the greatest potential to improve reliability, resilience and sustainability. This builds on the implications identified in subsection 3.9.1. In general, the WRS response across the majority of the strategies is largely similar in the both data-rich and sparse environments, except for those strategies where desalination is present. This implies a good level of model consistency, and a possible implication to justify the differences observed across the results where desalination was tested, extend to the schedule (planned) implementation of desalination; which is different for each FFH tested (N.B. the hydrological conditions in each of the FFH are different). An example of how this could impact on the results can be conceptualised in a data-rich environment where the desalination scheduled implementation was towards the end of the planning period. Therefore, in that case, a lower quantity of desalinated water would have been produced if the desalination operated within it design limits for large periods. In contrast, considering the same situation in a data-sparse environment, it is expected that the desalination facility would have been in operation at its maximum capacity for longer periods.

From an environmental perspective, the results illustrated in Figure 5.3 for MCDA weighting combination 1 are relatively consistent across the data-rich and data-sparse settings. The highest ranked strategy in the data-sparse setting was the *Demand Management and Goal-seeking Hedge,* which demonstrated a moderate improvement in its ranking when compared to the data-rich results. A factor which could have contributed to this result was an increased level of hedging attributed to the data-sparse conditions, which as discussed earlier, is a major contributor towards improving reliability and sustainability, which means a higher level of compliance with reservoir level of service (i.e. recall compliance with the level of service standard was set of 70% of the storage volume) and compensation releases. Unsurprisingly, combined strategies consisting of desalination technology are ranked poorly due to the contribution of high operating costs and limited contribution towards improving reliability as illustrated in Figure 3.20.

MCDA weighting criteria combination 2

The second weighting combination considers reliability and sustainability to be equally weighted (i.e. reliability = 0.5; resilience = 0; sustainability = 0.5). This indicates that the most reliable and sustainable strategies will be ranked higher. Referring to MCDA weighting combination 2 in Figure 5.2, the *Demand management, Temporary Use Bans* strategy *and Goal-seeking Hedge* was the best in terms of improved reliability and sustainability in a data-sparse environment. This result is key as it demonstrates the contribution of combined demand and supply typed strategies. In terms of sustainability, hedging results in lower abstraction costs and this extended to an improved reliability, as the hedging policy maintained reservoir levels and allowed for greater compliance with the reliability criteria. Moreover, no costs were incurred from the implementation of *Temporary Use Bans* (i.e. implies improves sustainability), and this strategy also contributed to reducing the required amount of abstractions (i.e. domestic consumption reduced and lower the stress placed on abstractions), which also contributed to lowering costs. Unsurprisingly, in both environments, strategies comprised of desalination technology were ranked poorly under this weighting combination, and this extends primarily to the implications associated with operating costs.

Moreover, from a data-sparse setting, and in the context of the environmental performance indicator, strategies consisting of desalination technology inherit the implications discussed. Results for the environmental performance indicator showed strategies comprising of the *Goal-seeking Hedge* recorded improved reliability due to the clear benefits of hedging (as a greater volume of storage was maintained in the reservoir), and reduced abstractions (low volumes of abstraction results in low abstraction costs).

MCDA weighting criteria combination 3

The third weighting combination considers resilience and sustainability to be equally weighted (i.e. reliability = 0; resilience = 0.5; sustainability = 0.5). The most striking result was the poor performance of the *Goal-seeking Hedge* under this weighing combination. The *Goal-seeking Hedge* which was ranked promising in the data-rich environment resulted in a lower rank (7th). However, this demonstrates the importance of the reliability concept which was applied in this work, but not considered in the MCDA weighting combination 3 analyses. This indicates that there is a trade-off between reliability and resilience in considering interventions for the Anyplace WRS between data-rich and sparse environments. For the remainder of the strategies, there are no large differences in their performance when compared across both environments.

In the context of the environmental performance indicator, strategies comprised of desalination technology in both settings were ranked lower due to the poor contribution associated with desalination in improving environmental performance, the high cost associated with this type of strategy, and the greater contribution to improved resilience and sustainability from strategies such as *Demand Management* where costs were not considered.

MCDA weighting criteria combination 4

The fourth weighting combination considers reliability and resilience to be equally weighted (i.e. reliability = 0.5; resilience = 0.5; sustainability = 0). In the context of the social performance indicator, all of the interventions comprising of desalination showed promise in terms of achieving high rankings. Desalination allowed for greater compliance with the social resilience performance indicator as desalinated water goes directly into supply. From a reliability perspective, there are less environmental abstractions as the reservoir levels is maintained and this contributed to the improvement in reliability achieved for this weighting combination.

In contrast, strategies that capture management on the demand side were ranked lower under this combination, and the *No Interventions* (i.e. Do-nothing) strategy was ranked last; and this suggests that it contributed the least towards improved reliability and resilience.

The positive benefits of desalination under this combination and also from a data-sparse perspective are well documented above (MCDA weighting combination 4); however, from an environmental performance perspective, the performance indicators captured under the MCDA weighting combination 4 also showed improvements from a reliability perspective. These improvements are demonstrated when strategies comprised of the *Goal-seeking Hedge* have been implemented, and is underpinned by general concept of hedging which allowed for less environmental abstractions and greater conservation of the reservoir storage. Interestingly, strategies comprised of desalination were ranked moderately (i.e. 3rd, 5th, 6th and 7th). These results underscore the shortcomings of desalination as an intervention that contributes in a limited way in the broad view of environmental resilience, and based on these results, also environmental performance.

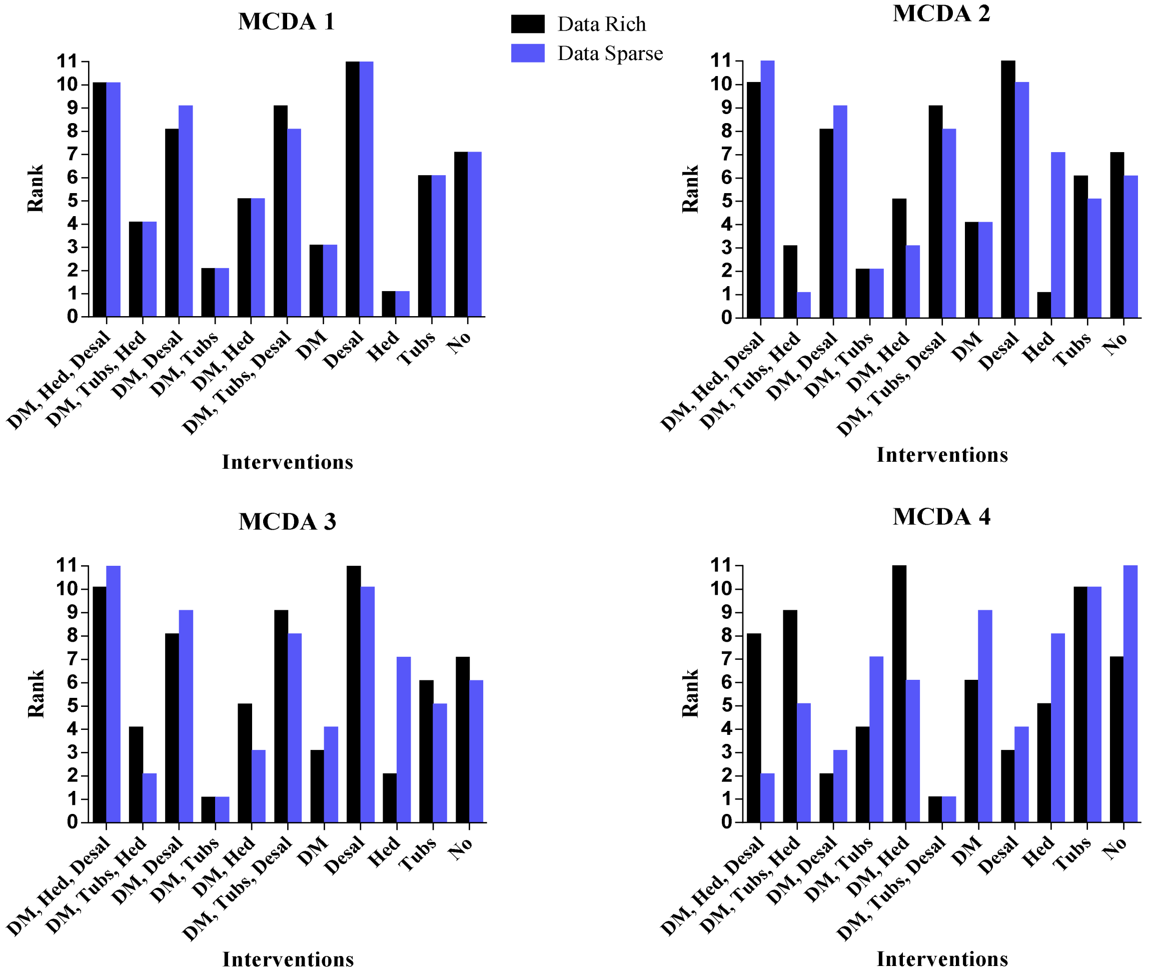


Figure 5.2 Illustration of the MCDA Anyplace data-rich and data-sparse comparison results for the social resilience indicator

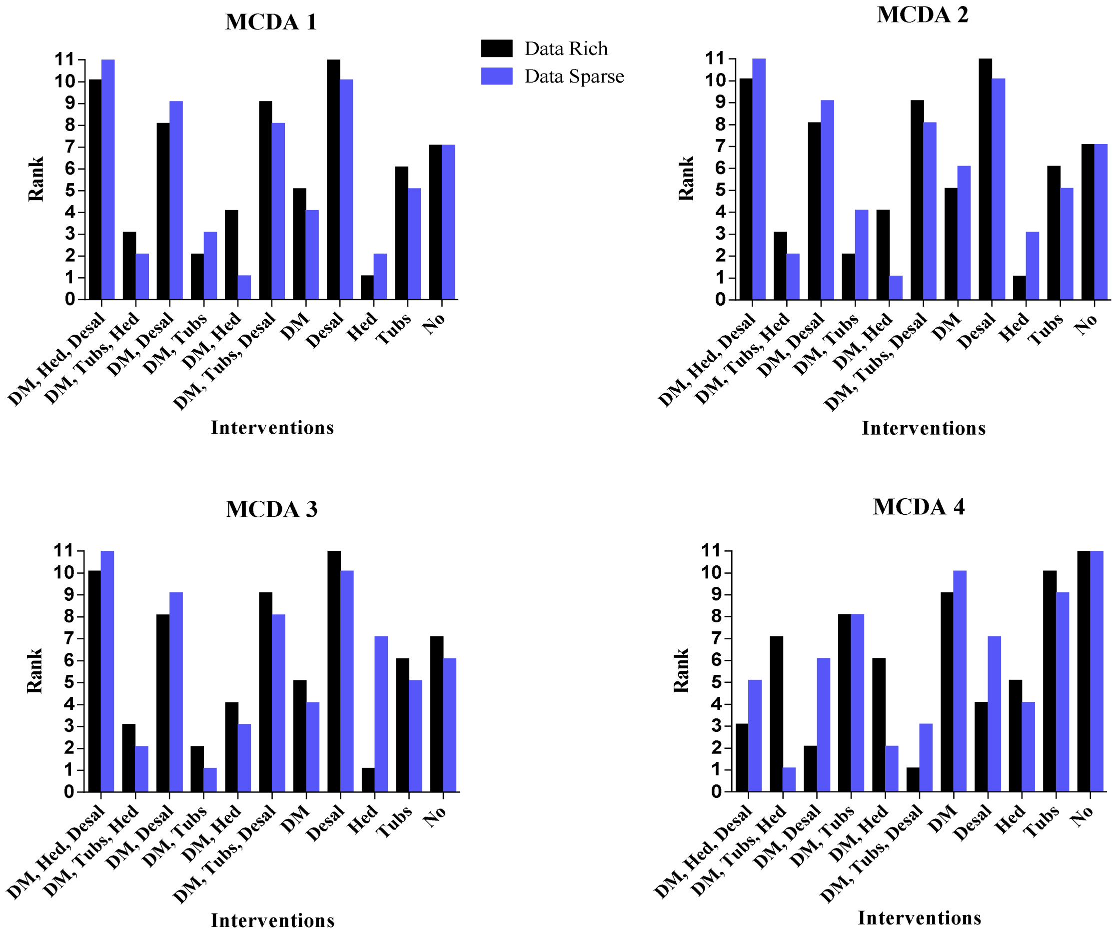


Figure 5.3 Illustration of the MCDA Anyplace data-rich and data-sparse comparison results for the environmental resilience indicator

MCDA weighting criteria combination 5

The fifth weighting combination prioritises the sustainability performance indicator (i.e. reliability = 0.1; resilience = 0.3; sustainability = 0.6). The results for this weighting criteria are shown in Figures 5.4 (illustration of the social resilience indicator results) and Figure 5.5 (illustration of the environmental resilience indicator results). When these are compared to the results obtained in MCDA weighting criteria combination 1 (i.e. the performance indicators are equally weighted) and weighting criteria combination 3, the ranking of the strategies show consistency in both a data-rich and data-sparse setting. These comparisons further highlight the important role of sustainability in the decision-making process, as in the weighting combinations where sustainability was weighted 0.5 or higher, strategies comprised of desalination continued to show no improvement.

MCDA weighting criteria combination 6

The sixth weighting combination prioritises the resilience indicators (i.e. reliability = 0.3; resilience = 0.6; sustainability = 0.1). In contrast to the results obtained for the MCDA weighting criteria combinations 1, 2 and 3, the strategies comprised of desalination except for the *Demand Management, Goal-seeking Hedge and Desalination* (ranked 8th) showed greater potential to improve reliability and resilience in both a data-rich and data-sparse environment when evaluated for the social resilience (Figure 5.4) and environmental resilience (Figure 5.5) performance indicators. The implications underpinning these results extend to the contribution of reliability and resilience discussed above in the MCDA weighting criteria combinations 4.

MCDA weighting criteria combination 7

The seventh weighting combination prioritises the reliability indicator (i.e. reliability = 0.6; resilience = 0.1; sustainability = 0.3). A review of Figure 5.4 and the graph titled 'MCDA 7' shows that the strategies are consistently ranked in both a data-rich and a data-sparse environment (in the context of the social resilience performance indicator). The most striking result was obtained for the *Goal-seeking Hedge* strategy which was ranked 1st in the data-rich environment and 7th in the data-sparse environment. A similar result was obtained for the MCDA weighting criteria combination 2 (i.e. resilience = 0). Similarly, from an environmental perspective, the consistency in the ranking of the strategies were the same in both a data-rich and data-sparse environment, with the exception of the *Goal-seeking Hedge* strategy which was ranked 1st in the data-rich environment and 5th in the data-sparse environment.

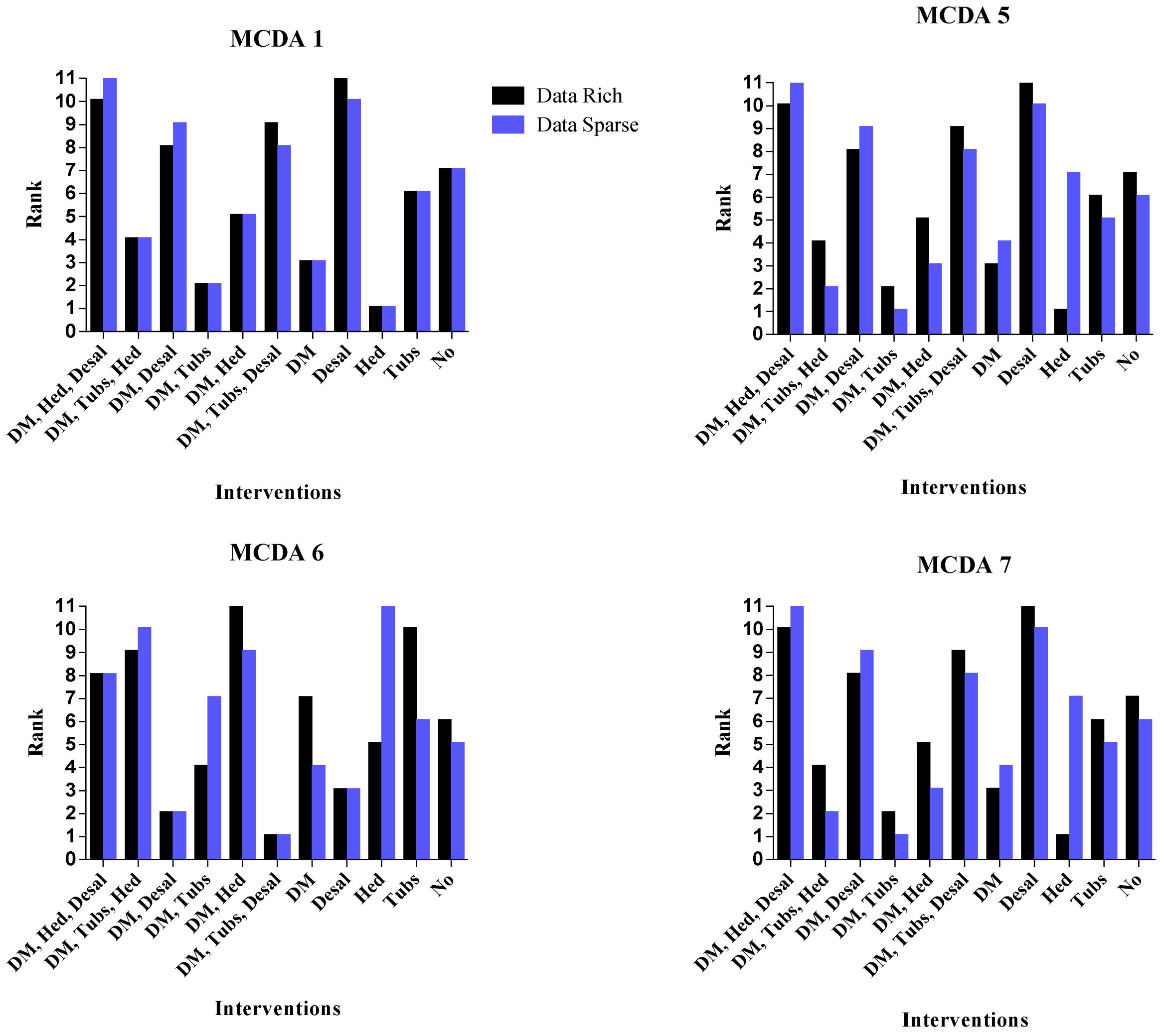
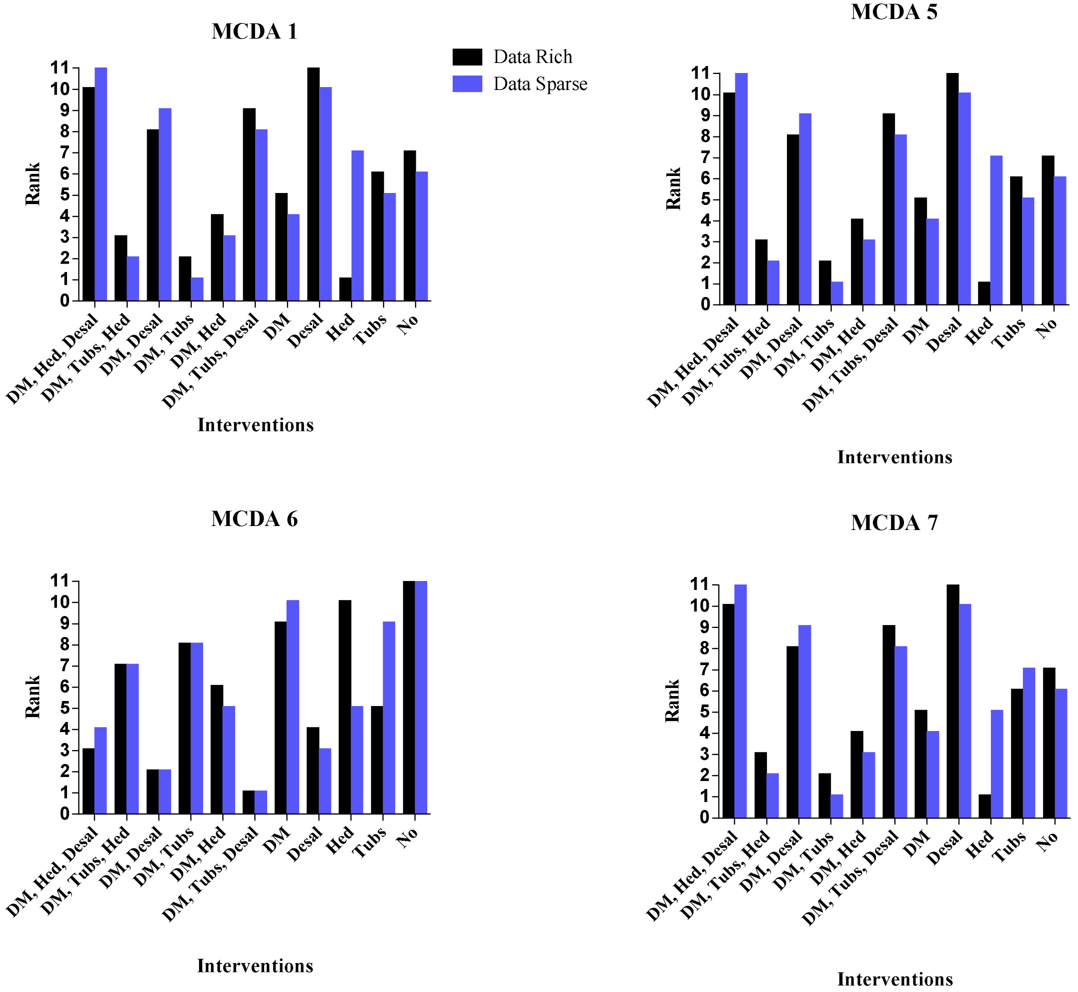


Figure 5.4 MCDA Anyplace data-rich and data-sparse unequal weighting results compared to the MCDA 1 equal weighting combination criteria result for the social resilience indicator.



194

Figure 5.5 MCDA Anyplace data-rich and data-sparse unequal weighting results compared to the MCDA 1 equal weighting combination criteria result for the environmental resilience indicator.

5.4. Conclusions

A MCDA approach for decision making under data-sparse conditions is demonstrated in this Chapter to explore the validity of the data-sparse results obtained in Chapter 4. To introduce this approach, the Anyplace water resources system was transformed from a data-rich to a data-sparse state, setting the scene for decision makers who can use this methodology to validate intervention strategies which have been planned for implementation in a data-sparse environment. This was performed by substituting the data gaps patterns identified in the KSA50 time series for the Yallahs pipeline, into the FFH time series representative of river flows at catchment CM1 during modelling. The 11 FFH are distinct in nature and present a snapshot of uncertain hydrological conditions associated with supply availability in the anyplace region between the evaluation period 2020 to 2049. To eliminate any bias during the data-sparse transformation process, Monte Carlo sampling was utilised to determine the start period for the data-sparse conditions. Results demonstrated the robustness of the intervention strategies tested in the data-rich and data-sparse case studies.

As a part of the MCDA approach, seven weighting combinations were tested to accommodate a wide realm of possibilities in terms of decision making. The seven MCDA weighting combinations were applied to the Aquator model results, and in each of them, a different performance indicator was assigned the highest ranked weighted value. The strategies were ranked from the highest (1) to lowest (11). For the MCDA weighting combination 1, the results for both the data-rich and data-sparse evaluation showed a high level of consistency. This has positive implications for the KSA50 data-sparse case study results, as it can be assumed that the KSA50 case study strategies would perform in a robust manner demonstrating the same level of consistency in terms of identifying the rank of the strategies that would contribute to improve reliability, resilience and sustainability in a Kingston and St. Andrew data-rich setting.

Moreover, the results from the remainder of the MCDA weighting combinations demonstrate how strategies can be prioritise for the water resources system planning process based on which types of performance indicators are of interest to stakeholders. The weighting combinations underpin the strengths and weaknesses of some of the performance indicators, and this is especially transparent in the MCDA weighting combinations 4 and 6, when sustainability was assigned the least weighting value, and strategies comprising of desalination were showed promising in the data-rich and data-sparse setting from a social resilience perspective – highlighting trade-offs amongst reliability, resilience and sustainability. In contrast, for the MCDA weighting combinations 2 and 3, combined strategies comprising of desalination contributed less towards improved sustainability when the sustainability indicator was captured. In those cases, the *Demand Management* strategies were ranked higher.

The most striking result occurred when the MCDA weighting combination 4 was analysed from an environmental perspective, as desalination which performed at a high level when evaluated for the social performance indicator, was not a feasible strategy to implement in the context of the combination tested. A factor which contributed to this relates to the configuration of the water resources model (i.e. desalinated water goes directly into domestic consumption).

CHAPTER SIX

6. Conclusions and Recommendations

6.1. Introduction

This chapter brings all of the results and discussion together to demonstrate how the thesis aim and objectives outlined in Chapter 1 were achieved. In section 6.2, the conclusions underpinning each of the research objectives are summarised and achievement of the aim highlighted. This is followed by the main contributions to research in subsection 6.3, and recommendations for future work in section 6.4.

6.2. Summary of objectives

Objective 1: Review literature on water resources management approaches.

As a result of the review of literature carried out in Chapter 2, the following conclusions can be drawn:

- An integrated approach to water management is considered a step in the right direction away from conventional water management approaches. In the early stages of this new paradigm, several questions were raised mainly in academia regarding transferring theory into practice. As a result, some of the early criticism has labelled the IWRM process as a ‘one size fits all' approach, calling for a more flexible approach. These critics pointed out that some of the principles outlined in the IWRM approach present challenges to implement in lesser developed countries, and named some challenges as overcoming political instability implying that the public may not be allowed to contribute in the integrated process. Moreover, other challenges pertained to the lack of familiarity with IWRM technical terminology and lack of investments. Overall, the lack of alternative guidelines in the integrated management process contributes to the gap in water resources management in those lesser developed regions when compared to more stable and developed regions.

- Moreover, recent studies suggest that some water planners and decision makers have resorted to constructing newly built infrastructure in developing countries (e.g. some parts of Asia) as solutions to management of available resources. However, no link was found between the challenges explored in Chapter 2, and this pattern of constructing newly built infrastructure. The studies that highlight the backward step called for the more widespread use of existing built infrastructure together with demand management strategies.

- In contrast, the literature review revealed that in some developed countries more passive and cautious approaches are being investigated. A large volume of water resources research is focused on water resources planning under uncertainty and is accomplished by conducting water resources assessments under various climate change scenarios and adopting the use of algorithms to explore various future conditions. A key aspect highlighted in those studies is the combined used of existing built infrastructure, and conservative approaches;

- Other literature highlighted the need for methods and approaches to assess WRS in data-sparse (e.g. developing countries) contexts.

Objective 2: Explore types of water resources models and software applications.

- A comprehensive review of water resources simulators was performed highlighting the strengths and weaknesses of commonly used simulators. This review led to Aquator being selected as the simulator software to assess the three water resources case studies. The literature review revealed key features and advantages of the Aquator water such as VBA scripting which allowed for strategies such as *Temporary Use Bans* and the *Goal-Seeking Hedge* to be modelled in the Anyplace case studies (Chapter 3 & 5).

Objective 3: Explore intervention strategies used in water resources management and select a feasible suite of strategies for the management of water resources systems in a data-rich and a data-sparse setting.

- A comprehensive review was conducted in Chapter 2 of supply-driven, and demand management interventions commonly used in water resources planning. The strategies were then prioritised using the decision matrix explored in Chapter 3 which led to the most feasible and realistic strategies being chosen for the Anyplace case studies. In the context of the Kingston and St. Andrew case study, the most feasible (in terms of comprising of sufficient parameters and relevant information) strategies planned in the 1990 Water Resources Authority, Water Resources Master Plan were tested in that case study.

Objective 4: Develop and model the implications of plausible future domestic demand and climate change scenarios representative of possible uncertainties which could affect the performance of the data-rich and the data-sparse water resources systems.

The main conclusions derived from the development of future scenarios carried out in Chapter 3 and 4 and from their application to the case studies are:

- Three demand scenarios (low, mid-range & high) were developed for the Anyplace case studies by adopting the framework used by the Office for National Statistics (ONS), and then extending those projections to the year 2049. These demand scenarios built plausible representations of the future demands covering a range of demographic possibilities (i.e. different levels of migration, mortality, etc.) assumed by the ONS for the UK. Moreover, supply uncertainty was addressed through the 11 Future Flow Hydrology time series between 2020 – 2049. These envision a range of possible hydrological conditions that could impact on supply availability as a result of climate change.

- The three demand scenarios developed for the Anyplace case studies in this thesis are flexible and can be adapted to different locations in the United Kingdom.

- Deployable output assessments were conducted to explore the data-rich water resources system performance and its response to demand increase under the three levels of demand uncertainty. This revealed that from a historical perspective (i.e. using historical parameters) the Anyplace data-rich water resources system performed well under the low and mid-range demand uncertainty. However, under extreme demand uncertainty, high levels of failure were observed especially during the historical drought periods. The implications of this performance under historical conditions led to the assessment of the Anyplace water resources systems assuming extreme demand uncertainty across the 11 plausible FFH hydrological conditions.

- In relation to the advantages of the water resources simulator, Aquator, explored in Chapter 2, different combinations of intervention strategies were scheduled and modelled using VBA. Moreover, a key contribution to water resources modelling is demonstrated through the modelling of the *Temporary Use Bans* and the *Goal-seeking Hedge* intervention strategies, which had not previously been modelled in Aquator at the time of writing.

Objective 5: Develop a method to evaluate and prioritise the reliability, resilience and sustainability of intervention strategies for the case studies.

The development of a method that allows the assessment of strategies for a wide range of objectives and future scenarios (presented in Chapter 3, 4 and 5) and its application to the three case studies revealed the following:

- Assessing resilience using social and environmental indicators was demonstrated to be useful for quantifying the possible consequences affecting customers and the environment in a future timeline. The water resources assessment demonstrated how these indicators can be operationalised (i.e. examining the period representing the maximum consecutive failure days or maximum consecutive period of non-compliance with a level of service criterion) to represent the overall performance of the water resources system before and after the implementation of the prioritised intervention strategies, informing decision makers in terms of deficits and durations under a range of scenarios representing uncertainty.
- The integration of different multi-criteria weighting combinations via an MCDA in the data-rich and data-sparse case studies helped identified the intervention strategies that performed the same when the performance indicators (i.e. reliability, resilience and sustainability) were equally weighted, from those strategies whose performances were different when analysed for the four weighting combinations. In the broader picture, this methodology demonstrated how intervention strategies could be prioritised to capture different performance requirements when multiple decision makers are involved in the decision-making process. Moreover, a key finding observed in the results relates to the sustainability indicator. When this indicator was assigned a weighted score of zero, the strategies that were ranked 4th to 6th under equal weighting (MCDA weighting combination 1), performed less desirable and resulted in lower rankings. In the context of the Anyplace case studies, this demonstrated the importance of sustainability during decision making.

Objective 6: Develop a methodology to facilitate the transformation of a water resources system from a data-rich environment to a data-sparse environment in order to demonstrate the validity of strategy selection in a data-sparse setting.

- The methodology developed in Chapter 5 transformed the Anyplace water resources system from a data-rich environment to a data-sparse environment. The assessment of Anyplace under data-sparse conditions revealed identical results to those observed in the data-rich case study, in terms of the three highest ranked strategies and the lowest ranked intervention strategy. The main implication of adopting this assessment methodology, and comparing the assessments results (in both a data-rich and data-sparse setting), improves the validity of the selection of strategies in a data-sparse setting.

Objective 7: **Assess the robustness of intervention strategies generated under data-rich circumstances as compared with data-poor.**

- The Kingston and St. Andrew case study was characterised as data-sparse due to the large number of assumptions utilised to fill data and information gaps. Aquator was utilised to test the feasible set of strategies identified in relation to Objective 3. The intervention strategies which contributed to improved reliability, resilience and sustainability were identified when the model performance was assessed using the performance indicators developed in Objective 4. Moreover, the Multi-criteria decision-making weighting combinations explored in this thesis have shown to be consistent from a comparative perspective in identifying the intervention strategies that contributed to improved reliability, resilience and sustainability for the Anyplace data-rich and data-sparse water resources system. These strong sets of results underpin the methodology developed in Objective 7, which transformed a data-rich water resources system to data-sparse.

As a result of the achievement of these objectives, the formal aim of the thesis, which was to develop a water resources system assessment methodology incorporating aspects of data sparsity through its application to a region of Kingston, Jamaica using insights from a data-rich setting in the UK, in order to assess the reliability, resilience and sustainability of interventions proposed for implementation in the data-sparse Jamaican case study. This has been accomplished by transforming the Anyplace (data-rich) case study into a data-sparse case study, then evaluating the intervention strategies implemented in the Anyplace data-rich setting, in a data-sparse setting (Anyplace data-sparse). A comparative analysis facilitated through the application of a multi-criteria decision analysis methodology, showed small differences in the rankings of the intervention strategies when the results of the Anyplace data-rich and the Anyplace data-sparse studies were compared. An implication of this result suggests that if the interventions tested in the data-sparse Kingston, Jamaica region were tested under data-rich settings, it could be assumed that it is possible that a similar trend in terms of the level of consistency observed in the Anyplace comparative assessment could be achieved. The results from comparative assessment were used to underpin the intervention strategy that showed greater potential to improve reliability, resilience and sustainability in the Kingston, Jamaica data-sparse case study.

6.3. Summary of Contributions

The work presented in this thesis has contributed to the field of water resources management in the following ways:

1) The extension of the Safe and SuRe framework and resilience definition allowed for the development of social and environmental performance indicators. These indicators characterise the compliance with domestic demands, and the level of service for compensation releases respectively.

2) The development of a method that transforms a water resources system from a data-rich environment to a data-sparse environment, and sets the scene for decision makers to validate intervention strategies which have been planned for implementation in a data-sparse environment.

3) The integration of MCDA through the weighting combinations explored in this thesis demonstrates how different stakeholder perspectives could influence the prioritisation of WRS intervention strategies.

4) Building on points (2) and (3) above, the comparison of the results obtained from the analysis of the MCDA results demonstrated the robustness of the intervention strategies tested in the data-rich and data-sparse case studies. In addition, the positive results obtained from this approach set the scene for decision makers who may be faced with similar challenges in the future, as this methodology could be used to validate intervention strategies which have been planned for implementation in a data-sparse environment.

6.4. Recommendations for future work

The work presented in this thesis can be complemented, expanded and built upon regarding the following themes for further research:

1) The degree of data-sparseness can be enhanced by converting the time series inputs at the remaining catchments (i.e. CM 2 and CM 3) to a data-sparse state based on the data-sparse methodology utilised in this thesis. This would allow for a more comprehensive comparison of strategies under data-rich and data-sparse conditions.
2) The Safe and SuRe framework could be extended to capture equity, especially in the context of low-income countries.

6.4.1 Recommendations for the Water Industry

1) The social and environmental resilience performance indicators are useful tools that can be extended beyond characterising compliance with domestic demands and the level of service for compensation releases. For instance, in water quality modelling, and from an environmental perspective, this concept could be used to characterise the length of time (failure duration) a water body remains in a deteriorated state (e.g. moving from a good status to moderate or lower status) by quantifying the amount of ammonia, phosphate or bio-dissolved oxygen (BOD) (failure magnitude) that resulted in the deterioration of the water body status. The implications of such an

approach could assist utility companies in complying with the Water Framework Directive guidelines.

2) Multiple stakeholders' perspectives can be captured from public surveys using the multi-criteria analysis approached utilised in this thesis to address the concerns and the objectives of all stakeholders during the selection of intervention strategies.

6.5. Closing Remarks

The three case studies presented in this thesis have accomplished the aim and objectives presented in Chapter 1. In addition, original contributions are demonstrated in the water resources assessments by proposing how performances under data-sparse conditions can be validated. The work in this thesis is especially relevant to lesser developed regions where poor data collection methods, poor infrastructure and lack of investment has contributed to data sparsity.

References

Abbotsford Mission Water & Sewer Services; (2012) *Abbotsford Mission Water and Sewer Services Report*. [online]Available at: https://abbotsford.civicweb.net/document/29592 [Accessed: 14 October 2017].

Abdelrhem, I.M., Raschid, K. and Ismail, A., 2008. Integrated groundwater management for great man-made river project in Libya. *Eur. J. Sci. Res*, *22*, pp.562-569.

Abdudayem, A. and Scott, A.H., 2014. Water infrastructure in Libya and the water situation in agriculture in the Jefara region of Libya. *African Journal of Economic and Sustainable Development*, *3*(1), pp.33-64.

Affinity Water. (2012) *Questions & Answers Temporary use ban*. [online] Available at: https://www.affinitywater.co.uk/docs/drought-QA-3-JUL-2012.pdf [Accessed: 15 October 2017].

Agrawal, N. and Desmukh, T.S., Rainfall Runoff Modeling using MIKE 11 Nam–A Review.

Ainuddin, S. and Routray, J.K., 2012. Community resilience framework for an earthquake prone area in Baluchistan. *International Journal of Disaster Risk Reduction*, *2*, pp.25-36.

Al-Mashaqbeh, O.A., Ghrair, A.M. and Megdal, S.B., 2012. Grey water reuse for agricultural purposes in the Jordan Valley: Household survey results in Deir Alla. *Water*, *4*(3), pp.580-596.

Al Radif, A., 1999. Integrated water resources management (IWRM): an approach to face the challenges of the next century and to avert future crises. *Desalination*, *124*(1-3), pp.145-153.

Amos, C.C., Rahman, A. and Gathenya, J.M., 2018. Economic analysis of rainwater harvesting systems comparing developing and developed countries: A case study of Australia and Kenya. *Journal of Cleaner Production*, *172*, pp.196-207.

Anderson, R.M. and Goodall, J.L., 2006. Regionalization of IHACRES Model Parameters for In-tegrated Assessment across the Lake Erie, northern Ohio USA basin.

Andreu, J., Capilla, J. and Sanchís, E., 1996. AQUATOOL, a generalized decision-support system for water-resources planning and operational management. *Journal of hydrology*, *177*(3-4), pp.269-291.

AP. (2012). Spring hosepipe ban announced for London and south-east. [online] Available at: https://www.theguardian.com/environment/2012/mar/12/hosepipe-ban-london-south-east-water [Accessed 15 Oct. 2017].

Arena, C., Cannarozzo, M., Fortunato, A., Scolaro, I. and Mazzola, M.R., 2014. Evaluating Infrastructure Alternatives for Regional Water Supply Systems by Model-Assisted Cost-Benefit Analysis–A Case Study from Apulia, Italy. *Procedia Engineering*, *89*, pp.1460-1469.

Arena, C., Cannarozzo, M., Fortunato, A., Scolaro, I. and Mazzola, M.R., 2015. Sensitivity of regional water supply systems models to the level of skeletonization–a case study from Apulia, Italy. *Procedia Engineering*, *119*, pp.535-544.

Bazza, M., 2002. Water resources planning and management for drought mitigation. In *Regional Workshop on Capacity Building on Drought Mitigation in the Near East, Rabat (Morocco), 1-5 Nov 2002*. FAO.

Beatty, R. and McLindin, M., 2012. Rainwater harvesting and urban design in Australia. *Proceedings of the Water Environment Federation*, *2012*(9), pp.6435-6447.

Bertsekas, D.P. and Tseng, P., 1994. RELAX-IV: A faster version of the RELAX code for solving minimum cost flow problems.

Beven, K., 1989. Changing ideas in hydrology—the case of physically-based models. *Journal of hydrology*, *105*(1-2), pp.157-172.

Biswas, A.K., 2004. Integrated water resources management: a reassessment: a water forum contribution. *Water international*, *29*(2), pp.248-256.

Borgomeo, E., Hall, J.W., Fung, F., Watts, G., Colquhoun, K. and Lambert, C., 2014. Risk-based water resources planning: Incorporating probabilistic nonstationary climate uncertainties. *Water Resources Research*, *50*(8), pp.6850-6873.

Bosshard, P., 2013. The World Bank is bringing back big, bad dams. *The Guardian*, *16*.

Boyce, S. (2011) D*ata Collection and Management Strategy for the Commonwealth of Dominica*. Belmopan, Belize.

Bruneau, M., Chang, S.E., Eguchi, R.T., Lee, G.C., O'Rourke, T.D., Reinhorn, A.M., Shinozuka, M., Tierney, K., Wallace, W.A. and Von Winterfeldt, D., 2003. A framework to quantitatively assess and enhance the seismic resilience of communities. *Earthquake spectra*, *19*(4), pp.733-752.

Bryant, B.P. and Lempert, R.J., 2010. Thinking inside the box: a participatory, computer-assisted approach to scenario discovery. *Technological Forecasting and Social Change*, *77*(1), pp.34-49.

Buettner, A. & (2002) *How to Select an Evidence-Based Intervention A Guide*. [online] Available at: http://cle.osu.edu/%0Awp-content/uploads/programselection.pdf. [Accessed 12 October 2017]

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

Butler, D., Ward, S., Sweetapple, C., Astaraie-Imani, M., Diao, K., Farmani, R. and Fu, G., 2017. Reliable, resilient and sustainable water management: the Safe & SuRe approach. *Global Challenges*, *1*(1), pp.63-77.

Cabinet Office (2011) *Keeping the Country Running: Natural Hazards and Infrastructure*. London.

Casal-Campos, A., Fu, G., Butler, D. and Moore, A., 2015. An integrated environmental assessment of green and gray infrastructure strategies for robust decision making. *Environmental science & technology*, *49*(14), pp.8307-8314.

Cashman, A., Nurse, L. and John, C., 2010. Climate change in the Caribbean: the water management implications. *The Journal of Environment & Development*, *19*(1), pp.42-67.

CCCCC (2015) *Caribbean Community Climate Change Centre (CCCCC)*. [online] Available at: http://clearinghouse.caribbeanclimate.bz/?db_type=Climate Model&country=&collection=V541&s=&sector=&topic=. [Accessed 13 June 2015]

Cohen, J., 1988. Statistical power analysis for the behavioral sciences . Hilsdale. *NJ: Lawrence Earlbaum Associates*, *2*.

Cole, E. (2013) *Caribbean Development Bank Final Draft Report Jamaica*.

Colorado Water Conservation Board (2010) *Guidebook of Best Practices for Municipal Water Conservation in Colorado SUMMARY 2*. [online] Available at: https://www.colorado.gov/pacific/sites/default/files/13WaterResources0927ColoradoWaterWiseSummaryofBestPractices.pdf [Accessed: 15 October 2017].

Crawford, N.H. and Burges, S.J., 2004. History of the Stanford watershed model. *Water Resources Impact*, *6*(2), pp.3-5.

Croke, B.F.W. and Norton, J.P., 2004. Regionalisation of rainfall-runoff models.

Crow-Miller, B., Webber, M. and Molle, F., 2017. The (Re) turn to Infrastructure for Water Management?. *Water Alternatives*, *10*(2), pp.195-207.

Davies, K., Doolan, C., van den Honert, R. and Shi, R., 2014. Water-saving impacts of Smart Meter technology: An empirical 5 year, whole-of-community study in Sydney, Australia. *Water Resources Research*, *50*(9), pp.7348-7358.

Deloitte LLP (2015) *Water trading - scope, benefits and options*. [online] Available at: https://064f1d25f5a6fb0868ac-0df48efcA31bcf2ed0366d316cab9ab8.ssl.cf3.rackcdn.com/wp-content/uploads/2015/12/rpt_com201512deloittewatertrading.pdf [Accessed: 18 October 2017].

Deltares (2015) *RIBASIM*. [online] Available at: https://www.deltares.nl/en/software/ribasim/ [Accessed 10 December 2015].

Desai, V.R., Ghosh, S.N. and Patro, J., 2005. National water grid—Its objective and challenge. *ISH Journal of Hydraulic Engineering*, *11*(1), pp.1-10.

DHI (2003) *MIKE 11 - A modelling system for rivers and channels*.

DHISoftware (2003) ‘MIKE 11 A modelling system for Rivers and Channels User Guide’, *DHI Software*. Available at: https://www.tu-braunschweig.de/Medien-DB/geooekologie/mike11usersmanual.pdf (Accessed: 29 October 2017).

Domínguez, I., Ward, S., Mendoza, J.G., Rincón, C.I. and Oviedo-Ocaña, E.R., 2017. End-user cost-benefit prioritization for selecting rainwater harvesting and greywater reuse in social housing. *Water*, *9*(7), p.516.

DTI ., 2002. Foresight Futures 2020–Revised scenarios and guidance. *UK Department of Trade and Industry*, p.32.

EA (2001) *A scenario approach to water demand forecasting*. West Sussex.

EA (2013) *Climate change approaches in water resources planning: overview of new methods*.

Egbuniwe, N. and Todd, D.K., 1976. Application of the Stanford watershed model to Nigerian watersheds. *JAWRA Journal of the American Water Resources Association*, *12*(3), pp.449-460.

European Commission (1999) *Towards sustainable water resources management : a strategic approach*. Luxembourg, Luxembourg: Office for Official Publications of the European Communities.

Evans, C.J. and Vickers, D.O., 1973. The Climate of Jamaica. *National Mvleteorological Service, Kingston*.

eWater (2004) *Rainfall Runoff Library User Guide - eWater Toolkit*. Melbourne.

Faber, B.A. and Stedinger, J.R., 2001. Reservoir optimization using sampling SDP with ensemble streamflow prediction (ESP) forecasts. *Journal of Hydrology*, *249*(1), pp.113-133.

FAO (2016) *Libya*. [online] Available at: http://www.fao.org/nr/water/aquastat/countries_regions/LBY/ [Accessed 18 October 2017].

Farrrell, D., Nurse, L. and Moseley, L., 2007. *Managing water resources in the face of climate change: A Caribbean perspective*. UWI.

Fawthrop, N. (1994) 'Modelling Hydrological Processes for River Management', in *The Rivers Handbook*. Two. Blackwell ScientificPublications, p. 523.

Felfelani, F., Movahed, A.J. and Zarghami, M., 2013. Simulating hedging rules for effective reservoir operation by using system dynamics: a case study of Dez Reservoir, Iran. *Lake and reservoir management*, *29*(2), pp.126-140.

Fink, A., 2003. The survey handbook (Vol. 1). Sage.

Fletcher-Paul, L.M., Madramootoo, C.A. and Thomas, H., Appropriate Technologies and Systems to respond to Climate Change, Improved Water Resources Management, Waste Management and Sanitation.

Forsythe, N., Fowler, H.J., Blenkinsop, S., Burton, A., Kilsby, C.G., Archer, D.R., Harpham, C. and Hashmi, M.Z., 2014. Application of a stochastic weather generator to assess climate change impacts in a semi-arid climate: The Upper Indus Basin. *Journal of hydrology*, *517*, pp.1019-1034.

French, P.N. and Taylor, M.R., Development and use of map-based simulation shells for creating shared-vision models.

Funke, N., Oelofse, S.H.H., Hattingh, J., Ashton, P.J. and Turton, A.R., 2007. IWRM in developing countries: Lessons from the Mhlatuze Catchment in South Africa. *Physics and Chemistry of the Earth, Parts A/B/C*, *32*(15), pp.1237-1245.

García-Montoya, M., Bocanegra-Martínez, A., Nápoles-Rivera, F., Serna-González, M., Ponce-Ortega, J.M. and El-Halwagi, M.M., 2015. Simultaneous design of water reusing and rainwater harvesting systems in a residential complex. *Computers & Chemical Engineering*, *76*, pp.104-116.

Georgakakos, A.P., Yao, H., Kistenmacher, M., Georgakakos, K.P., Graham, N.E., Cheng, F.Y., Spencer, C. and Shamir, E., 2012. Value of adaptive water resources management in Northern California under climatic variability and change: reservoir management. *Journal of Hydrology*, *412*, pp.34-46.

Gerrits, A.M.J., Savenije, H.H.G., Veling, E.J.M. and Pfister, L., 2009. Analytical derivation of the Budyko curve based on rainfall characteristics and a simple evaporation model. *Water Resources Research*, *45*(4).

Gleick, P.H. and Cooley, H.S., 2009. Energy implications of bottled water. *Environmental Research Letters*, *4*(1), p.014009.

Google Earth Pro 7.0 (2017). *Kingston, Jamaica, 18°00’10.12” N, 76°49’08.11”W, eye altitude 215.92 km, Terrain data layer viewed 7 May 2016.*

Google Maps (2017) *Google Maps*. [online] Available at: https://www.google.co.uk/maps/place/Jamaica/@16.9242153,-81.7587398,5z/data=!4m5!3m4!1s0x8eda2a1bc6cf719d:0x59a0d1c0b5120efa!8m2!3d18.109581!4d-77.297508 [Accessed: 30 October 2017].

Götzinger, J. and Bárdossy, A., 2007. Comparison of four regionalisation methods for a distributed hydrological model. *Journal of Hydrology*, *333*(2-4), pp.374-384.

Griffiths, J., Young, A.R. and Keller, V., 2006. Continuous Estimation of River Flows (CERF)–Technical Report: Task 1.3: Model scheme for representing rainfall interception and soil moisture. *CEH, Wallingford*.

GWP, (2000). Integrated Water Resources Management Global Water Partnership Technical Advisory Committee (TAC) Global Water Partnership. [online] Available at: http://www.gwp.org/Global/ToolBox/Case Studies/Americas and Caribbean/Guatemala. IWRM successful experiences, San Jeronimo Basin, Baja Verapaz %28%23321%29 ENGLISH.pdf [Accessed: 27 September 2016].

Hall, J. and Borgomeo, E., 2013. Risk-based principles for defining and managing water security. *Phil. Trans. R. Soc. A*, *371*(2002), p.20120407.

Hall, J.W., Watts, G., Keil, M., de Vial, L., Street, R., Conlan, K., O'Connell, P.E., Beven, K.J. and Kilsby, C.G., 2012. Towards risk-based water resources planning in England and Wales under a changing climate. *Water and Environment Journal*, *26*(1), pp.118-129.

Hamlet, A.F., Huppert, D. and Lettenmaier, D.P., 2002. Economic value of long-lead streamflow forecasts for Columbia River hydropower. *Journal of Water Resources Planning and Management*, *128*(2), pp.91-101.

Hansen, D.P., Ye, W., Jakeman, A.J., Cooke, R. and Sharma, P., 1996. Analysis of the effect of rainfall and streamflow data quality and catchment dynamics on streamflow prediction using the rainfall-runoff model IHACRES. *Environmental Software*, *11*(1), pp.193-202.

Heck, N., Paytan, A., Potts, D.C. and Haddad, B., 2016. Coastal residents' literacy about seawater desalination and its impacts on marine ecosystems in California. *Marine Policy*, *68*, pp.178-186.

Holness, C. (2017) *Parishes of Jamaica*. [online] Available at: http://www.my-island-jamaica.com/jamaica_parishes.html [Accessed 30 October 2017].

Huskova, I., Matrosov, E.S., Harou, J.J., Kasprzyk, J.R. and Lambert, C., 2016. Screening robust water infrastructure investments and their trade-offs under global change: A London example. *Global Environmental Change*, *41*, pp.216-227.

ICWE (1992) *Conference on Water and the Environment: The Dublin Statement*. [online] Available at: http://www.wmo.int/pages/prog/hwrp/documents/english/icwedece.html. [Accessed 18 October 2017]

Ifejika Speranza, C., Wiesmann, U. and Rist, S., 2014. An indicator framework for assessing livelihood resilience in the context of social–ecological dynamics. *Global Environmental Change*, *28*, pp.109-119.

IPCC (2000) *IPCC Special Report Emissions Scenarios Summary for Policymakers Emissions Scenarios*. [online] Available at: https://ipcc.ch/pdf/special-reports/spm/sres-en.pdf [Accessed 23 October 2017].

Ireson, A., Makropoulos, C. and Maksimovic, C., 2006. Water resources modelling under data scarcity: coupling MIKE BASIN and ASM groundwater model. *Water resources management*, *20*(4), pp.567-590.

Jamrah, A., Al-Futaisi, A., Prathapar, S. and Al Harrasi, A., 2008. Evaluating greywater reuse potential for sustainable water resources management in Oman. *Environmental monitoring and assessment*, *137*(1-3), p.315.

Johnston, R. and Smakhtin, V., 2014. Hydrological modeling of large river basins: how much is enough?. *Water resources management*, *28*(10), pp.2695-2730.

Jønch-Clausen, T., 2004. Integrated water resources management (IWRM) and water efficiency plans by 2005: Why, what and how. *Why, what and how*, pp.5-4.

Kayaga, S. and Smout, I., 2014. Tariff structures and incentives for water demand management. *Proceedings of the Institution of Civil Engineers*, *167*(8), p.448.

Kim, Y.O., Eum, H.I., Lee, E.G. and Ko, I.H., 2007. Optimizing operational policies of a Korean multireservoir system using sampling stochastic dynamic programming with ensemble streamflow prediction. *Journal of Water Resources Planning and Management*, *133*(1), pp.4-14.

Kinzig, L.,Gunderson, A., Quinlan, A., Walker, B., Cundhill, G., Beier, C., Crona, B. and Boden, Ö., 2010. Assessing resilience in social-ecological systems: workbook for practitioners. *Revised Version*, *2*.

Kraume, M., Scheumann, R., Baban, A. and El Hamouri, B., 2010. Performance of a compact submerged membrane sequencing batch reactor (SM-SBR) for greywater treatment. *Desalination*, *250*(3), pp.1011-1013.

Lach, D., Ingram, H. and Rayner, S., 2004. Maintaining the status quo: how institutional norms and practices create conservative water organizations. *Tex L. Rev.*, *83*, p.2027.

Lalozaee, A., Pahlavanravi, A., Bahreini, F., Ebrahimi, H. and Iezadi, H., 2013. Efficiency Comparison of IHACRES Model and Artificial Neural Networks (ANN) in Rainfall-Runoff Process Simulation Inkameh Watershed (a case study Inkhorasan Province, NE Iran). *International Journal of Agriculture*, *3*(4), p.900.

Lattemann, S. and Höpner, T., 2008. Environmental impact and impact assessment of seawater desalination. Desalination, 220(1-3), pp.1-15.

Li, H. (2007) *Hierarchical risk assessment of water supply systems*.

Liang, X., Lettenmaier, D.P., Wood, E.F. and Burges, S.J., 1994. A simple hydrologically based model of land surface water and energy fluxes for general circulation models. *Journal of Geophysical Research: Atmospheres*, *99*(D7), pp.14415-14428.

van der Linden, S. and Woo, M.K., 2003. Transferability of hydrological model parameters between basins in data-sparse areas, subarctic Canada. *Journal of Hydrology*, *270*(3), pp.182-194.

Liu, J., Chen, S., Wang, H. and Chen, X., 2015. Calculation of carbon footprints for water diversion and desalination projects. *Energy Procedia*, *75*, pp.2483-2494.

Liu, Z., Martina, M.L. and Todini, E., 2005. Flood forecasting using a fully distributed model: application of the TOPKAPI model to the Upper Xixian

Catchment. *Hydrology and Earth System Sciences Discussions*, *9*(4), pp.347-364.

Loucks, D.P., Van Beek, and E. Stedinger, J., 2005. *Water Resource Systems Planning and Management: An Introduction to Methods, Models, and Applications*. Springer.

Low, K.G., Grant, S.B., Hamilton, A.J., Gan, K., Saphores, J.D., Arora, M. and Feldman, D.L., 2015. Fighting drought with innovation: Melbourne's response to the Millennium Drought in Southeast Australia. *Wiley Interdisciplinary Reviews: Water*, *2*(4), pp.315-328.

Lund, J.R., 1996. *Developing seasonal and long-term reservoir system operation plans using HEC-PRM* (No. HEC-RD-40). HYDROLOGIC ENGINEERING CENTER DAVIS CA.

Lunduka, R., Bezabih, M. and Chaudhury, A., 2013. Stakeholder-focused cost-benefit analysis in the water sector: synthesis report.

Mackay, J.D., Jackson, C.R. and Wang, L., 2014. A lumped conceptual model to simulate groundwater level time-series. *Environmental modelling & software*, *61*, pp.229-245.

Makropoulos, C.K., Memon, F.A., Shirley-Smith, C. and Butler, D., 2008. Futures: an exploration of scenarios for sustainable urban water management. *Water Policy*, *10*(4), pp.345-373.

Mandal, D., Labhasetwar, P., Dhone, S., Dubey, A.S., Shinde, G. and Wate, S., 2011. Water conservation due to greywater treatment and reuse in urban setting with specific context to developing countries. *Resources, Conservation and Recycling*, *55*(3), pp.356-361.

March, H., Hernández Hernández, M. and Saurí Pujol, D., 2014. Assessing domestic water use habits for more effective water awareness campaigns during drought periods: a case study in Alicante, eastern Spain.

Matondo, J.I., 2002. A comparison between conventional and integrated water resources planning and management. *Physics and Chemistry of the Earth, Parts A/B/C*, *27*(11), pp.831-838.

Matrosov, E.S., Harou, J.J. and Loucks, D.P., 2011. A computationally efficient open-source water resource system simulator–Application to London and the Thames Basin. *Environmental Modelling & Software*, *26*(12), pp.1599-1610.

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*, *27*(4), pp.1123-1148.

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*, *494*, pp.43-58.

Micklin, P.P., 1988. Desiccation of the Aral Sea: a water management disaster in the Soviet Union. *Science*, *241*(4870), p.1170.

Milly, P.C., Betancourt, J., Falkenmark, M., Hirsch, R.M., Kundzewicz, Z.W., Lettenmaier, D.P. and Stouffer, R.J., 2008. Stationarity is dead: Whither water management?. *Science*, *319*(5863), pp.573-574.

Mirshahi, B., 2010. *Hydrological modelling in data-sparse snow-affected semiarid areas* (Doctoral dissertation, Imperial College London).

Moriasi, D.N., Arnold, J.G., Van Liew, M.W., Bingner, R.L., Harmel, R.D. and Veith, T.L., 2007. Model evaluation guidelines for systematic quantification of accuracy in watershed simulations. *Transactions of the ASABE*, *50*(3), pp.885-900.

Mugume, S., Gomez, D.E. and Butler, D., 2013. Statistical Downscaling Methods for Climate Change Impact Assessment on Urban Rainfall Extremes for Cities in Tropical Developing Countries–a Review.

Mugume, S.N., 2015. Modelling and Resilience-based Evaluation of Urban Drainage and Flood Management Systems for Future Cities.

Munoz, N.J., 2016. What Is The Economic Feasibility Of Implementing Grey Water Infrastructure At The Citywide Level?.

Murad, A.A., Al Nuaimi, H. and Al Hammadi, M., 2007. Comprehensive assessment of water resources in the United Arab Emirates (UAE). *Water Resources Management*, *21*(9), pp.1449-1463.

Murphy, J.M., Sexton, D.M., Jenkins, G.J., Booth, B.B., Brown, C.C., Clark, R.T., Collins, M., Harris, G.R., Kendon, E.J., Betts, R.A. and Brown, S.J., 2009. UK climate projections science report: climate change projections.

Naiga, R., Penker, M. and Hogl, K., 2012. From supply to demand driven water governance: challenging pathways to Safe Water Access in Rural Uganda.

Neal, J., Schumann, G. and Bates, P., 2012. A subgrid channel model for simulating river hydraulics and floodplain inundation over large and data-sparse areas. *Water Resources Research*, *48*(11).

Nielsen, S.A. and Hansen, E., 1973. Numerical simulation of the rainfall-runoff process on a daily basis. *Hydrology Research*, *4*(3), pp.171-190.

NLJ (2007) *Handbook of Jamaica*. [online] Available at: http://www.my-island-jamaica.com/rivers_in_jamaica.html [June 23 2016].

Norris, R., Dyer, F., Hairsine, P., Kennard, M., Linke, S., Merrin, L., Read, A., Robinson, W., Ryan, C., Wilkinson, S. and Williams, D., 2007. A baseline assessment of water resources for the National Water Initiative. Level 2 assessment. River and wetland health theme. Assessment of river and wetland health: potential comparative indices.

NWC (1993) *Addressing Jamaica's Water Needs Through to the 21st Century*. Kingston.

NWC (2011) *Kingston and St Andrew Water Supply Plans*.

NWC (2016) *National Water Commission*. [online] Available at: http://www.nwcjamaica.com/default.asp [Accessed: 23 February 2016].

OAS (2005) *Organisation of American States*. [online] Available at: http://www.oas.org/dsd/publications/unit/oea59e/ch10.htm [Accessed: 13 December 2015].

Observer, J. (2017) *Reduction of non-revenue water to boost drought resilience*. [online] Available at:

http://www.jamaicaobserver.com/news/Reduction-of-non-revenue-water-to-boost-drought-resilience [Accessed: 20 July 2017].

Office, M. (2017) *England and Wales drought 2010 to 2012*. [online] Available at: http://www.metoffice.gov.uk/climate/uk/interesting/2012-drought [Accessed: 7 June 2017].

Omar, M.M., 2013. Evaluation of actions for better water supply and demand management in Fayoum, Egypt using RIBASIM. *Water Science*, *27*(54), pp.78-90.

Ouda, O.K., 2015. Domestic water demand in Saudi Arabia: assessment of desalinated water as strategic supply source. *Desalination and Water Treatment*, *56*(11), pp.2824-2834.

Oxford Scientific software (2014a) 'A Guide to Aquator: Application'.

Oxford Scientific software (2014b) *A Guide to Aquator: Components*.

Oxford Scientific software (2015) *AQUATOR*. [online] Available at: http://www.oxscisoft.com/index.aspx. [Accessed 15 October 2015]

Palmer, R. N. and Lundberg, K. V (2004) *INTEGRATED WATER RESOURCE PLANNING*. [online] Available at: http://www.isws.illinois.edu/iswsdocs/wsp/iwrp_palmer_lundberg.pdf [Accessed: 27 November 2017].

Parmesan, C. and Yohe, G., 2003. A globally coherent fingerprint of climate change impacts across natural systems. *Nature*, *421*(6918), pp.37-42.

Perry, D. and Praskievicz, S., 2017. A new era of big infrastructure?(Re) developing water storage in the US West in the context of climate change and environmental regulation. *Water Alternatives*, *10*(2), p.437.

Phipps, C. (2017) *Hurricane Maria: Dominica 'in daze' after storm leaves island cut off from world | World news | The Guardian*, *The Guardian*. [online] Available at: https://www.theguardian.com/world/2017/sep/21/dominica-daze-hurricane-maria-island-caribbean-rescue [Accessed: 16 December 2017].

Pluntke, T., Pavlik, D. and Bernhofer, C., 2014. Reducing uncertainty in hydrological modelling in a data-sparse region. *Environmental earth sciences*, *72*(12), pp.4801-4816.

Prudhomme, C., Haxton, T., Crooks, S., Jackson, C., Barkwith, A., Williamson, J., Kelvin, J., Mackay, J., Wang, L., Young, A. and Watts, G., 2013. Future Flows Hydrology: an ensemble of daily river flow and monthly groundwater levels for use for climate change impact assessment across Great Britain. *Earth System Science Data*, *5*(1), pp.101-107.

Republic of Egypt (2005) *Arab Republic of Egypt Ministry of Water Resources and Irrigation*. [online] Available at: http://extwprlegs1.fao.org/docs/pdf/egy147082.pdf [Accessed 13 December 2017].

Roibás, D., García-Valiñas, M.Á. and Wall, A., 2007. Measuring welfare losses from interruption and pricing as responses to water shortages: an application to the case of Seville. *Environmental and resource economics*, *38*(2), pp.231-243.

Ryan, A.M., Spash, C.L. and Measham, T.G., 2009. Socio-economic and psychological predictors of domestic greywater and rainwater collection: Evidence from Australia. *Journal of Hydrology*, *379*(1), pp.164-171.

Saaty, T.L., 2008. Decision making with the analytic hierarchy process. *International journal of services sciences*, *1*(1), pp.83-98.

Sadashiva, Murthy; Parameshwara , Murthy; Kavya, S. (2016) ‘Greywater Treatment & Reuse: A Technological Review’, *GLOBAL JOURNAL FOR RESEARCH ANALYSIS*, 5(3), pp. 408–410.

Sanderson, M.G., Wiltshire, A.J. and Betts, R.A., 2012. Projected changes in water availability in the United Kingdom. *Water Resources Research*, *48*(8).

Saving Water Partnership (2017) *Rebates Residential*. [online] Available at: http://www.savingwater.org/rebates/ [Accessed: 15 October 2017].

Scottish Water (2005) *Scottish Water Drought Worksheet*. [online] Available at:
http://www.scottishwater.co.uk/assets/education/files/p67/p67worksheets

_drought.pdf. [Accessed August 27 2016]

Sechi, G.M. and Sulis, A., 2009. Water system management through a mixed optimization-simulation approach. *Journal of Water Resources Planning and Management*, *135*(3), pp.160-170.

Severn Trent (2014) *Appendix A - How much water do we have available?* [online] Available at: https://www.severntrent.com/content/dam/stw/ST_Corporate/About_us/Docs/Appendix-A-How-much-water-do-we-have-available.pdf [Accessed: 13 October 2017].

Sharif, H.O., Al-Zahrani, M. and Hassan, A.E., 2017. Physically, Fully-Distributed Hydrologic Simulations Driven by GPM Satellite Rainfall over an Urbanizing Arid Catchment in Saudi Arabia. *Water*, *9*(3), p.163.

Shatat, M. and Riffat, S.B., 2012. Water desalination technologies utilizing conventional and renewable energy sources. *International Journal of Low-Carbon Technologies*, *9*(1), pp.1-19.

Shih, J.S. and ReVelle, C., 1994. Water-supply operations during drought: Continuous hedging rule. *Journal of Water Resources Planning and Management*, *120*(5), pp.613-629.

Shirali, G.H.A., Mohammadfam, I., Motamedzade, M., Ebrahimipour, V. and Moghimbeigi, A., 2012. Assessing resilience engineering based on safety culture and managerial factors. *Process Safety Progress*, *31*(1), pp.17-18.

Simonovic, S.P., Fahmy, H. and El-Shorbagy, A., 1997. The use of object-oriented modeling for water resources planning in Egypt. *Water Resources Management*, *11*(4), pp.243-261.

Sønderlund, A.L., Smith, J.R., Hutton, C. and Kapelan, Z., 2014. Using smart meters for household water consumption feedback: knowns and unknowns. *Procedia Engineering*, *89*, pp.990-997.

Southern.Water (2014) *Water Resources Management Plan 2015–40*. Worthing.

SSR (2016) *Developing Countries*, *Society for the Study of Reproduction*.

[online] Available at: http://www.ssr.org/DevelopingCountries [Accessed: 16 December 2017].

Statinja (2011) *2011 Cencus of Population & Housing*. [online] Available at: http://statinja.gov.jm/Census/Census2011/Census 2011 data from website.pdf. [Accessed February 27 2016].

Sterbenz, J.P., Cetinkaya, E.K., Hameed, M.A., Jabbar, A. and Rohrer, J.P., 2011, January. Modelling and analysis of network resilience. In *Communication Systems and Networks (COMSNETS), 2011 Third International Conference on* (pp. 1-10). IEEE.

Sulis, A. and Sechi, G.M., 2013. Comparison of generic simulation models for water resource systems. *Environmental modelling & software*, *40*, pp.214-225.

Swayne, D.A., Yang, W., Rizzoli, A. and Filatova, T., Intercomparison of Generic Simulation Models for Water Resource Systems.

SWW (2017) *Save Water Save Money*. [online] Available at: https://www.savewatersavemoney.co.uk/southwest/free-water-saving-products [Accessed: 7 September 2017].

Syme, G.J., Nancarrow, B.E. and Seligman, C., 2000. The evaluation of information campaigns to promote voluntary household water conservation. *Evaluation Review*, *24*(6), pp.539-578.

Talebpour, M.R., Sahin, O., Siems, R., Stewart, R.A. and Hopewell, M., 2014. Evaluating rain tank pump performance at a micro-component level. *Alternative Water Supply Systems*, p.25.

Tarr, R. . S. (1900) 'Physiographic Notes', *Journal of the American Geographical Society of New York*, 32(1), pp. 52–54. Available at: http://www.jstor.org/stable/197283?seq=1#page_scan_tab_contents (Accessed: 23 February 2016).

Thames Water (2010) *Revised Water Resources Management Plan*. [online] Available at: https://corporate.thameswater.co.uk/-/media/Site-Content/Thames-Water/Corporate/AboutUs/Our-strategies-and-plans/Water-resources/Our-current-plan-WRMP14/WRMP14_Section_2.ashx?la=en [Accessed: 17 October

2017].

Thames Water (2014) *Our current plan (WRMP14) and annual performance*. Reading. [online] Available at: https://corporate.thameswater.co.uk/About-us/Our-strategies-and-plans/Water-resources/Our-current-plan-WRMP14 [Accessed: July 7 2017].

Thames Water (2016a) *Thames Water Utilities Draft Drought Plan*. [online] Available at: https://corporate.thameswater.co.uk/About-us/Our-strategies-and-plans/Our-drought-plan/-/media/5C8CE1DCEA0C4DA2AFC5DFEE9790CD9F.ashx?bc=White&db=web&la=en&thn=1&ts=ecdf36ab-8c4f-48e2-ae68-3639501cb459.pdf [Accessed: 15 October 2017].

Thames Water (2016b) *THAMES WATER UTILITIES LTD SEPTEMBER 2016 DRAFT DROUGHT PLAN APPENDIX I LONDON WRZ – WARMS2 MODELLING OF WATER RESOURCE AVAILABILITY Appendix I London WRZ – Modelling of Water Resource Availability Using the Water Resources Management System2 (WARMS2) THA*. [online] Available at: https://corporate.thameswater.co.uk/About-us/Our-strategies-and-plans/Our-drought-plan/-/media/5C8CE1DCEA0C4DA2AFC5DFEE9790CD9F.ashx?bc=White&db=web&la=en&thn=1&ts=ecdf36ab-8c4f-48e2-ae68-3639501cb459.pdf [Accessed: 19 October 2017].

The World Bank (2017a) *Climate Change Knowledge Portal Jamaica*. [online] Available at: http://sdwebx.worldbank.org/climateportal/index.cfm?page=country_historical_climate&ThisCCode=BRB [Accessed: 15 October 2017].

The World Bank (2017b) *Climate Change Knowledge Portal United Kingdom*. [online] Available at: http://sdwebx.worldbank.org/climateportal/index.cfm?page=country_historical_climate&ThisCCode=GBR [Accessed: 28 October 2017].

Tindigarukayo, J. (2014) 'The Impact of Rural-Urban Influx on Jamaican Society', *International Journal of Humanities and Social Science*, 4(9), p. 6.

Tsai, Y., Cohen, S. and Vogel, R.M., 2011. The impacts of water conservation strategies on water use: four case studies. *JAWRA Journal of the American Water Resources Association*, *47*(4), pp.687-701.

Turner, S.W.D., Blackwell, R.J., Smith, M.A. and Jeffrey, P.J., 2016. Risk-based water resources planning in England and Wales: challenges in execution and implementation. *Urban Water Journal*, *13*(2), pp.182-197.

Turner, S.W.D., Blackwell, R.J., Smith, M.A. and Jeffrey, P.J., 2016. Risk-based water resources planning in England and Wales: challenges in execution and implementation. *Urban Water Journal*, *13*(2), pp.182-197.

U.S. Army Corps of Engineers; (1998) *Water Supply Handbook*. Alexandria, VA: Institute for Water Resources Water Resources Support Center U.S. Army Corps of Engineers.

UK Government (2010) *The Water Use (Temporary Bans) Order 2010*. [online] Available at: http://www.legislation.gov.uk/uksi/2010/2231/made [Accessed: 15 October 2017].

Home, D., 2004. Catalyzing Change: A handbook for developing integrated water resources management (IWRM) and Water Efficiency strategies. *GWPT Committee (Ed.)*, p.56.

United Nations (1993) *United Nations Sustainable Development. United Nations Conference on Environment & Development*. Rio de Janeiro. [online] Available at: https://sustainabledevelopment.un.org/content/documents/Agenda21.pdf (Accessed: 10 October 2017). [Accessed: September 10 2017]

UNWater (2017) *Wasterwater The Untapped Resource. The United Nations World Water Development Report*. Paris. [online] Available at: http://unesdoc.unesco.org/images/0024/002471/247153e.pdf. [October 10 2017]

UWA (1990) *Water Resources Development Master Plan*. Kingston.

UWI (2012) *The State of Jamaica Climate 2012. Information for resileince*.

Vase, J., Jordan, P., Beecham, R., Frost, A. and Summerell, G., 2011. Guidelines for Rainfall-Runoff Modelling: Towards Best Practice Model Application.

Visessri, S. and McIntyre, N., 2016. Regionalisation of hydrological responses under land-use change and variable data quality. *Hydrological Sciences Journal*, *61*(2), pp.302-320.

Vogler, D., Macey, S. and Sigouin, A., 2017. Stakeholder Analysis in Environmental and Conservation Planning.

Vudhivanich, V. and Rittima, A., 2003. Development of Probability Based Rule Curves for a Reservoir. *Kasetsart Journal-Natural Science*, *242*, pp.234-242.

Vugrin, E.D., Warren, D.E. and Ehlen, M.A., 2011. A resilience assessment framework for infrastructure and economic systems: Quantitative and qualitative resilience analysis of petrochemical supply chains to a hurricane. *Process Safety Progress*, *30*(3), pp.280-290.

Walsh, C.L., Blenkinsop, S., Fowler, H.J., Burton, A., Dawson, R.J., Glenis, V., Manninga, L.J. and Kilsby, C.G., 2015. Adaptation of water resource systems to an uncertain future. *Hydrology & Earth System Sciences Discussions*, *12*(9).

Water, U. (2012) *Managing Water Under Uncertainty and Risk*. [online] Available at: http://unesdoc.unesco.org/images/0021/002156/215644e.pdf. [Accessed 23 October 2016].

WaterAid (2016) *Water: At What Cost? The State of the World's Water 2016*.

Water UK (2012) *National water grid*. [online] Available at: https://www.water.org.uk/policy/resilience/national-water-grid [Accessed: 18 October 2017].

Waterwise (2011a) *Evidence Base for Large-scale Water Efficiency Phase II Final Report*. London.

Waterwise (2011b) *Reducing Water Wastage in the UK*. [online] Available

at:
http://wwarchive.brix.fatbeehive.com/reducing_water_wastage_in_the_uk/the_facts/the_facts_about_saving_water.html [Accessed: 13 December 2015].

Waterwise (2012) *Waterwise - The Facts*. [online] Available at: http://www.waterwise.org.uk/data/resources/25/Water_factsheet_2012.pdf [Accessed: 1 September 2017].

Waterwise (2017) *National Water Grid*. [online] Available at: http://www.waterwise.org.uk/pages/national-water-grid.html [Accessed: 18 October 2017].

WCD (2000) 'DA MS A ND D E V E LOPM E NT'. [online] Available at: https://www.internationalrivers.org/sites/default/files/attached-files/world_commission_on_dams_final_report.pdf [Accessed: 14 October 2017].

WEAP (2017) *Water Evaluation And Planning*. [online] Available at: http://www.weap21.org/ [Accessed: 12 October 2017].

Welsh, L.W. and Endter-Wada, J., 2017. Piping water from rural counties to fuel growth in Las Vegas, Nevada: Water transfer risks in the arid USA West. *Water Alternatives*, *10*(2), p.420.

WRA (2012) *Jamaica Water Resources Authority Water Information System*. [online] Available at: http://webmapjam.dyndns.pro/webmap/app/db/index.php. [Accessed January 24 2016].

WRA (2016) *Water Resources Authority*. [online] Available at: http://www.wra-ja.org/ [Accessed: 23 February 2016].

WRI (2013) *World's 36 Most Water-Stressed Countries*. [online] Available at: http://www.wri.org/blog/2013/12/world's-36-most-water-stressed-countries [Accessed: 10 September 2017].

Wurbs, R.A., 1994. *Computer models for water resources planning and management* (No. IWR-94-NDS-7). ARMY ENGINEER INST FOR WATER RESOURCES FORT BELVOIR VA.

Wurbs, R.A., 1997. *Computer models for water-resources planning and management: National study of water management during drought*. Diane Publishing.

Yao, H. and Georgakakos, A., 2001. Assessment of Folsom Lake response to historical and potential future climate scenarios: 2. Reservoir management. *Journal of Hydrology*, *249*(1), pp.176-196.

Yates, D., Sieber, J., Purkey, D. and Huber-Lee, A., 2005. WEAP21—A demand-, priority-, and preference-driven water planning model: part 1: model characteristics. *Water International*, *30*(4), pp.487-500.

Yu, Z.L., DeShazo, J.R., Stenstrom, M.K. and Cohen, Y., 2015. Cost-Benefit Analysis of Onsite Residential Graywater Recycling: A Case Study on the City of Los Angeles. *Journal-American Water Works Association*, *107*(9), pp.E436-E444.

Zhao, T., Cai, X. and Yang, D., 2011. Effect of streamflow forecast uncertainty on real-time reservoir operation. *Advances in water resources*, *34*(4), pp.495-504.

Ziolkowska, J.R. and Ziolkowski, B., 2016. Effectiveness of water Management in Europe in the 21st century. *Water resources management*, *30*(7), pp.2261-2274.

List of Appendices

Appendix A: Anyplace Data-rich Modelling Results

Table A.1 Summary of the Social resilience and the Environmental resilience performance results for the Goal-Seeking Hedge strategies assessed for the high demand and the 11 Future Flow Hydrology[2].

	Hedging Policy					
	Social Resilience			**Environmental Resilience**		
FFH ID	**Consecutive Failure Days**	**Magnitude (ML)**	**Duration change**	**Consecutive Failure Days**	**Magnitude (ML)**	**Duration change**
Q0	611	5485.72	542	20	90.92	-35
Q16	585	2357.54	539	11	50.01	-7
Q14	*978*	*8975.09*	933	13	59.1	-14
Q13	943	10011.5	879	25	113.65	-12
Q11	554	5112.26	482	17	73.5	-52
Q10	950	9342.18	869	42	190.932	-38
Q3	960	9713.34	888	21	95.47	-51
Q9	571	8698.32	477	*57*	*255.2*	-36
Q4	1659	19006.73	1573	53	240.94	-23
Q6	906	6804.62	862	14	63.64	-15
Q8	**173**	**5282.04**	86	**0**	**0**	-60

Table A.2 Summary of the Social resilience and the Environmental resilience performance results for the Temporary Use Bans strategy assessed for the high demand and the 11 Future Flow Hydrology[2].

	Temporary Use Bans					
	Social Resilience			Environmental Resilience		
FFH ID	**Consecutive Failure Days**	**Magnitude (ML)**	**Duration change**	**Consecutive Failure Days**	**Magnitude (ML)**	**Duration change**
Q0	66	2432.11	-3	55	250.03	0
Q16	45	1636.05	-1	**12**	**51.18**	-6
Q14	45	1630.37	0	27	122.74	0
Q13	46	1687.31	-18	32	145.47	-5
Q11	72	2747.66	0	69	313.67	0
Q10	81	3152.74	0	80	363.68	0
Q3	72	2815.49	0	72	327.31	0
Q9	*94*	*3669*	0	*93*	*422.78*	0
Q4	78	2887.85	-8	64	290.94	-12
Q6	**35**	**1328.2**	-9	23	104.56	-6
Q8	86	3204.06	-1	59	268.21	-1

Table A.3 Summary of the Social resilience and the Environmental resilience performance results for the Demand Management strategy assessed for the high demand and the 11 Future Flow Hydrology[2].

	Demand Management					
	Social Resilience			**Environmental Resilience**		
FFH ID	**Consecutive Failure Days**	**Magnitude (ML)**	**Duration change**	**Consecutive Failure Days**	**Magnitude (ML)**	**Duration change**
Q0	65	2487.62	-4	54	245.48	-1
Q16	46	1760.85	0	**12**	**52.73**	-6
Q14	45	1745.19	0	19	82.53	-8
Q13	59	2040.39	-5	28	127.29	-9
Q11	72	2862.51	0	69	313.67	0
Q10	79	3211.82	-2	79	359.13	-1
Q3	70	2845.92	-2	70	318.22	-2
Q9	*94*	*3794.25*	0	*93*	*422.78*	0
Q4	84	3306.22	-2	72	327.31	-4
Q6	**39**	**1509.74**	-5	29	131.83	0
Q8	73	2842.13	-14	54	245.48	-6

Appendix B: Kington & St. Andrew Case Study

Fieldwork Investigation - Introduction

The challenges associated with gathering data for the KSA case study are outlined in section 4.6. In particular, in subsection 4.6.1 several approaches have been highlighted to fill the data gaps associated with the KSA case study, one of which refers to a fieldwork investigation conducted in Kington Jamaica from January 26th, 2016 to the 18th February 2016. This section examines the methods used and the information gathered during the fieldwork investigation.

Fieldwork Investigation – Overview

In this study, the first phase of the methodology involved distributing self-administered questionnaire surveys. These questionnaire surveys illustrated in Figure B.3 consist of questions where the respondents select the appropriate answers. For the fieldwork investigation, a total of two hundred questionnaires were administered. This sample size was chosen due to financial implications (refer to section 4.9) involved with printing a larger sample size. The locations where these were administered are shown in Figure B.1, and these are the University of the West Indies student accommodation (where the researcher was based), Downtown Kingston, Halfway Tree, Three Views, Mountain View and Mona Heights. These locations were identified following informal discussions with local residents.

Figure B.1 Illustration of the locations where the questionnaires surveys were administered in KSA

Downtown Kingston and Halfway Tree are the locations of two major bus terminals in Kingston, and these proved to be the most promising locations. At these locations, the researcher captured the views of residents who live in socially diverse communities across Kingston and St. Andrew. Figure B.2 shows the location of those communities.

Figure B.2 Illustration of the communities where the questionnaire respondents indicated they reside.

According to Fink, (2003), a response rate of 70% is considered adequate. All of the respondents to the questionnaires were eighteen years or older, is most cases, not all of the questions were answered. The main contributing factors to the response rate achieved is underpinned by some respondents preferred to engage in discussions with the researcher, as this allowed them to interject their own impressions which are summarised in Table B.1. Figure B.4 to B.6 shows the percentage distribution of the responses to the questions most answered by the respondents.

The second phase of the methodology involved conducting a building survey of 220 residential properties in the Mona Heights (where the researcher was staying for the duration of the fieldwork) community to observe how many of them have water storage tanks. (Figure B.7 show the most common type of storage used in Mona Heights). The building survey was not extended to other communities of Kingston as the researcher felt it was unsafe to conduct this survey in other communities during the period leading to the country's general elections. Data gathered from this survey was utilised as insight for considering the potential for seasonal rainwater harvesting as a possible intervention for the Kingston and St. Andrew case study.

The results of the building survey are summarised in Table B.2 show that approximately 42% of the residential properties have at least one water tank. Interestingly, all of these tanks observed were connected to the drinking water supply network as residents utilise these tanks to store potable water. This lack of rainwater harvesting in Kingston is underpinned by the statistics found in the UWA, (1990) water resources masterplan which shows that only 0.1% of the total rainfall in Kingston is harvested.

Water Use Questionnaire

Gender: Male / Female

Age: (12 – 17); (18 -24); (25 – 34); (35- 44); (45 – 54); (55 – 64); (65 – 74); (75 or older)

1) **How often do you experience restrictions regarding water supply?**

   Always ☐ Very Often ☐ Sometimes ☐ Rarely ☐
   Never ☐
   b) Please comment on the frequency of these restrictions (e.g. Once per week/month) ______________________________

2) **How long do these restrictions last?**

   1 to 2 hours ☐ 2 to 4 hours ☐ 4 to 6 hours ☐
   More than 6 hours ☐
   b) If more than 6 hours, please give an indication of the length of time:
   ____

3) **Indicate which of the following do you think contributes to water restrictions in Jamaica?**

   Climate change ☐ Poor water resources management ☐
   Lack of connections ☐
   Other ______________________________

4) **Indicate which of the following practices you use potable water for:**

   Cooking ☐ House cleaning ☐ Laundry ☐
   Personal use (shower, bath, washroom) ☐ Watering garden ☐
   Washing car ☐

5) **Please state any practices for which you use non-potable water.**

6) **Please indicate which of the following coping measures you are utilising in the current drought?**

   Bottled water ☐ Water reuse ☐ Water Supply from water trucks ☐
   Storage tanks/barrels ☐ Reduce water usage ☐ None ☐
   Others ______________________________

7) **Which of the following intervention strategies you think can potentially provide a continuity of water supply.**

   Rainwater harvesting ☐ Desalinated water ☐ Wastewater reuse ☐

UNIVERSITY OF EXETER

Artificial recharge ☐

Infrastructure upgrades (e.g *wells, water treatment plants, water mains, distribution mains, water metering*) ☐

b) Please indicate which of the above intervention strategies you are NOT knowledgeable about.

Rainwater harvesting ☐ Desalinated water ☐ Wastewater reuse ☐

Artificial recharge ☐

Infrastructure upgrades (e.g *wells, water treatment plants, water mains, distribution mains, water metering*) ☐

8) Do you know what water saving devices are?

Yes ☐ No ☐

b) (If NO, go to Question 9)

If yes, which of the following water saving devices do you have installed in your home?

Water efficient toilets (Dual flush) ☐ Low-flow shower heads ☐

Water flow valves (Flow restrictors) ☐ None ☐

Others ______________________________

c) If you selected "None", would you consider installing water saving devices?

Yes ☐ No ☐ Maybe ☐

Please explain your choice ______________________________

9) Are you willing to pay more for an improved water service?

Yes ☐ No ☐ Maybe ☐

Figure B.3 Residential Water Use Questionnaire

Table B.1 Notes from informal interviews with residents of Kingston and St. Andrew held during the period January 26th 2016 to February 18th 2016[13].

In response to the NWC water restrictions: "*By the time I arrive home from work, the water has already been turned off*"
In response to the NWC water restrictions: "*The restriction were imposed to reduce water theft in poorer areas, however the uptown areas always have water*"
In response to the NWC water restrictions: "*I am paying for a service I do not receive, because when I arrive how from work the water is off*."
"*The NWC needs to involve the community in the decision-making process. Investigate what strategies can help resident in order to make them feel comfortable*"
"*Rainwater harvesting is a good idea, however lack of promotion campaigns by government; Wastewater reuse is good for Agriculture*".
"*The only strategy Kingston needs is to construct another reservoir*"
"*Do not promote rainwater harvesting, build another reservoir*"
"*Water saving devices sounds like a good idea, but I am not sure it will work in Jamaica because what are we going to do with the old devices and who will help us pay for these new devices*"
In response to the question 'willingness to pay more for an improved water service': "*Yes, only if I can find employment*"
In response to the question 'willingness to pay more for an improved water service': "*Yes, as long as I get an uninterrupted service*"
"*NWC and WRA need to include the communities in the decision-making process; some of us University of the West Indies staff have to bring our own water to work, because some of the buildings here do not have running water*"

[13] As mentioned in section 4.7.2, the dry season in the Caribbean extends from December to April. The site visit was conducted during the dry period, and coincidently, at the time of the visit to Jamaica, the country was experiencing a drought. Perhaps, if the survey was conducted during the wetter months which are June to November, the responses recorded would not have reflected the the nature of the water situation in Jamaica. That is assuming that during the wetter months, the reservoirs in Kingston maintain higher water levels.

"*The University of the West Indies needs to invest in a WTW facility and storage for future droughts*"

"*KSA needs about major storage facility. The parish of St. Thomas has an abundance of streams. The NWC needs to conserve as well because they take about 3 or 4 days to respond to burst mains*"

"*The Mona reservoir was initially constructed to serve the Mona community only, now it serves also half of Kingston*"

"*Rainwater can be used to do laundry and water gardens*"

"*How often does it rain in Kingston to fill a storage tank?"*

"*Wastewater is a good idea but how safe is it to use for crop irrigation?"*

"*Desalinated water is a good idea but its damages the ecosystem and it sounds expensive*"

"*Convey water from less populated parishes where it is available in abundance to Kingston*"

Table B.2 Summary of the household properties in the Mona height community in Kingston that have water, January 2016.

Tanks	No. of households	Percentage of total No. of households (%)
0	93	42.27
1	92	41.82
2	28	12.73
3	1	0.45
4	4	1.82
5	0	0.00
6	1	0.45
7	1	0.45
Total	**220**	**100**

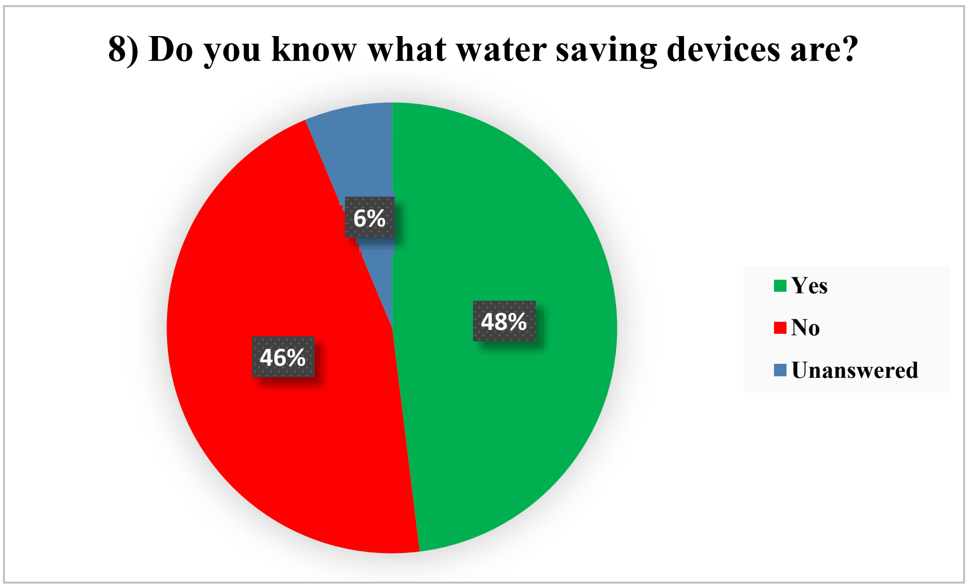


Figure B.4 Pie Chart of close-end question 8 from the Residential Water Use Questionnaire (n = 200 respondents)

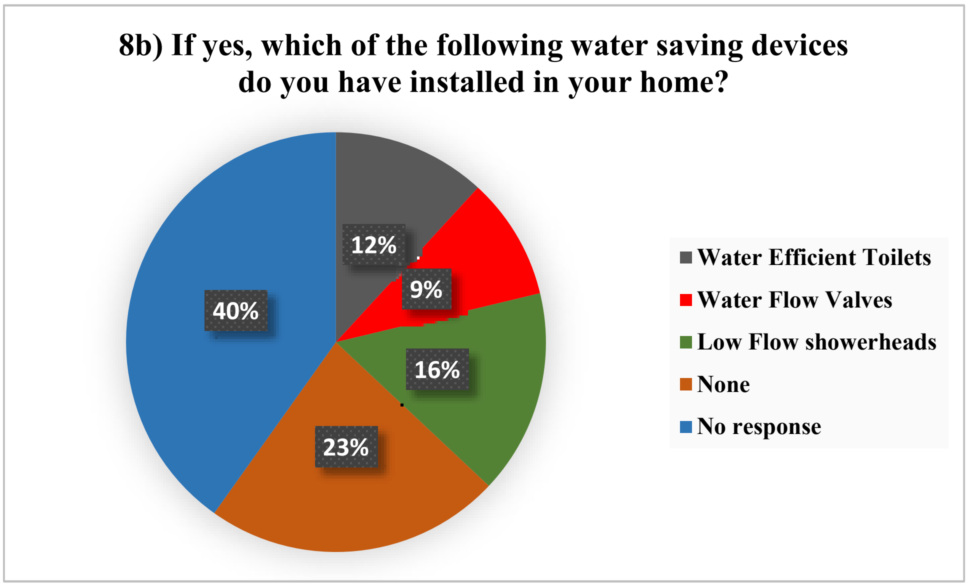


Figure B.5 Pie Chart of close-end question 8b from the Residential Water Use Questionnaire (n = 200 respondents)

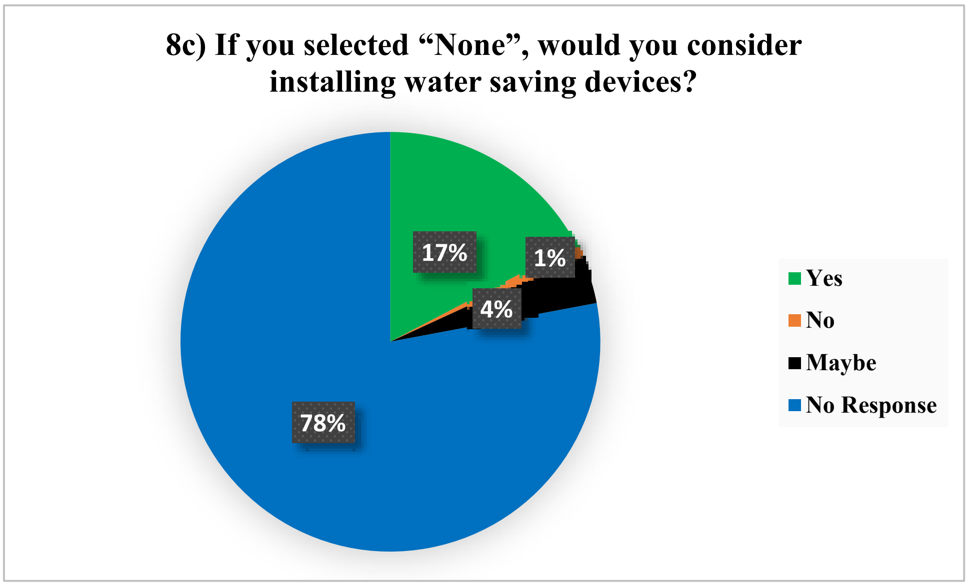


Figure B.6 Pie Chart of close-end question 8c from the Residential Water Use Questionnaire (n = 200 respondents)

Figure B.7 Types of water storage tanks found in Mona Heights Kingston, Jamaica

Table B.3 Summary of calculations demonstrating the production costs for desalinated water based on the unit cost to produce 1 ML of desalinated water in Jamaica, 1990 (UWA, 1990).

Site A - East			
Level of Uncertainty at the Yallahs Pipeline	**Production cost for 1 ML of Desalinated**	**Total Desalinated Water Utilised (ML)**	**Total cost of Desalinated Water (£ - thousand)**
-20%	0.34	1888	641.92
-15%		1888	641.92
-10%		1888	641.92
-5%		1888	641.92
0%		1888	641.92
5%		1888	641.92
10%		1888	641.92
15%		1888	641.92
20%		1888	641.92

Site A - East + Site B			
Level of Uncertainty at the Yallahs Pipeline	**Production cost for 1 ML of Desalinated**	**Total Desalinated Water Utilised (ML)**	**Total Production cost of Desalinated Water (£ - thousand)**
-20%	0.78	7044	5,494.32
-15%		6968	5,435.04
-10%		6873	5,360.94
-5%		6773	5,282.94
0%		6656	5,191.68
5%		6525	5,089.50
10%		6390	4,984.20
15%		6251	4,875.78
20%		6113	4,768.14