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Evaluating non-traditional water supply options through process-based modeling and risk assessment

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Evaluating non-traditional water supply options through process-based modeling and risk assessment

DISSERTATION

submitted in partial satisfaction of the requirements
for the degree of

DOCTOR OF PHILOSOPHY

in Civil & Environmental Engineering

by

Hunter Quon

Dissertation Committee:
Professor Sunny Jiang, Chair
Assistant Professor Adeyemi Adeleye
Professor Diego Rosso

2023

DEDICATION

To the family,
friends,
concerts,
iced coffees,
dog park trips,
and lofi beats to study to
that kept me motivated to keep going.

Trust your heart
Let fate decide
To guide these lives we see.

-Phil Collins

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ABSTRACT OF THE DISSERTATION

Evaluating non-traditional water supply options through process-based modeling and risk assessment

By

Hunter Quon

Doctor of Philosophy in Civil & Environmental Engineering

University of California, Irvine, 2023

Professor Sunny Jiang, Chair

The effects of climate change, population growth, and future hydrologic uncertainties necessitate increased water conservation, new water resources, and a shift towards sustainable urban water supply portfolios. Diversifying water portfolios with non-traditional water sources can play a key role. Rooftop harvested rainwater, stormwater, recycled wastewater and greywater, desalinated seawater and brackish water, and atmospheric and condensate harvesting are all currently utilized and rapidly emerging non-traditional water sources. Given the unique challenges and various pros and cons of each individual source, quantitative models and process analyses highlight the strength of comparative assessments across scenarios and water supply options to ultimately aid in decision making efforts. Larger frameworks including technoeconomic assessment (TEA) and quantitative microbial risk assessment (QMRA) paired with new, scenario specific efforts can quantify priority metrics to highlight areas of both improvement and concern. Through this research, three different non-traditional water sources were approached individually to demonstrate three aims: 1) assessment of processes, layouts, and local factors for specific facilities to identify opportunities for capital cost reduction and conditions required to adopt seawater reverse osmosis desalination as a water source; 2) locally conducted surveys and water sample collection used to both quantify health risks and local perception of water use and quality for harvested rainwater after major tropical

hurricanes; and 3) implementation of a dose-response model accounting for antibiotic resistance to produce a new outcome of quantified health risks to augment our understanding of risk-based regulations for non-potable reclaimed wastewater. The outcomes of this research will provide better understanding of cost origins, health risks, and where critical efforts are needed to improve design and use of these water sources.

Chapter 1

Introduction

1.1 Background

As population growth continues around the world, so, too does the need for potable water sources and infrastructure that can ensure its availability. Climate change, which includes extreme weather events and natural disasters, further exacerbates water stress due to its impacts to water quantity, quality, and local shortages (Schwabe et al. (2020); Van Aalst (2006)). Future climatic and hydrologic uncertainties that continue to widen the gap between water resource supply and demand have motivated water management decision-making towards increased conservation, technological advancements around water treatment, and a shift towards diversifying urban water portfolios with non-traditional, decentralized, or more "sustainable" sources (Brown et al. (2009)). In addition, the COVID-19 pandemic has spurred research on and increased emphasis in proper hygiene and water quality monitoring (Bogler et al. (2020); WHO (2020)). The water supply and treatment paradigm must handle and prepare for current and future impacts of climate, populations, and disease.

To date, it is most common in urban areas and developed countries around the world to depend on centralized drinking water systems that draw from traditional surface and groundwater sources. These systems provide clean water for consumers and abide by standardized environmental waste disposal requirements. Therefore, improvements to the system with regards to population growth and climate change are more difficult and tend to focus on infrastructure retrofitting for increased flows and to support larger populations. It is questionable whether this remains economically and environmentally feasible in the years to come, especially in water-stressed areas (Arora et al. (2015)). Conversely, onsite, and decentralized water systems remain the standard in many rural regions around the world, for water collection, storage, treatment, and use. For example, rooftop harvested rainwater (RHRW), cisterns, and water recycling are well-established practices and methods in rural areas and developing countries, globally. However, these non-traditional water supplies vary in quality, health risks, maintenance, and may be more impacted by weather events and disease agents than centralized infrastructure. Quantity, quality, and accessibility of water resources and treatment are complex challenges and a better understanding across non-traditional water resources and treatment designs is needed to ensure widespread availability and safety of the water supplies.

Non-traditional water sources come with unique implementation challenges and benefits of use. Future water supply security requires these challenges and benefits to be understood and researched for sustainable water management. Sustainable water management is the use of water in a way that provides adequate quality and quantity and addresses unique social and ecological needs while ensuring that these needs and standards will also be met in the future. Specific challenges depend on regional and socioeconomic factors, such as cost, land use, and differing perspectives on water governance and technology adoption. Water governance has been defined as the range of political, social, economic, and administrative systems in place to develop, manage, and deliver water resources at different levels of society (Rogers and Hall (2003)). The water industry in the U.S. is highly fragmented, with nearly 150,000 entities registered with the EPA Safe Drinking Water Information System (SDWIS) as drinking water providers (Environmental Protection Agency (2021)). Therefore, the decision to adopt new water sources or invest in technology is region specific, and dependent on local governance and conditions. As environmental and climatic factors exacerbate water issues, the effect is compounded on areas and communities that are at higher risk, such as semi-arid regions or areas of varying urbanization that lack the infrastructure to provide reliable, clean water. Therefore, there is no "one-size-fits-all" technology or approach to sustainable water management. Planning and design beyond the current systematic approaches and policies is necessary to begin to account for climate change, population growth, and water quantity and quality concerns across all groups and locations.

Quantitative models, tools, and frameworks are useful in planning and design by allowing researchers and stakeholders to pinpoint and highlight areas that deserve more attention. There are many existing and emerging quantitative models and frameworks that serve as tools for decision-making around water resources, as well as many areas where models have yet to be applied. Some well-known examples are technoeconomic assessment (TEA) and quantitative microbial risk assessment (QMRA). With these frameworks, priority metrics such as cost, energy, and health risk can be quantified. In addition, models within the frameworks can be utilized or created to pinpoint critical research areas to improve the design, regulation, and implementation of water source technologies. One can identify challenge areas of a specific non-traditional water source, apply and develop frameworks or models to a process, and quantify metrics for comparison against established benchmarks or measurements. Drawing such comparisons can guide policymakers and researchers towards better implementation strategies, intervention, and uncertainties as future research areas and data collection.

1.2 Objectives

My research interests have been primarily focused on non-traditional water sources and finding ways to apply modelling approaches within larger frameworks. Through this, results can expand and readjust our understanding of critical areas of cost and risk assessment, to suggest future data collection efforts and areas of improvement. Specific locations and case studies, large comparative assessments, and accounting for more detailed variables can open new areas of improvement and inquiry, to improve on past modelling techniques, monitoring efforts, and design decision making. I aim to investigate this as it applies to current non-traditional water sources through the following three approaches and objectives for three different water sources: 1) assessment of processes, layouts, and local factors for specific facilities can identify opportunities for capital cost reduction and conditions required to adopt seawater reverse osmosis desalination as a water source; 2) locally conducted surveys and water sample collection can be used to both quantify health risks and local perception of water use and quality for harvested rainwater after major tropical hurricanes; 3) a dose-response model accounting for antibiotic resistance can produce a new outcome of quantified health risks to augment our understanding of risk-based regulations for non-potable reclaimed wastewater.

The demands on freshwater continue to rise and are challenged by population growth and climate change. Coastal areas have begun to consider and turn to seawater desalination as a non-traditional source of drinking water, as it can be sourced from a limitless supply of seawater. However, despite significant advancements and a two-fold energy reduction in the state-of-the-art reverse osmosis process for treatment (Voutchkov (2018)), it continues to be a more expensive option of sourcing water. Price varies around the world, and the aim is to understand the price discrepancies in terms of overall design. In addition, quantification and subsequent comparison of metrics beyond cost such as technology reliability and robustness, especially in drought periods, is needed. Drawing such comparisons can demonstrate the role that seawater reverse osmosis can play in future water portfolios and aid in decision making efforts.

Rooftop harvested rainwater is another source of water that is an alternative to traditional surface and groundwater sources that plays a major role in many areas around the world. This water source is more variable in its water quality and therefore may pose unique health risks when used for daily domestic purposes. These health risks are of concern and less known after natural disasters, which may impact the surface water quality and/or damage the cisterns where the rainwater is stored for household use. In the tropical Virgin Islands, rainfall is plentiful, and all residential buildings have constructed rainwater cisterns, but back-to-back hurricanes in 2017 left many without power or access to reliably clean water. Differences in how people use their water and perceive its quality can vary socioeconomically. The concerns and uncertainties around possible risk and water use in such a disastrous time create a unique case study for investigation and quantification to aid in future planning and safety measures.

Treated and reclaimed wastewater has seen rapid growth as a non-traditional water source that can be utilized to more sustainably flush toilets and urinals, irrigate parks, golf courses, crops, and for other uses such as industrial cooling and ornamental fountains or water features (Chen et al. (2013)). It can alleviate water stresses in municipal and agricultural areas alike and diversify water supply portfolios, especially in urban areas. As a non-potable water source, it is regulated less stringently than drinking water, and there are concerns for a wide range of pathogens and impacts on human health upon exposure. Wastewater treatment plants have been identified as a hotspot for enriching antibiotic resistance and the transmission of ARB and antibiotic resistant genes (ARG) into the environment (Rizzo et al. (2013)). There is concern that reclaimed wastewater could facilitate the spreading of ARB and ARG and pose a threat to human health due to exposures with any of the applications, though there is uncertainty around their prevalence and the associated health risk impacts. The application of the Quantitative Microbial Risk Assessment (QMRA) framework in various scenarios of sustainable water uses has been documented (Hajare et al. (2021b)). Through application of a novel dose-response model based on simple death kinetics (Chandrasekaran and Jiang (2019)), an assessment of ARB associated health risks to multiple pathogens and exposure scenarios can shed some light on what is needed to better monitor and regulate pathogens in wastewater treatment, reuse, and reuse applications as well as in antibiotic resistance. The risk outcomes will be assessed beyond just the probability of risk including whether the risk is likely to be treatable or untreatable by an antibiotic and suggest areas of improvement for better understanding or monitoring of these pathogens/risks.

Chapter 2

Review of non-traditional water sources and approaches for assessment and decision-making efforts

2.1 Introduction

Identification of knowledge gaps on both the technologies and the socioeconomic side of quality and quantity of non-traditional water sources provides the motivation for this review and analysis, and is the first step to ensuring a more adequate and equitable water future. This chapter explores the status and trends around non-traditional water sources, and reviews models for prioritizing, predicting, and quantifying metrics of concern. First, a summary of water quantity and quality metrics is introduced. Next, a review of current and emerging non-traditional water sources is tabulated and described, including pros and cons and their respective metrics of interest. In parallel, the most demonstrated and established models for each non-traditional water source are explored and described, further identifying areas of priority application. Finally, a summary of key areas of future research is recommended. The analysis suggests that understanding the challenges of specific scenarios and water supply or management technologies is the crucial first step in establishing a model or framework approach to provide a strategy for improvement going forward. The multifaceted nature of decision-making for water management makes it important to compare, contrast, and weigh options and technologies for a sustainable water future. Therefore, this review is unique in that it defines and analyzes both a list of the major non-traditional water sources available and the quantitative methods for enumerating quantity and quality metrics to compare across sources. Such approaches and methods provide a toolbox for decision-makers, stakeholders, and researchers to better measure and predict trends and applications for more diverse water supply portfolios. This chapter serves to review non-traditional water sources, their challenges and areas of focused research, and unique approaches and assessments through established frameworks and quantitative modelling.

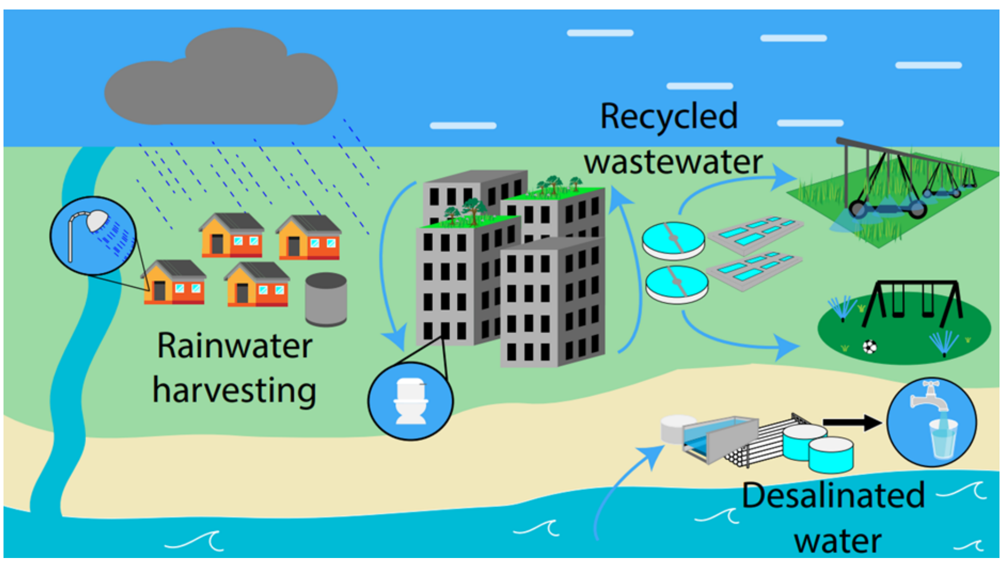

Figure 2.1: Illustration of some of the most common non-traditional water sources.

2.1.1 Non-traditional water sources

Non-traditional, or alternative, water sources are defined as sustainable methods of providing water from sources besides fresh surface water or groundwater that reduce or offset the demand for freshwater (Giammar et al. (2021)). Non-traditional water could mean onsite treatment and storage of rainwater or reclaimed greywater, or larger-scale treatment of municipal wastewater, seawater or brackish water to supplement the existing water supplies. Research and investment into non-traditional water sources have been underway especially in urban centers of developed countries. A list of the non-traditional water sources focused on in this paper and their definitions is provided in Table 2.1.

Table 2.1: A brief summary of non-traditional sources of water, their uses, challenges, and primary metrics of concern for implementation.

Water source	Definition	Examples of use	Challenges	Advantages	Priority metrics	Reference
Harvested rainwater from rooftops	Rainwater collected from the roof surface of a building	Non-potable: showering, toilet flushing, irrigation Potable: after treatment and disinfection	Variable water quality, health risks from in-premise plumbing, lack of maintenance and monitoring, region specific rainfall volume	Reduces runoff and pollution, removes the cost of water transport, better quality than wastewater and stormwater	Quality and quantity: possible risk of pathogen exposure and biofilm growth in rain tanks, climate change factors Cost: installation and maintenance of large storage tanks	Meera and Ahammed (2006)
Harvested stormwater	Urban stormwater runoff collected from road surfaces and storm drains	Non-potable: toilet flushing, irrigation, laundry, car washing	Variable water quality, high capital and maintenance cost for stormwater harvesting infrastructure	Reduces surface water pollution, takes advantage of natural treatment processes	Quality and quantity: possible risk of pathogen exposure, climate change factors Cost: capital and maintenance cost of stormwater harvesting infrastructure	Lim et al. (2015)
Reclaimed wastewater	Treated municipal wastewater effluent that is redistributed in separate pipes for non-potable or potable reuse	Non-potable: toilet flushing, irrigation, industrial cooling, wetlands. Potable: augments existing drinking water sources (groundwater or surface) or used directly for drinking water production	Poor water quality that requires multiple treatment steps, often requires a separate piping and distribution system from potable water. Poor public perception due to the "yuck factor"	Localized water resource in urban centers, reduces the need for long-distance water transport	Quality: efficiency of contaminants removal and maintaining disinfectant levels in storage and distribution, possible risk of pathogen and CEC* exposure Cost: capital, maintenance, and energy cost for potable water production, storage and distribution system cost for non-potable reuse	Furumai (2008)
Greywater	Household wastewater collected from sinks, showers, bathtubs, and laundry	Non-potable: localized irrigation and toilet flushing	Highly variable water quality and quantity that require flexible treatment system designs	Localized water supply, reduces the burden on wastewater treatment facilities	Variable quality, acceptable risks, and hazards	Vuppaladadiyam et al. (2019)
Seawater desalination	Freshwater produced by removing salts from seawater (10,000-40,000 mg/L TDS)	Potable: often blended directly with drinking water in the distribution pipes	High capital and energy cost per unit water production, challenges in brine waste disposal	Unlimited drought-proof source of water supply for coastal cities	Cost: capital, operation and maintenance, energy consumption	Greenlee et al. (2009)
Brackish water desalination	Freshwater produced by removing salts from brackish water (1,000-10,000 mg/L) sourced from groundwater or estuaries	Potable: often blended directly with drinking water in the distribution pipes or for industrial applications	High capital and energy cost per unit water production, challenges in brine waste disposal	Relatively large quantity of unexplored resource, lower energy demand than seawater desalination		
Condensate capture and atmospheric water harvesting	Water collected from condensed vapor from air conditioners or harvested from the atmosphere directly	Non-potable: can be used for irrigation, toilet flushing, cooling tower make-up water	Feasible only in hot, humid areas due to inefficiency in operation, potential for contamination if left in warm, stagnant, storage	Potential for large quantities of water to reduce fresh water demands	Cost: energy consumption per unit yield, water production per mass unit refrigerant	Al-Farayedhi et al. (2014)

*Contaminants of Emerging Concern (CEC)

A notable difference between existing water supplies and non-traditional sources is the cost. Cost structure for converting non-traditional waters to useful forms varies according to treatment and distribution requirements. The cost differences between several non-traditional water sources and traditional water supply can range from 1.5 to 4 times higher (Figure 2.2).

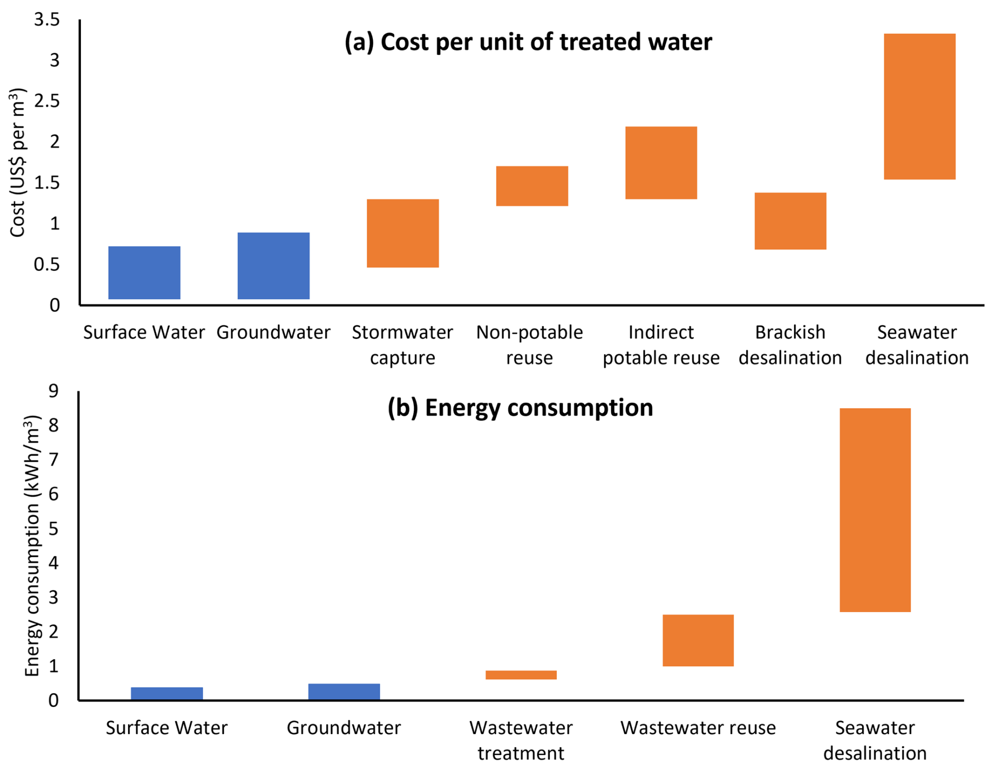

Figure 2.2: Total cost ranges per cubic meter (in 2015 dollars) water treated (a) and energy consumption[d] (b) for traditional (blue)[a,b] and non-traditional (orange)[c] water supply sources.
([a]Surface water cost Cooley (2020), [b]Groundwater cost Perrone and Rohde (2016); Cooley et al. (2019), [c]Non-traditional source cost Zhao et al. (2020), [d]Energy consumption Nassrullah et al. (2020))

Desalinated seawater has the highest cost with an upper end at U.S. \$3.3/m^3. Since they are potable non-traditional water supply sources, desalinated seawater and brackish water are regulated as part of the potable drinking water supplies. Therefore, the associated quality and health concerns are minimal. These waters vary instead with high capital cost and energy requirements for salts removal. While seawater desalination can provide a virtually limitless supply of freshwater, its cost and energy requirements pose a significant hurdle for many stakeholders and municipalities. Historically, desalination technology has been seen as a “last resort,” due to its energy intensive regime, but there has been a two-fold reduction of energy requirements over the last three decades and a booming of membrane seawater desalination plants around the world in the recent decade (Voutchkov (2018)). In addition, desalination (particularly of seawater) is only a feasible option for coastal regions, thus it is not readily accessible to landlocked countries experiencing water shortages (e.g., Jordan, Mongolia, and Nepal) (Schyns et al. (2015); Liu et al. (2017); Pandey (2021)). The varying quality, health risk, capital cost for treatment, and energy status of these water sources are the major knowledge gaps that require comparative quantification through models and simulations for better understanding of their origins, impacts, and management strategies for moving forward with water resource decisions.

2.1.2 Quality and quantity factors for non-traditional water sources

Water quality

Methods and practices for monitoring of physical and chemical properties of water are well-established, and tools for real-time or near-real-time monitoring of water quality indicators such as turbidity, conductivity, temperature, dissolved oxygen, and other parameters are generally available and adopted by water utilities in most of the developed countries (Zhao et al. (2020)). In addition, many types of mathematical models and numerical tools for water quality simulation and analysis have also been developed over the last several decades, with most research publications in this area beginning around 1970 (Fu et al. (2019)). However, microbiological constituents that represent major health risks for water reuse are more difficult to measure due to time consuming methods, the need for access to resources and testing materials, and uncertainty in model predictions for the level of pathogens or health risk present at a given time due to the non-conservative nature of the microbial pollutants. Currently, the U.S. EPA drinking water regulations are only applicable to traditional sources of water supplies, and there is no specific policy for non-traditional sources of water, such as reclaimed municipal wastewater for reuse or stormwater harvested for reuse, at the federal level. Thus, some states have their own reuse standards and regulations depending on water source and end use, while many others do not.

Microbial water quality currently relies on monitoring of fecal indicator bacteria, as it is well-established and utilized around the world as a marker of fecal contamination. In more recent years, increased attention has been placed on opportunistic pathogens that are not of fecal origin, such as *Legionella* and *Pseudomonas*, since they are commonly found in water storage and distribution systems and represent a significant contribution to worldwide disease burden (Price et al. (2017)). Additionally, since the COVID-19 pandemic begin, greater focus has been placed on viral pathogens in water sources and wastewater. They are particularly relevant for municipal wastewater reuse and rainwater/stormwater harvesting, in which viruses are present in a much higher concentration than in traditional water sources. The treatment technologies applied for water reclamation are much more variable and region-specific than in the highly standardized and centralized water management regime, and thus, the efficiencies of pathogen removal are highly uncertain. The decisions for implementing wastewater reuse and rainwater harvesting should focus on quantitative microbial risk management and water quality monitoring based on one or several pathogens to represent variability and uncertainty, to address various assumptions in these scenarios (Xagoraraki and O'Brien (2020)). Therefore, microbial risks should be a focus in the discussion of adopting rainwater, stormwater, or wastewater as new water supplies.

Contaminants of emerging concern

Besides pathogens, there are several contaminants of emerging concern (CECs) regarding water quality and for increased treatment and research efforts. Adverse effects of the CECs range from adverse health effects for humans and increased resistance in bacterial communities to impacts on marine life when discharged into the environment. Pharmaceuticals are one major class of CECs with origins of human and veterinary medicine. Pharmaceuticals include antibiotics, legal and illicit drugs, analgesics, steroids, stimulants, and beta blockers (Fatta-Kassinos et al. (2011)). They have been detected in wastewater from concentrations of 1 to 303,500 ng/L (Adeleye et al. (2022)). Pharmaceuticals in the environment can cause adverse effects on fish survival and behavior and are the contributor of the development of antibiotic resistance in bacterial populations (Sehonova et al. (2018)). Personal care products, like pharmaceuticals, are a major CEC and include a broad range of compounds from products used in health, beauty, and cleaning such as lotions, shampoos, sunscreens, toothpaste, and fragrances. Personal care products have also been detected in the ng/L to ug/L range in both surface waters and wastewater streams (Wang and Wang (2016)). One subset of pharmaceuticals and personal care products (PPCPs) known as endocrine disrupting compounds (EDCs) is a class of its own, coming from anthropogenic sources such as surfactants and pesticides. By definition, they are CECs due to their endocrine-disrupting effects for both wildlife and humans, such as declining male fertility, birth defects, and breast and testicular cancers.

Algal toxins, such as microcystins, are CECs that come from harmful algal blooms. Detected in the environment and in drinking water, they are toxic to fish, shellfish, and humans, and can cause illness, cancer, or death. Many countries worldwide have enforced concentration limits, from 1 to 1.5 $\mu g/L$(Richardson and Kimura (2019)). They have also become an issue for livestock, which experience poisoning and even death from these toxic blooms in their water supplies. Algal toxins are also problematic for desalination, causing increased chemical requirements and membrane fouling rates. Per- and polyfluoroalkyl substances (PFAS) have risen to a priority research area regarding their fate, abundance, and removal from water supplies. PFAS are environmentally persistent and have many origins including food packaging, fabrics, nonstick cooking pans and firefighting foams (Richardson and Kimura (2019)). Many studies and reviews have been conducted noting their sources and occurrence in water and wastewater (Vo et al. (2020); Crone et al. (2019); Scher et al. (2018)), their removal and techniques for remediation (primarily through adsorption) (Gagliano et al. (2020); Wanninayake (2021)), and human exposure and health effects (Domingo and Nadal (2019); Kotlarz et al. (2020); Podder et al. (2021). Recently, the US Environmental Protection Agency (EPA) officially announced the first federal drinking water standards for six PFAS at 4 parts per trillion (ppt) (EPA (2023)). This enaction will further the methods of remediation and monitoring of PFAS in drinking water sources around the world. Finally, microplastics are a growing topic area and CEC in water supplies and environmental waters. Caused by degradation from larger plastics, microplastics are defined as smaller than 5 mm in size. To combat microplastics, the Microbead-Free Waters Act banned microbeads in cosmetic products in the US (H.R.1321) where tap water has the highest worldwide microplastics concentration at 9.24 particles/L (Eerkes-Medrano et al. (2019)). In addition to traditional drinking water treatment methods for microplastics removal such as sedimentation and coagulation/flocculation, some successful methods are magnetic extraction, electrocoagulation, and membrane separation (e.g., ultrafiltration and reverse osmosis) (Shen et al. (2020)). Still, the presence of microplastics in drinking water and their fate in environmental waters will continue to be a challenge for years to come, as plastics are ubiquitous and long lasting.

Water quantity

When designing, implementing, and operating technological processes for water treatment and delivery, size (capacity) and cost are the priority metrics, especially at large urban scales. The economic trade-offs between conservation and investing in additional water supplies or technology need to be explored for a best management practice. In addition, energy requirements need to be considered, as scaling current efforts may not be energy-efficient or cost-effective. Many water technologies demonstrate economies and diseconomies of scale due to the cost, size, energy, and delivery requirements at different operating scales. Modeling efforts for these quantity metrics of non-traditional water sources are needed to illustrate these trade-offs and pinpoint areas of potential cost savings, energy requirements and capacity utilization for optimal use.

In addition to economic cost, the energy and associated carbon footprint and emissions must be considered when expanding or updating water treatment and resource technology. The carbon footprint and energy requirements can vary greatly depending on location, the source of energy, and the water source technology, ultimately impacting the overall cost and carbon emissions. Traditional water treatment has a carbon footprint of 0.11-0.16 kg CO_2eq m^{-3}. The carbon footprint is defined as the sum of individual greenhouse gas emissions, in which carbon dioxide (CO_2) and other greenhouse gases are expressed in carbon dioxide equivalents (CO_2eq) (Eggleston et al. (2006)). However, treatment for non-potable reuse of wastewater (Sections 2.2 & 2.2.2) has 0.3-2 kg CO_2eq m^{-3}. This is further increased by direct potable reuse and reverse osmosis (Sections 2.3 & 2.4) with 0.6-2.4 and 0.4-6.7 kg CO_2eq m^{-3}, respectively (Cornejo et al. (2014)). The energy requirements, cost, and carbon footprint are intertwined metrics with the quantity of water that a technology is designed to output.

The quantity of the respective non-traditional water supplies is highly region-specific. Rainwater and stormwater harvesting provides the most quantity in regions with ample rainfall, such as in tropical areas. However, in some areas where rainfall is less plentiful, other non-traditional water sources should be explored as alternatives to supplement existing water supply. For example, New Mexico and Texas have more brackish groundwater reservoirs, which makes brackish water desalination an option for non-traditional water supply (Buono et al. (2016)). Seawater desalination, in turn, is an apparent solution in coastal regions, such as California, Florida, Spain, and Israel (Quon et al. (2021b)). South Asia and the Arabian Peninsula may more effectively utilize captured condensate or atmospheric water harvesting due to humidity and air conditioner use (Loveless et al. (2013)). Furthermore, regional needs are very different and affluence level is not the same. This paper focuses on urban and developed regions where centralized water system have been in place. There are many unique challenges in developing rural regions that are not possible to be all inclusive within the scope of a single paper. Nevertheless, identification of a best non-traditional water source for inclusion in a water supply depends on local resource availability and water source quantity, highlighting the importance of diversification and the comprehension that no single water source is the solution for water shortages now or in the future.

2.1.3 Summary of modeling frameworks

Techno-economic assessment (TEA)

Techno-economic assessment (or analysis) (TEA) integrates a process with a cost model to ultimately estimate the capital cost and operating costs of the given process. Beginning with a process flow diagram (PFD), a process is outlined by unit. Previously established cost curves are useful in estimating the requirements and costs for units based on sizing, such as the volumetric flowrates needed for water treatment technologies, or the chemical addition needed based on the flow. Spreadsheets or process simulators such as WaterTAP3 (Miara et al. (2021a)), based in Python are most utilized for TEA. Once the process model is successfully implemented including all sizing and cost requirements, the final total capital cost, operating costs, and energy requirements can be summarized and compared between processes or locations (Burk (2018)).

Cost-benefit analysis (CBA)

Simply, cost-benefit analysis (CBA) is a systematic approach to weighing the benefits, such as benefits to the environment, with the costs of a process or policy. Unlike TEA, CBA can include benefits that are intangible or nonmonetary. In recent years, the externalities or environmental benefits are given a "shadow price" to establish a monetary value to such aspects that have no market value. A simple net profit equation can then be used to calculate the difference between the costs and priced benefits such as $NP = \sum B_i - \sum C_i$, where NP is the net profit, B is the benefit value of item i and C is the cost of item i, as outlined by Molinos-Senante et al. (2010) for wastewater treatment. As with TEA, a series of costs and values can be established for various processes, treatments, or pollutants and a CBA can be conducted and compared across different layouts or facilities. CBA is a useful tool in the decision-making process, particularly in adopting non-traditional water sources.

2.1.4 Life cycle analysis (LCA) and life cycle cost analysis (LCC)

Life cycle analysis (LCA) and life cycle cost (LCC) analysis are two additional tools for quantifying the costs and impacts of a system. LCA is a combination of the "inputs, outputs, and potential environmental impacts of a product system throughout its life cycle," also referred to as its impacts from "cradle to grave" (Finkbeiner et al. (2006)). Hellweg and Milà i Canals (2014) outlined LCA in four steps: 1) defining the goal and scope including the system boundaries such as resource extraction to end-of-life disposal of materials; 2) inventory analysis to compile all inputs, resources, and emissions; 3) impact assessment, or categorizing and converting impacts/emissions to a common unit such as CO2eq; and 4) interpretation of results, such as finding that a proposed water treatment technology has higher environmental impacts than the current system in place. Therefore, LCA is useful for assessing carbon footprint and emissions associated with non-traditional water sources. LCC extends the framework of LCA, which assesses the total impact of a system, to the costs associated with a system. LCC accounts for "all costs of acquiring, owning, and disposing of a building or building system" (Fuller (2010)). One area of uncertainty and challenge in utilizing TEA, LCC, CBA and LCA is a lack of a strict framework or methodology, resulting in models and approaches that are specific to individual case studies (Giacomella (2021)).

Quantitative microbial risk assessment (QMRA)

One of the most useful quantitative tools and frameworks for estimating health risk is the Quantitative Microbial Risk Assessment (QMRA). QMRA is a framework outlined by the National Academy of Sciences and often utilized by the U.S. EPA for evaluating microbial health risks of drinking water and water supply systems (Soller et al. (2016); NRC (1983)). The framework is comprised of five main components: hazard identification, exposure assessment, dose–response assessment, risk characterization, and risk management. First, a particular pathogen or toxin is identified as the hazard of concern for a modeled scenario of interest. Next, a specific exposure pathway and scenario is defined and modeled, such as drinking untreated water or eating produce which has been irrigated with recycled wastewater. This exposure assessment uses a quantitative model and/or behavioral data to estimate a dose of the pathogen one may be exposed to during a given scenario event. Then, a best-established dose-response model is utilized to calculate a probability of a response (such as illness or death) due to the range of possible exposed doses. Clinical dose-response data is useful in this endeavor and fitted to a model. Finally, the total risk of response is estimated, quantifying a daily, annual, or otherwise risk based on all inputs to provide the best course of action (risk management) going forward. QMRA is often utilized by the US EPA to establish or improve regulatory measures and monitoring practices.

2.2 Rainwater

2.2.1 Rooftop-harvested rainwater

Rooftop-harvested rainwater (RHRW) is defined as rainwater collected from the runoff of building rooftops and stored in engineered structures such as a rain tank or an underground cistern (Boers and Ben-Asher (1982); Campisano et al. (2017)). In comparison with rainwater that falls on the ground that can collect oil, grease, animal feces, trash, and other pollutants from the road, RHRW has relatively fewer contaminants and may serve as a good source of supplemental water for existing supplies (Boers and Ben-Asher (1982); Cook et al. (2014); Gurung and Sharma (2014); Imteaz et al. (2011)). In regions where rainfalls are plentiful, RHRW is a well-established water supply system with mandatory installation in countries such as Spain and Belgium (Domènech and Saurí (2011)). Additionally, drier countries such as Australia have utilized RHRW in part due to increased environmental awareness and mandatory water restrictions in urban areas (Rahman et al. (2012)). South Africa has utilized RHRW for generations, and tens of thousands of households use rainwater as their main water source (Kahinda and Taigbenu (2011)). The U.S. Virgin Islands has legal precedents in place, with building codes stating that buildings must consist of a "self-sustaining water supply system" such as a well or rainwater collection area and cistern (V.I. Code tit. 29, § 308, 2019). In more arid regions, RHRW systems can be used to capture and store water for temporary use to supplement other water supplies. Most RHRW is used for domestic purposes by individual household or apartment buildings, namely showering, toilet flushing, clothes washing, and outdoor watering (Figure 2.3). One of the main benefits of installation of RHRW systems is the reduced dependence on centralized water supplies. The secondary benefit of RHRW systems is to reduce the peak of the hydrograph during major storm events, thus, reducing stormwater runoff and surface water contamination (Sepehri et al. (2018)).

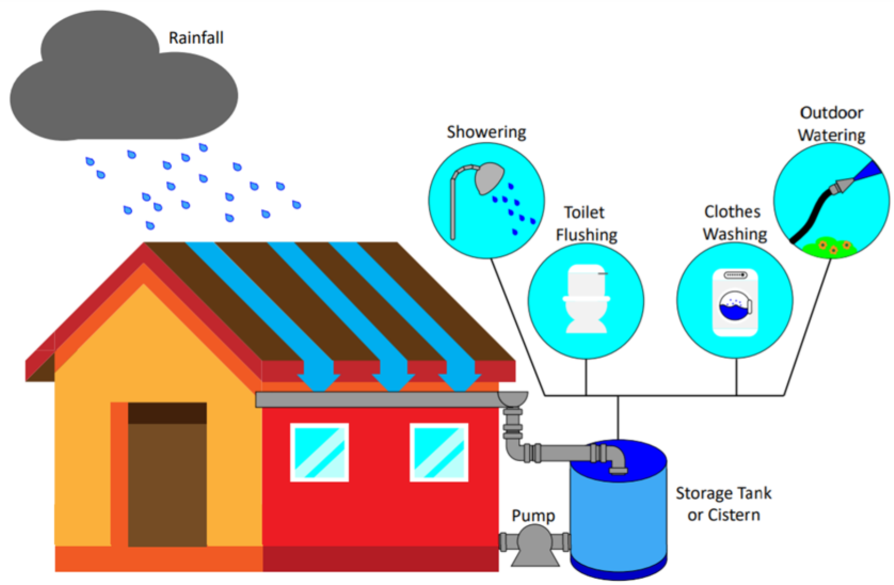

Figure 2.3: Diagram of RHRW and its application for domestic use.

The water quality of RHRW varies with system design, level of treatment, and local factors such as climatic conditions and regulation. Several studies have linked RHRW use with disease outbreaks and health risks for both drinking and household use (Ahmed et al. (2011); Fewtrell and Kay (2007); Crabtree et al. (1996); Dean and Hunter (2012); Simmons et al. (2008)). A suite of pathogens has been identified and associated with RHRW, with origins of dry deposition, wet deposition, and wildlife Campisano et al. (2017). Rainwater may contain *E. coli*, *Legionella* spp., *Salmonella* spp., *Mycobacterium avium*, and *Giardia*, according to limited investigations on water quality (Ahmed et al. (2011); Dobrowsky et al. (2014); Kaushik and Balasubramanian (2012)). Formation of natural biofilm and regrowth of bacteria in the rain tank are also major water quality concerns of RHRW)Hamilton et al. (2019a, 2017a); Zhang et al. (2021).)

RHRW has the potential to act as a potable source of water, however further levels of treatment are generally required to ensure that the supply meets potable quality standards, as determined by a review of recent developments in RHRW technology and management practices (Alim et al. (2020)). A study by Fuentes-Galván et al., (2018) found that after visual inspection and physical tests of RHRW in Guanajuato, Mexico, further treatment was required before consumption (Fuentes-Galván et al. (2018)). Another study by Keithley et al. (2018) found that RHRW systems in Texas, U.S. which utilized activated carbon filters and/or had chlorine residuals above 2 mg/L produced high quality potable water (Keithley et al. (2018)). Therefore, on-site testing and maintenance of RHRW collection and storage systems is suggested before recommending strategies for safe non-potable and potable water use or consumption.

Water quality and quantity estimation of RHRW

Quantitative models are useful for RHRW design and implementation. The design criteria are a balance of quantity of RHRW with water demand. Correlating, or matching, the water demand with the water supply availability (i.e., rainfall) is the main objective of quantitative models. A balance of these two variables would result in optimal storage and design recommendations for a RHRW system. Two possible approaches for this objective are based on empirical observations (Ghisi (2010); Rahman et al. (2011)) or stochastic rainfall analysis (Basinger et al. (2010); Cowden et al. (2008)). Water demand and use dynamics are highly variable and more difficult to accurately capture and model. Socioeconomic factors have a large impact on water use, even in similar areas or regions, as the demand and use can vary at the household level. Studies have been conducted to model water use and associated RHRW design based on empirical data and system configurations (Leonard and Gato-Trinidad (2021); Melville-Shreeve et al. (2016)). However, predicting water demand at the household level requires further research especially in the effects of various socioeconomic factors on water use and perceptions surrounding water quality and system maintenance. Mathematical models have also been implemented in water quantity analysis for the purposes of analyzing design and operational costs and optimal configurations (Melville-Shreeve et al. (2016); Morales-Pinzón et al. (2015)). Morales-Pinzón et al., (2015) compared the deployment of three economic and environmental models: Plugrisost, AquaCycle, and RainCycle for anyslsis of RHRW systems, and found that the urban scale being modeled (such as residential scale or neighborhood scale) is a critical factor. While RHRW is a relatively simple technological system, its implementation and quantity challenges rely on understanding of local water use and rainfall dynamics, socioeconomic factors, and a balance between supply and demand for optimal system design (Morales-Pinzón et al. (2015)). Accounting for the cost, footprint, and carbon emissions of constructing such systems has also been modeled and estimated. Hofman-Caris et al., (2019) modeled six scenarios of rainwater collection with various treatment methods specifically for potable use in the Netherlands and found impacts of 0.002-0.004 kg CO_2eq m^{-3}, as compared with around 1.16 kg CO_2eq m^{-3} for centralized, traditonal water supply (Hofman-Caris et al. (2019)). Non-potable systems that do not implement treatment methods such as reverse osmosis or UV disinfection, inherently would have even smaller carbon footprints to operate, which is the case for many countries.

Surveys and case studies that accommodate water analysis and modelling efforts can play a key role in identifying the impacts of location-specific and socioeconomic factors. Collection of information and data regarding income, water use, perception of water quality, and attitude towards treatment technologies is suggested and has proven successful. Such efforts were performed in Pakistan for RHRW, finding that the residents, especially the women, believed they could benefit from RHRW systems to improve their lives, but supported government subsidization as the income levels were generally low (Abbas et al. (2021)). In the U.S. Virgin Islands after disastrous hurricanes in 2017, surveys revealed that access to clean water was more limited for lower income groups. Higher income groups used bottled water most during this time of crisis, and there was a disparity in the local perception of water safety, also divided by income group. All groups and income levels felt the government should have intervened further and provided better access to clean water during this time (Quon et al. (2021a)). These analyses, one in a water-stressed country and one in a tropical region, both demonstrated the benefits and acceptance of RHRW, but highlight the impacts of socioeconomics and local perception on their water use and access. Since RHRW is a non-traditional water source used at the single household level, socioeconomics and public acceptance are critical factors regarding its use and implementation.

In terms of water quality, one of the most useful quantitative tools for understanding health risk of RHRW use is Quantitative Microbial Risk Assessment (QMRA). One example for RHRW would be using QMRA to quantify the annual risk of falling ill from or being infected by either *Legionella* or *Mycobacterium avium* complex while showering using RHRW (Quon et al. (2021a); Hamilton et al. (2017b)). A statistical outcome is modeled from a scenario based on probabilistic inputs along the framework and utilizing available data. Based on the results, risk management strategies can be recommended to reduce or mitigate future health risk. Many QMRA studies have been conducted on RHRW as a water supply for consumption (Dean and Hunter (2012)), gardening (Lim and Jiang (2013)), showering (Schoen and Ashbolt (2011)), and toilet flushing and faucet use (Hamilton et al. (2019b)). QMRA was also carried out for environmental exposures such as at a water park setting (De Man et al. (2014)). As such, the forefront of models for quantifying quality and health risk for RHRW are based on potable consumption and non-potable exposure through aerosolization (Bollin et al. (1985); Morawska et al. (2009)). Quantifying risk and water quality limits for risk thresholds is the first critical step in recommending proper disinfection and maintenance strategies for RHRW. When conducting QMRA, it is imperative to have data and information on 1) pathogen levels in the source water, 2) the type of exposure or water use (e.g., drinking, toilet flushing), and 3) the human dose-response relationship to a specific pathogen. The pathogen reduction based on chosen treatment or disinfection methods can also be modeled if the percent or logarthimic reduction is known. Based on the microbial risks from quantitative analyses, opportunistic pathogens in premise plumbing are identified as a critical future research area. Pathogen levels are often difficult to predict and measure, especially when RHRW storage and treatment is variable and seldom monitored. Impacts due to seasonality, presence of animals, and extreme weather events can all impact pathogen level, and bacterial growth and regrowth even with treatment interventions (Quon et al.

(2021a); Ahmed et al. (2018); Fiorentino et al. (2021)).

2.2.2 Harvested stormwater

Stormwater, or surface runoff collected from ground and in the storm drains, can become a non-traditional source of water with multiple benefits: 1) mitigating impacts to receiving surface water quality due to pollutants carried in stormwater runoff; 2) reducing risk of flooding in urban areas; and 3) increasing non-potable water supply when collected and managed appropriately (Philp et al. (2008)). Therefore, stormwater harvesting has gained attraction from an integrated urban water management perspective in the recent years. Without duplication of previous reviews on the benefits of stormwater harvesting for surface water quality protection and flood mitigation (Philp et al. (2008); Ahmed et al. (2019); Jiang et al. (2015); Akram et al. (2014); Mitchell et al. (2007)), this review will focus on models to determine the suitability of harvested stormwater as a non-traditional water supply.

Stormwater harvesting (SWH) is similar to RHRW, the distinguishing factor being RHRW is rainfall only from rooftops and stormwater is collected from drains, gutters, waterways, or engineered permeable infrastructure. SWH systems can comprise of various methods for collection and conveyance such as traditional drains and gutter systems or green infrastructure. A diagram illustrating integrated SWH in an urban setting is shown in Figure 2.4. Green infrastructure includes constructed systems, defined as "the range of measures that use plant or soil systems, permeable pavement or other permeable surfaces or substrates for stormwater harvest and reuse, or landscaping to store, infiltrate, or evapotranspirate stormwater and reduce flows to sewer systems or to surface waters," by the Water Infrastructure Improvement Act (H.R. 7279, 115$^{\text{th}}$ Congress). Notably, common types of green infrastructure for stormwater harvesting are bioswales, biofilters, and permeable pavements.

The quality and quantity of the stormwater runoff is critical in designing infrastructure

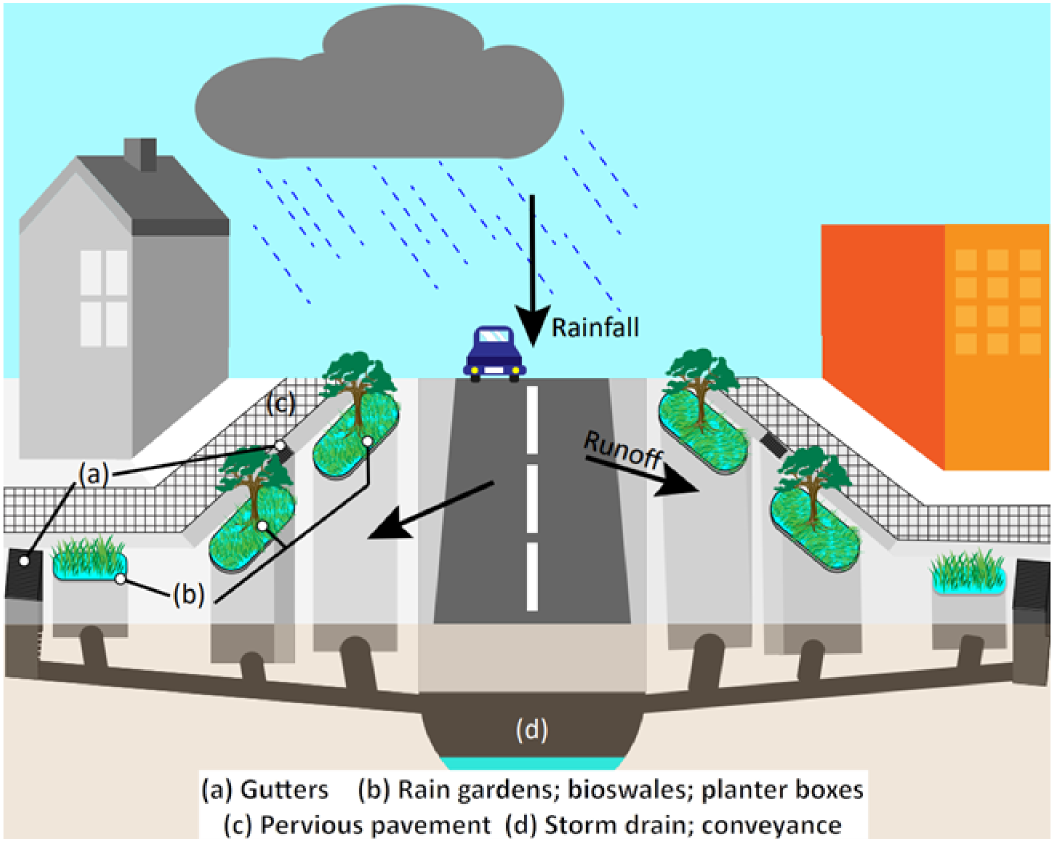

Figure 2.4: Diagram of stormwater harvesting in an urban setting.

and in determining management strategies for stormwater non-potable reuse. Treatment is a critical part of SWH for reuse, and the level and type of treatment depends on the reuse application. Stormwater that is collected in urban settings can contain a variety of contaminants, with many origins such as rainfall, irrigation and agricultural runoff, and car washes. The potential pollution sources for stormwater runoff could then be from vehicle oil and fuel (Hong et al. (2006)), organic matter (McCabe et al. (2021); McElmurry et al. (2014)), pesticides and fertilizers (Chen et al. (2019); Tsihrintzis and Hamid (1997)), heavy metals (Brown and Peake (2006)), and pathogens (Sidhu et al. (2012); Graham et al. (2021)). While SWH is useful in diverting these streams to prevent surface water contamination, water quality becomes the main concern when targeting stormwater for reuse and environmental applications. Some methods of stormwater capture such as biofiltration provide treatment of the stormwater but quantitative modeling should be done on a case-by-case basis to evaluate treatment needs and to determine the best method for removing pollutants and pathogens from the stormwater.

Quantity and quality estimation of harvested stormwater

Modeling for harvested stormwater can be conducted for simulating stormwater and hydrological movement for a watershed, or for predicting pollutants and quality metrics. Much like modeling for RHRW and rainwater harvesting systems, the quantity modeling is highly dependent on rainfall patterns. However, when not using rain tanks or cisterns, the subsequent runoff of stormwater must be modeled hydrologically, that is the simulation of stormwater transport across a watershed due to an inflow of rainfall. Stormwater models can be simple or complex, and deterministic or stochastic in nature. Many models for urban stormwater have been established in the field of hydrology. A review of 12 urban stormwater models was conducted by Zoppou (2001) (Zoppou (2001)). Here, the author also describes the different types of urban runoff models and the data requirements of each method. There are many more models for stormwater in urban catchments, as this area is not a new field of study. However, for the purposes of harvesting and reuse as a non-traditional water source, modeling the capture and storage of the stormwater is an area of focus. Fletcher et al., (2007) modeled various scenarios of implementing SWH in urban settings using the Model for Urban Stormwater Improvement Conceptualization (MUSIC) (Fletcher et al. (2007); Wong et al. (2002)). Model results showed that urbanization and pervious land coverage impact stormwater flow and runoff quality, and implementation of SWH and reuse regimes can address these impacts. Some of the most common models and tools for stormwater are the Storm Water Management Model (SWMM), STORM, and HEC-HMS.

In addition to understanding the benefits and quantity of stormwater reuse, stormwater-capture often utilizes "green infrastructure" as mentioned previously. As urban sustainability become more concerned with carbon emissions and footprint, green infrastructure can provide carbon offsets in the form of carbon sequestration. A study by Kavehei et al., (2018) compared across the literature the carbon sequestration potential of various stormwater infrastructure (Kavehei et al. (2018)). They found that rain gardens had the smallest net carbon footprint (carbon footprint – carbon sequestration) at -12.6 kg CO_2eq m^{-2} followed by bioretention basins, stormwater ponds, and vegetated swales, at 28.7, 108.9, 10.5 kg CO_2eq m^{-2} over a 30-year life, respectively. However, they did not account for the net footprint per volume of water treated or captured in this study. The referenced literature mainly utilized forms of life cycle analysis (LCA), a powerful methodology framework for quantifying the impacts (e.g., carbon footprint) of a process or technology. LCA is known as the "compilation and evaluation of the inputs, outputs, and potential environmental impacts of a product system throughout its life cycle," (Finkbeiner et al. (2006)) LCA analyses and quantifies the environmental aspects (e.g., emissions, resources) through all life cycle phases and processes of a technology, thus allowing for comparison across different scenarios or technology options that serve the same function, such as in the case of different non-traditional water source technologies. LCA requires the environmental impacts (e.g., CO_2eq) for all upstream processes in order to aggregate and conclude the final, or lifetime impacts. Thus, it is imperative that for a water treatment process, each unit of the treatment train is known such that the materials and impacts can be acquired in data form.

While the models mentioned above are used for predicting the quantity and quality of stormwater in urban catchments, fewer approaches have quantified the benefits of SWH for reuse purposes and extraction. Recently, Zhang et al., (2020) approached this gap by analyzing factors that could be impacted by SWH and quantified pollution mitigation and water quality improvements of different SWH scenarios (Zhang et al. (2020)). Using a sensitivity analysis approach, they found clear benefits of SWH in pollution reduction. Another study simulated a runoff model on a college campus by a stormwater capture tank with real-time control for capture and storage (Parker et al. (2021)). Here, the benefits of SWH for water supply and flood risk reduction were highlighted based on rainfall event prediction, precipitation, and tank size simulation.

As with RHRW, SWH quality for reuse has been approached based on modeling human health risk from water use and exposure. Ma et al., (2019) analyzed SWH pollutants based on hazard indices for drinking and swimming to create a hierarchy of hazard control for stormwater management (Ma et al. (2019)). Murphy et al., (2017) followed QMRA methodology to establish risk benchmarks for various stormwater harvesting scenarios and consumer uses and found that current (as of 2017) guidelines were inadequate for mitigating risk of *Campylobacter* (Murphy et al. (2017)). Risk-based QMRA was used by Schoen et al., (2017) to find targets for reduction of various pathogens in water sources, including stormwater for domestic use (Schoen et al. (2017)). These targets provide clear recommendations and standards for microbial risk and act as guidelines for disinfection and treatment of harvested stormwater for non-potable reuse.

Water quality, health risk, and quantity modeling of SWH is regional and scenario specific and is difficult to model for general cases. QMRA and risk-based modeling are critical for establishing treatment methods, guidelines, and regulations for SWH non-potable reuse. Hydrologic modeling and rainfall simulation play an important role in designing catchment and storage size requirements for SWH, and studies have shown that SWH systems are effective in reducing flood risk and improving water quality when using systems such as biofilters and bioswales to mimic the natural treatments.

2.3 Municipal wastewater

2.3.1 Reclaimed municipal wastewater

Reclaimed, or recycled, municipal wastewater (or sewage) is becoming an increasingly popular source of both non-potable and potable water. Partially treated wastewater from a sewage treatment facility that is normally discharged into the ocean, lakes, or rivers can be further treated for non-potable uses. Reclaimed non-potable reuse water is most often piped separately and used for irrigation and agriculture and other municipal purposes (Meneses et al. (2010)). Non-potable reuse is especially valuable in arid and semi-arid regions where rainfall is less common, and recycled water for irrigation and agriculture can reduce the demands on the conventional water supply. Water production for non-potable reuse only requires additional disinfection processes beyond traditional wastewater treatment for surface discharge. Potable reuse of wastewater requires advanced treatment technologies beyond the traditional biological treatment to meet potable drinking water standards. The finished water is often used to recharge groundwater aquifer or supplement surface drinking water reservoirs, which is referred to as indirect potable reuse. Directly potable water reuse of advanced treated wastewater is rare but is in consideration in highly water-stressed regions (i.e., California). So far, most of the wastewater reclamation plants in the U.S. are large, centralized municipal wastewater treatment utilities. The production of water for potable purpose often employs processes of biological treatment, microfiltration, ultrafiltration, reverse osmosis, UV disinfection, and advanced oxidation (Tang et al. (2018)). Therefore, it is much more costly in comparison with non-potable water reuse. Large-scale wastewater reuse is still emerging, often hindered by complex social and economic factors and management practices (Mizyed (2013); Garcia and Pargament (2015); Padilla-Rivera et al. (2016)). The most noteworthy successful implementations of wastewater reuse worldwide are in the United States (Rice et al. (2013)), Israel (Friedler (2001)), Singapore (Lafforgue and Lenouvel (2015); Tortajada

(2006)), Australia (Gude (2017)), and Namibia (Lahnsteiner and Lempert (2007); Jiménez and Asano (2008)).

One example of successful municipal wastewater reuse implementation is at the Groundwater Replenishment System (GWRS) in Southern California, where a portion of treated wastewater from a neighboring sanitation plant is diverted to a drinking water purification facility for full advanced treatment to produce drinking water quality water. The purified reclaimed wastewater is then infiltrated into local groundwater aquifers to increase water storage and for better public perception of the treated water through blending with the natural groundwater before withdrawn for drinking water treatment (Duong and Saphores (2015)). The GWRS allows for a reduced need for imported water sources to the local area, which were shown to cost more than water produced at the GWRS. In this case, the community involvement and transparency of treatment process, costs, and water quality of the produced water led to a successful addition to the local water portfolio.

The implementation of GWRS is highly costly as it requires expensive treatment processes, piping, pumping, and pathogen log-removal requirements defined by the regulatory agencies. As defined in the State of California, under Title 22, full advanced treatment is the treatment of wastewater using a reverse osmosis (RO) and an oxidation treatment process (22 C.C.R. §60320.201). Therefore, the high cost of water purification limits the broader implementation of wastewater as a non-traditional source of water supply in low economic regions outside California. Moreover, the application of reclaimed wastewater for domestic use can come with negative perception, often referred to as the "yuck factor." Duong & Saphores, (2015) explored this qualitative obstacle and found that it is one of the main reasons why purified wastewater is often not directly used to supplement drinking water supplies (Duong and Saphores (2015)). This factor requires public outreach efforts, such as other terminology besides "wastewater" and "sewage" in addressing and striving for public acceptance, or indirect potable reuse, such as with the case of the GWRS.

Wastewater treatment plants have been identified as a hotspot for enriching antibiotic resistance and the transmission of antibiotic resistant bacteria (ARB) and antibiotic resistant genes (ARG) into the environment (Rizzo et al. (2013)). This is due to the inputs of pharmaceuticals and antibiotics from fecal sources, and the conditions of biological treatment including high density of bacteria leading to horizontal gene transfer of ARG (Garner et al. (2021); Rizzo et al. (2013); Guo et al. (2017); Le et al. (2022)). The most common uses of this type of water source are to flush toilets and urinals, irrigate parks, golf courses, and agriculture, and other uses such as industrial cooling and ornamental fountains or water features (Chen et al. (2013)), all of which may involve human exposure.

The role of wastewater treatment in the prevalence and fate of ARB and ARG is now a research focus area (Bengtsson-Palme et al. (2019); Mao et al. (2015); Jiao et al. (2017)), but the role of reclaimed wastewater in this area is less understood. There is concern that reclaimed wastewater could facilitate the spreading of ARB and ARG and pose a threat to human health due to exposures with any of the aforementioned applications, though there is uncertainty. Several studies have begun to look into reclaimed water and distribution systems (Piña et al. (2020); Fahrenfeld et al. (2013)) but it is noted that further exploration is needed to establish better monitoring, future data availability, and a better understanding of the magnitude of ARB and ARG occurrence in environmental applications (Garner et al. (2021); Pepper et al. (2018)).

At the individual building level, sewage can also be reclaimed and recycled, most commonly for toilet flushing and overall water savings. This design concept for "green buildings" has been around for decades, for example in urban Japanese cities (Ogoshi et al. (2001)). Large office buildings, skyscrapers, or apartment complexes with an onsite wastewater treatment system can treat and recycle wastewater, and separately pipe and distribute it back through the building. One successful example in the U.S. is the Solaire building in New York City, an apartment building which recycles both onsite wastewater and stormwater for toilet flushing and irrigation, respectively (Figure 2.5). Due to the treatment technology and nature of municipal wastewater, the main challenges faced for decisions to adopt this non-traditional water source are cost considerations of treatment implementation and public perception of water quality and health risks associated with wastewater. Therefore, a combination of quantitative modeling and public outreach for approval is critical in increasing the capacity and utilization of reclaimed wastewater as a water source.

Figure 2.5: The Solaire building in New York, New York, USA (left) which is a residential apartment building that reclaims building wastewater for toilet flushing and cooling systems and reclaims stormwater for irrigation of a rooftop garden (right) (Consentini (2022)).

Quantity and quality estimation for reclaimed wastewater

Many of the challenges associated with reclaimed wastewater involve the impacts of constituents on human health either by direct exposure or indirectly through consuming food products irrigated by recycled water, or contamination of groundwater supply. Thus, quantitative modeling of reclaimed wastewater has tended to focus on economic analysis, risk assessment, and fate and transport models.

Human exposure to wastewater that is reclaimed and reused for irrigation of agriculture can be through direct exposure (inhalation or ingestion near an irrigation source) or indirect exposure (consuming irrigated produce). QMRA has been applied to wastewater reuse for inhalation risk, due to pathogens such as *Legionella*, which has been shown to experience regrowth in distribution networks and in biofilms (Hamilton et al. (2017a); Proctor et al. (2018); Caicedo et al. (2019)). Identification of health risk based on exposure, dose-response, and probabilistic thresholds aids in establishing proper treatment, implementation, and water use strategies. For consumption, Shahriar et al., (2021) modeled the fate of various pharmaceutical products in an agricultural setting with reclaimed wastewater based on biodegradation of the organic compounds in the soil, uptake by agricultural plants, and biotransfer from the plant (alfalfa) to cattle (Shahriar et al. (2021)). Based on this fate, a risk assessment was conducted to quantify the human exposure via consumption of the cattle. There have been other similar models including direct consumption of reclaimed wastewater irrigated lettuce (Chandrasekaran and Jiang (2018); Van Ginneken and Oron (2000)), of irrigated rice paddy (An et al. (2007)), and of kale, coriander, and spinach (Njuguna et al. (2019)). Various combinations of transport models and Monte Carlo methods of risk assessment were deployed for the studies, which is in line with the assessments conducted for other water supplies.

The tradeoffs of implementing wastewater reuse are most often quantified through cost-benefit analysis (CBA). Cost analyses primarily includes capital investment, operation and maintenance, and other project costs such as chemical requirements. However, in weighing the benefits of such an operation, estimations of environmental, public health, and groundwater impacts are quantified. This is an important tool in the decision-making process surrounding wastewater reclamation and reuse, and a necessary step in designing a possible treatment train and reuse plan. Regional cost, reuse standards, and climatic factors play important roles in determining the outcomes of such operations and are necessary data inputs for CBA modeling. For example, semi-arid regions with less rainfall may benefit differently from reclaiming wastewater for irrigation than an area where rainfall is plentiful year-round. Specific case studies of CBA for wastewater reuse are in Italy (Verlicchi et al. (2012)), Beijing (Fan et al. (2015)), Spain (Molinos-Senante et al. (2011)), and the semi-arid regions of the Mediterranean (Gonzalez-Serrano et al. (2005)). These models are process-based and data-driven, as opposed to the more probabilistic methods of risk assessment in this area.

2.3.2 Greywater reuse

Greywater, a sub-portion of municipal wastewater, is defined and characterized differently around the world. Generally, it is defined as wastewater from all non-toilet plumbing fixtures in the home, including kitchen, bath, and laundry wastewater (Christova-Boal et al. (1996); Ghaitidak and Yadav (2013)). In some cases, dishwasher, kitchen sink, and laundry wastewater are excluded from greywater classification because wastewater from these sources generally has a higher pollutant load than greywater from bathing and hand washing (Al-Jayyousi (2003)). In comparison with municipal sewage discussed in the previous section, greywater collection requires dual plumbing to separate the wastewater streams, which are generally installed at the household and single-building scale. Blackwater, which includes but is not limited to toilet water, is generally piped by sewer lines to centralized municipal wastewater treatment plants, while greywater is harvested and treated on-site for reuse (Friedler (2004)). Greywater classification, treatment requirements and standards, and separation from blackwater are highly dependent on local policy and laws.

Greywater recycling and reuse represents a significant opportunity for water savings for a domestic residence and follows the same basic principle and paradigm as reclaimed wastewater.

Unlike the large-scale municipal wastewater reuse approach, greywater reuse is decentralized and more like RHRW in design and implementation. The decentralized and on-site approach to greywater reuse has been referred to as a close-loop concept (Al-Jayyousi (2003)). The most common reuse purposes of greywater are replacing potable water for irrigation and toilet flushing in the household. Widespread greywater reuse towards toilet flushing in urban households and multi-story buildings can achieve a reduction of up to 10-25% of urban water demand (Friedler and Hadari (2006)).

Greywater treatment technologies vary in performance and complexity and may include direct reuse approaches such as diversion for toilet flushing, or treatment by physical, chemical, or biological processes for short term storage. Filtration and disinfection are commonly employed on-site treatments. For filtration, sand or membrane filters are often used, and disinfection is achieved using chlorine tablets or ultraviolet (UV) light. More complex treatment systems including biological treatment, similar to wastewater treatment trains, are also implemented in some cases. Such treatments are anaerobic sludge blanket (Elmitwalli and Otterpohl (2007)), sequencing batch reactors (Leal et al. (2010)), and membrane bioreactors (Merz et al. (2007)). Greywater can sometimes be diverted and drained to outdoor irrigation systems after a filtration step, and some systems divert the greywater to a constructed wetland system for additional treatment before disinfection (Allen et al. (2010); Gross et al. (2007)).

When used for irrigation, some larger size pathogens (e.g., helminths) are of less concern since they are easily filtered out through soil infiltration. However, bacteria and viruses have been known to be problematic. For example, *E. coli*, *Salmonella*, *Shigella*, *Legionella*, and enteric viruses are of concern and have been found in greywater sources and irrigated soils (Finley et al. (2009); Jahne et al. (2017)), and *Legionella* can be spread by aerosolization, such as through sprinklers for irrigation (Hamilton et al. (2017b)). Further research on greywater pathogen monitoring and health risks is recommended for advancing and improving its utilization and implementation as a water supply. A better understanding of the effects of greywater on irrigated soil and produce, and on human exposure through reuse is necessary for establishing policies and strategies for storage, treatment, and distribution requirements. Greywater as a water supply can significantly alleviate household water demands on traditional sources and provide a sustainable water management option for future utility portfolios.

Quantity and quality estimation for greywater reuse

The quality concerns with reusing greywater are not dissimilar from the previous non-traditional water supplies. Therefore, modeling efforts follow the same principles. For health concerns related to irrigation, priority is given to quantifying the health risks of consuming produce that has been irrigated with greywater (Morales-Pinzón et al. (2015)) and in human exposure to greywater that is airborne from irrigation sprinklers (Schoen et al. (2017); Blanky et al. (2017); Busgang et al. (2018)) or toilet flushing (Shi et al. (2018)). While these models have many exposure parameters such as physical transport and exposure distance and time, the most sensitive parameter is most often the number of pathogens being consumed or inhaled, and therefore the number of pathogens in the water source. Therefore, the quality of the recycled greywater and the type and thoroughness of treatment are all critical in minimizing risk.

Quantitative analysis is also like those applied to previous discussed non-traditional water sources. The primary metrics for modeling are cost, energy requirements, and the water supply-and-demand relationship. Studies have quantified the requirements and trade-offs of greywater reuse against demand for basic water use activities in households (Antonopoulou et al. (2013); Ghunmi et al. (2008)), airports (do Couto et al. (2013)), and schools (Alsulaili and Hamoda (2015); Godfrey et al. (2009)). Results demonstrate the benefits of utilizing greywater production to alleviate some domestic water demands to provide both water and financial savings. As mentioned previously, LCA and life cycle cost analysis (LCC) is another approach to understanding the investment, annual costs and impacts of implementing and maintaining designed systems. LCC follows the same basic principles and framework as LCA but with a focus on compiling and quantifying product life costs from "cradle to grave," (Asiedu and Gu (1998)). Leong et al., (2019) used LCA and LCC on both greywater recycling and rainwater harvesting systems, varying the water and electricity tariffs and installation costs for analyzing financial viability (Leong et al. (2019)). Software such as Impact 2002+, CMI 2001, and TRACI are often utilized for these efforts. One key difference between LCA and LCC is that LCC expressed results strictly in monetary terms, and the price (data point) of a given product or process serves as a measure for all aggregated upstream costs. Thus, upstream data is not as crucial, given the cost data across a process is acquired. As with LCA, the more detailed a design or treatment train is, the more accurate the LCA or LCC will be. Cost curves for treatment unit processes, chemical additions, and sizing are well-published in literature and can be combined to illustrate an entire treatment train.

Economic analysis is valuable for quantifying the cost of investment in a greywater reuse system. This is an important metric for stakeholders and investors particularly in the urban sector, such as for multi-story residential buildings where the systems cover many units and residents and would therefore be more expensive due to larger flows and distribution needs. Friedler & Hadari (2006) performed a cost-benefit analysis on such a scenario with estimations for capital investment and operation and maintenance costs, as well as annual savings (or benefit) of reusing greywater to reduce water demand (Friedler and Hadari (2006)). Their model found that a rotating biological contactor proved to be more economical than a membrane bioreactor system, becoming economically feasible for a building of 28 or more stories, versus the membrane system requiring 37 or more stories, with economy of scale as the building size increased. This type of economic analysis is typical for estimating costs and potential savings of water supply systems and depends on capacity, energy requirements, treatment train and process specifications such as chemical additions, and local subsidies, incentives, and interest rates. Similar studies were conducted for systems to be implemented in schools in India (Godfrey et al. (2009)) and Chile (Rodríguez et al. (2020)). While costs and benefits differed by location and system, one of the takeaways by Rodríguez et al. (2020) was that socioeconomic factors, feelings of improved quality of life, and a better understanding of societal roles should be considered in such studies and approaches when quantifying the impacts and decision-making around sustainability, water savings, and ecological systems (Rodríguez et al. (2020)). Cost-benefit analysis, LCA, and LCC are powerful approaches for estimating and quantifying financial viability but should not be the only consideration for policy- and decision-making around these non-traditional water resources.

2.4 Desalinated water

Desalinated water is brackish water or seawater from which the dissolved minerals, salts, and other contaminants are removed by purification processes. Brackish water is water with more salt than freshwater, but less than seawater. These waters are found where saltwater and freshwater mix, such as estuaries or in some groundwater aquifers. Typical salinity for seawater is around 35,000 ppm but can range between 30,000 and 50,000 ppm (Boerlage (2012)). Brackish water salinity covers a wider range, of around 1,000 ppm to 30,000 ppm (Cooley et al. (2019)). The most common processes for seawater and brackish water desalination are multi-effect distillation (MED), multi-stage flash distillation (MSF), or reverse osmosis membrane desalination. The distillation methods rapidly boil the brackish/seawater multiple times to collect the evaporated freshwater and separate the brine/waste stream. Reverse osmosis (RO) is a process that utilizes high pressure to move the water molecules cross a semi-permeable membrane, leaving the salts and other impurities behind the membrane as brine (Ghernaout et al. (2020)). These processes are typically designed to produce purified potable water, thus brackish water and seawater are non-traditional water sources for drinking water supplies.

The use of seawater and brackish water as non-traditional sources of potable water has become an increasingly attractive and viable long-term solution for water scarcity, particularly in semi-arid and coastal regions. Over the past 30 years, significant advances have been made in seawater desalination, including a 2-fold reduction in energy requirements for seawater reverse osmosis (SWRO). SWRO technology now accounts for the majority of desalinated water production worldwide, at 69% (Jones et al. (2019)). The state-of-the-art for SWRO plant installation includes three major engineering processes: pretreatment, reverse osmosis, and post-treatment. For SWRO, pretreatment prioritizes solids removal and pH adjustment, with chemical addition to prevent membrane scaling and fouling. Conventional pretreatment processes therefore may include screening, clarification through coagulation and flocculation, filtration, and chemical addition (Jamaly et al. (2014)). The RO process includes forcing pretreated seawater through a semipermeable SWRO membrane under pressure in a continuous flow condition to remove salts, the major constituent of concern for seawater, by rejecting ions (Greenlee et al. (2009)). Post-treatment is then used to meet drinking water requirements, which generally includes hardness adjustments to prevent pipe corrosion and disinfection (Fritzmann et al. (2007)). A typical SWRO treatment train is illustrated in Figure 2.6. Seawater desalination in general recovers approximately 50% of inflow as freshwater, discharging the other 50% with twice the salinity of seawater as reject brine. As brackish water salinity is much lower than seawater, the recovery is higher, up to 75-85% (Blanco-Marigorta et al. (2017)). As mentioned, there are high capital costs associated with desalination, and the substantial energy and brine discharge requirements generate high operational costs as well.

Beyond the energy requirement for RO operation, brine management is also a critical component of the seawater desalination process (Jones et al. (2019)). Brine is usually discharged back into the ocean as it is the most common and least expensive option. However, brine

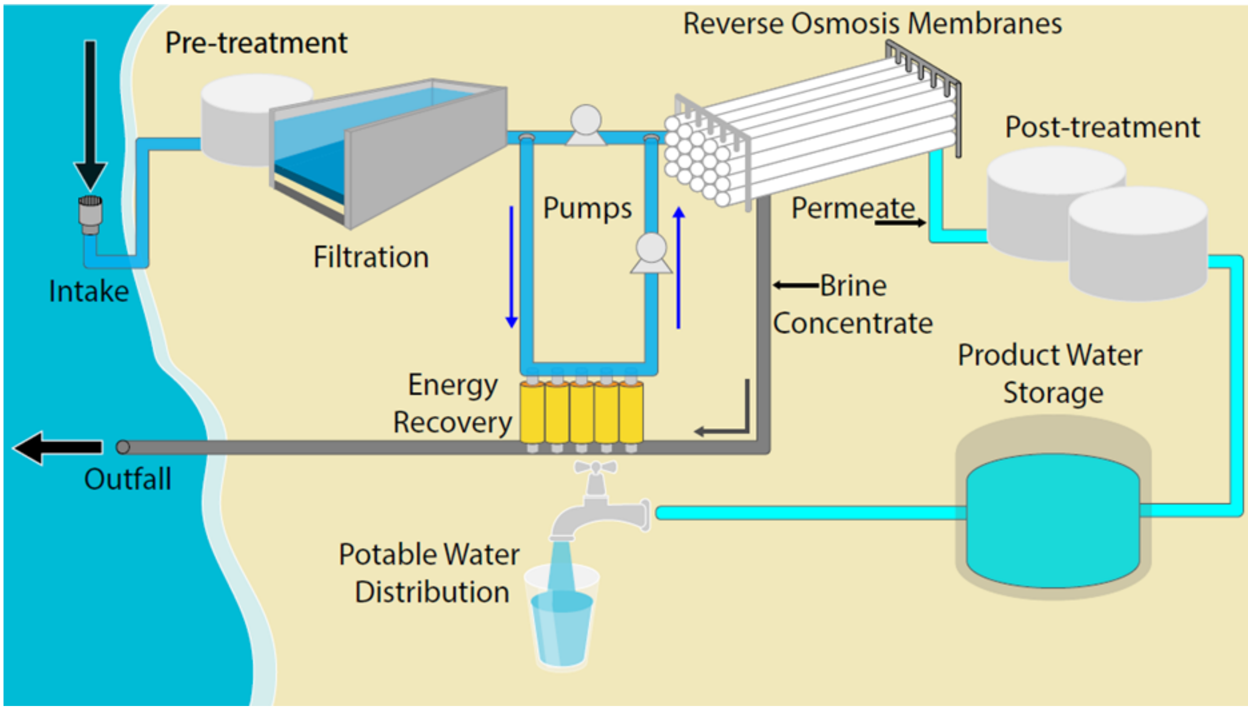

Figure 2.6: A typical treatment train (as outlined and illustrated by Kim et al. (2019)) for SWRO including offshore intake and brine discharge.

discharge raises concerns of impacts to marine life due to salinity, toxic substances, and temperature (in particular, when using distillation versus membrane desalination) (Ahmad and Baddour (2014)). These impacts are known and have been measured by several studies on various forms of marine life (Einav (2002); Gacia et al. (2007); Mabrook (1994)). In addition to a salinity and temperature differences, brine can also include chemicals from antiscalants, coagulants, and even heavy metals because of corrosion (Abdul-Wahab and Al-Weshahi (2009)). Inland desalination facilities have the added challenge of brine management without the means of ocean discharge. Common methods of inland brine management in the United States include evaporation ponds, zero liquid discharge systems involving evaporators, crystallizers, and spray dryers, and deep well injection (Mavukkandy et al. (2019); Mohamed et al. (2020)). Some methods and approaches are beginning to utilize recovery of salts to offset the high total costs of desalination, in which brine disposal could account for 5-33% (Giwa et al. (2017); Mohamed et al. (2005)). Cost, environmental, and regulatory concerns are major challenges for brine management.

Since desalination provides potable water to supplement the traditional supplies, acute illnesses such as microbial infections are not as much of an issue. Ultrafiltration membranes and RO membranes with pore sizes down to 0.0001 micron have been shown to significantly remove pathogens, and it is suspected that even viruses are significantly reduced due to adsorption onto particles (Cotruvo (2005)). In 2011, the World Health Organization issued a report on desalinated water health. Some of the major points were the recommendation for virus inactivation and disinfection after primary (RO membrane) treatment, and the challenge of maintaining microbial water quality during storage and distribution. Neither of these are unique to desalination and are general challenges for treating and delivering potable drinking water in a traditional centralized manner (WHO (2011)).

2.4.1 Water quality and quantity estimation for desalinated water

As noted in the previous section, cost and energy requirements are the main concern with desalination technologies to overcome in its adoption, so modeling efforts tend to focus on these areas. Techno-economic assessment (TEA) is a widely utilized method for quantifying the costs of a project, with the aim of minimizing costs and comparing across designs. In general, TEA is used to identify the areas or pathways for cost reduction and cost-effective implementation, the previously described LCC method quantifies cost impacts for all phases of its product life (Giacomella (2021)). TEA is valuable for desalination, as it is an energy and cost intensive process, and the assessment can compare between treatment methods (e.g., membrane, thermal). Additionally, TEA can assess the use of renewable energy sources (e.g., wind power) or co-location of desalination with power plants, for reducing cost (Loutatidou et al. (2017)). The latter reduces the overall cost, since the warmer source seawater collected from power plant discharge usually requires less energy for membrane separation than using ambient temperature seawater (Voutchkov (2018); Mezher et al. (2011)).

The most prevalent modeling software for cost and energy of desalination is The International Atomic Energy Agency's Desalination Economic Evaluation Program (IAEA DEEP). The DEEP model can be utilized for different configurations and power supplies for desalination processes and has been updated regularly since its creation (Kavvadias and Khamis (2010)). Another method of understanding cost and energy requirements for different desalination technologies used the cost database approach based on collating and correlating data from over 300 desalination plants worldwide (Wittholz et al. (2008)). More recently, the National Alliance for Water Innovation (NAWI) created The Water Technoeconomic Assessment Pipe-Parity Platform (WaterTAP3). This modeling tool can be used for user-created processes and treatment train configurations, including for seawater and brackish water desalination, to assess techno-economics of different options (Miara et al. (2021a)). Additionally, simulation models have been used to better understand and optimize specific processes, such as the RO process. Oh et al., (2008) simulated RO membrane performance based on solution-diffusion and fouling mechanisms to model permeate flux and recovery (Oh et al. (2009)). Models such as these have been valuable in improving desalination performance over the last few decades. A recent TEA study by Quon et al., (2021) conducted a baseline cost and energy analysis on several SWRO desalination plants and found that economy of scale plays a significant role in SWRO, with levelized cost of around U.S. $\$1 - 1.35/\text{m}^3$. In reality, actual costs are highly variable, made apparent by the $\$1.61/\text{m}^3$ cost of SWRO in Carlsbad, CA, USA versus the $\$0.53/\text{m}^3$ cost of SWRO in Ashkelon, Israel, despite the two facilities being nearly identical in design. With the aim of identifying cost-saving opportunities and discrepancies across large-scale SWRO facilities, this study suggested that future cost savings are most dependent on local socioeconomic factors and consistent plant operation; large RO seawater desalination plants with state-of-the-art technology have similar energy costs while total capital and operational costs vary (Quon et al. (2021b)).

The recognition of the impact of local factors to the cost and adoption of water technologies and supplies is of great significance for desalination. Economic analyses often lack the ability to properly capture externalities and local factors related to construction, permitting, financing, market regulations, and government subsidies, which have been identified as challenges of note in California (Cooley and Ajami (2012)). The risks associated with these areas and the economic feasibility of weighing them against the predicted costs of the facility (modeled through TEA, for example) are lacking based on the current state of knowledge and demonstrations (Ziolkowska (2015)). Two recent studies conducted technoeconomic analyses and produced results in line with these earlier conclusions. With the aim of identifying cost-saving opportunities and discrepancies across large-scale SWRO facilities, the study by Quon et al. (2021b) suggested that future cost savings are most dependent on local factors and consistent plant operation; large RO seawater desalination plants with state-of-the-art technology have similar energy costs while total capital and operational costs vary. A similar conclusion was drawn by a TEA on thermal desalination by Zheng and Hatzell (2020), who stated that "we cannot ignore many other factors that can affect the siting selection, such as local government subsidies, transportation fee of facilities, local land prices." In addition, the sociopolitical challenges of desalinating waters has been reviewed and explored (Ibrahim et al. (2021)). For example, studies have highlighted the disparities and vulnerabilities of border areas regarding water rights, namely at the Mexico-USA border (McEvoy and Wilder (2012)) and between Israel and Jordan (Aviram et al. (2014)). On one hand, increased water security shared between countries and the collaborative process is achievable (the case for Jordan and Israel) while on the other it may increase tensions (Mexico and USA). Such factors that ultimately affect the timelines and amenability of desalination are difficult to include from a modelling and design perspective and require further qualitative study and location-specific dives into how they inevitably impact the costs and benefits of including desalination in water portfolios.

In addition to overall performance, energy, and cost requirements, the rejection and fate of low-rejection ions is of increasing concern and can be estimated and accounted for in SWRO design. Boron, in the form of boric acid, exists in seawater with an average concentration of

4.6 mg/L, and experiences low rejection by RO membranes (Argust (1998)). This is particularly problematic as boron was identified as the cause of poisoned crops when desalinated waters are used for irrigation (Fritzmann et al. (2007)), which is a common and growing practice in Spain (Garcia et al. (2011); Zarzo et al. (2013)) and Israel (Yermiyahu et al. (2007); Avni et al. (2013)). Although in practice, multiple RO passes can complete boron removal as is the case in Ashkelon, Israel (Sauvet-Goichon (2007)), mathematical models can be utilized to better understand boron removal on a case-by-case basis. Models have been implemented based on solution-diffusion across membrane layers (Taniguchi et al. (2001)), irreversible thermodynamics (Mane et al. (2009)), and electrostatic and steric-hindrance (Wang et al. (1995)), which all vary in boron removal based on pH, boron concentration, and operating temperature and pressure (Tu et al. (2010)). Finally, determining boron removal is designed based on regulations and desalinated water concentration limits, which vary around the world from 0.5-5 mg/L. The ion removal and membrane rejection can be modeled to determine the size of membrane required and/or the number of passes needed to achieve desired product water chemical concentrations and purity.

2.5 Condensate capture and atmospheric water harvesting

Condensate capture and atmospheric water harvesting (AWH) are additional methods to provide non-traditional water. Captured condensate is the collection of condensate water generally from air conditioning cooling coils, rather than traditionally draining the condensate to sewer lines. Much like RHRW and SWH, it therefore relies on diverting and storing a previously wasted source of freshwater, making it a generally untapped water source, particularly in hot, humid regions. Captured condensate can be used for non-potable uses such as toilet flushing, irrigation, and cooling tower make up water (Algarni et al. (2018)). Atmospheric water harvesting (AWH) is the use of a device to extract water vapor directly from the air by various methods (Zhou et al. (2020)), namely condensation technology (Tu and Hwang (2020)), adsorption based technology (Ejeian and Wang (2021)), and cloud seeding/fog collection (Tu et al. (2018)).

Condensation technology for AWH requires a power source for cooling in order to condense the air to vapor. Adsorption technology can be designed to utilize day and night cycles, ambient temperatures, and solar heat for capturing and condensing vapor. Therefore, it is less energy intensive, but the yield of water harvested is less than with condensation technology (Tu and Hwang (2020); Ejeian and Wang (2021)). Cloud seeding is a form of weather modification to induce and collect rain, but only where water abundant clouds have gathered, thus it is difficult to perform in a routine and predictable manner (Ejeian and Wang (2021)). Fog collection is simply the capture of droplets on mesh-like material perpendicular to fog and wind. It has demonstrated a water production ability up to 3-7 kg day^{-1} m^{-2} but is best utilized in high elevations where fog and wind are regular (Klemm et al. (2012); Montecinos et al. (2018)).

2.5.1 Water quality and quantity estimation for captured condensate and AWH

Both sources of water are promising to alleviate water stresses on traditional sources, but research and efforts for their quantitative modelling and design are fewer than for the other sources outlined earlier. Currently, modeling has been conducted focusing on estimating theoretical yield (water quantity) of condensate based on thermodynamic principles and climate conditions (Al-Farayedhi et al. (2014); Rao et al. (2022); Hassan and Bakry (2013)). Regions around the world identified as having high potential for implementing condensate collection are the Arabian Peninsula, Sub-Saharan Africa, South Asia, and the Southeastern

United States (Al-Farayedhi et al. (2014); Loveless et al. (2013); Guz (2005)). Hassan and Bakry (2013) found that for 1 ton of refrigerant, the condensate recovery for a year of operation and typical weather conditions was highest in Singapore with 35.33 m^3 followed by 30.69 m^3 in Kuala Lumpur, Malaysia (Hassan and Bakry (2013)). Captured condensate as an onsite water supply offsets conventional water demand, similar to onsite greywater reuse, thus reducing the overall demand and footprint associated with potable water treatment. Khan (2013) estimated a reduction of 0.54 kg CO_2eq per kWh used for pumping of the conventional water supply associated with the implementation of captured condensate in residential buildings in Dubai, UAE (Khan and Al-Zubaidy (2013)). Conversely, atmospheric water harvesting was estimated to have a reduction of 0.3-0.35 kg CO_2eq per kWh based on the average footprints of traditional water sources in the United States and Middle East (Moghimi et al. (2021)). For water quality modeling, Loveless et al. (2012) conducted water quality testing on captured condensate systems throughout Saudi Arabia and found high quality, with all samples under the U.S. EPA recommended quality values (Loveless et al. (2013)). Based on their climate model and water quality findings, the authors suggested that industrial application of captured condensate could lead to cost savings and reduced impact on operations which already require highly pure water, and simple post-treatment methods could make the collected water drinkable.

2.6 Summary of key areas for future research

In this paper, several non-traditional water sources were described and compared, with a focus on the quantitative methods in place for estimating their respective quantity and quality metrics for implementation and water management. Computer modeling and analytical tools serve to pinpoint and predict metrics regarding capacity, cost, energy, microbial quality, and health risk. As each non-traditional water source varies in water quality, operation, size, and treatment level, there are still key areas that require further research to improve their use and management. This applies at all levels of society and water management, from the household scale of water use to the planning of government policy and regulatory measures. Below are the key areas identified in this study for each of the water sources explored.

- Rooftop harvested rainwater: RHRW has highly variable water quality and microbial concerns, therefore a clear and more uniform policy on onsite maintenance and upkeep for water quality concerns is needed in areas where RHRW is implemented or required.
- Stormwater harvesting: Due to the nature of stormwater and the variable effects of weather on its abundance and water quality, there are health concerns with utilizing it for reuse purposes. Further research is recommended in understanding the impacts of weather on stormwater quality and on the health risks of human exposure and consumption when using it as a non-potable water source.
- Reclaimed municipal wastewater: In general, reclaimed municipal wastewater is used for non-potable purposes, but recent advancements have demonstrated that direct potable reuse is possible. However, it is not readily accepted and there is a lack of policy for its implementation and regulation. Therefore, more research on potable reuse technologies in terms of cost and treatment capabilities is necessary, particularly in comparing RO with alternative treatment technologies. The health concerns with non-potable reuse due to aerosolization and the uncertainty around viral pathogens must be explored and compared to further develop an understanding of monitoring for pathogens in wastewater reuse. In addition, the occurence of antibiotic resistant bacteria in wastewater treatment plants has created concerns of their spread, prevalence, and subsequent effects on human health through exposure. A lack of sufficient ARB and ARG prevalence data in reclaimed wastewater calls for future efforts to better characterize these concentrations, information on antibiotics, and a reassessment of treatment criteria and regulation for possible associated health risks. These health

concerns apply to the variable water quality and pathogen levels in raw wastewater and in treated wastewater effluent that is used as a non-traditional and non-potable water source.

- Greywater reuse: The possible health risks and concerns of onsite greywater recycling, such as for toilet flushing and irrigation, continue to pose a hurdle for its wider spread implementation.

- Desalination: High costs and concerns of how to properly manage brine waste hinder its development and acceptance in the United States. More research is recommended on the origins of cost discrepancies in a comparative manner across desalination facilities around the world, including local costs and legal requirements. The effects of offshore brine discharge must continue to be studied, as well as other methods of brine waste processing and handling for inland desalination facilities.

- Condensate capture and AWH: These methods can be used to reduce the dependence on traditional, centralized water sources, but a better understanding of the quality requirements is suggested. The benefits are region-specific due to the pivotal impacts of temperature and weather, which should be well understood before any design and implementation. The design cost of post-treatment for potable water use is a key requirement and must be considered.

Chapter 3

Pipe Parity Analysis of Seawater Desalination in the United States: Exploring Costs, Energy, and Reliability via Case Studies and Scenarios of Emerging Technology

The contents of this chapter appear in the journal Environmental Science & Technology: Engineering Quon et al. (2021b).

3.1 Introduction

The demands for drinking water rise with population growth. Climate change-induced variables also further exacerbate water scarcities and water pollution in many regions around the world, including the Southwestern United States (Schwabe et al. (2020)). The use of seawater as a nontraditional source of potable water could provide a reliable long-term solution for water scarcity. Over the past 30 years, significant advancements have been made in seawater desalination technologies, including a 2-fold reduction in energy requirements for seawater-grade reverse osmosis (SWRO) membrane desalination (Voutchkov (2018)). SWRO technology has replaced thermal desalination as the leading desalination technology for drinking water production from oceans. The world has seen a boom of SWRO plants in the Middle East, Australia, and the Mediterranean regions since the late 1990s (Nair and Kumar (2013)). Compared with other alternative sources of water, seawater is an unlimited and unrestricted source of water. Therefore, large scale SWRO plants are a potential solution for providing a drought-proof source of water to coastal cities around the world. In severely water-stressed regions, such as Israel and the coastal areas of the Southwestern U.S., seawater desalination could be a reliable, climate invariable, and long-term solution to the escalating water crisis.

Globally, there are over 5000 operational seawater desalination facilities with an estimated capacity of almost 50 million m^3/day. Most plants use SWRO technology, followed by multistage flash (MSF) and multieffect distillation (MED) (Jones et al. (2019)). The United States has a dozen operational municipal-scale seawater desalination plants and has several others in the planning phase (see Table A.2 for existing and planned seawater desalination plants in the U.S.). In addition, small package SWRO plants are also common among island communities in the U.S., including the U.S. Virgin Islands and San Juan County in the state of Washington (Seven Seas Water (2023); Mayo (2009)). Nearly all U.S. seawater desalination plants for municipal water production are based on SWRO membrane technology.

The state-of-the-art SWRO plant installations include three major engineering processes: pretreatment, reverse osmosis, and post-treatment. For SWRO, pretreatment prioritizes solids removal and pH adjustment, with chemical addition to prevent membrane scaling and fouling. Conventional pretreatment processes therefore may include screening, clarification through coagulation and flocculation, filtration, and chemical addition (Jamaly et al. (2014)). The RO process includes forcing pretreated seawater through a semipermeable SWRO membrane under pressure in a continuous flow condition to remove salts, the major constituent of concern for seawater, by rejecting ions (Greenlee et al. (2009)).Post-treatment is then used to meet drinking water requirements, which generally includes hardness adjustments to prevent pipe corrosion and disinfection (Fritzmann et al. (2007)). Seawater desalination in general recovers approximately 50% of inflow as freshwater, discharging the other 50% with twice the salinity of seawater as reject brine. Therefore, brine management is also an important process for desalination (Jones et al. (2019); Mavukkandy et al. (2019)).

In comparison with traditional sources of water, seawater desalination is a more expensive form of drinking water treatment, with over one-third of overall cost represented by energy consumption (Lienhard et al. (2016)). While the costs of desalination processes have fallen significantly over the past three decades, seawater desalination costs still remain higher than conventional drinking water treatment methods (Zhou and Tol (2005); Bhojwani et al. (2019)). The costs of desalinated water range from \$1.53 to \$1.90/m^3, with a median cost of \$1.57 for larger facilities with at least 12 million gallon capacity per day. This price is higher for smaller facilities. The cost of the desalinated water is approximately 5 times higher than traditional surface water, which has an estimated average cost at \$0.27/m^3 (Cooley et al. (2019)). However, the desalinated seawater price varies significantly around the world. For example, the Ashkelon Desalination Plant in Israel produces desalinated seawater at less than \$0.6/m^3, while the Carlsbad Desalination Plant in Southern California produces desalinated water at a price of \$1.6/m^3 (Water Technology (1970)). Understanding the origins of the cost discrepancies between SWRO and traditional water supplies and between SWRO facilities around the world could guide the decision-making of desalination in the future.

Water resource planning depends on supply options, sectoral demand, and exposure to variable hydrological conditions. The water industry in the U.S. is highly fragmented, with nearly 150,000 entities registered with the EPA Safe Drinking Water Information System (SDWIS) as drinking water providers (Environmental Protection Agency (2021)).Therefore, the decision to adopt seawater desalination is region specific and dependent on local governance and conditions.The U.S. Department of Energy (DOE) recently introduced a concept of "pipe parity" metrics for evaluation of the adoptability of specific sources of water. Pipe parity is defined as technology solutions for treating and using nontraditional water that are competitive with conventional water sources for specific end-use applications. The framework comprises both quantitative and qualitative metrics (RFP #: NAWI-2-2021). A water source is considered to have achieved pipe parity when a decision-making body considers it to be the next-best option of water (Miara et al. (2021b)). The pipe parity concept embraces not just costs but metrics such as system robustness, reliability, and long-term sustainability that drive decision-making for investments in technology.

Building on the state of knowledge of existing desalination facilities, this study aims to assess the conditions required for seawater desalination to reach pipe parity and aid in decision-making for adopting it as a water source. This study focused on U.S. SWRO desalination plants with a production capacity greater than three million gallons per day (MGD) (or around 11,000 m^3/y) because large scale seawater desalination plants have the largest potential contribution to overall drinking water supply needs. We carried out a techno-economic analysis (TEA) of three U.S. SWRO plants in comparison with a SWRO plant in Israel to assess the current baseline for SWRO operation. "What-if" scenarios were incorporated in TEA to determine the cost savings associated with potential changes in SWRO operation. We examined how water reliability factors can influence the decision to opt for seawater desalination with a breakeven analysis. The role that SWRO may play in the future water supply portfolio was estimated based on the local water supply shortage, conservation efforts, and projected future demand in the U.S.

3.2 Methods

3.2.1 Data curation for case study facilities

Internet searches and literature reviews were first carried out to identify desalination facilities through state, regional, and private industrial Web sites and in research journals. Additional information on seawater desalination facilities was obtained through electronic communication, phone and video interviews, and discussions with technical staff and managers of regional water management departments, private industries, technical consultants, academic experts, and engineers focused on seawater desalination. Information was also obtained through virtual tours and field visits of select facilities. Finally, case studies were selected from a compiled list of facilities (Table A.2) based on operational status, location, history, capacity, treatment train, and unique aspects of the facility. A total of three U.S. facilities and one in Israel were chosen based on the capacity, geographic location, availability of detailed engineering processes, and data on cost and energy consumption from the plants. The three U.S. facilities are located in Carlsbad, CA; Tampa Bay, FL; and Santa Barbara, CA. The fourth plant located in Ashkelon, Israel, was chosen to represent a large SWRO facility outside of the U.S. The Ashkelon plant is one of the most well-studied SWRO plants that produces desalinated water at a price that is much lower than most SWRO facilities. Located in the aridic Middle East, Ashkelon has been in successful operation for over a decade. Technologically, it is nearly identical in design to the Carlsbad facility, and both facilities are operated by the same engineering firm (IDE, Inc.). By comparing this facility with the large SWRO facilities in the U.S., we hope to identify the gaps and opportunities to lower the energy and cost of seawater desalination in the U.S.

Data collected from each facility are summarized in Table 3.1 below. Additional information obtained from different sources are presented in Appendix Table A.3. Demographic and water buyer information are in Table A.4.

Table 3.1: Summary of case study SWRO facilities.

	Facility name			
	Ashkelon	**Claude "Bud" Lewis**	**Tampa Bay**	**Charles E. Meyer**
Location (country/state)	Israel	Carlsbad, CA	Tampa, FL	Santa Barbara, CA
Initial construction (expansion / redesign)	2005 (2010)	2015	2003 (2007)	1992 (2017)
Current capacity (MGD / m^3/year)	86 / 400,000	50 / 190,000	25 / 95,000	3 / 11,000
Operation	Year-round	Year-round	9 mo/y	Year-round
Total capital cost[a]	$561M	$1B	$197M	$106M
Feed salinity (ppm)	40,233	34,500	32,000	34,500
Water cost ($/$m^3$)[b]	0.53	1.61	0.66	1.08
No. of RO units	32	14	7	12
RO system	Multistage; multipass	Multistage; multipass	Partial two-pass	Single pass
Energy recovery	Pressure exchanger and Pelton turbine	Pressure exchanger	Turbo charger	Pressure exchanger
Power consumption (kWh/m^3)	3.8	3.3	3.0	3.6

[a]Total capital cost is retrieved from DesalData (GWI) for Ashkelon based on both the original and expansion costs and inflated to 2020 cost. For the other three facilities, values for initial cost and expansion costs were obtained from literature reports, which include adjustment to the current value. See Table A.3 for more information and literature sources.
[b]Water cost is retrieved from DesalData, (GWI) which includes CAPEX, OPEX, and financing costs. It is consistent with the levelized cost of water (LCOW) estimated by WaterTAP3 but differs in the use of water price since water price may include a subsidy.

3.2.2 Baseline cost and energy analysis using WaterTAP3

To dissect the capital cost, energy consumption, and LCOW at SWRO facilities in detail, the Water Technoeconomic Assessment Pipe-Parity Platform (WaterTAP3) developed by the National Alliance for Water Innovation (NAWI) was used to analyze data collected from each facility (Miara et al. (2021b)). Model input parameters include treatment train processes, daily treatment capacity, intake water salinity, chemical additions, local electricity tariffs, and annual operational days. The details of the WaterTAP3 methods and assumptions for inputs are presented in the Appendix Table A.3 and Tables A.5 - A.9. The current model provided four key outputs used in this analysis: capital cost (CAPEX), energy consumption, annual operation and maintenance cost (OPEX), and LCOW. The CAPEX estimate includes material cost, land acquisition, and labor and other expenses for construction and installation. The cost of permit acquisition was not included in the model due to limited data. The electricity estimates include the net consumption for intake and in-plant pumping, pressurized filtration and backwash in pretreatment, high pressure pumping to drive RO at constant permeation rate given the influent water salinity, single-pass, multipass, or multistage RO, energy recovery system, and post-treatment disinfection. The values were estimated based on treatment capacity and process configurations. OPEX includes electricity costs plus chemical costs (e.g., coagulant, disinfectant), operator and management labor, monitoring, consulting fees, and replacement and repair (R&R) of infrastructure costs per year during normal operation. Again, the values were determined by plant size, operation processes, and the state average electricity tariffs. In anticipation of RO membrane fouling, which causes a decline in water productivity over time, a membrane replacement rate was included in the OPEX estimates Kumar et al. (2006). Finally, the LCOW was computed using the outputs from each of the submodels by the equation:

$$\text{LCOW} = \frac{(\text{CAPEX} \times \text{CRF}) + \text{OPEX}}{\text{average annual yield}} \tag{3.1}$$

where the capital recovery factor $CRF = \frac{r(1+r)^n}{(1+r)^n-1}$, n is the plant service life (in years), and r is the weighted average cost of capital (WACC) or discount rate. The financing cost of a project is also included in the CRF. Therefore, LCOW includes all annual costs and financing normalized across a unit volume of water produced over the plant service life. A sensitivity analysis was conducted for each case study by varying several input parameters and evaluating changes in the results.

3.2.3 Scenario analysis for improvements

In addition to the baseline case modeled for the four facilities, a "what-if" scenario was applied to estimate the potential cost savings through increasingly automated technology and artificial intelligence (AI)-driven treatment processes in desalination. Increased implementation of automation and AI technology has been identified as one key research opportunity by NAWI and in the literature for enabling improved operations, process controls, and management (Al Aani et al. (2019)). We hypothesized that AI and automation of SWRO processes would result in cost reductions over time, specifically in fixed labor costs. To quantify potential cost savings, the base case fixed labor cost was reduced by 10–50% as hypothetical scenarios with implementation of such technology. These assumptions were consistent with other studies and estimates (Dixon (2020)), although the actual rate of cost reduction likely varies on a case-by-case basis. The scenario resulted in estimates of how much a facility could spend on automation systems (e.g., sensors, controls, software, training) without increasing their overall costs, assuming a given reduction in labor costs. The "what-if" outcomes were used as an exemplar to better understand a possible future path for SWRO to reduce costs and evaluate options to approach pipe parity.

3.2.4 Breakeven curve for SWRO adoption

In evaluation of pipe parity, it is necessary to assess the difference between the cost of SWRO (C_{SWRO}) and the cost of a less expensive water option (C). This margin is referred to as the SWRO cost premium. For SWRO, its primary advantage over traditional water resources is high reliability under drought; this means that a SWRO plant can be expected to provide water supply at full designed capacity under all conditions over the lifetime of the facility. Although there has not been a value placed on reliability and long-term sustainability of a water resource, we equated the water conservation cost to meet the water supply shortage as the value of a reliable water resource. We used R to represent a factor of water shortage-induced supply reduction (unitless) per year, and MC_{cons} (the marginal cost of water conservation) to represent cost per unit of water (\$/m^3) conserved under a given drought scenario (both R and MC_{cons} estimations are given below in the case study for the State of California). Thus, the total water conservation cost is estimated by $R \times MC_{\text{cons}}$. Therefore, SWRO is cost-effective or is considered to have reached pipe parity if $C_{\text{SWRO}} - C < R \times MC_{\text{cons}}$. The point when the SWRO premium ($C_{\text{SWRO}} - C$) equals $R \times MC_{\text{cons}}$ is the breakeven point. The conversion of water reliability, a nonmonetary factor, to monetary values contributes to pipe parity assessment, though it is not currently estimated in WaterTAP3.

The decision to develop the Carlsbad desalination plant in San Diego County, California, was used as a case study to illustrate incorporating the reliability factor of seawater desalination into pipe parity metrics. A breakeven curve based on the SWRO premium ($C_{\text{SWRO}} - C$) and $R \times MC_{\text{cons}}$ was developed. The cost premium of SWRO was estimated using the San Diego County Water Authority (SDCWA) contracted purchasing price for Carlsbad SWRO desalinated seawater at around \$2500/acre-foot [AF] (\$2/m^3) for 20 years of plant service life (Management (2021)), The average cost of the Metropolitan Water District water (C), the marginal water supply option for SDCWA for the next 20 years, was estimated at about \$1.40/m^3 based on the current cost and an escalation rate of approx. 4% per year (Metropolitan Water District of Southern California (2020)). Therefore, the cost premium of Carlsbad SWRO is \$0.60/m^3. In addition, an assumed value of \$0.40/m^3, which represents a hypothetical 10% future reduction of SWRO cost due to technological innovation over the same period of 20 years, was also used in establishing the second breakeven curve as a comparison.

We estimated the water supply reduction R in the entire State of California between 2001 and 2020 using U.S. Drought Monitor (USDM) data, which reports the percentage of California land area in one of five drought categories ($D_j, j = 0$ to 4), increasing in severity from D0 to D4 (Figure A.5) (NDMC (2021)). The USDM data were used to calculate an annual time series of the fraction of land area assigned to each drought category (w_{jy}). We assigned a water supply reduction factor (r_j) to each drought category, with $r_0 = 5\%$ and $r_4 = 50\%$. The statewide average water supply reduction factor R_y for each year was calculated as:

$$R_y = \sum_j r_j w_{jy} \tag{3.2}$$

The factor R_y represents the expected annual state-wide average reduction in raw water supply availability (surface water and groundwater), relative to a year with no drought, based on the fraction of land area assigned to each drought category.

As shown in Figure 3.7, R_y varies substantially from year to year. For use in the breakeven curve, we calculate the average annual reduction factor R over the 20-year data period as:

$$R = \frac{1}{N} \sum_y R_y \tag{3.3}$$

Here, $N = 20$ and the data spans the period $y = 2001$ to 2020.

The use of 20-year data to estimate R is to match the service life of a SWRO facility that was also used in the SWRO marginal cost and WaterTAP3. The water supply reduction for the entire state is used in the model because the system of large-scale water transfers used in California exposes most water districts to drought risk outside their local area. To validate the estimated R, we took a subportion of the data covering the 2012-2016 drought period to estimate a statewide average water supply reduction and compared the estimates with existing data for the period. The estimated R values agree with the observed data within 30% variability range, suggesting the validity of this approach over the study period. Additional details for R estimation are presented in the Appendix A, based on the observed relationship between drought conditions and water shortages (NDMC (2021); City of Santa Barbara PWD (2020); Lund et al. (2018)).

We assessed the MC_{cons} using a standard conservation supply curve (Stoft (1996); U.S. EPA (2016)). We defined four conservation cost categories—zero, low, moderate, and high—and estimated the magnitude of potential conservation in each category based on the associated conservation measures. The cost per unit of water conserved was estimated based on literature reports and summarized in Table A.3. The conservation supply curve is shown in Figure A.2. These values together with the SWRO premium were used to define the breakeven curve as a function of the shortage factor R.

3.2.5 Contribution of seawater desalination to water portfolios

The potential for expansion of seawater desalination in the future to address U.S. water needs depends on a combination of existing demand, growing demand (which include population growth and more, see (Shi et al. (2013)) for additional details), conservation potential, and projected water stress. We identified 40 coastal counties in 11 U.S. states that experience the highest level of future water stress according to a study conducted at the Columbia Water Institute (Shi et al. (2013)). Total municipal water demand for these counties was estimated based on USGS data (USGS (2020)). We incorporated categorical potential conservation metrics as low, medium, or high based on whether the state currently implements demand management policies (Alliance for Water Efficiency (2017)). Low conservation potential indicates that large conservation efforts are already in place, while the high conservation potential indicates that demand management policies have not been fully implemented in those states, therefore the high future potential for conservation. We eliminated the counties with estimated demand less than 3 MGD (11,000 m^3/d) from the analysis because of the relative high cost of small-scale seawater desalination plants in comparison with other alternative sources of water (i.e., wastewater recycle, brackish groundwater). The capacity of seawater desalination to meet future municipal drinking water demand was tabulated by individual state and the nation.

3.3 Results and discussion

3.3.1 Case study facilities

Each of the treatment trains for the case studies are shown in Figure 3.1. The greatest process similarities are between the Ashkelon and Carlsbad facilities, where multiple RO stages and passes are installed for boron removal and scaling reduction. Carlsbad has a lower chemical demand for pH adjustment and antiscalant dosing due to the lower salinity and boron concentration in the intake seawater from the Pacific Ocean. Dual-media filtration is used as pretreatment at Ashkelon, while trimedia filters are installed at Carlsbad, where they also serve as biofilters to remove excess organics from algal blooms. Despite the higher salinity in the intake water, the subsurface intake at Ashkelon provides water with a lower organic concentration and is less impacted by biofouling than at Carlsbad.

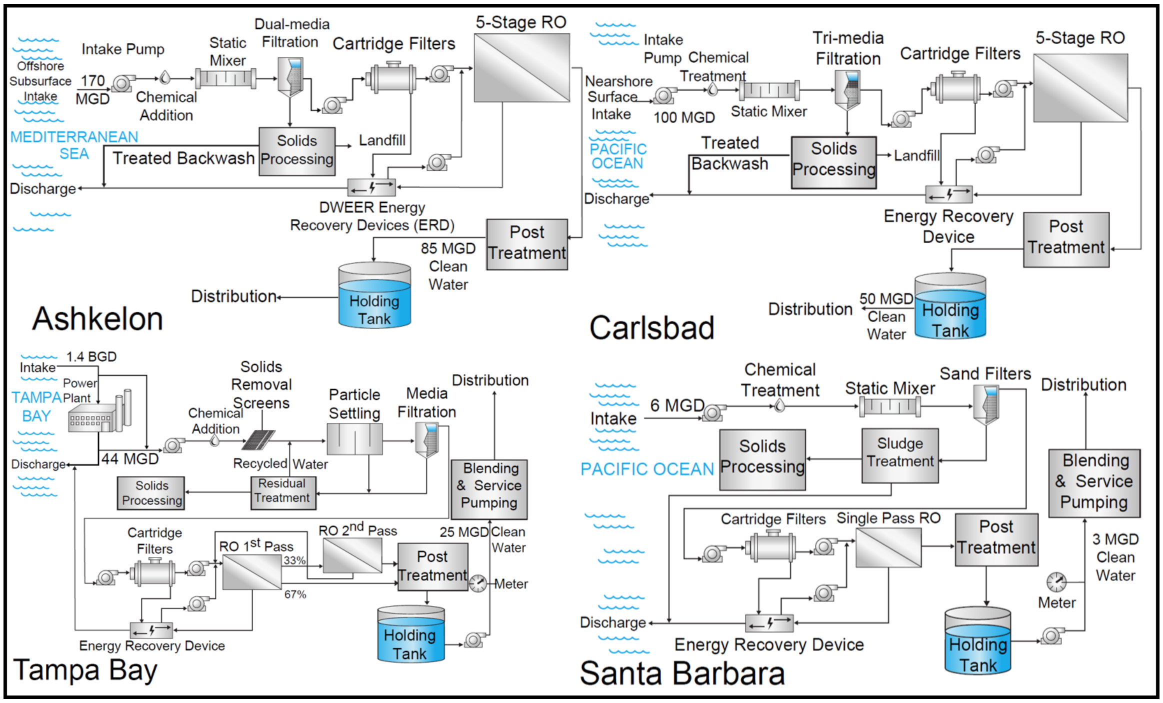

Figure 3.1: Process flow diagrams for the case study facilities.

Santa Barbara, the smallest plant at 3 MGD capacity, uses a near-shore surface intake and has a single-pass RO treatment with typical pre- (coagulation, flocculation, sand, and cartridge filtration) and post-treatment (chemical addition) processes. However, UV disinfection is added in Santa Barbara to treat the finished water before distribution. The Tampa Bay plant is collocated with Tampa Electric's (TECO) Big Bend Power Station and uses the warmer discharge from the power plant as intake water. A partial two-pass RO process is employed to blend the high purity second pass permeate with the first pass permeate. The split to the second pass RO varies slightly with the temperature and salinity of the intake. The brine discharge at Tampa Bay is mixed with the power plant discharge to reduce the salinity before final discharge into Tampa Bay.

3.3.2 Cost, energy, and LCOW

The WaterTAP3 cost and energy results are shown in Figure 3.2. The metrics are broken down categorically by major unit process in the treatment train modeled for each case study.

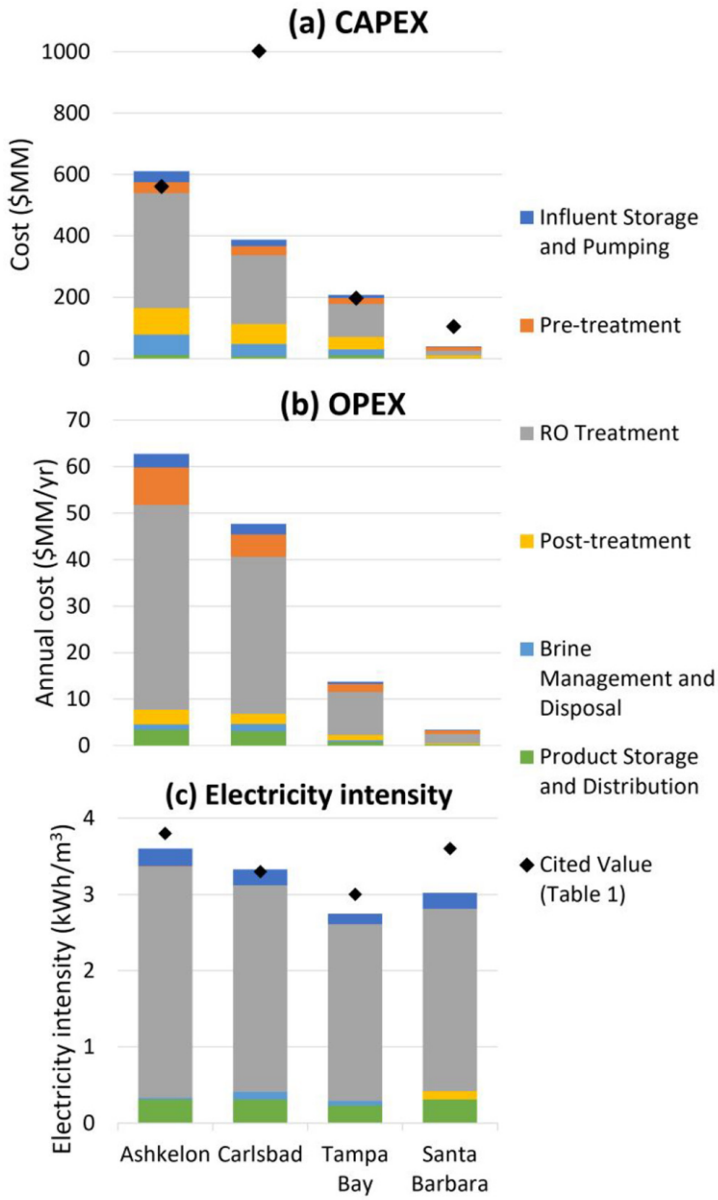

Figure 3.2: Comparison of WaterTAP3 results of capital investment (CAPEX, a), annual O&M cost (OPEX, b) adjusted to 2020 values, and electricity intensity per volume water production (c) across four case study plants based on the main treatment processes. Data from Table 3.1 are overlaid (black diamonds) for comparison of CAPEX and Energy intensity.

The CAPEX estimates were consistent with the expectations that they would increase with facility size (capacity), that the RO process would account for the most substantial portion of the capital costs, and that the relative proportion of each major process in the treatment trains would carry relatively consistent weight in CAPEX when compared across case studies. Currently, only fixed permitting costs are included in the WaterTAP3 model for all facilities (Pinto and Marques (2017)). The lack of differences in permitting costs in the WaterTAP3 model may account for the major discrepancy in CAPEX estimations in comparison with plant reported values (Figure 3.2a). WaterTAP3 overestimates the CAPEX for Ashkelon but significantly underestimates it for Carlsbad. This is illustrated in Figure 3.2a with the overlays of CAPEX values reported in the DesalData database or literature. DesalData estimates are based on lump sum values provided by facilities. Literature reports vary by the assumptions and source of data used and lack of uniformity across facilities. Only WaterTAP3 incorporates the detailed engineering process model and parameters collected for plant specific processes in generating the cost and energy values for each of the case study facilities. WaterTAP3 creates a uniformed baseline for different SWRO facilities. The overestimation of CAPEX for Carlsbad and Santa Barbara plants is attributed to the high permitting costs and extensive delays in the construction timeline due to California-specific policies for developing seawater desalination projects. The underestimation of this value at the Ashkelon plant is likely due to special governmental programs to support the development (Dayton (2019)). The discrepancies observed in this study emphasized the importance of factors other than technology that drive the cost of desalination.

The operation and maintenance (OPEX) costs are represented on an annual basis and included labor and fixed operational costs (such as energy and chemicals), also broken down by unit process (Figure 3.2b). As with CAPEX, OPEX scales with facility capacity and illustrates a similar breakdown by unit process for cost proportionality across case study plants. Again, the RO process is the largest contributor to cost. Across the case studies, post-treatment represented a larger portion of the CAPEX and pretreatment represented more of the OPEX. For SWRO, the modeled pre- and post-treatment OPEX costs are primarily in chemical additions, including coagulation in the pretreatment stage and lime softening in the post-treatment.

The WaterTAP3 results for electricity consumption per cubic meter of water production are comparable across all facilities (Figure 3.2c). As expected, the RO process is the highest consumer of electricity. Although there are differences in the RO configuration including single-pass, two-pass, and multistage and multi-pass layouts of RO arrays, the overall energy intensity for the RO process among different plants is not dramatically different (within 8% variability). One notable difference in the breakdown of energy intensity (Figure 3.2c) is the larger contribution of post-treatment to Santa Barbara's energy expenditure when compared with the other case studies. This is due to the unique inclusion of UV disinfection, a high-cost and energy intensive process, in the Santa Barbara plant before product water distribution, which is absent in the other treatment trains (Miklos et al. (2018)).

Breakdown of LCOW by unit processes and cost categories is shown in Figure 3.3. The model results showed that capital investments were the major portion of LCOW for small plants,while electricity costs are more significant in large plants (Figure 3.3b). Tampa Bay stands out with a relatively higher portion of allocated capital cost and lower electricity in comparison to the much smaller Santa Barbara plant (Figure 3.3b). This is due to the lower plant capacity utilization at Tampa Bay, which operates nine months of the year when desalinated water is needed and suggests that Tampa Bay behaves more like a smaller plant in terms of LCOW.

For the larger plants, the proportion of LCOW attributed to electricity is higher for Carlsbad than for Ashkelon (Figure 3.3b). The energy intensity is higher at Ashkelon than at Carlsbad (Figure 3.2c), indicating Carlsbad has higher electricity tariffs.

A comparison of LCOW results with the water cost reported by DesalData and other liter-

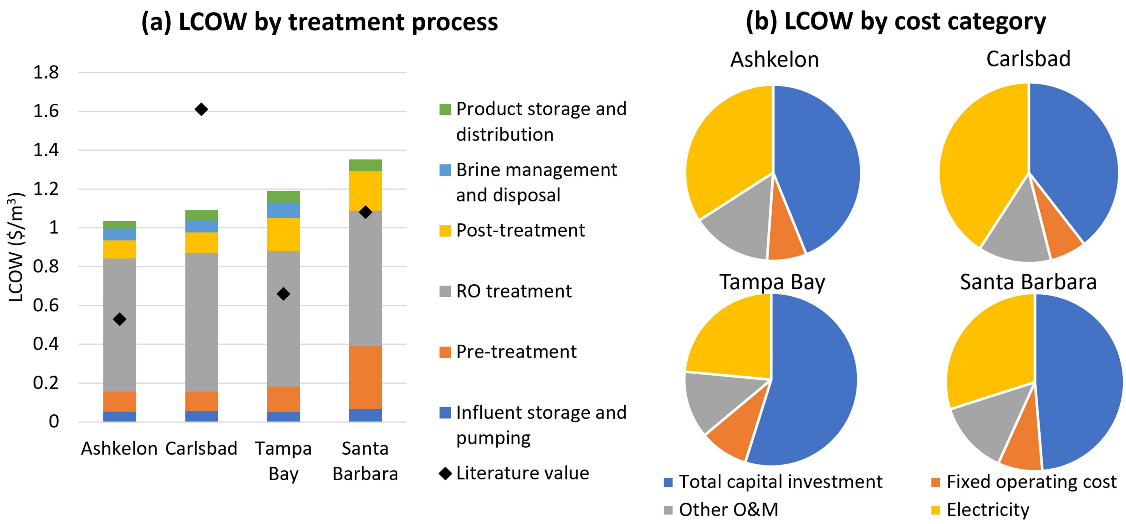

Figure 3.3: Estimated LCOW by unit process (a) and by cost category (b) for the case study facilities.

ature (Figure 3.3) highlights clear differences. To understand the discrepancies, we modeled hypothetical desalination plants across a range of product water capacities (Figure 3.4). The outcomes were compared with LCOW curves from several other existing works in the literature and models for the same design criteria. The International Atomic Energy Agency's Desalination Economic Evaluation Program (IAEA DEEP), Wittholz et al. (2008), and additional cases from Huehmer et al. (2011) along with curves created from Voutchkov (2016) were plotted together with WaterTAP3 LCOW results in Figure 3.4 (Kavvadias and Khamis (2010); Wittholz et al. (2008); Huehmer et al. (2011); Voutchkov (2016)). While Wittholz et al. and Huehmer et al. use a cost database approach, the DEEP model is a piece of software for cost and energy calculation developed by IAEA to represent different configurations and power supplies for desalination processes (Kavvadias and Khamis (2010)). The results of WaterTAP3 fit reasonably well with other estimations and indicate that desalination production follows economy of scale. Interestingly, none of the estimations match the reported cost of the Carlsbad plant, which is well above any pre-existing model results.

WaterTAP3 is a process-based data-driven model that captures the detailed desalination

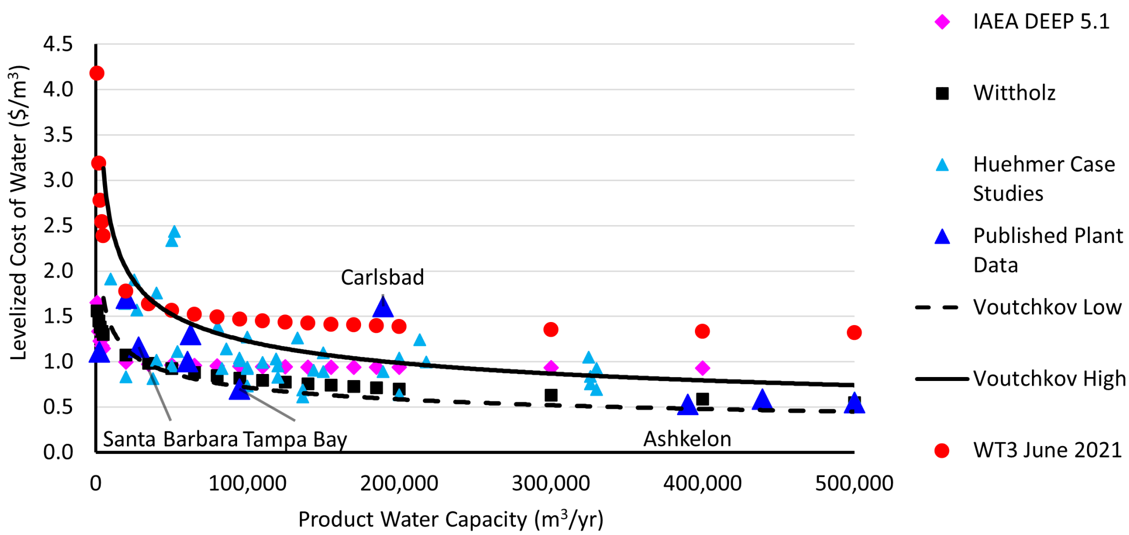

Figure 3.4: Comparisons for LCOW across hypothetical plants of different product water capacities and with other model predictions and literature data of SWRO plants worldwide.

engineering process and associated data. However, it does not include policy information such as the permit requirements and the time for permit approval. In the end, the goal of this work is not to produce a best model (there will not be a best model but a model that captures the inputs accurately) but to have a model to identify the discrepancies across different facilities. WaterTAP3 results clearly showed that the LCOW is heavily impacted by local factors, namely, cost structure for labor and materials, permitting and land acquisition rates, and electricity tariffs.

3.3.3 Sensitivity analysis

A sensitivity analysis was carried out to explore the degree to which critical input parameters influence model outcomes. The ranges defined for the sensitivity analysis were intended to capture not only the range observed in practice today but also the range of values that future projects similar to these particular case studies might experience with extensive innovation. The results shown in tornado plots for each facility (Figure 3.5) indicate that the plant capacity utilization (the percent of time during the year that the plant is operating at designed capacity) had the largest impact on the LCOW. This result suggests reducing plant downtime from fouling, cleaning, and replacement to ensure continuous water production at designed capacity is critical to reduce the overall LCOW over the plant service life.

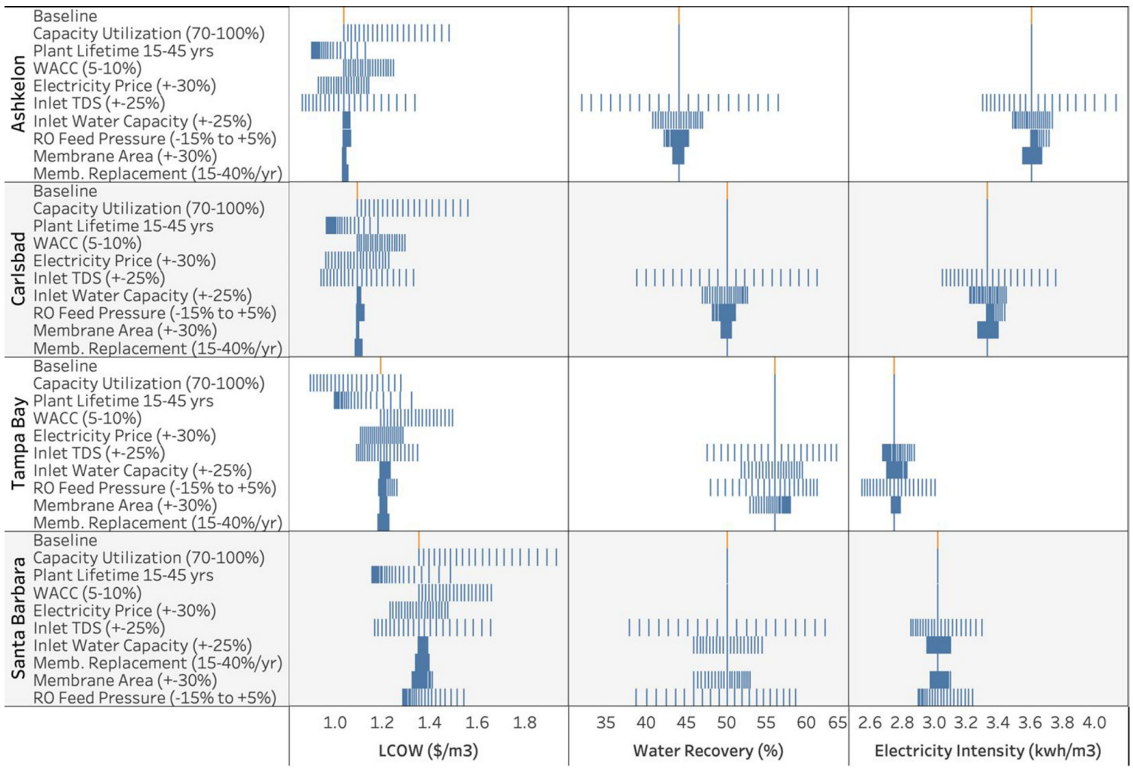

Figure 3.5: Tornado plots for sensitivity analysis of each case study. The charts are organized by total magnitude of % change to the LCOW, water recovery, and electricity intensity from varying each input parameter.

The water recovery and electricity intensity are most sensitive to the inlet TDS and the RO feed pressure in the model. Although lower seawater TDS associated with the seasonal storm runoff allows reduction in electricity and increase in water recovery in the model, storm runoff may also bring a higher concentration of suspended solids and organics that cause fouling. Therefore, while TDS impacts recovery to a large degree in the model results and in real life, its impact to water recovery and electricity intensity may be less well-captured since fouling factors are not linked to TDS in the model. Variation of ±15% inlet TDS reveals the model sensitivity more than it indicates a likelihood of such variation in the actual inlet

TDS of respective facilities. Though the sensitivity analysis indicates recovery could exceed 60% under certain unrealistically low-TDS or high-pressure conditions, recovery greater than 56% is not generally practiced due to fouling concerns.

3.3.4 "What if" scenario analysis

Current SWRO technology is highly efficient and approaches the thermodynamic limit for separating salts from water (Lim et al. (2021)). Although continuous improvements of RO technology, such as development of high permeability, high salts, and boron rejection RO membranes, can reduce the energy cost of SWRO for all plants, further improvements are likely to be slower and incremental in coming years (Anis et al. (2019)).

Automation of desalination processes could be another avenue for reduction of overall cost of seawater desalination (Giammar et al. (2021)). Automation reduces the need for manual measurements, user input, and repairs, which could result in reduction of fixed labor costs (Lior (2012)). This could be especially important for regions of high-cost labor and materials such as Southern California. We created a "what if" scenario to represent the automation of the desalination process through AI controls. Under this scenario, we assumed reductions of fixed annual labor costs through automation and process streamlining. Figure 3.6 illustrates the results based on fixed labor cost reductions ranging from 10-50% as well as the cost savings for each plant. Based on the model output, we estimated the incentive for this reduction. For example, 20% reduction of fixed labor cost (or 80% of the original cost) can result in $3.8M and $2.2M savings for Ashkelon and Carlsbad plants, respectively, over the plant service life, which can translate into the equivalent amount of funding available for capital investment for AI technology (Figure 3.6).

Our initial analysis captures only one aspect of potential savings from automation. Increased automation, sensing, and AI can also reduce chemical costs in pre- and post-treatment

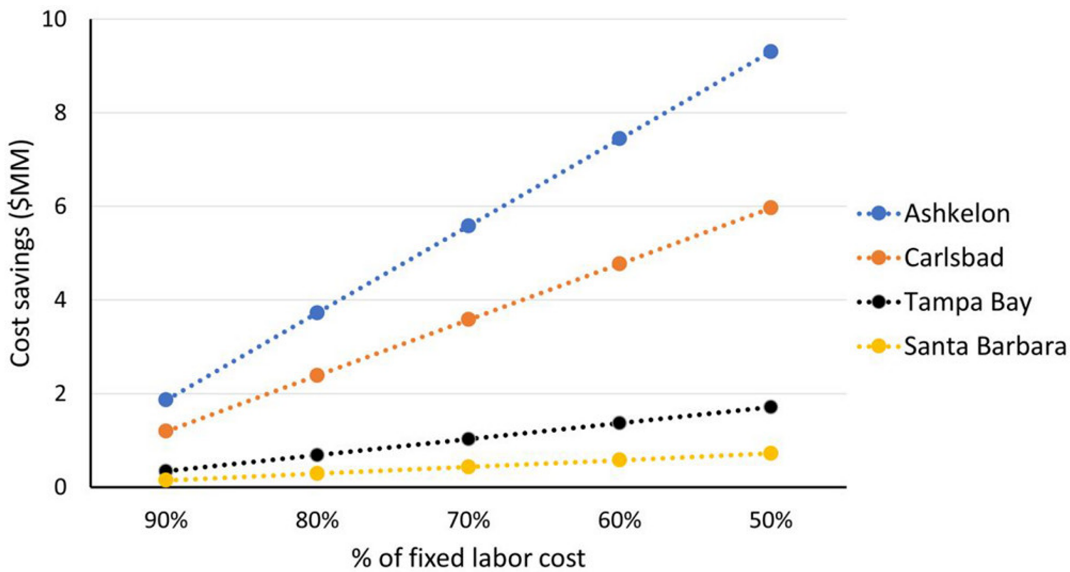

Figure 3.6: Potential cost savings through reduction of fixed labor cost of 10-50% for each of the four case studies.

through adjusting chemical dosage based on real-time water quality sensing to avoid over- or under-dosing. Other potential applications of automation and AI in seawater desalination include variable control of intake pumps, forecasting and predicting flows and water quality (i.e., algal blooms), and collection and integration of sensor data with autonomous controls throughout the entire process (Blondin et al. (2019)). Additionally, development of sensors and dynamic control systems could indicate membrane fouling propensity, potentially reducing the plant operation downtime and membrane cleaning and replacement. Other cost reduction opportunities in seawater desalination also include recovery of natural resources from desalination brine to offset the overall costs and reduction of intake salinity through blending seawater with wastewater treatment plant output or other lower salinity sources (Blondin et al. (2019); Quist-Jensen et al. (2016); Telzhensky et al. (2011); Wei et al. (2020); Amy et al. (2017)). There could be various "what-if" scenarios, but only a labor cost reduction scenario was used in this discussion because the relatively high labor costs in the U.S. significantly increased the plant operational cost in comparison with the plant in Israel. Detailed life cycle cost analysis is required to further understand the impacts of these opportunities, which is beyond the scope of this baseline analysis.

3.3.5 Breakeven curve of SWRO

The California water supply reductions R over the past 20 years and breakeven curves for SWRO are shown in Figure 3.7. The average statewide conventional water supply reduction varied by year, reaching the highest reduction in 2014-2015; the average water supply reduction over the past 20 years is around 8% (Figure 3.7a). Two breakeven curves are shown in Figure 3.7b; one represents an estimated SWRO premium of \$0.60/m^3 (solid line), and the other used an assumed lower future premium of \$0.40/m^3 (dashed line). For the given value of R, if MC_{cons} at a local district is below the breakeven curve, then for that district the cost of managing water shortages through conservation measures is less than the cost premium of SWRO and the conventional supply is preferred. If MC_{cons} at the local condition is above the breakeven curve, it is more cost-effective to adopt SWRO. As R increases, the value of the breakeven curve decreases. When large water supply reductions are needed, only inexpensive conservation measures can compete with SWRO.

The breakeven curves also showed that moderate decreases in the SWRO cost can have a large impact on the SWRO cost premium and, thus, on the cost-effectiveness of SWRO (Figure 3.7b). For the San Diego case with $C_{\text{SWRO}} = \$2/\text{m}^3$ and $C = \$1.4/\text{m}^3$, a 10% absolute cost reduction drops the premium from \$0.6/m^3 to \$0.4/m^3 (33%). For example, with regional shortage factor R equal to 15%, SWRO at \$2/m^3 is preferable if conservation costs more than \$4/m^3; SWRO at \$1.8/m^3 is preferable if conservation costs more than \$2.7/m^3 (Figure 3.7b).

The case study presented here illustrates several motivations for implementing SWRO, all of them related to supply shortage in arid or drought-prone climates. Review of water supply planning and management documents for Israel (Fernandes (2015); Lev (2012)), Santa Barbara (City of Santa Barbara PWD (2020)), the San Diego region (Authority (2021)), and the Tampa Bay region (The Southwest Florida Water Management District (2020); Water

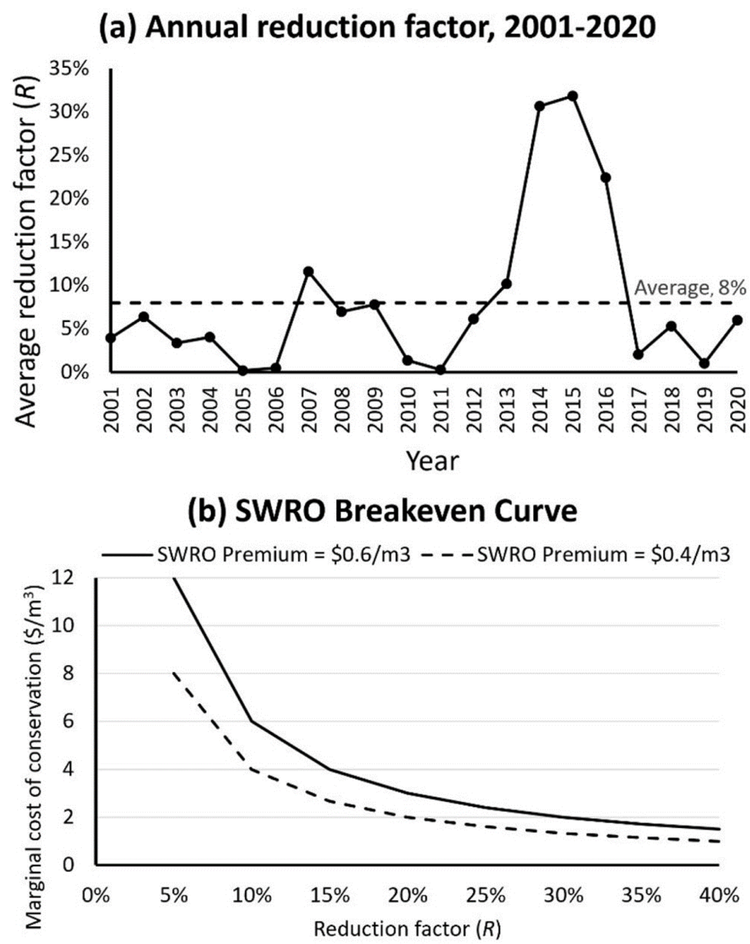

Figure 3.7: Water supply reduction R calculated based on California drought data between 2001 and 2020 (a) and breakeven curves, defined as $MC_{\text{cons}} \times R = \text{SWRO}$ Premium (b).

(2018); Barnett (2020)) revealed a difference in the use of SWRO between Israel and the U.S. In the Israeli case, expansion of seawater desalination is one of the few options to increase overall water supply to sustain future growth,46 whereas in the U.S., seawater desalination is one of many options including long distant water transport, brackish groundwater extraction, and wastewater reuse. In this context the principal advantage of SWRO is its reliability in drought conditions and the additional value desalination derives from self-reliance from imported water.

In quantifying the trade-off of SWRO and conservation costs, we included conservation measures such as reduction of water waste (e.g., fixing leaks), improved water end-use efficiency (e.g., efficient toilets), and changes to infrastructure that reduce demand (e.g., landscape conversion). These efforts can address drought-induced shortages and allow conventional water supplies to provide for users' daily needs appropriately. However, as resource use grows more efficient, the cost of additional conservation goes up. Therefore, conservation efforts do not remain cost-effective indefinitely, and ensuring that a water supply is sufficient, especially in semiarid and drought-prone areas, remains an ever-growing challenge.

Breakeven analysis provides a schematic approach to valuing the reliability of SWRO relative to alternative sources and shows how to connect the cost premium associated with SWRO to the cost of addressing supply shortage through conservation measures. We have also shown that cost reductions for SWRO technology can have an out-sized impact on the relative cost premium for this option. Cost reductions arise both from research into technology innovation and from the expansion of the market for this technology.

3.3.6 Contribution of seawater desalination to future water supplies

Based on a combination of existing demand, conservation potential, and projected water stress, we estimated the potential utilization of SWRO to meet future U.S. drinking water demand in coastal states. We used 3 MGD as a minimal water demand threshold to justify the development of a SWRO plant because a plant smaller than 3 MGD would be less economical. Figure 3.8 shows the potential SWRO desalination contribution to meet 5%, 10%, and 15% of future drinking water demand by state. The "variable" scenario assumed all water-stressed regions will develop a 3 MGD SWRO plant to meet the area's water demand even if the calculated future water demand is below the threshold. States are grouped according to future conservation potential (low, medium, and high).

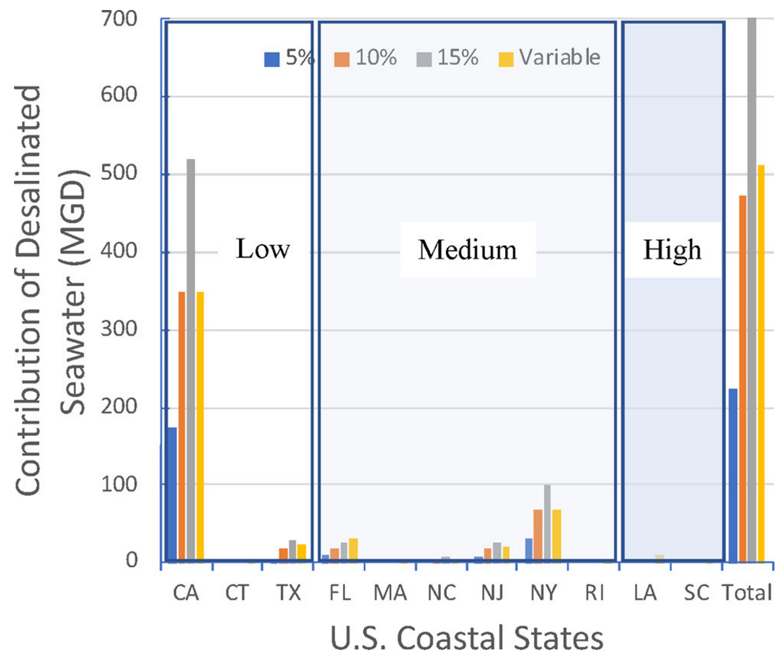

Figure 3.8: Potential contribution of SWRO to meet 5%, 10%, 15%, or a variable amount of drinking water demand in water-stressed coastal states. The states are classified by low, medium, and high future water conservation potential.

The results showed that, among the three states with low additional water conservation potential, SWRO desalination can provide the largest capacity to meet the municipal drinking water demand in California (Figure 3.8). Smaller scale SWRO desalination plants are also beneficial to eliminate water stress in Texas, where measures of water conservation have already been widely implemented. However, SWRO desalination is not a preferred option in Connecticut, where smaller scale water projects are more suitable. Similarly, SWRO is unlikely to be a preferred option for future drinking water supply in Louisiana and South Carolina, where high water conservation potential remains. Among the states that have the medium level of water conservation potential, New York stands out as a region that can benefit from high capacity SWRO, which is likely due to the projection of high water demand in the region (Figure 3.8). The overall potential for SWRO to meet future U.S. municipal drinking water demand is estimated in the range of 223 to 706 MGD (Figure 3.8).

Although these estimates simplify water demand and supply factors, they are based on sound historical data on water conservation, creditable data on future water demand, and detailed analysis of water supply-and-demand relationships by individual counties in each state. We acknowledge that individual counties may not be a perfect scale for analysis. Size of the counties is much smaller in the eastern U.S., so it is harder to achieve the required 3 MGD threshold to justify the SWRO plant. On the contrary, California counties are oversized; a single seawater desalination plant would not be able to serve the entire county of Los Angeles. Moreover, in a life cycle energy analysis of different water resource options, Lam et al. (2017) indicated that the decision to develop centralized supply sources such as seawater desalination should consider their likely long-term operating scenarios. Seawater desalination plant operation under the minimum production rate has been shown to be highly energy inefficient compared to operation at its design capacity. This agrees with our WaterTAP3 results indicating the importance of continuous operation. Therefore, the decision for adoption of seawater desalination should also consider if the project is developed for temporary drought adaptation or the long-term regular use of the facility. Overall, the results suggest SWRO should be considered as an important part of the water supply portfolio in the development of future water projects. Additional studies are needed to further incorporate greenhouse gas (GHG) emissions and decarbonization of future water systems in relation to reliability of water supplies considering climate change impacts (Cornejo et al. (2014); Lundie et al. (2004)).

3.4 Conclusions

Detailed analyses were carried out using case studies and scenario settings to illustrate the costs, energy, and reliability factors that influence the decision of adopting seawater desalination in the U.S. The following general conclusions can be made from the outcomes of this study.

- The seawater desalination cost is significantly impacted by local labor, material and electricity cost structure, and land value and permitting requirements. Seawater desalination project permeating requirement in California has significantly increased the LCOW of desalinated water in the state in comparison with the plant in Israel.

- The LCOW is highly sensitive to the plant capacity utilization; incremental reduction in annual water production below design capacity quickly drives up the LCOW. This result implies the importance of designing the "right size" desalination facility to meet water supply demands to avoid the need to halt plant production when desalinated water is not needed. Reducing plant operating downtime caused by membrane fouling, cleaning, and replacement is also critical to maintain the projected LCOW.

- Investments in automation and AI to reduce the fixed labor cost could result in long-term savings. This is especially important in regions with higher cost of labor or in remote areas where labor mobilization costs are high.

- The pipe parity value for seawater desalination is highly localized. It depends on the water supply reductions induced by natural drought, the marginal cost of water conservation, and the seawater desalination cost premium. A small change in the absolute cost of seawater desalination could cause a significant shift toward adoption of seawater desalination as a new source of water supply.

Chapter 4

Assessing the Risk of *Legionella* Infection through Showering with Untreated Rain Cistern Water in a Tropical Environment

The contents of this chapter appear in the journal Water, Quon et al. (2021a).

4.1 Introduction

Roof-top harvested rainwater (RHRW) is a major water resource to supplement both potable and non-potable water supplies in many parts of the world. Rainwater harvesting is beneficial in promoting water saving in water-stressed, semi-arid areas (Abdulla and Al-Shareef (2009)). It is especially useful in tropical climates, such as in the Caribbean Virgin Islands, where rainfall is plentiful but there is limited surface and groundwater storage capacity.

Ac-cording to U.S. Virgin Island Code (V.I. Code tit. 29, § 308, 2019), all residential buildings constructed in the Virgin Islands are required to install rain tanks in order to alleviate demands on surface waters and desalinated water. In fact, rainwater is the only source of tap water piped to rural homes in the Virgin Islands. In these tropical islands, rain tanks serve multiple functions: 1) providing storage of water for daily use, 2) lessening the impact of runoff on stormwater systems, and 3) allowing low-cost water access to rural homes that are separated from the municipal water delivery systems. RHRW provides both potable and non-potable water for indoor household uses, while non-potable use is especially common in the U.S. Virgin Islands (Ahmed et al. (2014)). However, the quality of RHRW water is not well documented and is seldom tested.

Health risks associated with RHRW could attribute to the occurrence of microbial contamination from wild animals' feces (Lim et al. (2015)). Natural soil bacteria and microbes from decaying leaf litters carried in rainwater can form biofilms in both the rainwater storage and distribution systems, which often harbor opportunistic human pathogens (Declerck et al. (2009)). Among diverse microorganisms, *Legionella* species, a bacterium commonly found in soil and plant litters in tropical regions, is of concern because they are the most documented causative agent of waterborne outbreaks (Shah et al. (2015)). The *Legionella* contamination is particularly problematic in warmer environments, such as tropical islands including the Virgin Islands. The daily temperature (22–32°C) range of these tropical islands is ideal for *Legionella* growth (25–42°C) (Fields et al. (2002)). *Legionella* infection in humans is primarily through inhalation of aerosols to the lungs from contaminated water sources. Shower water, the main application of rainwater for indoor non-potable use, can be a major vehicle for transmission of *Legionella* through water aerosols. The main symptoms of *Legionella* infection are respiratory illnesses, which are also known as Legionnaires' disease and Pontiac Fever (Abu Khweek and Amer (2018)).

In addition to the uncertainties of water quality associated with RHRW, severe storms and flooding can further exacerbate the contamination in the rain cisterns. In September 2017, two category-5 hurricanes, Irma and Maria, swept through the Caribbean Sea in what is now known as the region's most active hurricane season on record (Zolnikov (2018)). The wind, rain, and destruction delivered by the hurricanes impacted rain catchment systems and damaged many cisterns on the Virgin Islands. Excess loads of leaf litters, soil, and other organic debris that harbor opportunistic pathogens were washed from the rooftop to the underground cisterns. In addition, island-wide power outages halted treatment of human sewage, resulting in septic overflows that directly affected surface waters and possibly shallow groundwater. The underground cisterns compromised by fine cracks and poor seals may be impacted by sewage contamination through connection with surface and shallow groundwater (Jiang et al. (2020)). A boil water advisory was issued by the VI Water and Power Authority on September 27, 2017 to curtail public health risk (Consortium (2017)). However, the lack of access to fuel and electricity long after the passing of the hurricanes made the advisory impractical (The St. Thomas Source (2017)).

In an effort to assess the impact of hurricanes on water quality in the disaster-stricken region, water samples from 22 households' rain cisterns on the island of St. Thomas, Virgin Islands, were collected as soon as the island became accessible (Jiang et al. (2020)). Water samples were analyzed for microbial composition and contamination. Among 22 cisterns sampled, 86% were positive for *Legionella* spp. A household survey was also carried out alongside the water sample collection to understand the primary use of the rain cistern water and the public perception of water quality. Based on the survey outcomes, a quantitative microbial risk assessment (QMRA) was carried out using the *Legionella* contamination data and human exposure through showering water to understand the risk of *Legionella*-related disease in the post-disaster region. The outcomes of this study contribute to the decision of water quality management and disaster relief strategies.

4.2 Materials and methods

4.2.1 Household survey

This study was approved by the University of California, Irvine Institutional Review Board (IRB #2017-4032). Household surveys were conducted in St. Thomas, Virgin Islands, in November 2017, three months after the island was struck by Hurricane Maria. Verbal consent was collected from participants before survey questions were recorded. The purpose of the survey was to understand the island residents' perception of water quality and water use behavior. Household characteristics were also obtained, such as income and education, to test the correlation between risk perception and socioeconomic status. Surveys were collected from all residents who had given permission to sample their cisterns. Additional surveys were also collected from neighboring residents at nearby grocery stores, community gathering places, bars, and restaurants while water samplings were taking place in the neighborhood. Responses from a total of 107 complete surveys were included in this analysis. A copy of the survey questionnaires is included in supplementary information. The survey data were coded and binned, and the outcomes were plotted in Excel (Microsoft). The relationships between income level and awareness of water use, water quality, water safety, and perception of risk were assessed using χ^2 tests in RStudio (R Core Team (2022)). The χ^2 test compares categorical survey responses with the null hypothesis that there is no association between income level and water quality awareness, water use, and water safety perception. A $p < 0.10$ was considered as statistically significant.

4.2.2 Risk assessment

Quantitative microbial risk assessment (QMRA) was conducted based on the framework outlined by the U.S. National Academy of Sciences (Council (1983)). The four main components are hazard identification, exposure assessment, dose–response assessment, and risk characterization. A fifth supplementary component, risk management, was also included to provide management recommendations based on the simulated risk outcomes. A Monte Carlo simulation was used to analyze the range of the data and estimations of parameters, while providing randomization and variability in the selections. All calculations were performed using MATLAB (MathWorks) and RStudio (MATLAB (2022); R Core Team (2022)).

Hazard identification

According to CDC Waterborne Disease & Outbreak Surveillance Reports, *Legionella* has emerged as the most frequently reported etiology among drinking water-associated outbreaks. All waterborne outbreak-associated deaths reported in the most current surveillance period (2013–2014), including the outbreaks reported in hospital/health care settings or long-term care facilities, were caused by *Legionella*. *Legionella* can cause a serious type of pneumonia called Legionnaires' disease (Benedict et al. (2017)). Legionnaires' disease is remarkably similar to other types of pneumonia, with symptoms that include cough, shortness of breath, fever, muscle aches, and headaches. The bacteria can also cause a less serious illness called Pontiac fever. Pontiac fever symptoms are primarily fever and muscle aches; it is a milder infection than Legionnaires' disease. Symptoms begin between a few hours to 3 days after being exposed to the bacteria and usually last less than a week.

Legionella spp. in rainwater cisterns, especially in tropical environments, has been previously reported (Broadhead et al. (1988); Simmons et al. (2008)). *Legionella* is known to grow and persist within biofilms in engineered water storage and distribution systems such as cisterns (Declerck et al. (2009)), and spread by shower head, sink faucets, and other water handling devices that generate aerosols and water droplets. Inhalation of aerosols that harbor *Legionella* into the lungs is the major route of human infection (Diederen (2008)). There are no vaccines that can prevent Legionnaires' disease. *Legionella* spp. was detected in 86%

of rain cistern samples in 2017 post-hurricane Maria water quality study in St. Thomas, Virgin Islands (Jiang et al. (2020)), indicating a possible health risk through water aerosol exposure. Although the direct source of the *Legionella* was not clear, it is necessary to assess the health risk of such hazard because rain cistern water was the primary water used by the island residents for showering, the most common form of personal hygiene practice.

Exposure Assessment

The *Legionella* in rain cistern water in the 2017 post-hurricane season was collected from the study of Jiang et al. (2020). The study collected water samples from 22 households' rain cisterns on the island of St. Thomas and detected *Legionella* using NextGen sequencing of 16S rRNA gene. The concentrations of *Legionella* were reported as the fraction of total microbial population in each cistern water sample. To convert the fraction of *Legionella* to a range of concentrations that may be encountered in cistern water, the total viable heterotrophic bacteria determined by heterotrophic plate counts (HPC) collected from rain cisterns in St. Thomas from an earlier study were used to represent a baseline distribution of cultivable bacteria present in cistern water from RHRW (Crabtree et al. (1996)). The HPC was determined by SMEWW 9215C on R2A media using 0.1 mL of diluted sample (HPC (2018)). For HPC that was reported as a tabulated concentration in Crabtree et al. (1996), the detection limit was used as the upper bound. The HPC data were fitted with a non-parametric cumulative distribution function (CDF) curve. A second CDF curve was also created based on the reported *Legionella* data in the 2017 study. Then, the concentration of *Legionella*, C_{Leg}, in cistern water was estimated as follows:

$$C_{\mathrm{Leg}} = C_{\mathrm{bac}} \times \%_{\mathrm{Leg}} \tag{4.1}$$

where C_{bac} is the total viable bacterial concentration that is generated by randomly sampling from the CDF of HPC using the Monte Carlo sampling procedure; and $\%_{\text{Leg}}$ is the percent of *Legionella* in a cistern that is obtained by randomly sampling from the CDF of the fraction of Legionella detected in cisterns. The CDF for *Legionella* percentage values was left truncated at 0.

The human exposure to *Legionella* in this study was assumed to be through showering using cistern water only. Other aerosol exposures, such as through toilet flushing or water faucets, are also possible (Hamilton et al. (2016)). However, shower risk is considerably higher and is assumed to be a single daily exposure event. Since the data for *Legionella* are from water samples taken directly at the cistern, the additional growth of *Legionella* in indoor plumbing and shower heads was not considered (see Section 4 for additional details). In addition, no reduction in bacteria through physical chemical water treatment was included. The treatments of the cistern water on St. Thomas vary significantly from household to household ranging from a simple screen filter for litter removal to installation of reverse osmosis membrane filters. Since the use of high-end technologies for treating rainwater is rare based on our field observations, such treatment removal of *Legionella* was not considered in this estimation. Only thermal inactivation through a conventional water heater used for heating shower water to 60°C for a warm shower was included as an exposure scenario. According to Rogers et al. (1994), *Legionella* inactivation at temperatures below 50°C is negligible.

Two separate scenarios were considered in the exposure assessment: cold shower and hot (warm) shower. Cold shower assumes water drawn directly from the household cistern, at a temperature of 24 to 25°C. No mixing with water from heater and thus no thermal inactivation of *Legionella* was included. The warm shower scenario assumes a shower water temperature of 43.5°C, which includes mixing a portion of cold water from the cistern directly and a portion of hot water from the water heater (60 °C). For this warm water shower scenario, the $\%_{\text{hot}}$ was calculated as follows:

$$T = \frac{m_1 c_1 T_1 + m_2 c_2 T_2}{m_1 c_1 + m_2 m_2} \tag{4.2}$$

$$\%_{\text{hot}} = \frac{m_1}{m_1 + m_2} \times 100\% \tag{4.3}$$

where m is the mass of water, c is the water heat capacity, and T is the temperature of each respective water stream.

Human infection occurs through inhalation of aerosols containing *Legionella* into the lungs, where the bacteria can replicate in the alveolar macrophages of the lungs (Copenhaver et al. (2014)). Aerosols produced by common household shower heads have been found to contain *Legionella* when shower water is contaminated by the bacteria (Bollin et al. (1985)). The concentration of *Legionella* in water aerosols was assumed to equal the concentration in the water as used in a previous study (Lim et al. (2015)). Preferential aerosolization of *Legionella* from bulk water may occur as indicated in previous reports (Feazel et al. (2009)). However, the partition rate is highly variable and was not included here to reduce the uncertainty of the simulation. Inhalation of aerosols per minute of shower duration (mg/min), M_{AB}, is based on the volumetric flow rate of water from the shower head and aerosol deposition of mass in the bronchial and alveolar region according to previous experiments by Zhou et al. (2007). This mass deposition varies with shower temperatures, shower head flow rates, and human breathing habits (oral inhalation or nasal inhalation). A uniformed distribution was adopted to include the range of water aerosol mass deposition to human lungs for different shower head water flow rates and human breathing habits. The detailed data for deposition rates from Zhou et al. (2007) are summarized in Table B.1. The randomly selected mass deposition rates from the uniformed distribution for warm shower U(0.036, 0.364) and cold shower U(0.001, 0.008) were used in the Monte Carlo simulation. The input parameters for exposure assessments are listed and defined in Table 4.1.

The total dose of *Legionella* inhaled and deposited in the bronchial and alveolar region of a person's lungs during exposure in a single shower event (CFU) was adapted from the model for showering established by Lim et al. (2015) and was estimated as

$$Dose_{\mathrm{Leg}} = \left(C_{\mathrm{Leg}} \times \frac{(100 - \%_{\mathrm{heated}}) + \%_{\mathrm{heated}} \times 10^{-H}}{100} \right) \times \frac{M_{\mathrm{AB}}}{\rho_{\mathrm{w}}} \times t_{\mathrm{shower}} \tag{4.4}$$

where t is the duration of a single shower event (min), ρ_{w} is the density of water a temperature T, H is the log-reduction due to heat inactivation of *Legionella* at temperature T, and $\%_{\mathrm{heated}}$ is the portion of water from the water heater at 60°C used for mixing to heat the shower water to a final temperature of 43.5°C.

Dose-Response Assessment

We adopted a *Legionella* dose–response model established through clinical trials on guinea pigs (Berendt et al. (1980)). The endpoint of response is infection due to exposure to a known dose of *Legionella* through inhalation. An exponential model, shown in Equation 4.5, is the best-fitted model to the clinical data based on the dose inhaled.

Table 4.1: Input parameters for the Monte Carlo simulation to calculate the daily and annual infection risk of *Legionella*

Parameter definition	Symbol	Point estimate or distribution	Unit	Source
Concentration of total heterotrophic bacteria measured in an untreated rainwater cistern	C_{bac}	Empirical distribution	CFU/mL	Crabtree et al. (1996)
Percent of total bacteria DNA represented by *Legionella*	$\%_{\text{Leg}}$	Empirical distribution	unitless	Jiang et al. (2020)
Shower water temperature	T_{unheated} T_{heated}	24.5 43.5	°C	This study
Percent of shower water that is heated to 60°C by a conventional water heater	$\%_{\text{heated}}$	54	unitless	This study, Equation (3)
Density of water	ρ_{unheated} ρ_{heated}	997 991	g/cm^3	This study
Thermal inactivation of *Legionella* at temperature T = 60°C	H	3	unitless	Cervero-Aragó et al. (2015)
Shower duration	t	Normal distribution ($\mu = 7.8, \sigma^2 = 0.02$), left-truncated at zero	min	DeOreo et al. (2016)
Aerosol mass inhaled and deposited in the alveolar-bronchiolar region, heated shower	$M_{\text{AB,heated}}$	Uniform distribution, $U(0.036, 0.364)$	mg/min	Zhou et al. (2007)
Aerosol mass inhaled and deposited in the alveolar-bronchiolar region, unheated shower	$M_{\text{AB,unheated}}$	Uniform distribution, $U(0.001, 0.008)$	mg/min	Zhou et al. (2007)
Dose-response curve constant	k	0.0599	unitless	28
Number of exposures per year	n	365	no. per year	This study

$$P_{\text{inf}} = 1 - \exp\left(-k \times Dose\right) \tag{4.5}$$

where $k = 0.0599$ as determined by Armstrong and Haas based on the guinea pig trial data (Armstrong and Haas (2007)).

Risk Characterization

The dose calculations represent a dose inhaled for a single, daily shower event. Therefore, the response, P_{inf}, represents a daily risk in this case. Annual risk represents the risk of a single infection for the duration of one year, or in this case, 365 consecutive daily exposure events. This annual risk, P_{annual} , is calculated as

$$P_{\text{annual}} = 1 - \prod_{i=1}^{365} \left(1 - P_{\text{inf}}\right) \tag{4.6}$$

A sensitivity analysis was conducted in order to identify the model parameters that had the greatest contribution to uncertainty and variability in the results. The analysis was based on the warm or cold shower scenarios. Using 10,000 iterations of the input parameters, the Spearman rank correlation coefficient was computed in MATLAB in order to determine the strength and direction of a presumed monotonic relationship between input parameters and model output, where a coefficient of 0 indicates no influence of the variable on the results, and a value of + or - 1 indicates a positive or negative influence on the output. The sensitivity analysis was conducted for the following model input parameters: *Legionella* concentration (C_{Leg}), the total viable bacterial concentration (C_{bac}), the fraction of *Legionella* ($\%_{\text{Leg}}$) in the cistern water, exposure time (t_{shower}), and mass inhalation rates (M_{AB}) for each of the scenarios.

4.3 Results

4.3.1 Household Survey

The household survey results showed that bottled water was the primary source of drinking water for the island residents after the hurricanes. Tap/rain cistern water was mainly used for non-potable purposes including washing hands, dishes, and food (4.1). Approximately 80% of the survey participants reported using tap/cistern water for showering. The survey did not differentiate between municipal piped tap water and cistern tap water. However, it is important to mention that most of the rural homes in St. Thomas were not connected to the municipal piped water system. Rain cistern water was plumbed into houses during home construction. Municipal tap water from desalination of seawater was piped to downtown commercial area (hotels, shops and restaurants) and residential communities near the city center. However, municipal water was unavailable for an extensive period after the 2017 hurricanes hit. Cisterns became the only source of tap water. The survey results confirmed the priority of conducting risk assessment of daily *Legionella* exposure through shower water.

Moreover, survey results also showed that most island residents perceived their water to be safe or somewhat safe (4.2), yet this perception differed by income group. Lower-income households ($<$ \$40K per annual) overwhelmingly felt that their water was only somewhat safe, while higher-income households ($>$ \$40K per annual) tended to be more confident about water safety. This difference in perception of water safety between the two income categories is statistically significant at the 10 percent level ($p < 0.10$) based on χ^2 tests. It should be noted that the median household income in the U.S. Virgin Islands is \$37,254 based on the

2010 census, which is significantly lower than the median household income of $57,617 in the United States (U.S. Census Bureau (2010)).

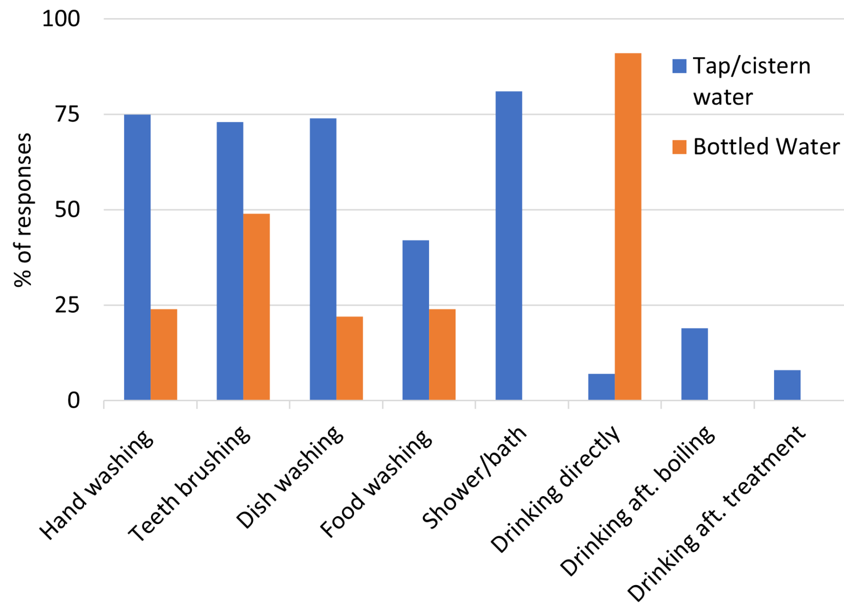

Figure 4.1: Survey responses (n = 107) of water usage in St. Thomas, Virgin Islands, after the hurricanes in 2017.

Survey results also indicate that the majority of the local residents (55%) were aware of the governmental advisory for boiling water (Figure 4.2), but less than half of residents (41%) believed that the government had done enough to let them know the safety of their water supply after the disaster. A similar fraction of the residents (43%) thought that the local and federal governments had done enough to provide them with safe sources of water (Figure 4.2). Satisfaction regarding government management of water after the disaster did not differ by income level. Results of χ^2 tests indicate that there were no significant differences between the two income groups in regard to the above questions (Figure 4.2).

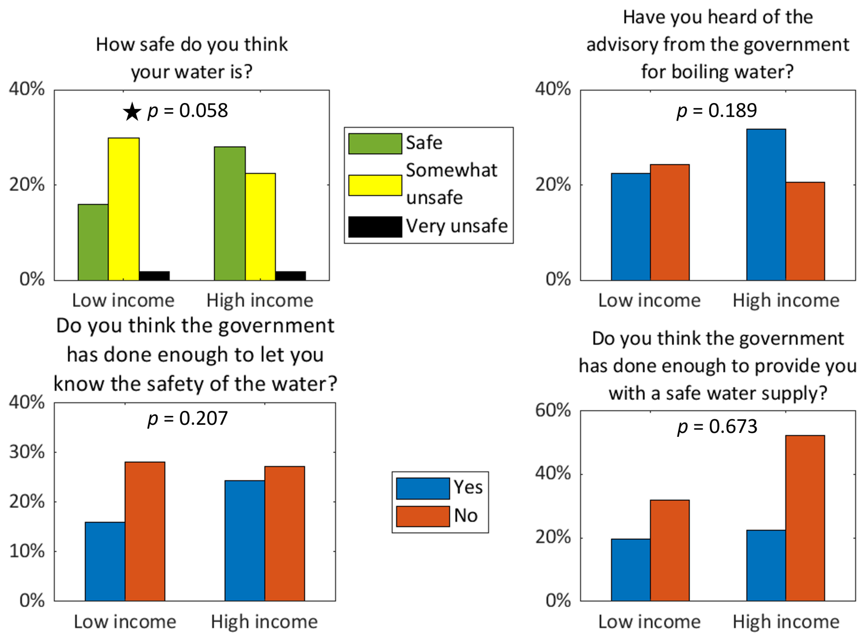

Figure 4.2: Survey responses to questions of perceived water quality and government role, grouped by income level (low and high). The statistical comparison of responses by the two income groups is indicated for each survey question. The starred p-value indicates statistical significance defined in this study.

4.3.2 Quantitative Microbial Risk Assessment

The Monte Carlo simulation of *Legionella* concentrations in rain cisterns using the distribution of total viable heterotrophic bacteria (Figure 4.3a) and the fraction of *Legionella* bacteria among total bacterial community (Figure 4.3b) in cisterns showed that the *Legionella* concentration was distributed over a large range, with a median value of $8.8 \times 10^3 \frac{CFU}{L}$ (Figure 4.3c). The daily risk of infection from aerosol inhalation during show-ering with rain cistern water was estimated by randomly sampling (10,000 iterations) for the parameters from 4.1 and simulated *Legionella* concentration in rain cisterns (under the curve of (Figure 4.3)c). The daily risk of infection varied and had a median value of 3.5×10^{-6} for a cold shower and approximately 100 times higher for a warm shower (Figure 4.4). For both scenarios, outliers approached a risk value of 1. The median annual risk based on theory of independence was estimated as 1.3×10^{-3} for cold showers and 2.5×10^{-2} for hot showers.

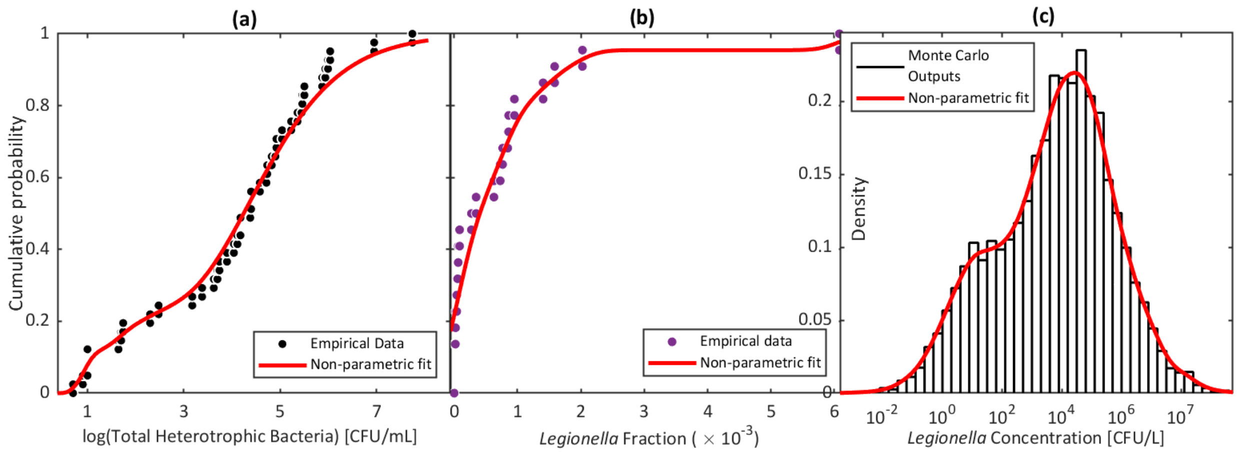

Figure 4.3: Non-parametric cumulative distribution fits of total heterotrophic bacterial data in cisterns (a), the fraction of *Legionella* bacteria among total microbial community in cisterns (b), and Monte Carlo simulation outputs of *Legionella* concentration in cisterns of St. Thomas, Virgin Islands (c). Empirical data for (a) and (b) were collected by Crabtree et al. (1999) and Jiang et al. (2020), respectively.

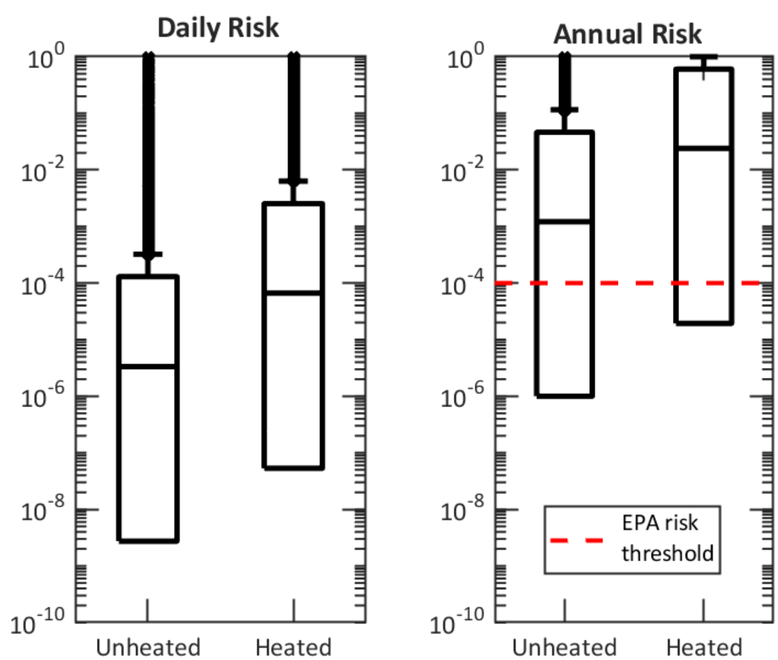

Figure 4.4: Boxplots of daily and annual infection risk based on aerosol inhalation of *Legionella*-contaminated cistern water during unheated and heated shower scenarios. The median is shown as the mark inside the box, 25th and 75th percentile values are the bottom and top edge of the box, respectively.

The results of the sensitivity analysis (Figure 4.5) indicated that for both scenarios, the concentration of *Legionella* in the cistern was the most influential model parameter(C_{Leg}, ρ $= 0.99$) regardless of the shower water temperature. This was further broken down to fraction of *Legionella* among the total microbial community ($\%_{\mathrm{Leg}}$, $\rho = 0.60$) in the cisterns and total heterotrophic bacterial counts (C_{bac}, ρ =0.65). Spearman rank coeffi-cients indicated that shower duration (t) and aerosol mass deposited in the alveolar-bronchiolar region (M_{AB}) were not sensitive input parameters in influencing the model output (ρ is close to 0).

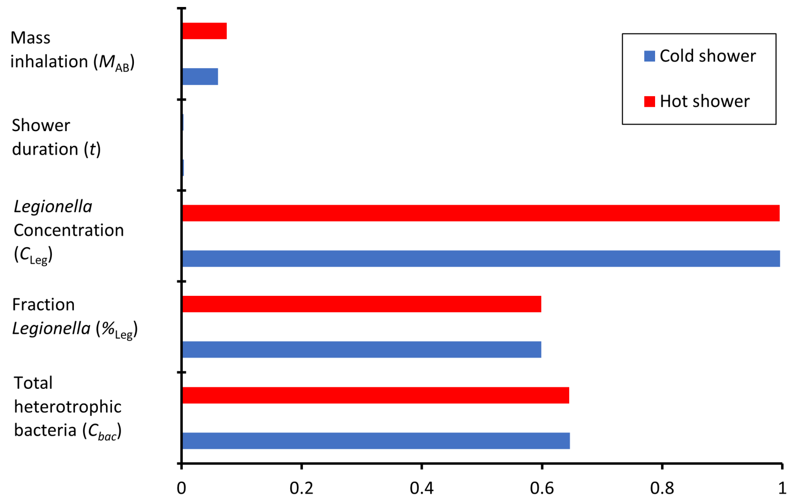

Figure 4.5: Sensitivity analysis for annual infection risk by scenarios and input parameters as defined in 4.1

4.4 Discussion

4.4.1 *Legionella* risk after the hurricanes

Legionella risk in captured rainwater or recycled water has been discussed in the previous studies (Ahmed et al. (2014); Hamilton et al. (2016); Kim et al. (2016)). Yet not much is known of the impact of the hurricanes on rain cistern water quality and associated micro-

Table 4.2: Comparison of simulated *Legionella* concentrations in rain cisterns and estimated annual health risk from showering using cistern water with literature values of similar studies.

	25th Percentile	**Median**	**75th Percentile**	**Unit**	**Source**
Legionella cistern concentration	3.0×10^{3}	9.0×10^{4}	1.3×10^{6}	Gene copies/L	Hamilton et al. (2016)
	4.0×10^{3}	8.5×10^{4}	3.1×10^{6}		Hamilton et al. (2016)
	1.6×10^{4}	2.5×10^{4}	1.0×10^{5}		Ahmed et al. (2014)
	2.5×10^{2}	8.8×10^{3}	1.2×10^{5}	CFU/L	This paper
Annual Risk		1.0×10^{-2}		pppy (per person per year)	Hamilton et al. (2017b)
	1.1×10^{-3}		3.1×10^{-3}		Ahmed et al. (2011)
	2.9×10^{-6}	4.5×10^{-3}	2.1×10^{-1}		This paper

bial infection risk. High occurrence of *Legionella* spp. discovered in rain cistern water on St. Thomas, VI, post hurricane season promoted the investigation of infection risk of this ubiquitous pathogen. The QMRA results indicated that *Legionella* risks post hurricanes were not significantly higher in comparison with two previous studies estimating the risks in captured rainwater (Table 4.2). The median annual risk of the current study is slightly higher than the value reported by Ahmed et al. (2011) but is an order of magnitude lower than the risk estimated by Hamilton et al. (2017b). At first glance, the comparative results suggest that hurricanes do not appear to increase the *Legionella* risk through shower water in the disaster-stricken region. However, discussions are warranted to better understand the contribution of the current study to the knowledge field and the limitation of the risk estimation.

The *Legionella* concentrations in cistern water were estimated based on the fraction of *Legionella* spp. identified among the total bacterial community by NextGen sequencing of 16S rRNA gene (Jiang et al. (2020)). The underlying assumption used in this study is that the fraction of *Legionella* spp. among the total microbial community in cistern water is the same regardless of assay methods (genome-based versus culture-based). Therefore, combining the $\%_{\text{Leg}}$ with C_{bac} (CFU/L) in cisterns could estimate the concentration of viable *Legionella* spp. in the water. However, a direct comparison between the sequencing-based approach (# *Legionella* spp. gene copies/# total gene copies) and the culture-based methods (# *Legionella* spp. CFU/# total CFU) has not been performed. No relationship between HPC counts and *Legionella* concentration or prevalence currently exists. Therefore, the assumption used here is an important limitation to the estimation of viable *Legionella* spp. in the water. Past research in estimating the risk of *Legionella* relied on PCR-based approaches to quantify *Legionella*-specific genes, which assumed the genetic fragments of *Legionella* equal to infectious *Legionella*. Therefore, a PCR-based approach is also not a perfect solution to identify infectious *Legionella*. Despite the limitations of 16S rRNA gene based NextGen sequencing, they are powerful data to understand the microbial composition in water. The HPC baseline data served to model a range of probabilistic values of *Legionella*. High HPC counts can result from cistern stagnation, lack of disinfection, or inadequate temperatures, which can also lead to *Legionella* growth (Fields et al. (2002)). The HPC values, therefore, reflect the condition of the cisterns, which bounds the range of *Legionella*. The HPC data were also used to correct for over-estimation of infectious *Legionella* by the genetic method (Ahmed et al. (2014); Hamilton et al. (2016); Lee et al. (2011)).

Significant progress has been made in recent years on improving the method for culture detection of *Legionella* in drinking water and the water distribution network (Scaturro et al. (2020); De Giglio et al. (2020); Ditommaso et al. (2011)). The *Legionella* monitoring methods by ISO 11731 and the U.S. CDC also provide a reliable framework for culturing *Legionella* in drinking water (11731:2017 (2017); for Disease Control (2018)). However, these methods are still labor intensive and time consuming in comparison with genetic-based methods, which limit their implementation in a field study in the absence of a functional microbiology lab during the post-hurricane period. Nevertheless, culture-based methods should be considered whenever possible. Future studies should also carry out side-by-side investigations of culture-based vs. genetic-based *Legionella* detection in drinking water under various conditions. The outcomes of these comparisons will improve our understanding on the limitations of genetic-based methods and develop credible correlations between the two different monitoring approaches to improve future risk estimations.

A comparison of the range of *Legionella* concentrations from this study with those from other related studies is shown in Table 4.2. Hamilton et al. (2016) reported *Legionella* concentrations between 3×10^3 and 3.1×10^6 gene copies/L by qPCR based on 134 roof-top harvested rainwater samples collected in Southeast Queensland, Australia. Ahmed et al. (2014) reported *Legionella* concentrations by qPCR to be between 1.6×10^4 and 1.0×10^5 copies/L, with a median concentration of 8.8×10^3 copies/L in 72 rainwater tank samples also from the same region in Australia. The simulated concentrations in this study ranged between 2.5×10^2 and 1.2×10^5 CFU/L, which are roughly one order of magnitude lower than qPCR results by Hamilton et al. (2016). The high-end values in this study are similar to the report of Ahmed et al. (2014). However, those previous studies reported concentrations as gene copies/L, whereas in this study the concentrations are CFU/L to represent viable counts. In drawing comparisons between the *Legionella* concentrations, it should be noted that qPCR results may overestimate the viable *Legionella*. Previous studies concluded that qPCR is useful for rapid detection and risk assessment, but often detect higher amounts than by culture methods, especially from water tanks which have been disinfected (Lee et al. (2011); Scaturro et al. (2020). On the other hand, Hamilton et al. (2017b) noted in a study on seasonality and RHRW premise plumbing pathogens that culture-based methods can underestimate concentrations due to the presence of viable but nonculturable (VBNC) cells. These distinctions are important in interpreting risk results as conservative or liberal estimates. Nevertheless, in the absence of a better method to estimate the infectious *Legionella*, the genome-based approach in combination with the culture-based assessment of total HPC presents a useful method to estimate the viable *Legionella* for risk quantification.

Comparison of the annual risks for combined warm and cold shower scenarios with those by Hamilton et al. (2016) and Ahmed et al. (2014) revealed a much wider range of estimated risks than those in the previous studies. This is due to a very large variability of the viable HPC detected in different cisterns. This variability could be attributed to the unevenness in cistern maintenance on the island. Alternative, the SMEWW 9215C method for HPC could also generate viable results because it uses a very small volume of water (0.1 mL) that could hit or miss of particle bound bacteria. The fractions of *Legionella* detected also varied among cisterns. Some of the cisterns may be impaired by the hurricane-induced storms as noted in the study by Jiang et al. (2020). During the sample collection effort, we covered as a broad of an area on the island as we could, but we did not have a pre-existing knowledge of income level of the households at the time of sample collection. Future study design should consider water quality assessment across different income levels. It should also be noted that both previous studies Ahmed et al. (2014) and Hamilton et al. (2016) equated qPCR genome copies with the viable CFU in the dose–response model, which may overestimate the risk of infectious *Legionella*.

Moreover, the uncertainty of the dose–response model is not only limited to ambiguity of the infectious *Legionella* concentration. Both this and previous studies adopted a dose–response model developed using guinea pigs rather than humans. Human infectivity requires clinical trials of exposing humans to *Legionella*, which are highly un-likely due to the ethical concerns. Research and data on Legionella infectivity in humans would be useful to further improve the risk assessment. Detailed epidemiology investigations of human exposure to contaminated water and health outcomes could be useful data to refit the human dose–response model.

The results of this study suggest investigations of HPC in cisterns can significantly further the understanding of the water quality and *Legionella* risk since HPC can reflect the condition of cisterns. The HPC is relatively simple to perform, but an identical HPC method should be used for comparison across seasonal and spatial samples. Moreover, identification of *Legionella* to the species level could improve our understanding of the pathogenic vs. non-pathogenic species in water. Additionally, our model does not account for additional *Legionella* that may be growing in the plumbing and showerhead due to biofilm release or the presence of amoeba (De Giglio et al. (2020)). There have also been reported differences in *Legionella* concentrations between the cistern and the in-home faucet due to fluctuations in water age and in uncertain chlorine residuals from chlorinated cisterns, causing pipes to act as *Legionella* reservoirs (Kim et al. (2016)). The growth in the plumbing and showerhead is especially important in stagnant water when the home is abandoned during a time of disaster. However, this situation was not applicable to this study. All households sampled during this study were occupied during the hurricane season because evacuation from an isolated island far from the mainland was more challenging.

It is unclear whether the hurricanes exacerbated the *Legionella* risk due to the lack of historical data on the cistern water quality for the Virgin Islands. The impact of the hurricanes on the water safety in the Virgin Islands may be reflected through the lack of access to chlorine or other disinfection methods after the disaster struck. Regardless of the source of the *Legionella*, the outcomes of this risk analysis suggest the need of a routine water quality monitoring and maintenance program to reduce the risk of *Legionella*. Since rain cisterns are considered private property, a public education program should be put in place to promote the self-monitoring and routine cleaning of the cisterns.

4.4.2 Risk perception and risk management

Although the majority of islanders perceive their water to be safe or somewhat safe for household uses based on the on-site surveys, the QMRA results indicate otherwise. The median annual risk values for both warm and cold showers exceed the EPA recommended threshold of 10^{-4} pppy (per person per year). The perceived water safety may be related to water use patterns because over 90% of the respondents answered that they used bottled water for drinking. Washing water is considered “less risky”, and the aerosol transmission of pathogen through shower mist is not well known. We found that the perceived risk was divided by income levels; twice as many high-income participants deemed their water, ”safe,” whereas most low-income participants answered “somewhat unsafe.” These low-income families may not have access to treatment methods such as filtration, or chlorine or UV light methods of disinfection when electricity is compromised, which was the case during and after the hurricanes. They generally live in older, poorly maintained housing communities with aging water infrastructure and lack of economic resources to perform routine upkeep of the cisterns. They rely more on the cistern water that is freely available, especially in times of crisis.

The large discrepancy between the risks estimated based on the QMRA and the perception of adequate water quality suggests that the prevalence of *Legionella* in cistern water and its risks are not always apparent. Public education and routine monitoring programs are necessary for public health protection. HPC monitoring could be a simple solution for reducing the risk of *Legionella*.

Temperatures in the range of 20°C (68°F) to 45°C (113°F) favor the growth of *Legionella*. Therefore, finding *Legionella* in the cisterns on the Virgin Islands, where the temperature is around 24 °C year-round, is not surprising. *Legionella* can be inactivated when temperatures rise above 50°C (Ditommaso et al. (2011)). Heating water to 60°C is effective at reducing *Legionella* in shower water. However, thermal inactivation is only effective on the heated portion of the water, while *Legionella* may still be present in the cold-water portion mixed to achieve a desired final shower water temperature. In fact, our results showed the warm shower risk was higher than the cold-water risk. This is because, as shown in Table 4.1, a hotter shower produces more aerosols per minute in the shower stall resulting in a higher concentration of aerosols within the shower stall. A shower temperature that is reasonably hot and much higher than room temperature causes a chimney effect in the shower stall, in which aerosols are carried upward with convective flow (Zhou et al. (2007)). A higher aerosolization rate therefore results in more aerosols inhaled for the duration of the shower. The increase in aerosolization between warm- and cold-water showers is up to a factor of 100, and is directly proportional to the increase in the dose for each shower event. Although the high temperature provided by a conventional water heater is adequate for heat inactivation of the bacterium, it is not enough to reduce the ultimate risk. The amount of hot water needed from the water heater to mix with ambient temperature water is low since the ambient water temperature in the Virgin Islands is relatively warm due to the warm, tropical climate. The amount of inactivation for a reduction in the concentration of *Legionella* is small in comparison to the increase in aerosolization.

To better mitigate risk associated with RHRW household use, routine cleaning of cisterns and flushing of premise plumbing should be planned on a fixed schedule to reduce the opportunistic pathogens in shower water. In anticipation of an increase in *Legionella* prevalence in the rainwater by hurricane-induced storms, infrastructure damage (i.e., connection of underground cisterns with surface floodwater), and loss of power, stocking up on chlorine tablets before hurricane season and organizing quick transport and distribution of chlorine tablets immediately after the hurricanes to the disaster area could be helpful to reduce the waterborne and water-related illness. Other water treatment methods, such as UV and reverse osmosis membrane filtration, are less effective in the case of loss of power from electricity grids. Other types of fuel (no natural gas, gasoline is in significant shortage) are hard to access on the islands during and after the hurricanes.

Public education is an important tool to enhance the awareness of *Legionella* risks. Currently, there is no routine monitoring program, cistern water quality standard, nor uniformed recommendations for cistern management on the islands. Cisterns are considered private property; the governmental "interference" on the cistern water was not embraced by the local residents. There is a general mistrust of governmental agencies in advising of water quality and water use. Such mistrust and dissatisfaction among the public was reflected in our survey results, regardless of the household income levels. Public outreach programs using the research outcomes from objective QMRA could instill trust in the local residents about the governmental role in cistern water management. Development of transparent public policy with sufficient time for residents' input and buy-ins are necessary to improve the relationship between the government and the citizens. This trust is critical to improve cistern water quality through monitoring, routine cleaning, and addressing technological treatment requirements. Cistern water quality management is above and beyond the hurricane seasons.

4.5 Conclusion

A QMRA was carried out to estimate the *Legionella* risk through daily showering using untreated cisterns water on St. Thomas, Virgin Islands, following the 2017 hurricane season. The results showed that

- The estimated *Legionella* annual risks exceed the EPA guideline of 10^{-4} pppy.
- The model outputs are sensitive to the concentration of *Legionella* in individual cisterns, which was highly variable and uncertain due to unevenness in the cistern water management on the island.
- There is a disparity between perceived risk and QMRA estimated risk of cistern water, suggesting the *Legionella* risk associated with the shower water is not apparent to the local residents.
- Moreover, the perception of water safety is divided by income group. Most people in the high-income group considered their water "safe", while people in the low-income group only considered their water "somewhat unsafe".
- Both income groups believed that the government could have done more to help them understand the water quality and water safety at the time of natural disaster. A fact-based public education program should be developed to bring residents onboard to manage the cistern water quality collaboratively.

Chapter 5

Application of a dose-response model for risk assessment of antibiotic resistant bacteria: a reverse QMRA for non-potable urban reuse

5.1 Introduction

Antibiotic resistance has become a major threat to public health and modern medicine. The World Health Organization has recognized the need to better understand antibiotic resistance in water and wastewater as an area of concern in their action plan (WHO (2015). Antibiotic resistant bacteria (ARB) numbers continue to increase (Segura et al. (2009)) in accordance with the increase in the use of pharmaceuticals and consequently their abundance in the environment (Khetan and Collins (2007); Adeleye et al. (2022)).

Wastewater treatment plants have been identified as a hotspot for enriching antibiotic resistance and the transmission of ARB and antibiotic resistant genes (ARG) into the environment (Rizzo et al. (2013)). This is due to the inputs of pharmaceuticals and antibiotics from fecal sources, and the conditions of biological treatment including high density of bacteria leading to horizontal gene transfer of ARG (Garner et al. (2021); Rizzo et al. (2013); Guo et al. (2017); Le et al. (2022)). Globally, there has been increased interest in advancing and expanding sustainable water use, including the recycling and reusing of treated wastewater. The most common uses of this type of water source are to flush toilets and urinals, irrigate parks, golf courses, and agriculture, and other uses such as industrial cooling and ornamental fountains or water features (Chen et al. (2013)). Reclaiming wastewater as a non-traditional water source can alleviate water stresses in municipal and agricultural areas alike and diversify water supply portfolios.

The role of wastewater treatment in the prevalence and fate of ARB and ARG is now a research focus area (Bengtsson-Palme et al. (2019); Mao et al. (2015); Jiao et al. (2017)), but the role of reclaimed wastewater in this area is less understood. There is concern that reclaimed wastewater could facilitate the spreading of ARB and ARG and pose a threat to human health due to exposures with any of the aforementioned applications, though there is uncertainty. Several studies have begun to look into reclaimed water and distribution systems (Piña et al. (2020); Fahrenfeld et al. (2013)) but it is noted that further exploration is needed to establish better monitoring, future data availability, and a better understanding of the magnitude of ARB and ARG occurence in environmental applications (Garner et al. (2021); Pepper et al. (2018)).

The application of the Quantitative Microbial Risk Assessment (QMRA) framework in various scenarios of sustainable water uses has been documented (Hajare et al. (2021b)). Through the main steps of hazard identification, exposure assessment, dose-response assessment, and risk characterization, a likelihood of risk (i.e. of infection, illness, or death) can be quantified and better understood. Previously established dose-response models or exposure scenarios are helpful in this framework, especially when paired with a collection of pathogen concentration data for specific locations or water fixtures. Previous QMRA applications for sustainable water sources include the use of reclaimed wastewater for toilet flushing (Hamilton et al. (2018); Lim et al. (2015)), and spray irrigation (Hamilton et al. (2018); Pepper et al. (2018)), irrigation of golf courses (Simhon et al. (2020)), and irrigation of agriculture (Hamilton et al. (2006); Amha et al. (2015); Pang et al. (2017); Van Ginneken and Oron (2000)). However, there is a significant gap in the assessment of risk and exposure to ARB. This is due to both the current limited availability of data characterizing ARB in these reclaimed wastewater systems and the previous lack of models to quantify the human dose-response to these bacteria. Regarding this latter point, a recent approach to ARB dose-response modelling was created by (Chandrasekaran and Jiang (2019)). This model uses simple death kinetics to establish a relationship with existing dose-response models to predict both the risk of response (infection) as well as the likelihood that such an infection is treatable by antibiotics.

The main aim of this study is to apply the ARB DRM to an array of exposures to better understand the risks and risk outcomes across different pathogens, exposure routes, and dose-response relationships. This novel application will shed some light on what is needed to better monitor and regulate pathogens in wastewater treatment, reuse, and reuse applications as well as in the area of antibiotic resistance. Current regulations do not account for ARB, thus their contribution to overall risk is of concern and interest for regulators, water resource planning and decision-making, and the general public who may be at risk. Given these research needs, the goals of this study are to: 1) apply the antibiotic resistant bacteria dose response model to exposure models for non-potable wastewater reuse applications; 2) develop risk-based concentrations of pathogens representative of these applications; 3) compare the concentrations and risks against literature ranges or regulatory measures currently in place for the selected pathogens; 4) assess the risk outcomes beyond the probability of risk including whether the risk is likely to be treatable or untreatable by an antibiotic and suggest areas of improvement for better understanding or monitoring of these pathogens/risks. The results of this study will provide insight on ARB monitoring targets and whether current regulations are sufficient for acceptable health risks.

5.2 Methods

5.2.1 Hazard Identification: Pathogen Selection

In order to explore the comparative impacts of the models and parameters on risk outcomes and estimations across scenarios, four pathogens were chosen for this assessment. *E. coli* and *Campylobacter* were selected as representative pathogens of fecal contamination and indicators, both with infection risks associated with ingestion of contaminated food or water sources. Pathogenic *E. coli* and *Campylobacter* can cause gastroentiritis when ingested, evidenced by outbreaks from infected water storage systems (Palmer et al. (1983); Kuusi et al. (2005)) and irrigated produce (Hilborn et al. (1999); Ackers et al. (1998); Söderström et al. (2005)). *Legionella penumophila* is known to cause respiratory illnesses, namely Legionnaire's Disease, a type of pneumomia, and Pontiac fever (Benedict et al., 2017). *Mycobacterium avium* complex (MAC) is a type of nontuberculosis *mycobacteria* (NTM) commonly found in water and soil and known to cause skin, tissue, and pulmonary infections (Busatto et al. (2019)), accounting for up to 80% of NTM-related pulmonary diseases (Prevots and Marras (2015)). Existing regulations and guidelines from the EPA regarding pathogens in wastewater or reclaimed wastewater are based on fecal indicators (such as *E. coli*), but do not include opportunistic waterborne pathogens such as *Legionella* and MAC (EPA (2012)). *Legionella* and *Mycobacterium* were chosen due to their potential for persistence and regrowth in water distribution systems for non-potable reuse water, and to represent possible inhalation exposure risks. *Legionella* have been detected in reclaimed wastewater systems at concentrations

Table 5.1: Reference concentration ranges for selected pathogens based on regulation or measurements in treated wastewater reuse systems. Concentrations were measured in the United States unless otherwise noted.

Pathogen	Exposure route	Concentration in reclaimed wastewater (CFU/L)	Units	Source
E. coli	Ingestion	22	MPN/L	CA Title 22 (2018)
		10	CFU/L	Jjemba et al. (2010)
Campylobacter	Ingestion	10^1 to 10^4	CFU/L	Farhadkhani et al. (2020)
Legionella	Inhalation	10^5	CFU/L	Caicedo et al. (2019)
		10^6		Whiley et al. (2015)
		10^3		Jjemba et al. (2010)
		10^3 to 10^5		Ajibode et al. (2013)
Mycobacterium avium	Inhalation	10^5	CFU/L	Ajibode et al. (2013)
		10^1 to 3.5×0^2		Jjemba et al. (2010)
		10^8		Whiley et al. (2015)

*measurements from reclaimed wastewater distribution systems in Australia

ranging from 10^3 to 10^5 CFU/L in the U.S. (Ajibode et al. (2013); Jjemba et al. (2010); Johnson et al. (2018)), and up to 10^6 in Australia (Whiley et al. (2015)). *Mycobacterium* concentrations have been measured from 10^1 to 10^5 in the U.S. and as high as 10^8 CFU/L in Australia. Measured reclaimed wastewater concentration ranges or maximums, including current EPA regulation for fecal coliforms, are listed in Table 5.1. to serve as a reference for this reverse QMRA. The results of the infection risks and outcomes will circle back to these measured ranges as guidelines for possible exposures.

5.2.2 Exposure Scenarios and Models

The pathogens mentioned above can cause infections from exposure through either inhalation or ingestion. To best compare between pathogens and exposure routes, three model scenarios were chosen. The exposure scenarios vary in exposure route, as well as how direct and/or often such exposures may take place. The US EPA has minimum water quality requirements for unrestricted urban reuse, and notes that common applications are spray irrigation, urinal and toilet flushing, ornamental fountains and other water features, and cooling tower make-up water (EPA (2012)). Toilet flushing, consumption of produce irrigated with recycled wastewater, and accidental ingestion of water used for irrigation (golf courses and public parks) were chosen. Each scenario was modeled to quantify possibly exposure doses per event (or day), and are outlined below. All calculations were done in MATLAB MATLAB (2022).

Toilet Flushing

The method used to assess infection risks from exposure to aerosols expelled from toilet flushing is based on calculation of the aerosol dose from the concentration of aerosols, as demonstrated by Lim et al. (2015). As adapted by Hamilton et al. (2018), this method includes data from Johnson et al. (2013) for aerosol diameters and concentrations for modern flush toilets. The following equation is used for dose calculation of toilet flushing exposure:

$$Dose_{\mathrm{tf}} = C_{\mathrm{w}} I t_{\mathrm{tf}} \sum_{i=1}^{10} C_{\mathrm{aero,i}} V_{\mathrm{aero,i}} DE_{\mathrm{i}} \tag{5.1}$$

Where $Dose_{\mathrm{tf}}$ is the dose of pathogens inhaled or ingested after one toilet flush (number of pathogens), C_{w} is the concentration of pathogen in the treated and recycled wastewater $\#/m^3$, I is the mean inhalation rate (m^3air/min), t is the duration of exposure (min), $C_{\mathrm{aero,i}}$ is the concentration of aerosols of mean diameter i with diameters from 1 to 10 μm, $V_{\mathrm{aero,i}}$ is the volume of each aerosol of diameter i (calculated as the volume of a sphere, $V_{\mathrm{aero,i}} = 4/3\pi r_i^3$) ($m^3$, and DE_{i} is the deposition efficiency of the aerosols of each size bin i (unitless). For *Legionella* and MAC, the route of exposure is inhalation so the depositon efficiencies are for the alveoloar-bronchiolar region of the lungs. Meanwhile, *E. coli* and *Campylobcater* cause infection through ingestion, so it was assumed that aerosols deposited in the extrathoracic (nasal and laryngeal) region are cleared to the gastrointestinal tract, thus are ingested through this pathway. Different deposition efficiencies for each pathway are utilized here and listed in Table 5.2.

Accidental ingestion: public parks and golf courses

As mentioned previously, reclaimed wastewater is also commonly used to irrigate public areas, namely parks and golf courses. Both of these areas are highly recreational, thus direct contact with irrigation water is possible. This scenario assumes that one could ingest a dose of pathogens through plant contact, such as playing in the grass or handling and cleaning a golf ball, followed by hand-to-mouth motion and transfer. Previous studies have made similar assumptions (Asano et al. (1992); Ryu (2003); Chhipi-Shrestha et al. (2017)). This scenario is more conservative and assumes that these behaviors could lead to a dose as follows:

$$Dose_{\text{acc}} = C_{\text{w}} Vol_{\text{acc}} e^{-k_{\text{solar}} t_{\text{acc}}} \tag{5.2}$$

Where Vol_{acc} is the volume of water ingested (mL) and k_{solar} is the solar decay rate min^{-1} for a time t_{acc} of 12 hours. This inclusion of solar decay assumes that the fields are watered overnight and exposure would occur the next day.

Consumption of irrigation produce

As reclaimed wastewater is also commonly used for irrigation of agriculture, the risk of consuming irrigated produce is also estimated here. For adequate representation of this scenario and the possibility of exposure to irrigation water, consumption of lettuce was chosen as lettuce is consumed raw and irrigation water capture in and on its leaves has been observed. The model applied for this scenario is as follows:

$$Dose_{\text{cons}} = C_{\text{w}} W(BM_{\text{cons}}) 10^{-(t_{\text{hold}}\lambda_{\text{hold}}+\lambda_{\text{wash}})} \tag{5.3}$$

Where $Dose_{\text{cons}}$ is the dose of pathogens ingested through lettuce consumption per exposure (day), W is the amount of water captured on the lettuce leaves from irrigation, B is the average body weight (kg) with M_{cons} the average mass of lettuce consumed per capita per day (g/kg d) based on bodyweight (both from EPA food intake distributions, λ_{hold} is the bacterial decay per day, t_{hold} is the holding or storage time before consumption, and λ_{wash} is the reduction due to washing the lettuce at home. For this final parameter, literature reported that about 88% of people wash their vegetables at home prior to consumption, this the reduction was only applied randomly to this percentage of exposures (Li-Cohen and Bruhn (2002)). Parameters for all exposure models are listed and defined in Table 5.2.

5.2.3 Dose-response

Typically, the dose-response model utilized in QMRA studies is a best-fit single-hit model applied to clinical data, following the exponential function (Equation 5.4) or beta-Poisson function (Equation 5.5).

$$P(d) = 1 - \exp(-r \times d) \tag{5.4}$$

Table 5.2: Parameters for the three exposure models.

Exposure scenario	Definition	Parameter	Value or distribution		Units	Source
Toilet flushing	Inhalation rate	I	U(0.013,0.017)		m^3/min	EPA (2011)
	Time after flush	t_{tf}	U(1,5)		min	Lim et al. (2015)
	Concentration of aerosols of median diameter i (μm)	$C_{aero,i}$	Lognormal distribution		# aerosols/ m^3air	Johnson et al. (2013)
		$i = 1$	$\mu = 10.53, \sigma = 0.87$			
		2	$\mu = 10.43, \sigma = 0.87$			
		3	$\mu = 10.33, \sigma = 0.89$			
		4	$\mu = 10.30, \sigma = 0.90$			
		5	$\mu = 10.31, \sigma = 0.90$			
		6	$\mu = 10.31, \sigma = 0.89$			
		7	$\mu = 10.30, \sigma = 0.90$			
		8	$\mu = 10.30, \sigma = 0.91$			
		9	$\mu = 10.29, \sigma = 0.91$			
		10	$\mu = 10.28, \sigma = 0.91$			
	Deposition efficiency for aerosols of median diameter i	λ_{hold}	Alveolar-bronchiolar	Extrathoracic	Unitless	Heyder et al. (1986)
		$i = 1$	U(0.23,0.25)	U(0,0)		
		2	U(0.40,0.53)	U(0,0)		
		3	U(0.36,0.62)	U(0.01,0.02)		
		4	U(0.29,0.61)	U(0.03,0.08)		
		5	U(0.19,0.52)	U(0.06,0.15)		
		6	U(0.10,0.40)	U(0.07,0.27)		
		7	U(0.06,0.29)	U(0.08,0.38)		
		8	U(0.03,0.19)	U(0.08,0.49)		
		9	U(0.01,0.12)	U(0.08,0.58)		
		10	U(0.01,0.06)	U(0.07,0.65)		
Golf course/public park water ingestion	Volume of water ingested during activity	Vol_{acc}	U(0.9,1.1)		mL	Asano et al. (1992) Ryu (2003) Chhipi-Shrestha et al. (2017)
	Solar decay	k_{solar}	*Campylobacter*	*E. coli*	h^{-1}	Mattioli et al. (2017) Bae and Wuertz (2012)
			0.0625	0.127		
	Time since watering (daytime)	t_{hold}	12		h	Assumption
Consumption of irrigated lettuce	Water holding on lettuce	W	Normal distribution $\mu = 0.108, \sigma = 0.019$		mL/g	Hamilton et al. (2006)
	Body weight (average, adult)	B	U(60,80)		kg	EPA (2003)
	Mass lettuce consumed per capita	M_{cons}	Normal distribution $\mu = 0.219, \sigma = 0.013$		g/kg d	EPA (2003)
	Log reduction from holding time	λ_{hold}	*Campylobacter*	*E. coli*	Unitless	Kärenlampi and Hänninen (2004) Bezanson et al. (2012)
			0.59	0.61		
	Log reduction from washing	λ_{wash}	*Campylobacter*	*E. coli*	Unitless	Singh et al. (2002)
			0.51			
	Lettuce holding time before consumption	t_{hold}	U(0, 2)		d	Mok et al. (2014)

$$P(d) = 1 - \left(1 + \left(\frac{d}{\beta}\right)\right)^{-\alpha} \tag{5.5}$$

. Where d is the dose, P is the probability of response, and r, α, and β are fit parameters.

The dose-response model (DRM) adopted in this study for risk assessment and characterization is the novel model from Chandrasekaran and Jiang (2019). This model is based on simple death kinetics, thus over some time there is a probability of extinction for each bacterium in a population (or dose) once they enter the host. This key assumption also includes the probability that there is a host response (illness or infection) is equal to the probability that there is not a bacterial extinction $1 - P_{\mathrm{ext}}$. The model accounts for the prior implementation of either the exponential or the beta-Poisson DRM as a best-fit to clinical dose-response data. The best fit parameters are used to calculate the simple death rate μ. More detailed methodology of how this is done can be found in Chandrasekaran and Jiang (2019). The data and fit parameters for the traditional DRMs are listed in Table 5.3 below.

The distinction is made between the two populations, antibiotic susceptible bacteria (ASB) and antibiotic resistance bacteria (ARB). By assuming that fitness cost does not affect the death rate of the populations in the presence of antibiotics, the ARB population is then unaffected by AB while the death rate of ASB would increase. Therefore, this model and Equation 5.6. depends on both the fraction of the population that is ARB f_{r} and the presence of any residual concentration of antibiotics C. While the unaffected death rate $\mu = \mu_{\mathrm{ARB}}$ for the resistance population, the increased ASB death rate is calculated by:

$$\mu_{\mathrm{ASB}} = \mu + \frac{E_{\mathrm{max}} C}{EC50 + C} \tag{5.6}$$

From the sigmoidal Emax function, as noted by Nielsen et al. (2011), where μ_{s} is the ASB death rate in the presence of AB (day^{-1}), $E_{\max}$ is the maximum killing rate (day^{-1}), EC50 is the AB concentration. For the pathogens selected in this study, these parameters were obtained from literature of experiments of exposing the bacteria to various antibiotics (Schaper et al. (2005); Park et al. (2022); Lemaire et al. (2009); Ferro et al. (2015)) (Table 5.4). The probabilities of extinction of each population are given by:

$$P_{\mathrm{ext,ASB}} = (1 - \exp(-\mu_{\mathrm{ASB}} t_{\mathrm{fs}}))^{d \times (1 - f_{\mathrm{r}})} \tag{5.7}$$

$$P_{\mathrm{ext,ARB}} = (1 - \exp(-\mu t_{\mathrm{fs}}))^{(d \times f_{\mathrm{r}})} \tag{5.8}$$

Where $P_{\mathrm{ext,ARB}}$ is the probability of extinction of the ARB and P_{ASB} is the probability of extinction of the ASB, based on respective doses d_{ASB} and d_{ARB} and death rates μ_{ASB} and μ_{ARB}. To account for variable population sizes, the fraction of the dose that is ARB $f_{\mathrm{r}}$is multiplied by the dose d, while $1 - f_{\mathrm{r}}$ is applied to d for the estimation of ASB risk. Finally, the probability of infection based on this approach is the complement of the probability of both populations going extinct, given by:

$$P(d) = 1 - P_{\mathrm{ext,ARB}}(d_{\mathrm{ARB}}, C) P_{\mathrm{ext,ASB}}(d_{\mathrm{ASB}}, C) \tag{5.9}$$

A probability of infection was considered for all pathogens and models. For MAC, the DRM and best-fit are based on an endpoint of pulmonary infection, so the dose was reduced by a

Table 5.3: Dose-response parameters, including for previously established DRM fits and the ARB DRM.

Pathogen	DRM parameter	(days)	DRM source	mu (day-1)	$E_{\max}$ (d^{-1}), $EC50$ ($\mu g/L$)	MIC ($\mu g/L$)	Antibiotic	Source
E. coli	$r = 1.07 \times 10^{-8}$	1	Tacket et al. (2000) Chandrasekaran and Jiang (2019)	18.35	50.4, 0.62	1	Cefixime	Schaper et al. (2005)
Campylobacter	$\alpha = 0.145$, $\beta = 7.59$	3	Medema et al. (1996)	Beta distributed	22.53, 4.07	0.5	Gentamicin	Park et al. (2022)
Legionella pneumophila	$r = 0.0599$	2	Armstrong and Haas (2005)	1.4224	0.956, 2.86	0.1	Azithromycin	Lemaire et al. (2009)
Mycobacterium avium	$r = 1.10 \times 10^{-6}$	3	Hamilton et al. (2017c) Tomioka et al. (1993)	4.5734	0.262, 0.149	4	Clarithromycin	Ferro et al. (2015)

factor of 500 to convert the response to a respiratory infection, as this study only considers inhalation route for MAC (Hamilton et al. (2017c)). A table of all parameters for the death rate calculations and dose-response model are listed in Table 5.3.

In addition to the quantification of the probability of response (infection), this ARB DRM by Chandrasekaran & Jiang (2019) has the unique addition of varying outcomes of infection. If the ASB population survives and causes infection, it is likely that the ASB-caused infection would be susceptible (treatable) by the AB being utilized and modelled. However, if that ARB population survives, the infection it causes is likely to be untreatable by the AB. Once a risk of infection is estimated and quantified through QMRA and using this ARB DRM, this final equation comparing probabilities provides a risk outcome of "AB treatable" if:

$$(1 - P_{\text{ext,ASB}})\,(P_{\text{ext,ARB}}) > (1 - P_{\text{ext,ARB}}) \tag{5.10}$$

It holds that if this statement is not true, then the outcome is AB untreatable.

A sensitivity analysis was also conducted on the variable exposure and dose response parameters to assess and identify the parameters which contribute most to the infection risk across pathogens and scenarios. The sensitivity analysis was performed in MATLAB using 10,000 Monte Carlo iterations for the parameters and ranges listed in Table C.1 (MATLAB (2022)). The Spearman rank correlation was used in this case, where the rank is from -1

Table 5.4: Risk characterization parameters for calculating annual risk.

Definition	Parameter	Value			Units
		Toilet flushing[a]	Accidental ingestion (golf course/ public park)[b]	Consumption of irrigated lettuce[c]	
Daily exposures per year	N	365	26	365	Days per year
Daily frequency of exposures	f	5	1	1	Exposures per day

[a]Friedler et al. (1996)

[b]Assumption, biweekly golf or park sessions.

[c]The lettuce consumption data provided by EPA (2003) already includes an estimate of daily consumption, this the rate was applied once per day, annually.

to +1. A rank of 0 means the parameter variation has no influence on the output, +1 is a perfect dependence of the output on the parameter, and -1 means the results are fully dissimilar to parameter variation.

5.2.4 Risk Characterization

Annual risks were calculated with Equation 5.11.

$$P_{\text{annual}} = 1 - \prod_{i=1}^{N_{\text{j}} f_{\text{j}}} (1 - P(d)) \tag{5.11}$$

Where $N_{\mathbf{j}}$ is the number of daily exposures in one year and $f_{\mathbf{j}}$ is daily frequency of the exposure event j.

5.3 Results

The risk results for the toilet flushing scenario are shown in Figure 5.1. This figure shows the annual risk profile over different hypothetical point estimates of pathogen concentration in reclaimed wastewater for all four pathogens. As the exposure parameters were variable, the median annual risk was plotted against the concentration for all risk results. Variations in f_{r} and C (as a percent of the minimum inhibitory concentration, or MIC) are plotted for each pathogen as well. A similar illustration of results is shown for the accidental ingestion and lettuce consumption models for *E. coli* and *Campylobacter* in Figure 5.2.

The toilet flushing annual risks were low (far less than the EPA recommended threshold of 10^{-4} per person per year (pppy)) for all *E. coli* and MAC, even at high concentrations (>10^4 pathogens/L). The *Legionella* annual risk for inhalation of flushwater aerosols is comparable with other QMRA studies, passing the 10^{-4} pppy threshold at around 10^4 CFU/L. This is to be expected as the exposure model is based on other similar studies and the variation of parameters is relatively consistent (Lim et al. (2015); Hamilton et al. (2018)). The *Campylobacter* risk (triangles in Figure 5.1.) is similar to but slightly lower than the *Legionella* risk, falling above 10^{-4} pppy at 4×10^4 CFU/L.

To assess the impacts of ARB on the risk and the risk outcomes, the fraction of ARB in the total bacterial dose (f_{r}) and the concentration of residual AB (C) were also varied. A change from red to black in each figure signifies a change in the risk outcome from AB Treatable to AB Untreatable. In Figure 5.1a, by keeping the AB constant at 5% of MIC, but increasing f_{r}, the infection risk most noticeably increases for *E. coli*, by nearly one order of magnitude. The opposite is true for the case of holding f_{r} constant, but increasing the AB concentration, C (Figure 5.1b) in which the *E. coli* risk decreases by nearly two orders of magnitude. In some cases, these variations produced a differing outcome of AB untreatable across all concentrations, as seen in Figure 5.1a for the higher ARB fractions of *E. coli*,

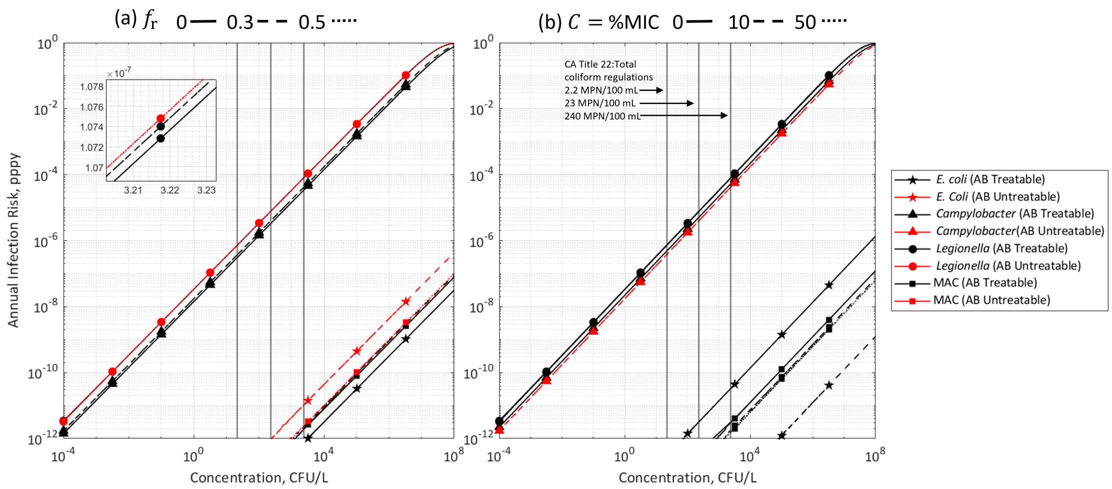

Figure 5.1: Annual infection risk for the toilet flushing for *E. coli*, *Campylobacter*, *Legionella*, and MAC. For (a), the AB concentration was kept constant at 5% MIC while displaying different ARB fractions, f_r. The f_r in (b) was held constant at 0.05.

MAC, and *Legionella*. Thus, a higher fraction of ARB at low AB concentrations leads to a higher infection risk and a worse, untreatble by the AB in question, outcome. Increasing this concentration (Figure 5.1b) can also lead to an untreatable outcome, but a lesser risk, due to higher ASB extinction at the higher C.

The annual risk results for the two ingestion-based models are shown in Figure 5.2. The annual risk for *E. coli* through lettuce consumption was over 10^{-4} pppy at 74.8 CFU/L and 5.8 CFU/L for f_r of 0 and 0.3, respectively. The annual risk for *E. coli* through accidental gold course or park ingestion exceeded 10^{-4} pppy for the same fractions at 616.3 CFU/L and 44.5 CFU/L, respectively. This corresponds to exceeding the EPA annual risk benchmark at concentrations under the 7-day regulation for fecal coliforms as set by the EPA and CA Title 22 for unrestricted urban reuse. These results and calculations assume that the entire concentration is pathogenic, which is not the case for total coliform detection. As before, increasing the ARB fraction increased the risk (by about half an order of magnitude) and led to an untreatable outcome for *E. coli* in both ingestion scenarios. Despite conservative probabilities, the significance of the impacts on annual risk by varying the ARB fraction should be taken into account, especially for *E. coli*. The *Campylobacter* risk for both scenarios was considerably higher, exceeding the benchmark at 1.9×10^{-4} CFU/L for the golf/park scenario and 1×10^{-7} for lettuce consumption. The changes in f_{r} or C had less of an impact on the quantitative risk, but it is apparent that at higher concentrations, ($> 10/L$, or around the regulation of 22 MPN/100 mL) the infection became untreatable.

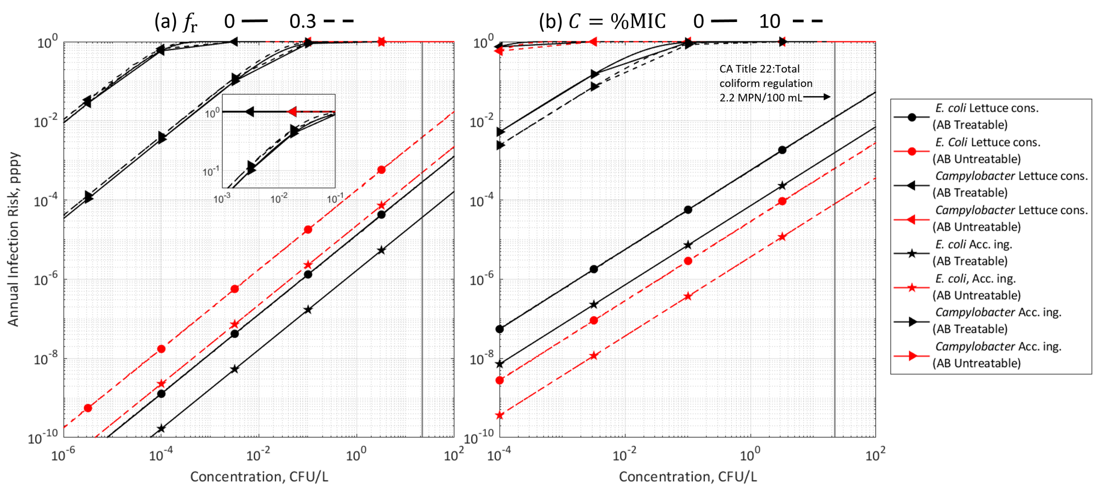

Figure 5.2: Annual infection risk *E.coli* and *Campylobacter* for the ingestion-based scenarios: consumption of irrigated lettuce and accidental ingestion at an irrigated golf course or public park. For (a), the AB concentration was kept constant at 5% MIC while displaying different ARB fractions, f_{r}. The f_{r} in (b) was held constant at 0.05.

For the toilet flushing scenario, the concentration (C_{w}) and the volume of water inhaled (expressed as the summation of aerosol concentration and deposition efficiency per size bin) were the most influential parameters (Figure 5.3). The time spent in the bathroom after flushing was also an important indicator of risk (> 0.4). The volume of water ingested at the golf course or park was critical in influencing the risk for that scenario (> 0.5). The final scenario (lettuce consumption) was far less sensitive to variation in other parameters than to the concentration of pathogens in the water itself (C_{w}). The risk of *E. coli* proved to be the most sensitive to changes in the ARB DRM parameters f_{r} and C, while the other pathogens were far less affected. This is due to the reduced impact of these changes on the ARB and ASB death rates for each respective pathogen and model, based on the chosen AB and killing rate parameters (Equation 5.6 and Table 5.2).

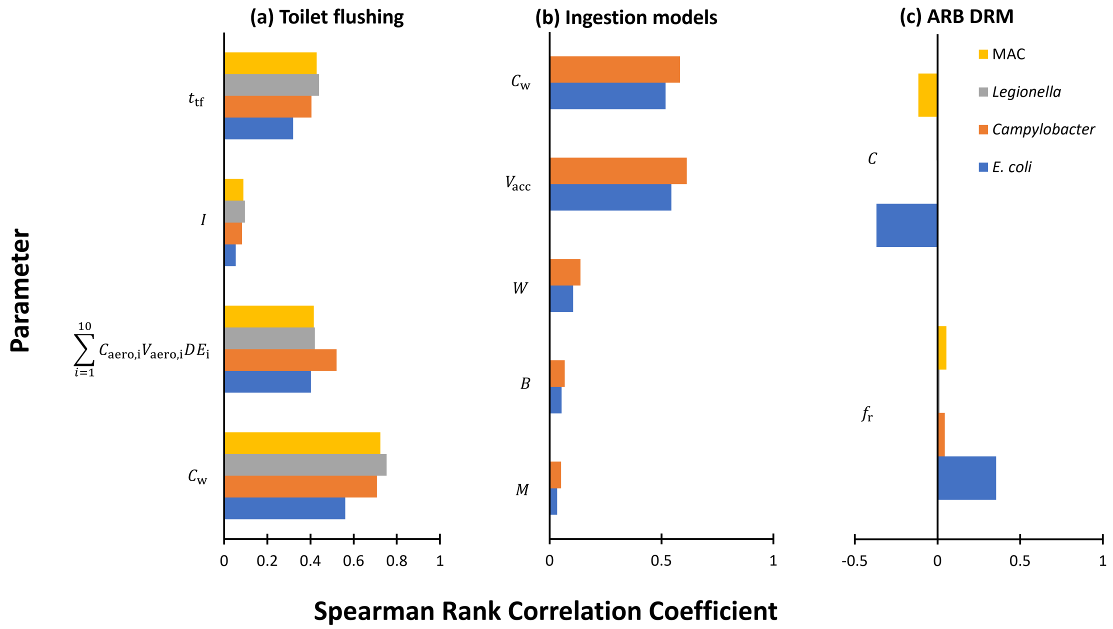

Figure 5.3: Sensitivity analysis for exposure and ARB dose-response parameters.

To further explore this final aspect of sensitivity (Figure 5.3c), a graph showing the binary heatmap of all combinations of f_r between 0 and 0.25 and C between 0 and 10% MIC and the risk outcome is shown in Figure 5.4. It is apparent that at single doses, based on the AB-specific kinetic parameters utilzed in this study, these variations impacted *E. coli* the most. In other words, at lower combinations of either parameter, the risk outcome at the illustrated low or high dose sooner became untreatble. The dose itself had a more pronounced impact on *Legionella* and *Campylobacter*, effectively shifting this critical untreatable boundary down to a much lower fraction of ARB (< 0.1).

5.4 Discussion

This QMRA was aimed at implementing the new Simple Death dose-response model to water reuse exposure scenarios for different pathogens with different exposure routes and param-

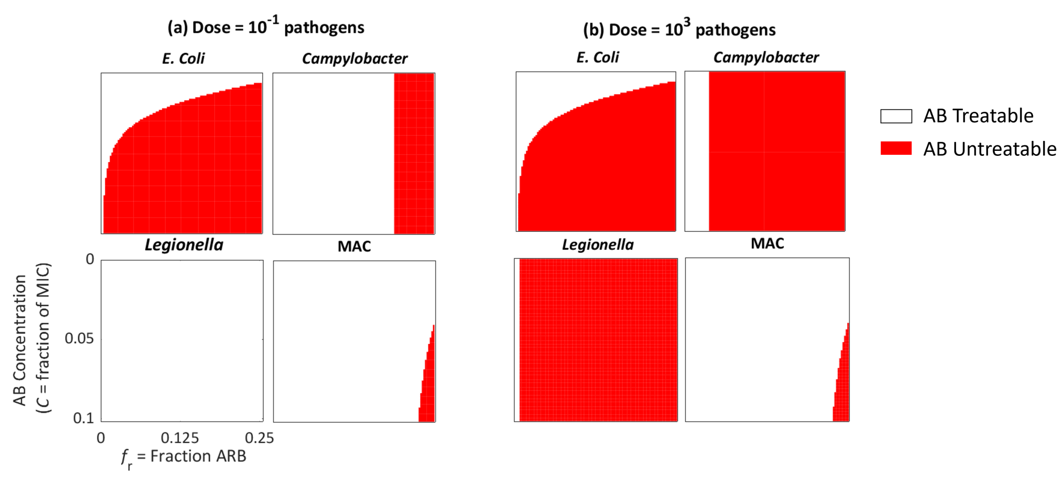

Figure 5.4: Illustration of the binary outcomes of infection (AB treatable or untreatable) by varying f_r and C for constant doses of all four pathogens.

eters. The risks of each pathogen presented in this study has been characterized before for various applications of water reuse or alternative water sources. Examples include the assessment of *E. coli* in agriculture, toilet flushing, and garden irrigation (Kouamé et al. (2017); Hajare et al. (2021a)), *Campylobacter* through water wells and rainwater harvesting (Hora et al. (2017); Murphy et al. (2017)), and *Legionella* and MAC through inhalation of aerosols for different non-potable water fixtures (Hamilton et al. (2017b); Blanky et al. (2017); Quon et al. (2021a)). However, the majority of these studies are not for large scale non-potable urban reuse due to uncertainties in pathogen concentrations beyond the current regulatory measures of monitoring total coliforms (EPA (2012)). Unrestricted urban reuse water is used in public areas. The exposure scenarios selected here were chosen to demonstrate possible exposures to the general public. Toilet flushing and produce consumption are ubiquitous behaviors. Accidental ingestion of irrigation water can affect different populations which may be more at risk such as older people (golf courses) or children (public parks).

The advantage of utilizing this Simple Death DRM is that it works well with the QMRA framework and existing dose-response models. The fundamental approach of QMRA and utilization of exponential and beta-Poisson models and fit parameters, which are established based on data for many pathogens, is essentially unchanged and adapted here. The ability to still estimate the dose and risk per exposure event allows for this model to be used to quantify the combined risk of pathogens or multuiple exposures/water fixtures, assuming that the risk for each is independent. However, this model relies on information on the kinetics of ARB in the presence of AB, which are AB-specific and require empirical data, which is a gap in this area. In addition, it is well accepted that bacteria which mutate to acquire antibiotic resistance often do so at a fitness cost, or a higher death rate (Gagneux et al. (2006); Martinez and Baquero (2000); Schrag et al. (1997)). However, it can also occur at little-to-no cost (Melnyk et al. (2015)), or even greater fitness than prior to the mutation (Blot et al. (1994)). While these gaps and assumptions lead to a more conservative approach, the model is adaptable to the proposed changes, as each population or death rate (such as in the case of fitness-increased death rate for some ARB) can be included while maintaining the model structure.

The risk of *Campylobacter* was high, particularly in the two ingestion scenarios (Figure 5.2) in which nearly all concentrations corresponded to an annual risk well above 10^{-4}. The calculated doses for each scenario were situationally the same between *Campylobacter* and *E. coli*, yet the *Campylobacter* risk was higher. This is due in part to its dose-response parameters, in which a much lower dose contributes to a higher likelihood of illness, and to the reduced sensitivity to ARB in the overall risk (Figure 5.3). Based on the DRM and AB-specific parameters utilized here based on *Campylobacter jejuni*, (Medema et al. (1996); Park et al. (2022)) the effect of the AB gentamicin on the overall death rate at low concentrations was less than half that for *E. coli* with cefixime ($E_{\max}$ of 22.5 vs 50.4 d^{-1}). This demonstrated a more profound impact of the concentration on the overall risk and outcomes (Figure 5.4) than changes in f_{r} and C. While monitoring and regulations in the U.S. center on a limit of total coliforms (7-day median of ≤ 2.2MPN/100mL), the presence of these bacteria may not accurately predict the presence of pathogenic microorganisms. Bonetta et al. (2016) isolated pathogenic *E. coli* and *Campylobacter* in wastewater, but the total coliform count will generally not be harmful to humans (Edberg et al. (2000)). It is unlikely that *Campylobacter* itself would exceed this concentration, but concentrations as low as 10^{-1}/L corresponded to annual risk approaching 1 (Figure 5.2). Thus, even at concentrations of 1% of the total coliform limit (of 22 MPN/L), there is high risk of infection. Farhadkhani et al. (2020) detected *Campylobacter* in concentrations of 0.2-10 CFU/mL in treated wastewater samples, and even found no correlation between the concentration of *E. coli* and the presence of *Campylobacter* in their samples. Our results indicating that exposure to irrigated crops or public areas is a possible route of exposure for campylobacteriosis, which is known to cause an estimated 1.5 million illnesses every year in the United States (CDC (2021)).

Currently, the main uses of reclaimed wastewater of toilet flushing, spray irrigation for both public areas and agriculture, and even ornamental fountains and cooling towers all generate aerosols, which can be transported and inhaled. Therefore, the inhalation route of exposure is important to consider, including pathogens which cause respiratory illnesses, such as *Legionella* and MAC presented in this study. These pathogens are not regulated for non-potable reuse (U.S. EPA 2012) and their occurence in reclaimed wastewater distribution systems has been observed (Table 5.2) (Caicedo et al. (2019); Whiley et al. (2015); Jjemba et al. (2010); Ajibode et al. (2013)). Transport, distribution, and storage of reclaimed wastewater see decreased water quality when compared with the final effluent at the point of treatment, or the point of compliance when it comes to monitoring. This is due to loss of disinfectant residuals, residence time, and differences in temperature (Johnson et al. (2018)). These factors also contribute to the growth of biofilms in distribution pipes, and all contribute to the presence and persistence of *Legionella* and *Mycobacterium* in these networks and supplies (Lau and Ashbolt (2009)). Just as wastewater treatment plants have been identified as hotspots for propogating ARB, the abundance and biodiversity of bacterial populations in these biofilms appears to drive the antibiotic resistance within them (Brienza et al. (2022)). Both the effect of (Pappa et al. (2020); Carter et al. (2004)) and the resistance against (Jia et al. (2019); Heifets et al. (1993)). the antibiotics azithromycin and clarithromycin used in this study (Table 5.3) have been observed for *Legionella* and *Mycobacteria*, respectively. However, there is still a lack of robust data for ARB in reclaimed wastewater, particularly including opportunistic pathogens and biofilms in distribution systems, due in part to a lack of regulation. Further examination and data collection is recommended to advance risk assessment in this area, especially given the non-trivial risk of *Legionella* in the inhalation scenario presented here (Figure 5.1).

The risk outcomes including in this risk assessment are a unique result, as it supplements the quantified annual risk results (pppy). The outcomes were affected by the ARB fraction f_{r} and the AB concentration C. Variations in the ARB fraction were below 50% in this study, but some studies have seen resistance in *E. coli* strains up to 95% in treated wastewater effluent and reclaimed wastewater samples (Aslan et al. (2018); Pignato et al. (2009)). The annual risk was the same when there is no antibiotic (C) present in the host body, as the death rates are unaffected (comparison illustrated in the Supplementary Information, Figure C.1). The development of infections that are untreatable by antibiotic is depending on the trace level of antibiotic in the human host. There is a critical antibiotic concentration for the transition from a treatable to an untreatable infection (Figure 5.4). Based on the selected pathogens and antibiotics in this study, ARB were more influential in the overall risk and risk outcomes for *E. coli* (Figures 5.3 & 5.4). This is due to its higher death rate and larger impact of C on increasing the death rate for ASB. Although the trace antibiotic concentration C lowers the overall infection risk (Figures 5.1 & 5.2), its presence increases the risk of developing an infection that is not treatable by the antibiotic. Residual antibiotics, such as those found in food or water (Anthony A et al. (2018); Jin et al. (2022)) can be the source of these trace antibiotics in the body, in addition to use of antibiotics for past infections or surgeries. These trace levels have not been reported, except for studies in the hours or days following treatment or injection. These clinical studies, however, suggest that the antibiotics are rapidly cleared out of the body on the timescale of hours based on plasma detection (Meng et al. (2005); Liu et al. (2002); Taninaka et al. (2000)). In our study, the antibiotic concentration C was varied at low concentrations to demonstrate its impact on the results. Therefore, the overall results of this study indicate that high concentration of pathogen exposure and high fraction of ARB increase the overall risk of infection, while the residual concentration of antibiotic in human system impact the transition from antibiotic treatable to an antibiotic untreatable infection. Therefore, the outcomes of this study call for further anaylsis of water quality at the point of use and the consideration of a new risk threshold for antibiotic untreatable infections and for better protection of human health.

5.5 Conclusion

A novel dose-response model for the inclusion of antibiotic-resistant bacteria was applied to a quantitative microbial risk assessment for the first time across three different exposure models and four different pathogens. The annual risk of scenarios was calculated across a wide range of microbial concentrations for applications of reclaimed wastewater for unrestricted reuse. Given the differences in exposure route and infection pathway, the scenarios were modeled in parallel for final comparison, including for the possibility of the infection resulting in being able to be treated or not by antibiotics. Based on this, the conclusions are:

- The highest risk was for *Campylobacter* through consumption of lettuce or accidental ingestion at irrigated public areas, even at low concentrations.
- The development of antibiotic resistant infection is a considerable threat to consumers of irrigated lettuce and groups who utilize irrigated public areas, such as children.
- The risk of infection was most sensitive to the concentration of pathogens in the reclaimed water used for each scenario.

- The fraction of ARB has a much higher impact on quantitative risk for *E. coli* than for the other pathogens.

Chapter 6

Conclusions

6.1 Summary

The broad questions and objectives outlined in Chapter I concern non-traditional water sources and application of both new and existing models to identify critical areas of improved understanding. Frameworks were outlined for each project, which included both location-specific case studies, wide comparative assessments, and new approaches to better understand how a non-traditional water source may be perceived or fit in future water supplies. In some cases, existing data sets were utilized, fitted, or modelled, demonstrating that quantitative approaches can still be developed and improved upon with "old" data and to reaffirm where future data may be needed. In Chapter V, the framework was reversed given the lack of empirical data, and reflected upon with literature and illustrations of critical parameters for future areas to highlight. These various objectives and individual projects served to demonstrate that there is no "one-size-fits-all" supply of water that yet satisfies the niche of a sustainable water future, as there are a number of quantity and quality metrics, health concerns, and costs associated with each.

I began with seawater as a non-traditional water source in Chapter III. After reviewing the main sources in Chapter II, I focused on the cost and energy requirements as a major hurdle for seawater reverse osmosis (SWRO). This amounted to the use of a new technoeconomic assessment model and platform called WaterTAP3 for a comparison across four facilities to try and pinpoint cost discrepancies and sensitive areas. After selecting one facility in Israel and one nearly identical process design in the U.S., the model was unable to single out a process or technological difference that amounted to the variation in levelized cost of water (LCOW), concluding that the overall capital cost of SWRO is impacted by localized factors: labor, materials, electricity cost structure, namely land and permitting requirements which were not yet accurately included in the model. I found that the LCOW was also highly sensitive to plant capacity utilization, that is, if the plant is operated seasonally rather than year-round, the LCOW is quickly and dramatically increased. Therefore, an oversized facility in which production would be halted in off seasons is not economically productive. Reduced operation and increased downtime could also lead to costs associated with membrane fouling, cleaning, and replacement. An approach using drought data and conservation cost curves found further "local" value and reliability aspects for SWRO that would vary by local climate and water supplies. This top down look at large scale SWRO facilities demonstrated the hidden costs of design beyond the technological process and the impacts of local factors that require further investigation.

Chapter IV transitioned from seawater to rainwater as the target water source. The case study and location-specific approach was preserved through looking at the late-2017 disastrous hurricane season in the Virgin Islands. Metagenomic data collection and a socioeconomic survey were conducted in tandem to assess the water quality impacts of the hurricanes and local perception and water use. High prevalence of *Legionella* prompted further investigation. I conducted a QMRA using the prevlance data in conjunction with an older dataset of plate counts to produce a range of possible concentrations for the untreated rain cistern water. Showering was used to represent a likely daily exposure to the non-potable water.

The survey responses were further analyzed for any significance related to income, water use, government intervention opinions, and other post-hurricane measures related to their safety. The findings of this study were high health risks, exceeding the EPA risk guideline and an overall disparity between this estimated risk and the local percieved risk (survey results), suggesting that the health concerns were not readily apparent to the local residents.Both high and low income groups believed that the government could have done more to help them understand the water quality and water safety at the time of natural disaster. A fact-based public education program should be developed to bring residents onboard to manage the cistern water quality collaboratively and more consistently.

Finally in Chapter V I demonstrated the first implementation of a novel dose-response model for including antibiotic resistant bacteria through a wide QMRA approach. In this case, the utilization of treated reclaimed wastewater as a source of non-potable water was under assessment. Based on the health concerns and increased research around antibiotics and antibiotic resistant bacteria in both wastewater and reclaimed water, a QMRA was the best framework for this study. As with Chapter III, I thought a broader and comparative approach across different scenarios would serve best to pinpoint critical areas and sensitivities for different antibiotic-bacteria combinations. Three different exposure scenarios were modelled based on some of the most common uses of reclaimed wastewater, and based on the state of California's and the US EPA's guidelines around "unrestricted urban reuse." Toilet flushing, irrigation of produce, and irrigation of public areas were chosen, and two pathogens each for ingestion risk (*E. coli* and *Campylobacter*) and inhalation risk (*Legionella* and *Mycobacterium avium*) were selected. The different infection exposure routes, dose response parameters, concentration ranges, and pharmacodynamics of each pathogen and scenario led to a wide set of results with different outcomes. The highest risk was for *Campylobacter* through consumption of lettuce or accidental ingestion at irrigated public areas, even at low concentrations. The Simple Death model included the unique risk outcome distinction of the quantified infection being treatable or not by antibiotics, after the given exposure and resistance parameters.

The development of an antibiotic resistant infection is a considerable threat to consumers of irrigated lettuce and groups who utilize irrigated public areas, such as children. The overall risk of infection was most sensitive to the concentration of pathogens in the reclaimed water used for each scenario and the fraction of ARB had a much higher impact on quantitative risk for *E. coli* than for the other pathogens. The outcomes of this study call for further analysis of water quality at the point of use and the consideration of a new risk threshold for antibiotic untreatable infections and for better protection of human health.

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Appendix A

Pipe Parity Analysis of Seawater Desalination in the United States: Exploring Costs, Energy, and Reliability via Case Studies and Scenarios of Emerging Technology

Table A.1: Conservation supply relationship based on cost categories, conservation potential as percent of baseline demand, and estimated conservation cost.

Cost Category	**Example Measures**	**% Demand (reduction of demand)**	**Cost (c) ($/m3)**
Zero	Ineffective irrigation, simple repairs	5%	0
Low Cost	Efficient fixtures, irrigation management	10%	0.1 to 1
Moderate Cost	Landscape conversion, minor infrastructure	15%	1 to 3
High Cost	Major infrastructure	10%	3 to 7.5

Table A.2: Existing and planned ocean desalination facilities in the US considered in case study.

Facility	Location	State	Capacity (MGD)	Year implemented	Source
Monterey Bay Aquarium	Monterey Bay	CA	0.04	late 1990s	Aquarium (2014)
San Nicolas Island	San Nicolas Island	CA	0.042	1990	Gorman (1990)
Sand City	Sand City	CA	0.27	2010	Herrera (2019)
Marina Coast Water District	Marina	CA	0.3	1997	MCWD (2020)
Gaviota Oil Heating Facility	Gaviota	CA	0.3		**?**
Pebbly Beach	Catalina	CA	0.325		Board (2017)
Marathon	Florida Keys	FL	1	1960s	Pearson (2010)
Morro Bay	Morro Bay	CA	1.2	1992 SWRO, 2009 BWRO	Wilson (2017)
Diablo Canyon Nuclear Power Plant	Avila Beach	CA	1.5		Cunningham (2019)
Stock Island	Key West	FL	2	1960s	Harn R/O Systems (2020)
Charles E. Meyer	Santa Barbara	CA	3		PWD (2020)
Claude "Bud" Lewis	Carlsbad	CA	50		June 15 (2012)
Tampa Bay Water	Tampa	FL	25	2007	District (2020)
Cape May	Cape May	NJ	2.98		Hurdle (2020)
Cape Coral	Cape Coral	FL	15		Harvey and Missimer (2020)
Caribbean Islands	Various				Balch (2015)
Morro Bay Power Plant	Morro Bay	CA	20		Staff (2017)
Moss Landing	Moss Landing	CA	15 to 28		Adamson (2015)
Huntington Beach	Huntington Beach	CA	50	2023	Water (2021)
Corpus Christi	Corpus Christi	TX	20-30	2025	of Corpus Christi (2020)
Brownsville Pilot Demonstration	Brownsville	TX	2.5	2020s	Norris (2008)
Brownsville Implementation	Brownsville	TX	25	2060s	Norris (2008)
Yacht Haven Water Cooperative	San Juan County	WA	0.01		WRM (2009)

A.1 Water-TAP3 model details

The Water-TAP3 model inputs specific to the case studies presented in the main text are provided in Tables A.5-A.9. Other model details not specific to specific case studies, including

Table A.3: Cost and energy data collected from literature and Global Water Intelligence (GWI) reports for case study facilities.

Facility	Data source	CAPEX ($MM)	OPEX ($/yr)	Electricity (kWh/m^3)	LCOW ($\$/m^3$)
Ashkelon	Literature value	272*		3.8	0.66
	GWI DesalData	561.3		3.8	0.53
	GWI Cost Estimator Tool				
Carlsbad	Literature value	1003	49 to 54	3.56 to 3.97	1.61
	GWI DesalData	646		3.3	1.61
	GWI Cost Estimator Tool		31.5	3.3	0.93
Tampa Bay	Literature value	197**			0.6
	GWI DesalData	158		3.0	0.66
	GWI Cost Estimator Tool	167.7	15.7	3.0	1.18
Santa Barbara	Literature value	106***	4.1	3.6	1.08
	GWI DesalData	78.8			1.08
	GWI Cost Estimator Tool	22	2	3.0	1.28

*CAPEX for 2006 construction inflated to 2020 cost. Refit not included in this estimate due to lack of source.
**Combined $110MM cost plus $48MM for remediation, inflated to 2020 cost.
***Combined cost of 1991 construction plus 2017 refit, inflated to 2020 cost.

chemical costs and unit-level cost calculations, are provided in the main manuscript are also documented in this supplementary material section.

For the baseline case studies, water treatment recoveries were known from facility data and used as constraints in the model (Ashkelon: 44%, Carlsbad: 50%, Santa Barbara: 50%, Tampa Bay: 56%).

Table A.4: Additional information for case study facilities reports for case study facilities.

	Facility	Ashkelon	Carlsbad - Claude "Bud" Lewis	Santa Barbara - Charles E. Meyer	Tampa Bay
Demographic Info	Per capita water use (gal/day)	60	100	100	100
	Average household income (2018)	53,000 USD	90,000 USD	90,000 USD	70,000 USD
	residential water price ($/kgal)	13	10	15	7
	residential wastewater price ($/kgal)	4	7	5	6
	residential total ($/kgal)	17	17	20	13
Water Buyer Info	Service Population	9 million	3 million	0.1 million	2.5 million
	Number of Member Agencies	n/a	29	n/a	6
	Regional Planning Entity	Israel Water Authority	SDWA, City and County of San Diego	County of Santa Barbara	Southwest Florida WMD
	Drought-prone	yes	yes	yes	no
	Water supply stresses	Drought; no other new supply sources	Drought; dependence on imported water	Drought; dependence on surface water storage	GW depletion; seawater intrusion

Table A.5: Source water flow rate by case study.

Case Study	Source Water	Flow (m3/s)
Ashkelon	Seawater	7.782
Carlsbad	Seawater	4.583
Santa Barbara	Seawater	0.309
Tampa Bay	Seawater	1.928

A.2 Assumptions used in water reduction analysis.

The following assumptions were used in the estimation of water supply reduction R:

Table A.6: Source water constituent levels (mg/L) by case study.

Constituent	Ashkelon	Carlsbad	Santa Barbara	Tampa Bay
Boron	5	4	3	3
Bromide	75	66	66	56
Calcium	468	409	409	351
Chloride	21899	19162	19162	16424
Magnesium	1460	1278	1278	1095
Potassium	451	395	395	338
Sodium	12204	10679	10679	9153
Strontium	1	1	1	1
Sulfate	3062	2680	2680	2297
TDS	40700	35000	35000	30000
TSS	34	30	30	25

Table A.7: Technoeconomic assumptions by case study for the entire treatment train.

Variable	Ashkelon	Carlsbad	Santa Barbara	Tampa Bay
Analysis Year	2020	2020	2020	2020
Location Basis	Israel	California	California	Florida
Plant Life Years	20	20	20	20
Land Cost Percent	0.0015	0.0015	0.0015	0.0015
Working Capital Percent	0.05	0.05	0.05	0.05
Salaries Percent	0.001	0.001	0.001	0.0015
Employee Benefits Percent	0.9	0.9	0.9	0.9
Maintenance Cost Percent	0.008	0.008	0.008	0.008
Laboratory Fees Percent	0.003	0.003	0.003	0.003
Insurance and Taxes Percent	0.002	0.002	0.002	0.002
Default Cap Scaling Exp.	0.7	0.7	0.7	0.7
Default Opex Scaling Exp.	0.7	0.7	0.7	0.7
Cap. by Equity	0	0	0	0
Debt Interest Rate	0.05	0.05	0.05	0.05
Expected Return on Equity	0.1	0.1	0.1	0.1
Default TPEC Multiplier	3.4	3.4	3.4	3.4
Default TIC Multiplier	1.65	1.65	1.65	1.65
Base Salary Per Fixed Cap Inv.	0.0005	0.0005	0.0005	0.0005
Plant Capacity Utilization	1	1	1	0.75

- The probability of the occurrence of drought conditions for each drought category is equal to the average percent of land assigned to that category. For example, if the 20-year average percentage of land assigned to category D1 is 10%, then we assume that the probability of a D1-level drought is 10%. While this is clearly an over-simplification, it is a way to use the available data to provide useful rough estimates of drought probability.

- For each USDM drought category we define a factor rj which represents the percent reduction of conventional supply that would occur if the entire state were in the drought category. While these factors are difficult to calculate precisely, we constructed a simple order-of-magnitude estimate in two steps. First, because drought impacts accumulate in a nonlinear fashion, we assumed that $r_j = 2r_j - 1$ and that $r_5 = 0.5$. This implies that, if the entire state of California were in the highest drought category D4, then supply would be reduced by 50%.

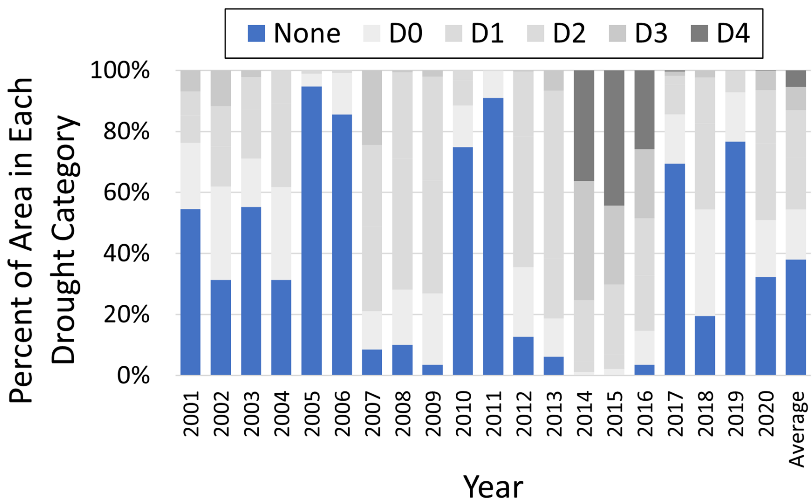

Figure A.1: The percentage of land in California that fell under each US Drought Monitor (USDM) drought category (D0-D4), measured over the years 2001-2020.

A.3 Details of conservation supply relationship.

The conservation supply estimates summarized in Table A.1 are based on a study of efficiency programs in California Cooley et al. (2019). The moderate and high-cost categories are associated with conversion to drought-tolerant landscapes. Porse et al. (2018) estimated measure costs that range from $\$1 - 7.5/m^3$ Porse et al. (2018). Here these are broken into two categories, $\$1 - 3/m^3$ and $\$3 - 7.5/m^3$. The percent of normal-year demand reduction associated with each measure is estimated based on existing studies for a total conservation potential of 40% Buck et al. (2016). This is consistent with data for San Diego and Santa Barbara, where under conditions of severe drought saw a maximum of about 40% demand reduction. For a specific water district, the relevant cost measure is the cost of conserving the next unit of water since the district may have already implemented conservation measures. Hence, each water district will be at a distinct point along the conservation supply curve; the unique value appropriate to that district is called the marginal cost of conservation.

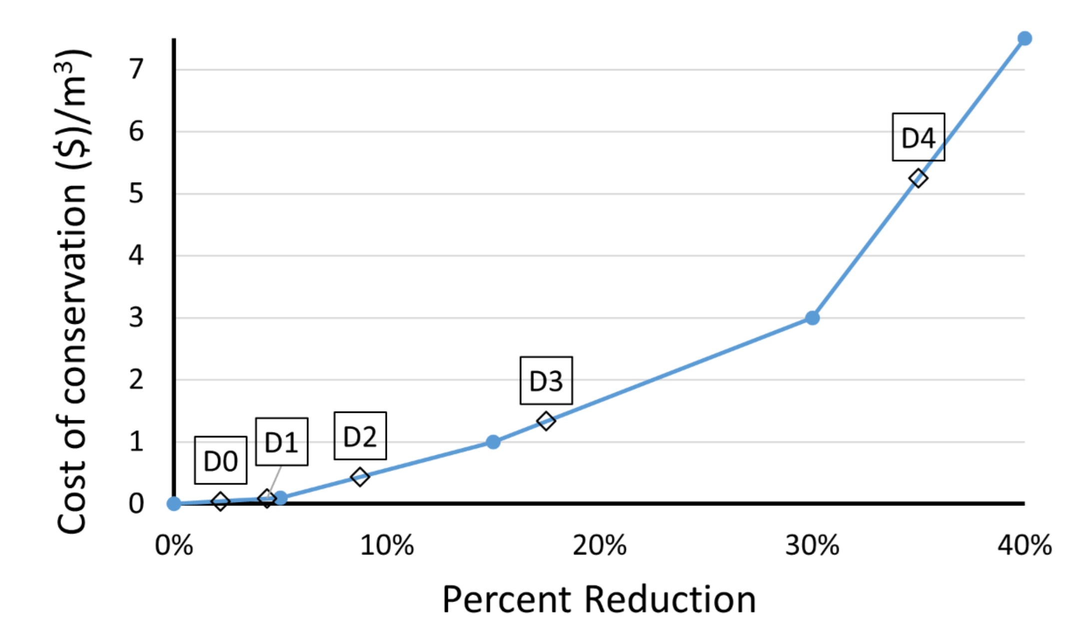

Figure A.2: Conservation supply curve illustrated as the cost of implementing various tiered categories of conservation based on the estimations and assumptions from Table A.1. Drought category labels mark the typical cost tier associated with that level of drought.

Table A.8: Water recovery (%) and constituent removal (%) model results. All factors are constants unless marked with an asterisk.

Case Study		TSS	Water Recovery	Bromide	Calcium	Chloride	Magnesium	Potassium	Sodium	Strontium	Sulfate	TDS	Boron
Ashkelon	SW Onshore Intake	0	100	0	0	0	0	0	0	0	0	0	0
	Sulfuric Acid Addition	0	100	0	0	0	0	0	0	0	0	0	0
	Ferric Chloride Addition	0	100	0	0	0	0	0	0	0	0	0	0
	Chlorination	0	100	0	0	0	0	0	0	0	0	0	0
	Static Mixer	0	100	0	0	0	0	0	0	0	0	0	0
	Tri Media Filtration	95	90	0	0	0	0	0	0	0	0	0	0
	Cartridge Filtration	50	100	0	0	0	0	0	0	0	0	0	0
	RO First Pass*	99.3	24.9	99.3	99.3	99.3	99.3	99.3	99.3	99.3	99.3	99.9	0
	RO Second Pass*	99.3	32	99.3	99.3	99.3	99.3	99.3	99.3	99.3	99.3	99.8	0
	Lime Softening	0	99.9	0	0	0	0	0	0	0	0	0	0
	Chlorination B	0	100	0	0	0	0	0	0	0	0	0	0
	Caustic Soda Addition	0	100	0	0	0	0	0	0	0	0	0	0
	Treated Storage	0	100	0	0	0	0	0	0	0	0	0	0
	Backwash Solids Handling	95	95	0	0	0	0	0	0	0	0	0	0
	Municipal Drinking	0	100	0	0	0	0	0	0	0	0	0	0
	Surface Discharge	0	100	0	0	0	0	0	0	0	0	0	0
	Landfill	0	100	0	0	0	0	0	0	0	0	0	0
Carlsbad	SW Onshore Intake	0	100	0	0	0	0	0	0	0	0	0	0
	Sulfuric Acid Addition	0	100	0	0	0	0	0	0	0	0	0	0
	Ferric Chloride Addition	0	100	0	0	0	0	0	0	0	0	0	0
	Chlorination	0	100	0	0	0	0	0	0	0	0	0	0
	Static Mixer	0	100	0	0	0	0	0	0	0	0	0	0
	Tri Media Filtration	95	90	0	0	0	0	0	0	0	0	0	0
	Cartridge Filtration	50	100	0	0	0	0	0	0	0	0	0	0
	RO First Pass*	99	31.8	99.2	99.2	99.2	99.2	99.2	99.2	99.2	99.2	99.8	0
	RO Second Pass*	99	34.9	99.2	99.2	99.2	99.2	99.2	99.2	99.2	99.2	99.7	0
	Lime Softening	0	99.9	0	0	0	0	0	0	0	0	0	0
	Chlorination B	0	100	0	0	0	0	0	0	0	0	0	0
	Caustic Soda Addition	0	100	0	0	0	0	0	0	0	0	0	0

Table A.9: Water recovery (%) and constituent removal (%) model results. All factors are constants unless marked with an asterisk, cont'd.

Treated Storage	0	100	0	0	0	0	0	0	0	0	0	0	
	Backwash Solids Handling	95	95	0	0	0	0	0	0	0	0	0	0
	Municipal Drinking	0	100	0	0	0	0	0	0	0	0	0	0
	Surface Discharge	0	100	0	0	0	0	0	0	0	0	0	0
	Landfill	0	100	0	0	0	0	0	0	0	0	0	0
Santa Barbara	SW Onshore Intake	0	100	0	0	0	0	0	0	0	0	0	0
	Ferric Chloride Addition	0	100	0	0	0	0	0	0	0	0	0	0
	Chlorination	0	100	0	0	0	0	0	0	0	0	0	0
	Static Mixer	0	100	0	0	0	0	0	0	0	0	0	0
	Holding Tank	0	100	0	0	0	0	0	0	0	0	0	0
	Media Filtration	50	100	0	0	0	0	0	0	0	0	0	0
	Anti Scalant Addition	0	100	0	0	0	0	0	0	0	0	0	0
	Cartridge Filtration	50	100	0	0	0	0	0	0	0	0	0	0
	Reverse Osmosis	99	50.1	99.3	99.3	99.3	99.3	99.3	99.3	99.3	99.3	99.6	0
	Holding Tank B	0	100	0	0	0	0	0	0	0	0	0	0
	UV AOP	0	100	0	0	0	0	0	0	0	0	0	0
	CO2 Addition	0	100	0	0	0	0	0	0	0	0	0	0
	Lime Addition	0	100	0	0	0	0	0	0	0	0	0	0
	Treated Storage	0	100	0	0	0	0	0	0	0	0	0	0
	Backwash Solids Handling	95	95	0	0	0	0	0	0	0	0	0	0
	Landfill	0	100	0	0	0	0	0	0	0	0	0	0
	Municipal Drinking	0	100	0	0	0	0	0	0	0	0	0	0

Table A.10: Water recovery (%) and constituent removal (%) model results. All factors are constants unless marked with an asterisk, cont'd.

Tampa Bay	SW Onshore Intake	0	100	0	0	0	0	0	0	0	0	0	0
	Sulfuric Acid Addition	0	100	0	0	0	0	0	0	0	0	0	0
	Ferric Chloride Addition	0	100	0	0	0	0	0	0	0	0	0	0
	Chlorination	0	100	0	0	0	0	0	0	0	0	0	0
	Static Mixer	0	100	0	0	0	0	0	0	0	0	0	0
	Tri Media Filtration	95	90	0	0	0	0	0	0	0	0	0	0
	Cartridge Filtration	50	100	0	0	0	0	0	0	0	0	0	0
	RO First Pass*	99	62.8	99.1	99.1	99.1	99.1	99.1	99.1	99.1	99.1	99.5	0
	RO Second Pass*	99	97.8	99.1	99.1	99.1	99.1	99.1	99.1	99.1	99.1	93.2	0
	Lime Softening	0	99.9	0	0	0	0	0	0	0	0	0	0
	Chlorination B	0	100	0	0	0	0	0	0	0	0	0	0
	Caustic Soda Addition	0	100	0	0	0	0	0	0	0	0	0	0
	Ammonia Addition	0	100	0	0	0	0	0	0	0	0	0	0
	Treated Storage	0	100	0	0	0	0	0	0	0	0	0	0
	Backwash Solids Handling	95	95	0	0	0	0	0	0	0	0	0	0
	Municipal Drinking	0	100	0	0	0	0	0	0	0	0	0	0
	Surface Discharge	0	100	0	0	0	0	0	0	0	0	0	0
	Landfill	0	100	0	0	0	0	0	0	0	0	0	0

Table A.11: System and unit level configuration assumptions.

Case Study	Unit Process	Parameter
Ashkelon	SW Onshore Intake	Water Type: Seawater
	Sulfuric Acid Addition	Dose: 10
	Ferric Chloride Addition	Dose: 20
	Chlorination	Chemical Name: Chlorine
	Static Mixer	
	Tri Media Filtration	
	Cartridge Filtration	
	RO First Pass*	ERD: No
	RO Second Pass*	ERD: Yes
	Lime Softening	Lime Dose: 2.3
	Chlorination B	Chemical Name: Chlorine
	Caustic Soda Addition	Dose: 30
	Treated Storage	Hours: 1
	Backwash Solids Handling	Recovery: 0.95
	Municipal Drinking	
	Surface Discharge	Pump: No
	Landfill	
Carlsbad	SW Onshore Intake	Water Type: Seawater
	Sulfuric Acid Addition	Dose: 10
	Ferric Chloride Addition	Dose: 20
	Chlorination	Chemical Name: Chlorine
	Static Mixer	
	Tri Media Filtration	
	Cartridge Filtration	
	RO First Pass*	ERD: No
	RO Second Pass*	ERD: Yes
	Lime Softening	Lime Dose: 2.3
	Chlorination B	Chemical Name: Chlorine
	Caustic Soda Addition	Dose: 30
	Treated Storage	Hours: 1
	Backwash Solids Handling	Recovery: 0.95
	Municipal Drinking	
	Surface Discharge	Pump: No
	Landfill	

Table A.12: System and unit level configuration assumptions, cont'd.

Santa Barbara	SW Onshore Intake	Water Type: Seawater
	Ferric Chloride Addition	Dose: 20
	Chlorination	Chemical Name: Chlorine
	Static Mixer	
	Holding Tank	Hours: 2
	Media Filtration	
	Anti Scalant Addition	Dose: 5
	Cartridge Filtration	
	Reverse Osmosis	ERD: Yes
	Holding Tank B	Hours: 1
	UV AOP	Chemical Name: Hydrogen Peroxide; Dose: 5; UV Dose: 350; AOP: True; UVT In: 0.95
	CO2 Addition	
	Lime Addition	Lime: 2.3
	Treated Storage	Hours: 1
	Backwash Solids Handling	
	Landfill	
	Municipal Drinking	
Tampa Bay	SW Onshore Intake	Water Type: Seawater
	Sulfuric Acid Addition	Dose: 10
	Ferric Chloride Addition	Dose: 20
	Chlorination	Chemical Name: Chlorine
	Static Mixer	
	Tri Media Filtration	
	Cartridge Filtration	
	RO First Pass*	ERD: Yes; Split Fraction: 0.67, 0.33
	RO Second Pass*	ERD: Yes
	Lime Softening	Lime Dose: 2.3
	Chlorination B	Chemical Name: Chlorine
	Caustic Soda Addition	Dose: 30
	Ammonia Addition	Dose: 3
	Treated Storage	Hours: 6
	Backwash Solids Handling	Recovery: 0.95
	Municipal Drinking	
	Surface Discharge	
	Landfill	

Appendix B

Assessing the Risk of *Legionella* Infection through Showering with Untreated Rain Cistern Water in a Tropical Environment

Table B.1: Aerosol inhalation deposition rates in the alveolar-bronchilar region of the lungs (Zhou, Benson, Irvin, Irshad, & Cheng, 2007).

	Hot shower inhalation deposition rate (mg/min)		Cold shower inhalation deposition rate (mg/min)	
Showerhead flow rate (L/min)	Nasal Breathing	Oral Breathing	Nasal Breathing	Oral Breathing
5.1	0.036	0.297	0.002	0.005
6.6	0.049	0.357	0.003	0.008
9	0.044	0.364	0.001	0.007

UCI University of California, Irvine

Household Water Resource Survey
Date: ______________
Location: ___________________

Survey Information:
The University of California, Irvine is conducting a survey about Hurricanes Irma and Maria. Your responses will help us understand how people are coping with the disaster impacts. The study aims to find ways to improve disaster preparation and recovery efforts. All information you provide will be kept confidential. The interview will last about 30 mins. You have to be 18 or older to participate. If at any time you wish to stop the interview or not answer a specific question, this is entirely up to you. If you have any questions or concerns about the study, please contact Dr. Sunny Jiang of the University of California, Irvine at sjiang@uci.edu or 949-824-5527.

1. After the hurricanes, do you have running water in your home?
   - o Yes
   - o No

2. After the hurricanes, where do you now get your water from for daily use (check all that apply)?
   - o Tap water
   - o Bottled water
   - o Collect rainwater
   - o Use water at neighbor's home
   - o Other sources; please list: ____________________________

3. If you get tap water, what do you use it for (check all that apply)?
   - o Drinking without treatment
   - o Drinking after boiling
   - o Drinking after other treatments in my house
   - o Washing hands
   - o Brushing teeth
   - o Washing dishes
   - o Washing food that eating raw
   - o Showering and bathing

4. Do you use bottled water for any of the following (check all that apply)?
   - o Drinking
   - o Brushing teeth
   - o Washing dishes
   - o Washing food that eating raw
   - o Washing hands

5. If you use bottled water, how much do you spend?
   $ _____ per week
   Do you wait in line to get bottled water? ______ minutes waiting in line each week

6. How many hours per day that you have running water in your tap in the past two weeks?
   - Less than two hours per day
   - Less than 4 hours per day
   - Between 4 and 23 hours per day
   - 24 hours per day

7. If you use tap water for drinking or cooking, do you treat it?
   - Boil water Time per day: ____ minutes
   - Add chlorine (or other chemical)
   - Use a water filter Cost of filter: $______
   - Other, please list: ____

8. Do you store water?
   - No
   - Yes → If yes, how do you store water: ☐ In bathtub ☐ In sink ☐ In containers

9. How safe do you think your water is?
   - Very unsafe (serious health risk)
   - Somewhat unsafe (some health risk)
   - Safe (no health risk)

10. Does your water look dirty or taste bad (check all that apply)?
    - Looks bad
    - Tastes bad
    - Neither

11. Do you think the government has done enough to let you know the safety of the water?
    - Yes
    - No

12. Do you think the government has done enough to provide you with the safe water supply?
    - Yes
    - No

13. Have you heard of the advisory from the government for boiling water?
    - Yes
    - No

14. Normally, what is your monthly water bill? $___ per month

15. Please rank the issues below that you think government should address first after the events of Irma and Maria (rank 1 to 5; 1 is the most important).
    - ____ Restore communication (cell phone, internet)
    - ____ Restore electricity
    - ____ Make sure local water is safe for people to use
    - ____ Restore roads and transportation
    - ____ Restore police service and security situation

Your household

16. How many people in your household? ____ people
    How many adults (18+ years old): ____ adults
    How many children: ____ children

17. Has anyone in your household been sick that you think might be related to food or water?
    - Yes → How many days were they sick? _____ days
    - No

18. What is your annual income for the entire household? (Normal income, before the hurricanes)
    - less than $10,000
    - $10,001- 25,000
    - $25,001- 40,000
    - $40,001- 60,000
    - $60,001- 80,000
    - more than $80,000

19. What is the main source of income in your household (check all that apply)?
    - Food services
    - Hotel services
    - Business owners of service industry
    - Tourist services
    - Other

20. What type of income do you have?
    - Wage (hourly or daily)
    - Salary (fixed amount, each year)
    - Own your business

21. Has anyone in your household been out of work after hurricanes without pay?
    - No
    - Yes → If yes: How many people are out of work? ______ people
      For how many weeks? ________ weeks
      How much income have they lost? $ ________ per week

22. Have you completed:
    - Elementary school (8th grade)
    - High school
    - College
23. Type of home you live in:
    - Apartment
    - Single-family home, detached
    - Multiple family home; townhouse; condo
    - Other, please list: ______________________________

24. Do you own or rent your home:
    - Own
    - Rent
    - It's my relatives or friends' home
    - Other

25. How many times have you been affected by a hurricane (not counting the hurricanes this year)? _____ times

Impacts of Hurricanes

26. Were any of your motor vehicles damaged during Hurricanes Irma and Maria?
    - No
    - Yes → If yes: Vehicle type: ☐ Car ☐ Truck ☐ Motorcycle
      Age of vehicle: ____ years
      Cost to repair: $______________
      Too damaged to be repaired (completely destroyed)? ☐ No ☐ Yes

27. Have you repaired, cleaned, or replaced items in your home?

	Time your household spent cleaning & repairing (days)	**Money** you spent to clean, repair, and replace ($)
TOTAL you've spent:	**____ days**	**$ ______**
☐ Furniture (*Tables, chairs, sofa, beds, cabinets*)	____ days	$ ______
☐ Appliances (*refrigerator, stove, dishwasher, washing machine, clothes dryer*)	____ days	$ ______
☐ Fans, air-conditioner, lighting, electrical wiring, plumbing, septic	____ days	$ ______
☐ Electronics: television, computer	____ days	$ ______
☐ Clothing; Kitchenware (pots, plates)	____ days	$ ______

☐ Flooring, carpet, walls, doors, windows	____ days	$ ______
☐ Roof; exterior or interior painting; garden	____ days	$ ______
☐Other, please list: _____________	____ days	$ ______

28. What items did you need to buy or spend money on because of the disasters?

	Total Spent ($)	**Time** Waiting in Line
☐ Generator	$	______ minutes
☐ Fuel for generator	$	______ minutes
☐ Batteries, Solar powered devices, lighting, flashlights	$	______ minutes
☐Gas for cooking; cookstove; barbecue grill	$	______ minutes
☐ Food	$	______ minutes
☐Water filter, water treatment (chlorine, other chemical)	$	______ minutes
☐ Container to store water	$	______ minutes
☐Tents; other temporary dwelling	$	______ minutes
☐ Other, please specify 1…………………………………………………. 2…………………………………………………. 3………………………………………………….		

29. Did your household receive aid (check all that apply)?

Who provided aid:	**Aid received:**
☐ Government *(U.S. Government; Virgin Islands; Federal Emergency Management Agency)*	☐ Money: $ __________ ☐ Loan: $ __________ ☐ Supplies or services
☐ Charity or Non-profit *(Red Cross, church group, etc.)*	☐ Money: $ __________ ☐ Loan: $ __________ ☐ Supplies or services
☐ Family or friends ☐ Neighbors; Community	☐ Money: $ __________ ☐ Loan: $ __________ ☐ Supplies or services
☐ Your employer	☐ Money: $ __________

	☐ Loan: $ __________
	☐ Supplies or services
☐ Flood insurance	Money received from insurance: $ ________
☐ Bank	☐ Loan: $ ________

30. Did you evacuate from your home due to the hurricane?
   - ○ No
   - ○ Yes → If yes: How long did you leave your home? ____ weeks
     Have your returned to your home? ☐ No ☐ Yes
     Why did you evacuate (check all that apply)?
     - ☐ No electricity in home
     - ☐ No water in home
     - ☐ Crime
     - ☐ Home too damaged to live in
     - ☐ Other, please list: ___________________________

Are you willing to answer 5 minutes of follow-up questions, 4 months from now? If so, please let us know how to contact you: Email: ______________ Cell phone: _______________.

Thank you again for participating!

UCI University of California, Irvine

If you have questions or concerns about your rights as a research participant, you can contact the UCI Institutional Review Board by phone, (949) 824-6662, by e-mail at IRB@research.uci.edu or at 141 Innovation, Suite 250, Irvine, CA 92697.

What is an IRB? An Institutional Review Board (IRB) is a committee made up of scientists and non-scientists. The IRB's role is to protect the rights and welfare of human subjects involved in research. The IRB also assures that the research complies with applicable regulations, laws, and institutional policies.

Appendix C

Application of a dose-response model for risk assessment of antibiotic resistant bacteria: a reverse QMRA for non-potable urban reuse

Figure C.1 below illustrates a comparison of risk results after direcly inputting various doses into the cited and previously established exponential or beta-Poisson models and the results after using the Simple Death model from Chandrasekaran and Jiang (2019). The results are nearly identical, which is to be expected as there was no inclusion of antibiotics or antibiotic resistant bacteria.

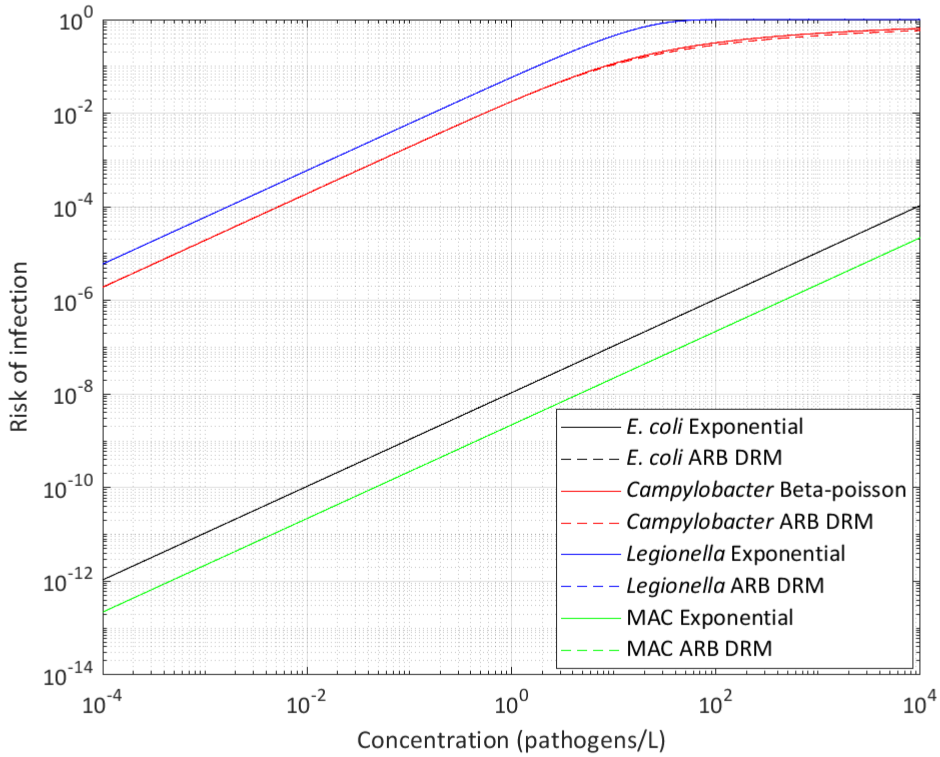

Figure C.1: Comparison of risk results between the ARB DRM applied in this study and the cited exponential or beta-Poisson fits previously used for each pathogen.

Table C.1: Parameter ranges for sensitivity analysis.

Parameter	**Units**	**Lower bound**	**Upper bound**
t_{tf}	min	1	5
I	m$^3/min$	0.013	0.017
$C_{\mathrm{aero,i}}$	aerosols/m$^3 air$	See Table 1.	
DE_{i}	Unitless		
C_{w}	pathogens/m^3	10^{-2}	10^6
Vol_{acc}	mL	0.9	1.1
W	ug/mL	Normal distribution $\mu = 0.108$ $\sigma = 0.019$	
B	kg	60	80
M_{cons}	g/kg d	Normal distribution $\mu = 0.219$ $\sigma = 0.013$	
f_{r}	Unitless	0	0.3
C	ug/mL	0	0.1