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Improving Local Water Supply Resilience and Reliability During Drought: Computational Analysis and Identification of Alternative Water Management Issues and Solutions Under Deterministic and Stochastic Environments

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Improving Local Water Supply Resilience and Reliability During Drought: Computational Analysis and Identification of Alternative Water Management Issues and Solutions Under Deterministic and Stochastic Environments

A Dissertation submitted in partial satisfaction
of the requirements for the degree of

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by

Quynh Kim Tran

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Dedication

This dissertation is dedicated to amazing my parents, my aunt, uncle, my in-laws, and my beloved husband. Your sacrifice for my better future has led me to where I am today. Thank you for your unconditional love and unwavering support during the past years. I am nothing without you.

ABSTRACT OF THE DISSERTATION

Improving Local Water Supply Resilience and Reliability During Drought:
Computational Analysis and Identification of Alternative Water Management Issues and Solutions Under Deterministic and Stochastic Environments

by

Quynh Kim Tran

Doctor of Philosophy, Graduate Program in Chemical and Environmental Engineering
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Dr. David Jassby, Co-Chairperson
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Water scarcity has become a critical problem in many semi-arid and arid regions. California is located in the arid southwest and is expected to experience more frequent and intense droughts under climate change. Currently, residents of Southern California rely on groundwater and imported water from both the state water project, which transports water from the Bay-Delta, and Colorado River Aqueduct. With concerns over current and future levels of water availability, municipalities and state governments are focusing significantly more attention and resources towards groundwater management strategies and alternative water supplies via desalination and the reuse of municipal wastewater. While the reuse of treated wastewater is not a new concept, concerns over the rising demand for water from population growth, coupled with challenges—both economic and environmental—confronting agencies in their efforts to appropriate new supplies, have made this option significantly more attractive. Consequently, the reuse of treated wastewater presents

municipal water and irrigation agencies, including farmers, with the possibility of a low-cost, reliable and environmentally friendly local water source whose value will only increase under expected climate change conditions.

Currently, information relating to groundwater extraction, groundwater use, managed and natural recharge throughout California is limited. Unconstrained use of this source has led to groundwater table depletion, land subsidence, and impact of water quality. Groundwater depletion and degradation of groundwater aquifers results from a lack of effective governance. Moreover, climate change conditions have an immediate impact on the natural recharge in some regions. The coupling of climate change and a growing population presents a challenge to sustainable management of groundwater resources; as demand increases and recharge decreases groundwater levels drop, which results in increased production costs and potential depletion. Thus, municipalities are exploring adding additional resources to their resource portfolios, with cost, quality, and reliability concerns serving as guidelines in this quest. Desalination water, therefore, may become an answer to water shortages due to continual depletion of many groundwater basin and unreliability of imported water—amplified by the climate change.

This research will build upon current research and further explore water supply alternatives that are intended to improve local water supply reliability and provide greater resilience to drought and climate change conditions by evaluating different wastewater treatment technologies and their impacts on municipal wastewater effluents for varying degrees of agricultural applications. In addition, the research will also study the impacts of droughts on municipal wastewater quality and treatment technologies in compliance with state and federal regulations. This research is designed to give water agencies the tools needed to make informed management decisions under future water and climate uncertainties.

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Introduction

INTRODUCTION AND MOTIVATION

Water Resources in Southern California

Water resources are under increasing stress due to excessive population growth, climate change and its impact on the frequency and intensity of drought. As residential and commercial water demands increase alongside little concerted or coordinated effort to recharge groundwater aquifers, groundwater basins become heavily depleted. Therefore, it is important to identify and evaluate the role other water supply alternatives might play in improving local water supply reliability and resilience to cope with water shortages.

The main water sources supplied to Southern California residents are groundwater and imported water from State Water Project (SWP) and Colorado River via the Colorado River Aqueduct (CRA). In fact, during the drought confronting California in 2014, only 5% of the SWP allocation water was delivered to agencies in southern California. This is one of the driest years in California's history[1]. Decreases in surface water supply due to prolonged drought have increased reliance on groundwater. This in turn, impacted groundwater availability and/or have increased pumping costs due to deeper water levels. Consequently, opportunities to augment groundwater supplies through the reuse of wastewater and desalination are increasingly being explored and adopted.

Currently, groundwater is an inexpensive source of water relative to other options[2]. For instance, the cost of pumping groundwater in San Bernardino is approximately $147.61/AF in 2015.[3] However, as groundwater becomes limited, recycled water or treated wastewater and desalination water become more attractive water supply alternatives due to their availability locally and cost[4-10]. Recycled water, on the other hand, is much cheaper than desalination water[2, 9, 10]. An average unit costs of disinfected, tertiary treated recycled water and desalination water are approximately \$900/AF[2] and \$2,100/AF[9, 10] in 2015 U.S. dollars, respectively. Currently, recycled water can only be used for non-potable purposes, such as irrigation, groundwater recharge, and for maintaining local stream flows[11, 12]. Recycled water becomes significantly more attractive, especially during drought when the availability, and, hence, reliability of imported surface water supplies decreases substaintially[2].

In terms of who might benefit the most from the further development of recycled water, irrigated agriculture is one of the largest consumers of water, especially in arid and semi-arid regions[13]. In fact, during the 2012-2015 drought period, imported water supply decreased significantly[14-22]; farmers, in response, over-drafted groundwater systems, reduced water application rates, fallowed additional lands but also explored opportunities to augment supplies from nontraditional sources of water. The reuse of treated wastewater for irrigation presents water agencies and farmers with the possibility of a low-cost, reliable and environmentally friendly local water source whose value will only increase under expected climate change conditions[13].

Nevertheless, increasing augmentation through the use of treated municipal wastewater, especially during drought periods, results in the generation of a more concentrated wastewater stream with elevated concentrations of total dissolved solids (TDS), nitrogen species, and carbon[23, 24]. Conventional wastewater treatment plants are not designed to remove certain constituents effectively; thus, wastewater quality is likely impacted, which can subsequently impact streams and groundwater systems depending on disposal source[25, 26]. Therefore, it is important to consider the economics and sustainability of investing in recent treatment technologies to mitigate such impacts.

In addition to alternative water supply options, recycled water is also being used for indirect potable reuse (IPR) purpose. The Groundwater Replenish System (GWRS) in Southern California is the world's largest water purification for this purpose[27-36]. It takes secondary treated recycled water (i.e., wastewater passes through the primary screening process and is then conveyed to the activated sludge where it breaks down organic matters) from Orange County Sanitation District (OCSD) that would have been previously discharged to the ocean and purifies it using advance treatment steps[29]. The produced treated water exceeds all state and federal drinking water standards[29] via microfiltration, reverse osmosis, and Ultraviolet light with hydrogen peroxide. Half of this water is used to recharge basins in Anaheim, California where it will serve residents of north and central Orange County while the other half is being pumped into injection wells where it serves as seawater intrusion barriers[34]. GWRS is leading the way to provide cost-effective, drought-proof, high-quality, and locally reliable water supply under future uncertainty.

Moreover, as groundwater and SWP water resources becomes limited, desalination water may be needed to supply freshwater to arid regions[7, 8]. Desalination water is abundant and locally available but comes with significant economic and environmental costs. While this process is energy intensive to remove unwanted constituents to meet drinking water standards[9, 10], desalination will increasingly be the answer to water shortages under future water and climate uncertainties[7-10].

Wastewater Reuse for Agriculture

Due to current and future freshwater availability, water districts are paying more serious attention toward the reuse of municipal wastewater[4]. While the reuse of treated wastewater is not a new concept, concerns over the rising demand for water from population growth, coupled with both economic and environmental challenges have made the reuse of treated wastewater significantly more attractive[4-6]. For instance, the estimated costs for Metropolitan Water District (MWD) to recycle treated sewage water is approximately $900 per acre-foot, significantly less than the current cost of $1,400 per acre-foot cost for imported water[42]. In addition to offering a less costly alternative to imported water, reuse of treated wastewater is also seen as a more environmentally friendly and locally reliable option, particularly during drought when the reliability of imported supplies decreases substantially. During the drought confronting California in 2014, only 5% of the SWP allocation water was delivered to agencies in southern California. The allocations have varied significantly from 80% of the full allocation in 2011 (a relatively wet year) down to only 5% in 2014, one of the driest years in California's history[1]. Consequently, the reuse of treated wastewater for irrigation, which constitutes 80-90% of the nation's consumptive water use[13], presents water agencies and farmers with the possibility of a low-cost, reliable and environmentally friendly local water source whose value will only increase under expected climate change conditions.

Currently, most wastewater treatment plants in California treat municipal wastewater to secondary or tertiary standards, followed by disinfection and discharge of the treated effluent to the ocean[6]. Increasingly, disinfected secondary and tertiary effluents are used to irrigate restricted and unrestricted access irrigation areas, such as golf courses and freeway medians. However, secondary and tertiary treated wastewaters are suitable for highly salt-tolerant crops only, such as turfgrass, and are rarely used on crops that are consumed raw by humans due to high level of salts and pathogens[12]. Treated wastewater can also contribute an appreciable amount of the necessary nutrients for plants, thereby reducing the need for synthetic fertilizers and the associated costs[43]. The suitability of reclaimed wastewater for specific applications depends on water quality and usage requirements. The main factors that impact the suitability of recycled water for irrigation are salinity, heavy metals, and pathogens, which may cause adverse effects on human, plants and soil health.

Groundwater Management

Currently, groundwater is the lowest-cost water source in Southern California, with the cost to produce this water dependent on its quality and the depth from which it is withdrawn[2, 44]. Nonetheless, there is limited collaboration on groundwater management between agencies that share an aquifer[45-47]. Information relating to groundwater extraction, groundwater use, managed and natural recharge throughout California is limited[46]. Unconstrained use of this source has led to groundwater table depletion, land subsidence, and impact of water quality[45]. Groundwater depletion and degradation of groundwater aquifers results from a lack of effective governance[47].

On the other hand, due to climate change conditions, patterns of precipitation and storm events are also likely to change[37, 38]. Less precipitation is expected in some regions while some may have more or no change in precipitation. Extreme variations/intensity in average annual and seasonal precipitations are also observed[39]. As snow is collected on the mountaintops during winter season, water is gradually released and infiltrate the groundwater basins during spring and summer months to ensure water storage throughout the year. However, as recharge shifts from snowmelt-dominated to rainfall-dominated events[40], the amount of natural recharge is expected to decrease and water shortages are expected to be more frequent[41]. The major groundwater recharge source in mountain regions of the western U.S. is natural precipitation via snowmelt and rain. As a result, groundwater resources are reduced, and municipalities are presented with management challenges to sustain groundwater resources. As demand increases and recharge decreases groundwater levels drop, which results in increased production costs and potential depletion[40].

The increased withdrawals resulted in rapid depletion of groundwater resources, and led to the passage of the landmark Sustainable Groundwater Management Act (SGMA)[48]. SGMA is created to raise awareness of the importance of groundwater resources and promote groundwater management to achieve the sustainable management of groundwater resources in California[46, 47]. SGMA requires governments and water agencies to bring groundwater basins into balanced levels of pumping and recharge. Before SGMA, there was no statewide governance regulating groundwater pumping. Effective and successful groundwater management requires significant efforts, commitment, and collaboration from water managers, public water agencies, and communities to protect and sustain groundwater resources, especially under future water uncertainty[45, 47].

Wastewater Treatment Technologies

Conventional municipal wastewater treatment plants typically use primary, secondary, tertiary, and disinfection processes to meet state and federal regulations[49-62]. Primary treatment stage includes the mechanical screening of large and coarse objects, followed by a clarifying step where large inorganic particles settle out. Water from the clarifier continues to the secondary treatment stage consisting of biological reactors (such as activated sludge or membrane bioreactors (MBRs)) coupled to another clarification stage[63]. In MBRs, clarification is achieved through the incorporation of membranes inside the reactors[64, 65]. The purpose of the secondary treatment is to break down nutrients and organic matters biologically, typically through the use of activated sludge. Effluent from the secondary clarifier is either disinfected and discharged to surface water or further treated with tertiary treatment such as granular filtration, membrane separation and/or advanced oxidation processes (AOP), followed by disinfection (chlorine, UV, or Ozone)[61].

Recently, due to more stringent state and federal regulations, increasing water demand, and dwindling water resources, membrane separation processes, such as reverse osmosis (RO), are now accepted worldwide[12]. This combined physical and chemical technology is based on size exclusion, charge

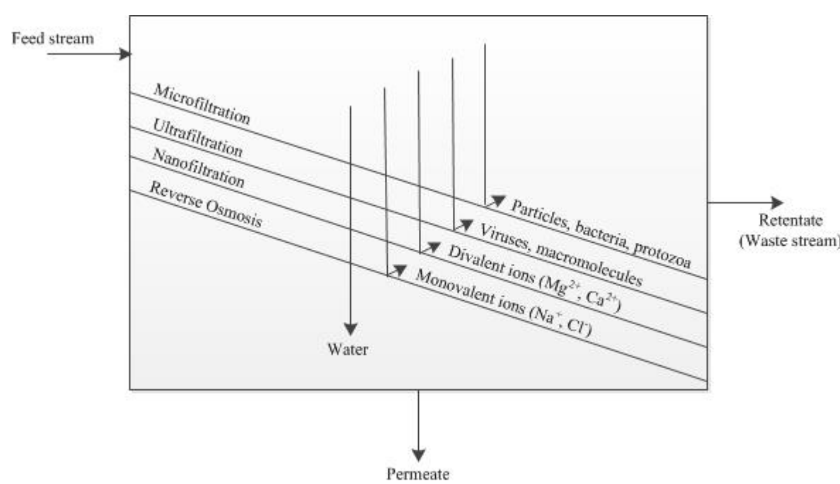

Figure 1.1. Membrane Separation Processes

exclusion, and/or differences in diffusion rates of contaminants[62]. There are four types of membrane processes commonly used in industry, as illustrated in Figure 1.1: microfiltration (MF), ultrafiltration (UF), nanofiltration (NF), and RO. The predominant removal mechanism in MF and UF is straining (size exclusion); MF can remove large suspended particles (>0.1 μm), achieving between 3-6 log removal of bacteria, whereas UF provides nearly complete removal of organic macromolecules, bacteria and protozoan cysts, and about 4 to 6 log removal of viruses, due to its smaller pore size (0.01 μm)[62]. On the other hand, the rejection mechanism of NF and RO membranes relies on differences in diffusion rates between dissolved solutes (such as ions) and water, with water diffusion rates being significantly higher. The diffusion rates through these membrane materials are a function of size and charge, with small, uncharged molecules (such as water) enjoying higher diffusion rates[66]. NF and RO are used to remove smaller dissolved species, such as ions and small organic molecules, down to the size of 0.001 and 0.0004 μm for NF and RO, respectively. NF is typically used to remove sugars and divalent ions, while RO is used to remove almost all dissolved salts, including monovalent ions[62]. Due to their structural and material characteristics, NF and RO require higher operating pressures to produce a reasonable flux, compared to MF and UF[12].

RESEARCH OBJECTIVES

The objectives of this dissertation are to (i) illustrate the flexibility of the water reuse decision-support model (RWRM) that allows blending of different wastewater treatment effluents to produce a water supply (i.e., *purposed* water) to meet specific water demands at an affordable cost; (ii) to investigate the impacts of drought on surface water/groundwater/wastewater quality and quantity and efforts to mitigate such impacts. Lastly, (iii) this work will identify opportunities to provide long-term, low-cost reliable water as well as groundwater management within urban scarce water environments under future uncertainty. Consequently, this research seeks to explore water supply alternatives, and to identify water supply management strategies to improve local water supply reliability and provide greater resilience to drought and climate change conditions. These goals will be accomplished by analyzing the following objectives:

(1) Develop a regional water reuse decision-support model (RWRM) to evaluate the impact of blending different wastewater streams (from different treatment systems) on water quality parameters and treatment costs with the goal of assisting water district managers and irrigators to make informed and cost-effective decisions. This will be done by matching specific water quality requirements of certain crops with an optimized wastewater treatment train that ensures the crops receive irrigation water tailored to their specific needs while minimizing the wastewater treatment cost and meeting California's strict wastewater reuse regulations (Title 22). The key element of this project is to identify cost-effective blending combinations across treatment processes that maintain crop yield and soil health.

(2) Study the effects of drought on water/wastewater quality and how it impacts water districts, state and federal agencies, wastewater treatment facilities and their current treatment processes. As a particular drought progresses and agencies enact water conservation measures to cope with drought, influent flows likely decrease while influent pollution concentrations increase, particularly salinity, which adversely affects wastewater treatment plant (WWTP) costs and effluent quality and flow. Consequently, downstream uses of this effluent, whether to maintain streamflow and quality, groundwater recharge, or irrigation may be impacted. This work investigates how drought and water conservation strategies combine to reduce flow and quality of wastewater and also identifies mitigation strategies to mitigate the impacts of drought on effluent water quality for reuse purposes.

(3) Develop a constrained supply-demand optimization model of a regional water supply system that can identify the costs and groundwater system implications associated with alternative economic, biophysical, and institutional / management scenarios. In particular, we describe a complex supply-demand model that evaluates the role of recharge from treated municipal wastewater on local water supply reliability, groundwater sustainability, and recommends cost-effective water management alternatives to water agencies. The model is a complex water balance that considers trade-offs between water supplies and demands, while taking into account both climate-change scenarios that affect local recharge rates and evapotranspiration rates, as well as changes in population, treatment costs, and regional supplies. The model predicts water availability, groundwater extraction, technological needs, and supplemental water sources designed to meet the demands of a municipality over a 100-year period to give water agencies the tools needed to make informed management decisions under climate uncertainty.

DISSERTATION OUTLINE

This dissertation is organized into five sections: Chapter 2 explains the concepts of wastewater reuse for agriculture and the development of the RWRM to identify the cost-effective irrigation sources. Chapter 3 is a study on the impacts of drought and water conservation on the reuse of municipal wastewater and mitigation strategies identified by the RWRM. Chapter 4 investigates the trade-off relationship between water supplies and demands and also offers water agencies tools to make informed and cost-effective water decisions under future water and climate uncertainties. Summary and conclusions of this research are described in Chapter 5.

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Chapter 2

Wastewater Reuse for Agriculture: A Development of a Regional Water Reuse Decision-Support Model (RWRM) for Cost-Effective Irrigation Sources

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ABSTRACT

Water scarcity has become a critical problem in many semi-arid and arid regions. The single largest water use in such regions is for crop irrigation, which typically relies on groundwater and surface water sources. With increasing stress on these traditional water sources, it is important to consider alternative irrigation sources for areas with limited freshwater resources. One potential irrigation water resource is treated wastewater for agricultural fields located near urban centers. In addition, treated wastewater can contribute an appreciable amount of necessary nutrients for plants. The suitability of reclaimed water for specific applications depends on water quality and usage requirements. The main factors that determine the suitability of recycled water for agricultural irrigation are salinity, heavy metals, and pathogens, which cause adverse effects on human, plants and soils. In this paper, we develop a regional water reuse decision-support model (RWRM) using the general algebraic modeling system to analyze the cost-effectiveness of alternative treatment trains to generate irrigation water from reclaimed wastewater, with the irrigation water designed to meet crop requirements as well as California's wastewater reuse regulations (Title 22). Using a cost-minimization framework, least-cost solutions consisting of treatment processes and their intensities (blending ratios) are identified to produce alternative irrigation sources for citrus and turfgrass. Our analysis illustrates the benefits of employing an optimization framework and flexible treatment design to identify cost-effective blending opportunities that may produce high quality irrigation water for a wide range of end uses.

INTRODUCTION

With concerns over water availability and reliability, municipalities and state governments are focusing more attention and resources on the reuse of municipal wastewater. California's Orange County Water District (OCWD), which operates the largest groundwater replenishment system in the world, has invested $142 million to increase its current wastewater recycling plant capacity from 70 to 100 million gallons per day (MGD)[1]. Meanwhile, neighboring Metropolitan Water District of Southern California (MWD), the country's largest water district, is considering investing $15 million in a demonstration project that is intended to pave the way for the development of a 150 MGD wastewater recycling plant[2]. Such investments are consistent with national trends as revealed in a recent National Academy of Sciences report, which notes that the volume of wastewater reused in the U.S. is increasing at an annual rate of 15%[3]. California, Texas, and Arizona—three of the top four states in terms of total volume reused—are located in the arid southwest and are expected to experience more frequent and intense droughts under climate change[4].

While the reuse of treated wastewater is not a new concept, concerns over the rising demand for water from population growth, coupled with both economic and environmental challenges, have made this option more attractive[3, 5, 6]. For instance, the estimated costs for MWD to recycle wastewater is approximately $0.72/m^3$ ($900 per acre-foot), significantly less than the current cost of $1.13/m^3$ ($1,400 per acre-foot) for imported water from northern California via the State Water Project (SWP) which supplies approximately 30% of water used in Southern California[2]. In addition to offering a less costly alternative to imported water, reuse of wastewater is also seen as a more environmentally friendly and locally dependable option, particularly during drought when the reliability of imported supplies decreases substantially. Indeed, during the drought confronting California in 2014—one of the driest periods in California history—only 5% of the SWP allocation water was delivered to agencies in Southern California as compared to 80% in 2011[7]. As farmers in California are heavily reliant on SWP deliveries for irrigation, low allocations mean farmers must turn to other sources. The reuse of treated wastewater for irrigation, which constitutes 80-90% of the nation's consumptive water use (water lost to the environment through evaporation, crop transpiration, or incorporation into products), is one such possible source that presents water agencies and farmers with the possibility of a low-cost, reliable and environmentally friendly local water source whose value will only increase under expected climate change conditions[8].

Currently, most wastewater treatment plants in California treat municipal wastewater to secondary or tertiary standards, followed by disinfection and discharge of the treated effluent to a surface water body[6]. Increasingly, disinfected secondary and tertiary effluents are used to irrigate restricted and unrestricted access areas, such as golf courses and freeway medians. The degree to which treated wastewater may broaden the water supply portfolios of particular water agencies and substitute for other water supply sources depends largely upon the chemical and biological composition of the effluent. Different water quality parameters (which often vary geospatially) can impact the suitability of recycled wastewater for irrigation, including salinity (expressed as total dissolved solids (TDS)), nutrient load, heavy metals, and pathogens, which might cause adverse effects on human, plants and soil health[5, 9-12]. The concentrations of these elements in wastewater impacts the treatment approach, and hence cost, necessary to generate irrigation water of appropriate quality.

Yet such limitations on the expanded use of treated wastewater need not be the case. For example, the removal of salt would generate usable irrigation water that meets minimum salinity thresholds for crops[9]. For this alternative treatment, then, some degree of desalination is necessary[13]. While adding desalination will increase costs, recycled wastewater can contribute an appreciable amount of the necessary nutrients for plants, which reduces the need for—and costs associated with—synthetic fertilizers[14]. So as the demand for treated wastewater increases from different sectors of society (e.g., agricultural, municipal potable and non-potable, environmental), the benefits of tailoring wastewater treatment plant operations to the intended use can reduce unnecessary expenditures on capital and O&M expenditures, energy, greenhouse gas emissions, and provide utilities with a more competitive option relative to other water supply sources[15]. As such, a flexible treatment approach that pairs specific wastewater treatment steps (arranged in treatment trains) and resulting water quality, with specific crop water quality demands, might offer utilities at the urban/agricultural interface a cost-effective means of transforming wastewater from a waste product to a valuable commodity.

The objective of this paper is to illustrate, through the development of a regional water reuse decision-support model (RWRM), how flexible wastewater treatment processes that allow blending can be optimized to produce a water supply that meets and surpasses a variety of water quality requirements at an affordable cost. While regulations such as Title 22 tend to focus on pathogen removal to ensure safe reuse of wastewater, this study instead focuses on producing irrigation water with tailored *chemical* properties, with the assumption that a disinfection step is needed to maintain safe wastewater reuse. In particular, we estimate and compare the costs and water quality characteristics of treated wastewater under a wide array of feasible treatment combinations that meet irrigation guidelines for two different types of products—citrus and turfgrass. The model, which has been developed and calibrated to reflect current wastewater treatment plant processes, costs, irrigation guidelines, and regulatory requirements in California, also identifies the cost-effective solution under alternative nutrient and bicarbonate constraints. Our solutions illustrate that when wastewater treatment management matches the water quality requirements of certain crops with an optimized wastewater treatment train that ensures the crops receive irrigation water tailored to their specific needs, an affordable water supply can be created that can meet and even exceed current water quality standards and practices. Consequently, the key contribution of this paper is to identify cost-effective blending combinations across treatment processes that maintain crop yield and long-term soil health.

METHODS

Modeling Methodology

Here we discuss the modeling procedures and input data used to analyze the cost-effectiveness of alternative treatment trains to generate irrigation water from wastewater while meeting specific crop requirements. A unique element of our model is its ability to consider and identify cost-effective blending combinations across processes, and the resultant water quality parameters that meet crop requirements. That is, rather than irrigating crops with one type of recycled water, e.g., disinfected secondary or tertiary effluents, the model provides water managers the opportunity to identify blended streams with higher quality at a lower price[5]. The blended effluents can be used on different crops, depending on specific irrigation guidelines.

Current wastewater treatment practices often consist of exposing the entire volume of influent to secondary or tertiary treatment prior to discharge[16]. Our analysis consists of considering the cost-effectiveness of a more flexible system in which fractions of plant influent, primary, secondary (membrane bioreactors (MBR) or activated sludge), and tertiary (filtration, membrane separation, and desalination) effluents are combined to produce a blend that meets particular demand criteria. When the demand for the effluent is for irrigation of particular crops (e.g., citrus or turfgrass), as it is in our current analysis, the criteria are based on plant requirements (i.e., nutrients) and thresholds (i.e., bicarbonate, salinity), along with any additional regulatory restrictions—such as a disinfection requirement—at the least cost. For example, under California's Title 22 regulations, recycled water used for the irrigation of food crops (where the recycled water comes in contact with the edible portions of the crops) and unrestricted access golf courses must meet tertiary treated quality in which the total coliform concentration must not exceed a 7-day median measure of 2.2 MPN/100 ml nor a 23 MPN/100 ml value in more than one sample in any 30-day period[17].

The RWRM was evaluated by analyzing three different scenarios corresponding to different model constraints: (a) with nutrient (N, P, K) and bicarbonate constraints (baseline), (b) without crop nutrient constraints, and (c) without nutrient and bicarbonate constraints. Crop nutrients and bicarbonate constraints were developed based on published data regarding the average amount of nutrients and bicarbonate in irrigation water (i.e., irrigation guidelines) typically used for each crop, which, through long-term experimental projects, caused no adverse effects on crop yield and soil health[9, 18]. Highly concentrated bicarbonate in irrigation water can cause soil permeability problems[9]. However, the bicarbonate concentration in typical municipal wastewater is, in fact, below the maximum allowable bicarbonate concentrations suitable for citrus and turfgrass (less than 8.5 meq/L or 519 mg/L HCO_3^- for moderate restriction)[9, 18]. While 519 mg/L of bicarbonate concentration is far higher than what is typically found in wastewater (and irrigation water) this level of bicarbonate will result in a decrease in crop yield, which is beyond the scope of our current modeling effort[12]. We include the nutrients and bicarbonate constraints here—which are, in essence, a conservative design meant to represent concentration levels of these factors typical in effluent—to evaluate the sensitivity of the model to different blending combinations, as well as the treatment costs associated with varying restrictions on input parameters. By increasing nutrient concentrations in terms of NH_4-N, NO_3-N, PO_4-P, and K in irrigation water, i.e., removing the nutrient constraints, crop demand for synthetic fertilizers can be reduced, which offers irrigators further possibilities to reduce costs.

We begin by discussing the alternative treatment trains that comprise our analysis, followed by a presentation of the optimization framework employed to identify the cost-effectiveness of each possible train. Results showing optimal blending ratios and cost savings associated with the reduced use of synthetic fertilizers for specific crops (citrus and turfgrass) are then highlighted. Data on the effectiveness of each process and its costs are reported as well.

Development of Regional Water Reuse Decision-Support Model (RWRM) and Optimization Framework

The RWRM was developed by focusing on combinations of treatment processes that are compatible with one another while avoiding redundancy. Seven feasible wastewater treatment technologies, arranged in eleven different wastewater treatment trains, were identified (Figure 2.1). These trains include: (1) activated sludge-granular filtration (TERT); (2) activated sludge-microfiltration-RO (MF-RO); (3) activated sludge-MF-nanofiltration (MF-NF); (4) activated sludge-ultrafiltration-RO (UF-RO); (5) activated sludge-UF-NF (UF-NF); (6) MBR-RO; (7) MBR-NF; (8) activated sludge-granular filtration-MF-RO (TERT-MF-RO); (9) activated sludge-granular filtration-MF-NF (TERT-MF-NF); (10) activated sludge-granular filtration-UF-RO (TERT-UF-RO); and, (11) activated sludge-granular filtration-UF-NF (TERT-UF-NF). While typical wastewater treatment plants do not employ a desalination step (NF or RO), this technology is emerging in certain instances where direct or indirect potable reuse is implemented. However, in contrast to potable reuse, to produce irrigation quality water, only a portion of the effluent stream may require desalination, which will depend on the water quality characteristics of the particular wastewater stream. When evaluating the blending ratios (i.e., the proportion of water in a given volume from each treatment process), the final irrigation water can be composed of effluent from any of the technologies in a treatment train. For example, the final blended water could be composed of primary effluent, secondary effluent, and RO effluent (or any other combination). These eleven treatment trains represent current municipal wastewater treatment technologies and cover almost all possible treatment configurations. Each of these trains provides for the removal of nutrients (carbon, nitrogen, phosphorous) through the activated sludge or MBR process, and the removal of large particles and molecules along with bacteria and viruses through the use of the MF and UF processes. The NF and RO processes, meanwhile, effectively remove divalent and monovalent ions, respectively. The removal efficiencies of different wastewater constituents by the different treatment steps are provided in the Supporting Information (Table A.A.1).

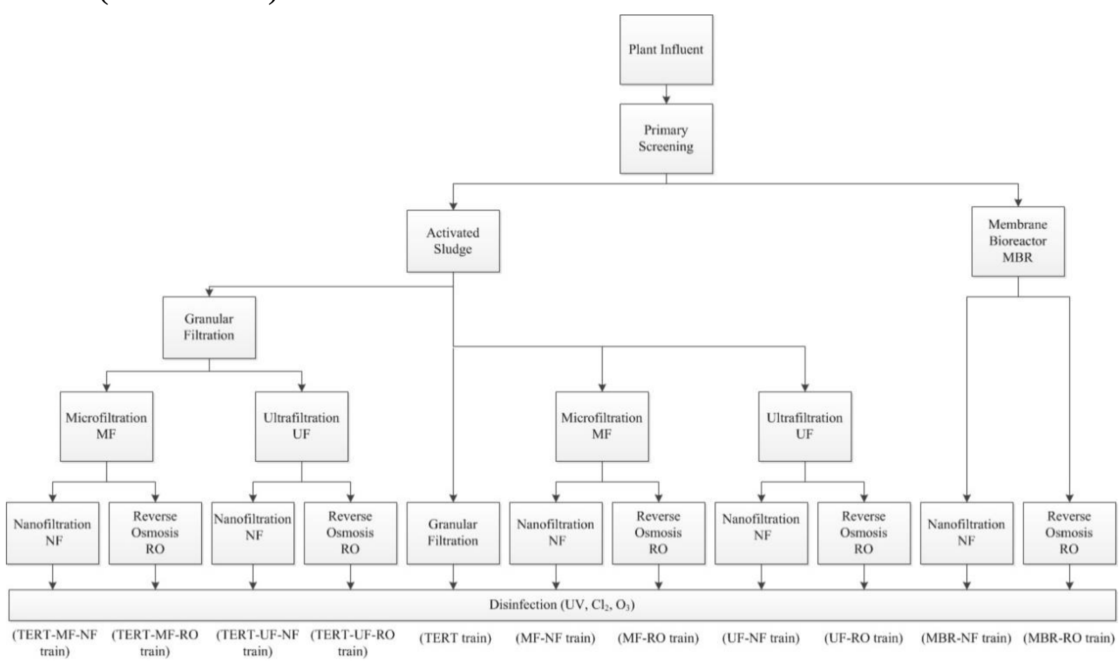

Figure 2.1. Different treatment train processes corresponding to different GAMS logic

The cost of each treatment train was estimated as the product of the unit treatment cost ($\$/m^3$) for individual processes within a particular train multiplied by the blending fraction of the produced water associated with that process, as shown in equation 1:

$$C_T = \sum_{j=1}^{J_T} b_T(j) * c_T(j) \quad (1)$$

where C_T is the total treatment cost per m^3 of wastewater treated, T is the particular treatment train, and j is the specific process within a treatment train T, which consists of J_T unique processes. The blending ratio of the produced water associated with process j for treatment train T is designated by $b_T(j)$, whereas the unit cost of each process is $c_T(j)$. The treatment cost is subject to two constraints. First, the total blending fractions of the produced water from each treatment train is restricted to one, corresponding to 100% of a given effluent volume as in equation 2, assuming all effluent is used by the crop.

$$\sum_{j=1}^{J_T} b_T\,(j) = 1 \quad (2)$$

Second, crop-specific upper limits are imposed on irrigation water quality parameters from each treatment train. The final concentrations of constituents resulting from the blending process, as shown in equation 3, must be less than those specified in a certain crop's irrigation guidelines, assuming water quality parameter values are additive:

$$\sum_{j=1}^{j_T} p_T(i,j) * b_T(j) \leq a(i) \quad (3)$$

where i is a specific wastewater constituent associated with process j within a given treatment train T, $p_T(i,j)$ is the concentration of specific constituent i resulting from process j, and a(i) is the concentration threshold for constituent i so as to not impact crop yield.

Using the General Algebraic Modeling System (GAMS), the lowest cost solution is identified by choosing the blending ratios and treatment train that minimize the cost of producing a particular volume of effluent under the blending and irrigation guideline constraints. Our optimization framework, thus, seeks to minimize $C_T = \sum_{j=1}^{J_T} b_T(j) * c_T(j)$ subject to equations (2) and (3).

Model Inputs

Model inputs are comprised of four categories: concentrations of water quality parameters in a given unit volume of wastewater, concentrations of water quality parameters associated with effluent from each treatment process, concentration limits of water quality parameters to meet crop requirements, and the unit cost for each treatment process. Wastewater influent, or raw wastewater, typically contains high levels of nutrients, specifically nitrogen (NH_4-N, 40.3 mg/L), phosphorus (P, 9.7 mg/L) and potassium (K, 15.9 mg/L), which can be utilized by plants[19, 20]. However, to meet state and federal water quality regulations, most conventional municipal wastewater treatment plants subject this raw influent to primary, secondary, tertiary, and disinfection processes, which results in significant removal of nutrients (typical treated effluent concentrations of 5.2 mg/L NH_4-N and 6.6 mg/L P)[5, 20-22]. However, wastewater may contain a high load of TDS, which are not effectively removed by conventional treatment processes[22]. To reduce salinity and meet more stringent effluent quality regulations, membrane-based treatment methods are becoming more common in wastewater treatment[22]. Membrane separation processes commonly used in water treatment processes include MF, UF, NF and RO. These processes rely on physical (size exclusion, charge exclusion) and chemical (differences in diffusion rates) phenomena to remove contaminants from waste streams[23].

A summary of water quality parameters in the wastewater, as well as the concentration after each treatment process can be found in Table A.A.2. These water quality parameters are represented by $p_T(i,j)$ in equation (3). Average removal rates of contaminants by each treatment process are displayed in Table A.A.1. The data in Table A.A.2 were obtained from sources including the Inland Empire Utilities Agency, the U.S. EPA, and OCWD[19, 20, 22, 24]. In instances where data were not available (italicized entries in Table A.A.2), parameters were estimated based on typical removal percentages (e.g., MF and UF are known to not effectively remove TDS). RO results in the lowest concentrations of salts, nutrients, and other constituents compared to NF, UF, and MF. It is important to note that TDS levels in the model wastewater used here (522 mg/l in secondary effluent and 496 mg/l in tertiary effluent) are near or below the maximum recommended TDS values for citrus and most turfgrass. Thus, salinity was not a limiting factor when determining the optimal blending ratios. However, many wastewater plants generate high salinity (TDS > 500 mg/l) effluents. For example, the wastewater from the city of Carlsbad in Southern California contains TDS in excess of 1000 mg/l[25].

Specific water quality standards and thresholds associated with irrigation water quality parameters for citrus and turfgrass can be found in Table A.A.3[9, 22, 26-28]; the concentration thresholds for each parameter correspond to the $a(i)$ values in equation 3. In general, citrus has lower threshold concentrations than turfgrass. For comparison purposes, Table A.A.3 also includes the range of concentrations typically found in standard irrigation water. In considering the model outcomes (left-hand side of equation 3), many blending ratios (model solutions) generated water with high levels of pathogens. In such instances, an additional disinfection step using chlorine, ozone and/or UV radiation was necessary to meet California Title 22 regulations for water reuse on agriculture.

A summary of estimated individual treatment costs associated with the individual processes within a treatment train for small-medium (≤ 20000 m^3/d (5 MGD)) and large (> 20000 m^3/d) treatment facilities is provided in Table A.A.4. Capital costs were amortized over 15 to 20 years, depending on the process, and included construction and administrative costs. Operation and maintenance (O&M) costs reflect the cost of energy, chemicals, and maintenance[29]. The information in Table A.A.4 spans several plant capacities and time intervals, with data adjusted to 2013 U.S. Dollars using the Engineering News Record Construction Cost Index and the Consumer Price Index[30, 31]. Finally, given the importance of choice of interest rate to the magnitude of the amortized costs (and thus unit costs), sensitivity analysis over a range of interest rates (3%-10%) was explored, yielding no appreciable qualitative differences.

MBR and RO processes are relatively expensive compared to the other processes (Table A.A.4). Additionally, economies of scales associated with the larger plants significantly reduce the unit treatment costs for nearly all processes (Table A.A.4). For turbid water, a higher dosage of disinfectant is needed to comply with discharge regulations, a step that increases our disinfection costs by two to three-fold.

RESULTS AND DISCUSSION

The RWRM was used to determine the most cost-effective treatment train and associated blending ratios that meet both regulatory standards and crop thresholds. This section presents the costs of each of the treatment trains used to meet irrigation guidelines for citrus and turfgrass. For each crop, three different scenarios were analyzed—(a) with nutrient and bicarbonate constraints (baseline), (b) without nutrient constraints, and (c) without nutrient and bicarbonate constraints. The RWRM screened the treatment systems and eliminated those that did not produce any feasible solution. For example, granular filtration and NF systems were never selected due to their inability to produce blends with sufficiently low ion concentrations (Na^+, HCO_3^-, Cl^-, K^+, and B) to meet the irrigation guidelines for either citrus or turfgrass. A model sensitivity analysis based on plant size, removal performances and treatment costs is provided in Tables A.A.7-A.A.15.

Treated Wastewater for Citrus Irrigation

Figures 2.2 and 2.3 present the results from the RWRM when the treated wastewater from small-medium and large treatment facilities is applied to citrus while meeting Title 22 regulations and citrus irrigation guidelines. Title 22 specifies that recycled wastewater used for orchard irrigation must meet or exceed *undisinfected secondary effluent* water standards[17, 22]. Figure 2.2 presents the blending ratios (in percentage terms) for the least cost scenarios with the unit costs listed at the top of each pie chart. All solutions contained a certain percentage of desalinated (RO effluent) wastewater due to the excess amounts of P, Na^+, K^+, and CO_3^{-2}. Under baseline conditions (scenario A), in which nutrient constraints are applied to maintain NH_4-N, PO_4-P, and K concentrations typically found in irrigation water, the cost-effective solutions were \$0.87/m^3 and \$1.83/m^3 for large and small-medium plants, respectively; these results were generated from an MF-RO train comprised of a blend of 88% RO effluent and 12% secondary effluent.

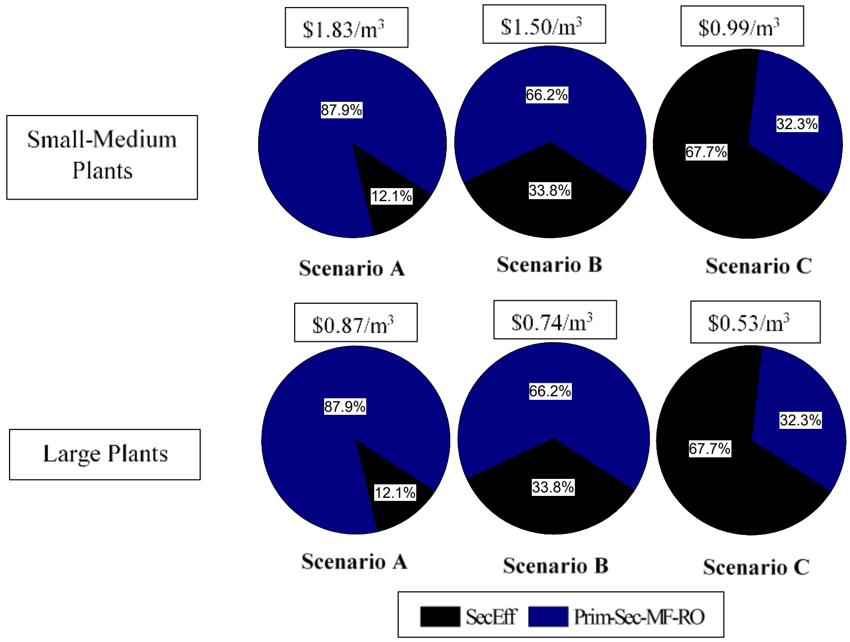

Figure 2.2. Optimized blending ratios for citrus irrigation from the MF-RO treatment train for small-medium and large treatment facilities; due to TDS restrictions, all model solutions require some degree of desalination (RO). Three scenarios were investigated: (A) with crop nutrient and bicarbonate constraints (baseline); (B) without crop nutrient constraints; and (C) without crop nutrient and bicarbonate constraints

Figure 2.3 presents the concentrations of the different water quality parameters for the least cost MF-RO train (yellow bars) relative to water quality parameters found in typical irrigation water (black bars). As can be seen, the least cost MF-RO solution produces irrigation water with water quality parameters far superior to crop guideline levels, an outcome driven by the large percentage of RO effluent in the final blended product. For example, TDS concentrations were 83% below the threshold for citrus (86.683 mg/L vs. 500 mg/L TDS). The most cost-effective irrigation solution in scenario A is constrained by the amount of phosphorus in the blended product (Figure 2.3). While the MF-RO train was the least cost solution under the baseline (scenario A), $0.87/m^3 and $1.83/m^3 for large and small-medium plants, respectively, are relatively high costs for irrigation water, making this solution challenging in real world applications. For example, agricultural operators in the Westlands Water District—the largest irrigation district in the country—pay around \$0.28/m^3 for contract water when it is available[32], but may find water prices on the spot market during drought in excess of \$1.62/m^3 [33]; alternatively, the price MWD charges its retail agencies for tier 1 water, which is water that comes from the Colorado River Aqueduct or SWP, is \$0.76/m^3 [34].

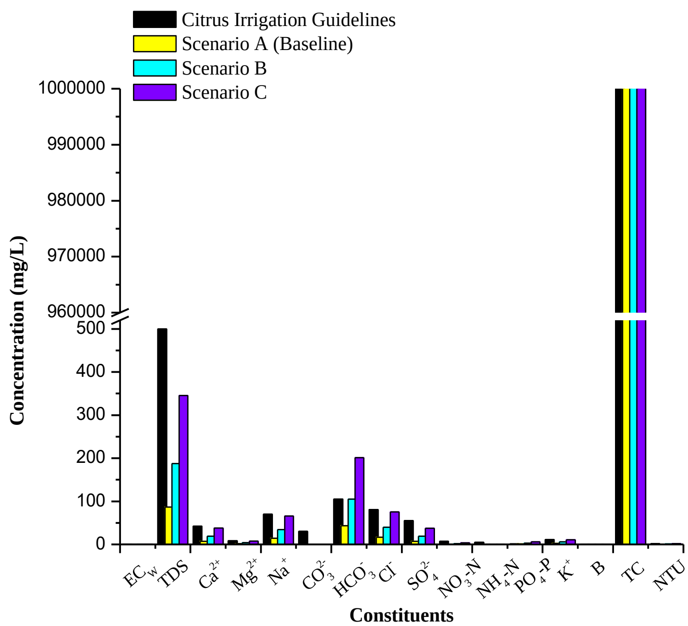

Figure 2.3. Comparison of irrigation guidelines for citrus (black bars) with the water quality parameters of the different blending ratios from the MF-RO treatment train under the three different constraints: (yellow bars) with crop nutrient constraints, i.e., baseline (RSC = 0.22 meq/L); (blue bars) without crop nutrient constraints (RSC = 0.42 meq/L); and (purple bars) without nutrient and bicarbonate constraints (RSC = 0.74 meq/L)

Removing the nutrient concentration limits (scenario B) and bicarbonate constraint (scenario C) leads to a significantly different cost-effective solution. While the MF-RO train still provides the least-cost solution relative to other trains, removing the nutrient constraint reduces treatment costs by 18%—\$0.74/m^3 for large plants and \$1.50 for small-medium plants—under scenario B (Figure 2.2). These lower costs relative to scenario A are directly related to a blend that relies less on RO effluent (25% reduction) and more on secondary effluent. By removing the bicarbonate constraint, alternatively, treatment costs under the least-cost MF-RO train are reduced by 46% relative to the baseline ($0.53/m^3 for large plants and $0.99 for small-medium plants), again a result of a blend that relies even less on RO effluent (only 32%) and more on secondary effluent (68%). Importantly, by removing these constraints, the fraction of water treated with the high cost RO process in the final blended product can be significantly reduced while still meeting water quality guidelines for these crops.

The water quality impacts of relying less on the RO process under scenarios B and C is illustrated by the light blue and purple bars in Figure 2.3. The concentrations of different constituents increase relative to the baseline scenario (scenario A) yet are still significantly lower than the threshold levels associated with crop requirements. Consequently, even though the overall water quality declines when the fraction of RO effluent decreases, the quality of the blended product still meets citrus irrigation quality requirements at a significantly lower cost. Importantly, in each of the three scenarios, bicarbonate concentrations were below 210 mg/L (Figure 2.3), well within the range of maximum allowable bicarbonate concentration[18]. The tolerable levels of bicarbonate in irrigation water, expressed as residual sodium carbonate (RSC), are RSC $\leq$ 1.25 meq/L[9, 18]. In all cases, the blended effluents were safe for citrus and soils, with RSC values < 0.75 meq/L for both citrus and turfgrass.

Two final observations worth noting. First, the treatment costs for large facilities were nearly 50% less than the costs under small-medium plant sizes, highlighting the role economies of scale can play in making treated wastewater more competitive to implement in real world applications. Second, the results from blending the effluent from different treatment processes generated a feasible and cost-effective irrigation source for citrus while complying with Title 22 regulations and without affecting soil and crop health. However, significant numbers of total coliform (TC) bacteria (10^6 MPN/100 ml) are projected to be present in the effluents (Figure 2.3). Therefore, appropriate precautions should be taken when applying this water to citrus groves.

Treated Wastewater for Restricted Turfgrass Irrigation

Figures 2.4 and 2.5 present the results from RWRM when the treated wastewater from small-medium to large treatment plants is used for turfgrass irrigation. Under Title 22 regulations, restricted access turfgrass requires at least disinfected secondary-23 treated water[17, 22]. While many turfgrass fields (such as golf courses) do not typically use desalinated wastewater for irrigation, if the treated wastewater contains elevated salinity levels, these fields require periodic flushing with fresh water to remove accumulated salt[9]. Under the baseline (scenario A), the MF-RO-Cl train with a blend of 77% MF-RO effluent and 23% secondary effluent with chlorine disinfection was the lowest cost solution at $\$0.81/m^3$ and $\$1.67/m^3$ for large and small-medium plants, respectively (Figure 2.4). Notable is the superior quality of the blended water relative to the parameters under the turfgrass irrigation guidelines, specifically in TDS and salts, as shown in Figure 2.5 (black bars), a consequence of the significant RO fraction in the final blend.

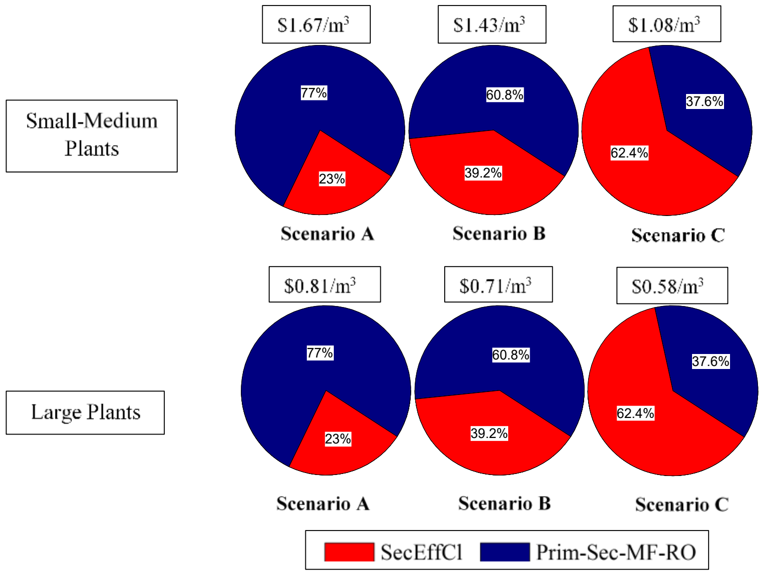

Figure 2.4. Optimized blending ratios for restricted access turfgrass irrigation with chlorine disinfection from MF-RO treatment train for small-medium to large treatment facilities; due to TDS restrictions, all model solutions require some degree of desalination (RO). Three scenarios were investigated: (A) with crop nutrient and bicarbonate constraints (baseline); (B) without crop nutrient constraints; and (C) without crop nutrient and bicarbonate constraints

When the nutrient constraint was removed, the low-cost treatment train is again MF-RO-Cl, with a blend consisting of 61% MF-RO effluent and 39% secondary effluent with chlorine disinfection. Compared to the baseline, RO effluent was reduced by approximately 21% while the amount of secondary effluent was almost doubled, resulting in a total treatment cost of $0.71/m^3$ and $1.43/m^3$ for large and small-medium plants, respectively (Figure 2.4). Even with the large reduction in the percentage of RO effluent, which was replaced with secondary effluent, the resulting irrigation water quality significantly exceeds irrigation guidelines for turfgrass (Figure 2.5). When the bicarbonate constraint was removed, the RO portion of the blended water was further reduced to 38% (a 38% decrease from scenario B), and the portion of secondary effluent increased to 62% (a 60% increase from scenario B), with a final treatment cost of $0.58/m^3$ for large plants and \$1.08/m^3 for small-medium plants.

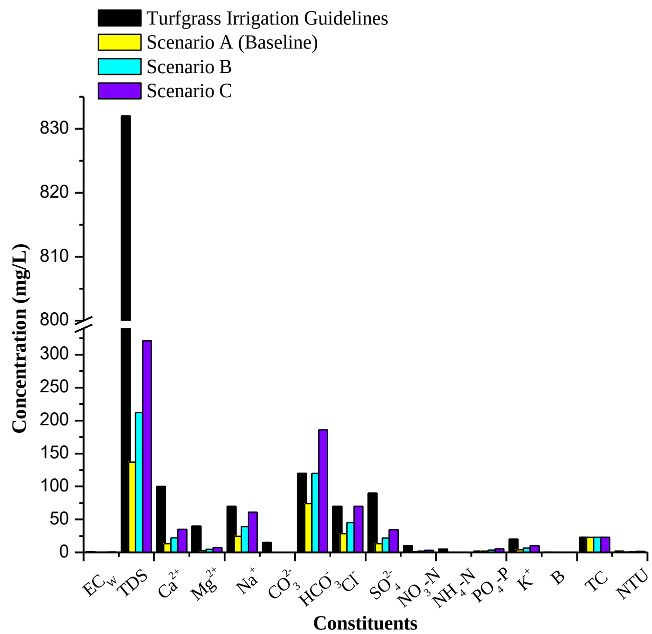

Figure 2.5. Comparison of irrigation guidelines for restricted access turfgrass (black bars) with the water quality parameters of the different blending ratios from the MF-RO treatment train under the three different constraints: (yellow bars) with crop nutrient constraints, i.e., baseline (RSC = 0.32 meq/L); (blue bars) without crop nutrient constraints (RSC = 0.47 meq/L); and (purple bars) without nutrient and bicarbonate constraints (RSC = 0.69 meq/L)

In each of the solutions under this scenario chlorine disinfection was necessary to ensure the water quality was compliant with Title 22 regulations. Also, and similar to the citrus case above, economies of scale reduced the treatment costs associated of larger plants by up to 50% relative to the small-medium sized plants.

Treated Wastewater for Unrestricted Turfgrass Irrigation

The major difference under Title 22 between recycled wastewater used to irrigate turfgrass with restricted and unrestricted access is that wastewater used to irrigate the latter must be irrigated with at least disinfected tertiary treated wastewater with a total coliform count below 2.2 MPN/100 ml[17, 22]. The RWRM found that the MF-RO-Cl train produced the least cost solution, with a cost of \$0.86/m^3, \$0.80/m^3 and \$0.72/m^3 for scenario A, B and C, respectively, for large plant sizes. Treatment costs under the small-medium plants sizes were nearly double those relative to the larger plants ($1.76/m^3, $1.56/m^3 and $1.27/m^3 for scenarios A, B, and C, respectively). A detailed analysis of treated wastewater for unrestricted turfgrass irrigation can be found in the Supporting Information (Figure A.A.F1 and A.A.F2).

Treated Wastewater vs. Traditional Irrigation Sources

The blending method used in this work produced alternative irrigation options in place of the traditional irrigation sources, which typically rely solely on secondary/tertiary effluent and natural sources.[5] The water quality of surface/groundwater sources is typically better than conventionally treated wastewater, with low electrical conductivity (typically less than 0.522 dS/cm) and low ion concentrations (0.1 mg/L B, 23.6 mg/L Na^+, 28 mg/L Cl^-, and 88.3 mg/L Ca^{2+})[35]. Secondary and tertiary effluents have relatively high salinity, with TDS in excess of 500 mg/L, electrical conductivity exceeding 0.85 dS/m, and high ion concentrations (0.3 mg/L B, 95.9 mg/L Na^+, 130 mg/L Cl^-, and Ca^{2+} in excess of 49 mg/L)[20, 21] (Table A.A.2). Compared to natural water sources, the blended effluent provides competitive or even superior water quality at an affordable cost. And while groundwater pumping costs are relatively cheap ($0.115/m^3), groundwater supplies are often under stress or may be entirely unavailable[36-39]. Furthermore, as groundwater levels drop, the cost of bringing this water to the surface increases, incentivizing irrigators and municipalities to look elsewhere for their water supplies[38, 40].

While disinfected secondary/tertiary effluents have been used to irrigate landscapes in restricted/unrestricted areas for some time, particularly on golf courses, the sustainability of such practices is sometimes problematic. That is, the salinity of certain conventionally treated wastewater is above the maximum irrigation tolerance for turfgrass, which, over time, harms both the soil and the crop due to salt accumulation[9, 28, 41]. The results from our analysis suggest the possibility of a high-quality irrigation source at a competitive price relative to what is typically observed today. Based on the literature, the cost of disinfected secondary and tertiary (granular filtration) effluents is around $\$0.25/m^3$ and $\$0.19/m^3$ (thus, the total cost of tertiary effluent is $\$0.25 + \$0.19 = \$0.44/m^3$), respectively (Table A.A.4) whereas the cost-effective blending ratio for turfgrass from our model was $\$0.58/m^3$ (restricted access; large plants) and $\$0.72/m^3$ (unrestricted access; large plants). Our low cost/higher quality solution is achieved through the use of secondary effluents in the blend, with chlorine disinfection of the final product ensuring compliance with Title 22 regulations.

Nutrients in Treated Wastewater and Cost Savings Associated with Lower Synthetic Fertilizer Usage

In addition to providing optimal irrigation solutions to citrus and turfgrass, reusing wastewater for agriculture also offers cost savings on synthetic fertilizers due to the appreciable amount of nutrients present in the wastewater streams (PO_4-P and K). To illustrate this, we assume synthetic fertilizers costs of \$0.86/kg nitrogen (N), \$1.19/kg phosphate (P_2O_5), and \$0.72/kg potash (K_2O) based on average wholesale prices[45-47], and irrigation rates of 2.54 cm/week[48] and 5.08 cm/week for citrus and turfgrass, respectively[49,50]. For citrus, recommended fertilizer application rates to meet plant requirements using synthetic fertilizer alone range from 100-400, 0-208 and 135-224 kg/ha-yr for N, P, and K, respectively, resulting in fertilizer costs of \$777.94/ha-yr[42-44, 51-53]. For turfgrass, the recommended application rates range from 98-195, 49, 146 kg/ha-yr for N, P, and K, respectively for a cost of \$332.08/ha-yr[42-44, 54-57]. Before presenting the cost comparisons, though, it should be noted that Phosphorous demand by citrus is not uniform, and largely depends on the available phosphorous in the soil, evaluated by measuring phosphorous in both soil and leaves[42,43]. Thus, the recommended phosphorous fertilization rate can range from zero to 228 kg P_2O_5/ha-yr, depending on the age of the trees and assuming fruit production of 1483 boxes/ha-yr[44].

Now we consider the degree to which the nutrient loads from our cost-effective solutions may substitute for synthetic fertilizers. Tables S5 presents the list of nutrient loads corresponding to each cost-effective blend. For the three different blending solutions identified by the model (corresponding to scenarios A, B, and C), treated wastewater under scenario C provides the largest concentration of nutrients to citrus (2.00, 172, and 174 kg/ha-yr of N, P, and K, respectively (Table A.A.6)). Comparing these rates with the recommended rates above, we see that phosphorous concentrations in the blended irrigation water may meet, and in some cases exceed, the crop's demand for phosphorous. And because NH_4-N removal during the wastewater treatment process is so effective, little NH_4 is available as fertilizer. Given these contributions, fertilizer costs can be reduced by \$33.134/ ha-yr under scenario C. Under scenario B, treated wastewater can provide 1.32, 87.2, and 90.8 kg/ha-yr of N, P, and K, respectively, providing cost savings of \$170.41/ha-yr, while under scenario A the cost savings can be as high as \$67.15/ha-yr (Table A.A.6). It must be stated here that some wastewater treatment plants now include a biological phosphorous removal step, which removes nearly all phosphorous from the waste stream. Thus, irrigation water generated from these plants will not have significant P concentrations.

For turfgrass the irrigation water generated under scenarios B and C had phosphorous and potassium concentrations that exceeded average demand for turfgrass (Table A.A.5). Under scenario A, the annual value stemming from the presence of nutrients in the irrigation water was $95/ ha-yr, replacing 71% of the cost of synthetic fertilizer (Table A.A.6), where synthetic fertilizer requirements were based on an average value across different turfgrass species with different N/P/K tolerances and requirements[54,55]. Consequently, the concentrations of potassium and phosphorous in our solutions may be too high for low-tolerance strains. For instance, for sand-based soil Kentucky blue grass and perennial ryegrass (cool-season lawn), the ideal potash application ranges from 195 to 390 kg/ha whereas bentgrass and fine fescue lawns only require 98 kg/ha of potash annually[56, 57]. This problem can be solved by reducing potassium and phosphorous concentrations in the final blended product through increasing the RO portion of the irrigation water, a solution that will increase the cost of the irrigation water.

The RWRM was shown to optimize blending combinations of different treated municipal wastewater effluents to provide irrigation water tailored to meet crop requirements at minimum costs. Alternative scenarios related to water quality parameter thresholds and guidelines were evaluated to demonstrate the model's flexibility and usefulness; in all cases, a certain fraction of the water required desalination to reduce salinity in the final blended product. When more restrictions were placed on irrigation water quality, the model responded with a large portion of RO water in the final blended product, which resulted in a significantly higher treatment cost. By gradually relaxing each of the constraints, the RO portion of the final blended product was reduced, lowering the treatment costs to a level appropriate for adoption.

Limitations

While there were a number of simplifying assumptions in the model presented above, two in particular are worth discussing. First, we assumed a cost-minimization framework that identified the least-cost solution to produce treated wastewater given particular water quality constraints. This framework overlooks possible solutions in which the users of the water may be willing to accept poorer quality water in certain dimensions (along with the consequent yield reductions) for a lower price. To represent this alternative, crop-water production functions could be incorporated into the model along with their associated prices and costs so that the model could identify efficient solutions in which net benefits are maximized, where net benefits are defined by the profits to the water user less the costs to the wastewater treatment plant operator. Crop-water-salinity production functions developed and utilized in Letey et al. (1985), Kan et al. (2002), and Schwabe et al. (2006) are being further developed for such an extension[58-60]. Second, our model focuses on the water quality requirements associated with two separate crops in isolation. In reality, demand for the treated municipal wastewater nor the current distribution will likely allow wastewater treatment managers to produce water for a single type of crop or product. There are a number of ways the model could be adjusted to reflect this, including incorporating the net benefits framework mentioned above. Alternatively, the current cost-minimization framework could be modified to generate solutions subject to constraints imposed on the most sensitive parameters across the array of crops. As this model is intended to be flexible and allow for the evaluation of a wide variety of treatment processes and output scenarios, such explorations are easily incorporated into the current framework.

Using the RWRM, wastewater treatment trains can be optimized to produce irrigation water suitable for a wide range of crops with varying salinity tolerance, reducing the impact on soil and crop quality that is currently experienced by irrigators using conventionally treated wastewater. Salinity, heavy metals, and pathogens were minimized to comply with existing regulations and safe agriculture practices. By utilizing this blending technique as an alternative irrigation source for agriculture, freshwater resources would be reserved to cope with drought-induced extreme water scarcity.

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Chapter 3

The Implications of Drought and Water Conservation on the Reuse of Municipal Wastewater: Recognizing Impacts and Identifying Mitigation Possibilities

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ABSTRACT

As water agencies continue to investigate opportunities to increase resilience and local water supply reliability in the face of drought and rising water scarcity, water conservation strategies and the reuse of treated municipal wastewater are garnering significant attention and adoption. Yet a simple water balance thought experiment illustrates that drought, and the conservation strategies that are often enacted in response to it, both likely limit the role reuse may play in improving local water supply reliability. For instance, as a particular drought progresses and agencies enact water conservation measures to cope with drought, influent flows likely decrease while influent pollution concentrations increase, particularly salinity, which adversely affects wastewater treatment plant (WWTP) costs and effluent quality and flow. Consequently, downstream uses of this effluent, whether to maintain streamflow and quality, groundwater recharge, or irrigation may be impacted. This is unfortunate since reuse is often heralded as a drought-proof mechanism to increase resilience. The objectives of this paper are two-fold. First, we illustrate—using a case study from Southern California during its most recent drought—how drought and water conservation strategies combine to reduce influent flow and quality and, subsequently, effluent flow and quality. Second, we use a recently developed regional water reuse decision support model (RWRM) to highlight cost-effective strategies that can be implemented to mitigate the impacts of drought on effluent water quality. While the solutions we identify cannot increase the flow of influent or effluent coming into or out of a treatment plant, they can improve the value of the remaining effluent in a cost-effective manner that takes into account the characteristics of its demand, whether it be for

landscaping, golf courses, agricultural irrigation, or surface water augmentation.

INTRODUCTION

In response to water shortages and drought, municipalities are exploring a broad array of options to improve local water supply reliability and resilience. Demand management strategies that include both price (e.g., increasing block rate pricing) and nonprice approaches (e.g., low flush toilets, low flow showerheads) to reduce water consumption are one option[1, 2]. As Figure 3.1 illustrates, demand management is likely a significant contributor to the noticeable reductions in per capita water use throughout the U.S.[3-14]. Concurrently, and on the supply-side of the water scarcity ledger, agencies increasingly are augmenting local supplies through the use of treated municipal wastewater (Figure 3.1)[3-34]. As an increasingly important element of a municipality's water portfolio, treated municipal wastewater can and often is used to irrigate local agriculture, golf courses, landscapes, and maintain local stream flows.

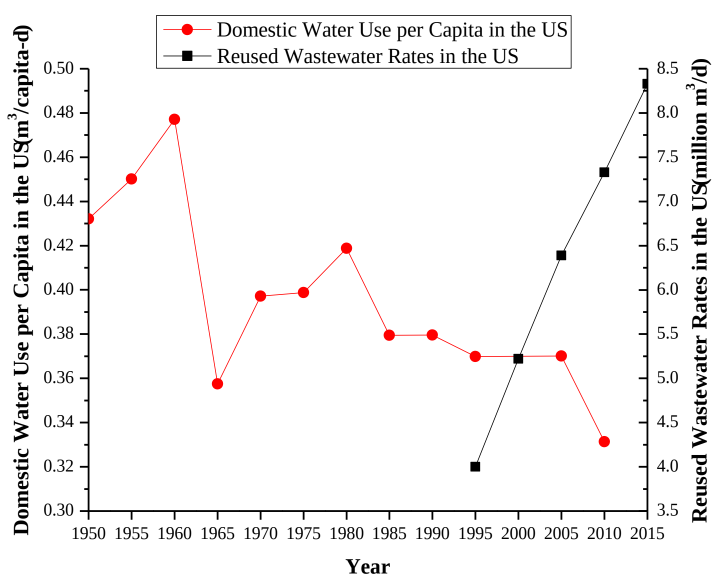

Figure 3.1. Domestic water use per capita versus water

However, a simple water balance exercise reveals that demand management techniques and the use of treated municipal wastewater are linked, particularly through residential indoor water use[35, 36]. Consequently, if residential indoor water use decreases, the ability of treated municipal wastewater to help mitigate water shortages decreases while its cost likely increases. Specifically, indoor conservation can result in the generation of a more concentrated wastewater stream, with elevated concentrations of total dissolved solids (TDS), nitrogen species, and carbon[37]. This increase can present challenges to wastewater treatment plants, which were not designed to remove certain constituents[37-39]. Dissolved solids, in particular, represent a significant treatment challenge, as traditional wastewater treatment facilities are primarily concerned with nutrient removal. Therefore, conventional wastewater treatment facilities may have to invest significant resources to upgrade their treatment technologies, with the goal of complying with state and federal discharge limits, as well as to generate treated municipal wastewater that can be safely and reliably reused[22, 28, 38, 40-43].

Without investments to address more concentrated wastewater streams, the water supplies of downstream communities might be adversely affected via multiple pathways. During extreme drought, natural flows in rivers decline, and wastewater discharges become a larger portion of surface water flow, resulting in a deterioration of surface water quality[44-47]. The deterioration of water quality is illustrated by elevated concentrations of wastewater-derived contaminants, such as salts, nutrients, and/or other contaminants in the wastewater discharges, due to human and natural factors (e.g., population, water conservation, droughts, storms, etc.)[45, 46, 48]. An example of this phenomenon can be found in the Santa Ana River Basin, which is the largest stream system in Southern California[48,

[49]. The flow in the Santa Ana River Basin is dominated by tertiary treated wastewater discharges generated by utilities throughout the basin[49]. The water quality in the Santa Ana River Basin becomes poorer as the water flows downstream due to several factors, including recycling and reuse of wastewater within the basin[48]. Consequently, any deterioration in treated wastewater effluent quality from upstream discharges has an immediate impact on downstream users.

The increase in wastewater contaminant concentrations may violate discharge limitations set by state and federal regulations (i.e., National Pollutant Discharge Elimination System (NPDES) permit) and result in penalties. Consider, again, the case of California in which violations result in mandatory minimum penalties from the Regional Water Quality Control Board[50]. According to the Water Quality Enforcement Policy, California Water Code section 13385(h) requires that a mandatory minimum penalties of $3,000 be assessed by the Regional Water Quality Control Board for each serious violation. A serious violation is any discharge that exceeds the effluent limitation by at least 40% for Group I pollutants (Biochemical Oxygen Demand (BOD), Chemical Oxygen Demand (COD), Total Oxygen Demands (TOD), Total Organic Carbon (TOC), Total Suspended Solids (TSS), TDS, nutrients, detergents and oil, minerals, and metals), or by 20% for Group II pollutants (cyanide, total residual chlorine, all organic substances not listed under Group I).

Finally, poorer quality effluent can negatively impact sectors that rely on this water for irrigation, including golf courses, municipal and residential landscapes, and agriculture. Without mitigative actions within these sectors, plant growth and yields may suffer.

Mitigative efforts to avoid such damages might include flushing with higher quality water, a response that can be equally costly and limited given freshwater supplies may be curtailed.

There have been several studies on the impacts of climate change on water/wastewater quality and availability[44, 46, 49, 51-56]. To date, however, there have been few case studies investigating how drought may impact wastewater influent and effluent flow and quality, and how modernizing wastewater treatment trains can address these impacts. As the frequency and intensity of drought increases along with agency adoption of demand side management strategies, further understanding of how influent and effluent flows are impacted and cost-effective strategies to reduce such impacts is warranted.

The objective of this research is to investigate the impacts of drought and water conservation on the availability and quality of treated municipal wastewater. In particular, we investigate the impacts of drought on influent and effluent concentrations and identify cost-effective treatment responses to mitigate such impacts. After illustrating how the drought in California, including the state's response that mandated local water suppliers reduce water consumption by between 4 to 36%, impacted influent and effluent concentrations at a representative wastewater treatment plant in Southern California, we demonstrate how wastewater treatment trains can be upgraded to ensure continued high quality effluent discharges in the face of declining influent water quality.

MATERIALS AND METHOD

Development of the regional water reuse decision-support model (RWRM) and optimization framework

The RWRM uses constrained linear programming methods to identify wastewater treatment trains optimized to deliver treated wastewater with specific water quality properties[32]. Importantly, the programming model uses a constrained cost-minimization framework that identifies not only specific treatment processes, but also the relative volume fractions from individual parts of the treatment train that, when blended, provide for specific water quality attributes in the treated wastewater effluent. Constraints are imposed on various effluent water quality parameters to ensure that the specific water quality attributes meet regulatory and end-use demand parameters (e.g., threshold concentrations limits).

Seven feasible wastewater treatment technologies capable of providing a certain measure of salt removal were identified and arranged in eleven different wastewater treatment trains (Figure 3.2): (1) activated sludge-granular filtration-MF-NF (TERT-MF-NF); (2) activated sludge-granular filtration-MF-RO (TERT-MF-RO); (3) activated sludge-granular filtration-UF-NF (TERT-UF-NF); (4) activated sludge-granular filtration-UF-RO (TERT-UF-RO); (5) activated sludge-granular filtration (TERT); (6) activated sludge-MF-nanofiltration (MF-NF); (7) activated sludge-microfiltration-RO (MF-RO); (8) activated sludge-UF-NF (UF-NF); (9) activated sludge-ultrafiltration-RO (UF-RO); (10) MBR-NF; and, (11) MBR-RO. Details on each treatment train is described in Supplementary Information. Critically, the model's flexibility allows for certain treatment steps to be bypassed in a given train in the interest of meeting particular effluent objectives/constraints at minimum cost. For example, if salt concentrations are not a concern, the model will not choose any desalination treatment technologies within a particular train given their relatively high cost.

The eleven treatment trains represent current and possible municipal wastewater treatment configurations, which could be implemented by Inland Empire Utilities Agency – Regional Water Recycling Plant #1 (IEUA – RP1) and many if not most other treatment plants throughout the U.S. to achieve the desired water quality. In these treatment trains, the removal of nutrients is achieved through the activated sludge or MBR processes, whereas MF and UF effectively remove large particles/molecules and bacteria/viruses and provide pre-treatment for the desalination steps. The desalination processes (NF and RO) effectively remove divalent and monovalent ions, respectively.

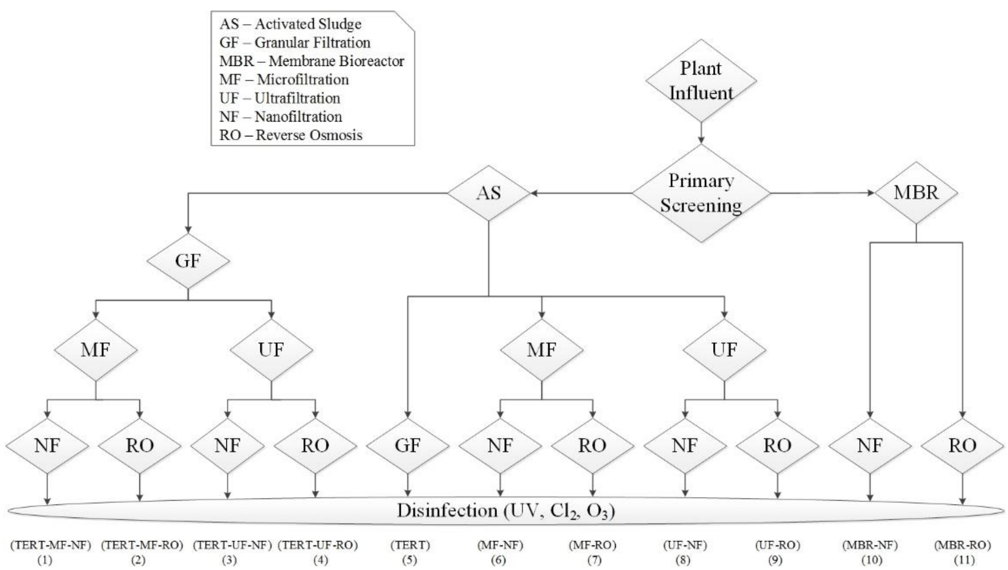

Figure 3.2. Different treatment train processes corresponding to different GAMS

The cost associated with generating a unit volume of treated wastewater from each treatment train was estimated as the product of the unit treatment cost ($/m^3) for individual processes within a particular train multiplied by the blending fraction of the produced water associated with that process, as shown in equation 1:

$$C_T = \sum_{j=1}^{J_T} b_T(j) * c_T(j) \tag{1}$$

where C_T is the total treatment cost per m^3 of wastewater treated, T is the particular treatment train, j is the specific process within a treatment train T, which consists of j_T unique processes. The blending ratio of the produced water associated with process j for treatment train T is designated by $b_T(j)$, whereas the unit cost of each process is $c_T(j)$. The treatment cost is subject to two constraints. First, the total blending fractions of the produced water from each treatment train is restricted to one, corresponding to 100% of a given effluent volume as in equation 2, assuming no water loss in the treatment processes.

$$\sum_{j=1}^{J_T} b_T\,(j) = 1 \tag{2}$$

The concentration of different wastewater constituents (e.g., TDS, phosphorous) in the final blended product must be below specific threshold limits. In the current work, wastewater used for irrigation must comply with general irrigation guidelines established by the Food and Agriculture Organization of the United Nations (FAO) and the NPDES permit or, if discharged, must meet pre-drought treated effluent concentrations (which in our case study presented below is the year 2011)[57]. Assuming water quality parameters are additive, the concentration of specific constituents can be calculated by:

$$\sum_{j=1}^{j_T} p_T(i,j) * b_T(j) \leq a(i) \tag{3}$$

where, i is a specific wastewater constituent associated with process j within a given treatment train T, $p_T(i,j)$ is the concentration of specific constituent i resulting from process j, and $a(i)$ is the specific water quality threshold.

The optimization framework minimizes $C_T = \sum_{j=1}^{J_T} b_T(j) * c_T(j)$ subject to equations

(2) and (3) using the General Algebraic Modeling System (GAMS) software. Thus, the RWRM is capable of producing a particular volume of effluent with specific water quality parameters at lowest cost by blending fractions of different treated wastewater effluents within a particular treatment train.

Modeling methodology

By modifying the RWRM's inputs and constraints, we applied the model to influent entering the IEUA – RP1 plant between 2011-2015, with the goal of identifying the lowest-cost solution to maintain pre-drought discharge quality, ensure treated effluent is suitable for reuse (irrigation), and protect downstream water quality. By identifying cost-effective blending combinations across different treatment processes (which could require infrastructure upgrades to the existing plant), the model suggests a method of restoring plant effluent quality to pre-drought conditions (i.e., 2011 effluent quality). In addition, the resultant water quality was evaluated for regulatory compliance (NPDES permit and Title 22 regulations) and safe agricultural practices based on general irrigation guidelines established by the FAO. Water quantity and quality data were collected from IEUA – RP1 to (a) investigate the changes in influent and effluent quality during drought and pre-drought periods (2011-2015) and (b) make recommendations on treatment train upgrades that would maintain pre-drought discharge quality for safe irrigation practices and state/federal compliance.

Model inputs

The RWRM inputs are comprised of four categories: the concentrations of water quality parameters in a given unit volume of influent wastewater, the removal performance associated with each treatment process, the concentration limits of water quality parameters, and the unit cost for each treatment process (Table A.B.1-A.B.7). Conventional wastewater treatment processes consist of preliminary screening, primary, secondary, tertiary, and disinfection steps, which provide significant removals of nutrients and particles[58]. However, these conventional processes do not remove salinity (i.e., TDS) effectively[40, 58]. On the other hand, membrane separation processes, such as NF and RO, are commonly used in the water treatment industry to remove divalent and monovalent ions, respectively. The mechanisms responsible for ion removal are physical (size exclusion) and chemical (differences in diffusion rates)[59, 60]. Therefore, to achieve a degree of desalination, these membranes must be incorporated into the wastewater treatment train.

IEUA – RP1's treatment train consists of screening, primary, secondary (activated sludge), tertiary (granular filtration), and chlorine disinfection[61]. According to California's Title 22 regulations, recycled water used for food crops, parks and playgrounds, school yards, residential landscaping, and unrestricted access golf courses shall be at least disinfected tertiary-recycled water, in which the turbidity does not exceed 2 NTU and the total coliform concentration must not exceed a 7-day median of 2.2 MPN/100 ml nor a 23 MPN/100 ml value in more than one sample in any 30-day period[62]. The plant's NPDES permit limits total nitrogen concentration in the water discharges to less than 10 mg/L, and limits ammonia-nitrogen (NH_4-N) to 4.5 mg/L to prevent excessive algae growth[57, 62, 63]. IEUA is in compliance with both NPDES and Title 22 regulations for the discharge and reuse of disinfected tertiary-recycled water[64]. Between 2011-2015, IEUA – RP1 effluent water quality was reported with NH_4-N concentrations of approximately 0.1 mg/L, which is far lower than the maximum concentration listed in the plant's permit[63, 65]. Currently, there is no limitation on total phosphorus concentrations in IEUA – RP1 effluent. However, to prevent plant damage, phosphorous concentrations in irrigation water should be kept below 2 mg/L PO_4-P[66]. In addition to phosphorous, elevated HCO_3^- concentrations exceeding 1.5 me/L (91.5 mg/L) in irrigation water can, over time, cause soil damage due to decreased soil permeability[66].

The 2011 and 2015 IEUA – RP1 removal efficiencies of different wastewater constituents by different treatment steps are provided in the Supplementary Information (Table A.B.1 and A.B.2). Wastewater quality parameters after each step of the IEUA – RP1 treatment process are listed in Table A.B.3 and A.B.4; these data were obtained directly from the treatment plant for 2011 and 2015[65]. Since IEUA – RP1 operates a conventional wastewater treatment facility, data on membrane performance (i.e., % removal) were obtained from the literature[40-42, 67]. In instances where data were not available (italicized entries in Table A.B.1-A.B.4), parameters were estimated based on typical removal percentages reported in the literature (e.g., activated sludge, microfiltration, ultrafiltration and granular filtration do not effectively remove salinity). Title 22 specifies a total coliform concentration of less than 2.2 MPN/100 ml for disinfected tertiary-recycled water; therefore, an additional disinfection step using either chlorine, ozone, or UV radiation must to be applied to the final treated effluent to comply with state guidelines.

The individual treatment costs associated with each process within a treatment train for large treatment facilities (> 20000 m^3/d or 5 MGD) are reported in Table A.B.5[32].

Capital costs for each treatment step, including construction and administrative costs, were amortized over 15 to 20 years, depending on the process, plant capacities and time intervals. Operation and maintenance (O&M) costs cover the energy, chemicals, and maintenance costs[68]. Data in Table A.B.5 were adjusted to 2013 U.S. Dollars using the Engineering News Record Construction Cost Index and the Consumner Price Index[69, 70]. To summarize our general cost categories, currently it costs aapproximately $0.69 to produce a cubic meter of chlorine disinfected, tertiary-treated effluent at large conventional wastewater treatment facilities, such as IEUA – RP1; this cost consists of primary treatment ($0.02), activated sludge ($0.32), granular filtration ($0.33), and chlorine disinfection ($0.02)[32]. In our analysis, sensitivity analysis over the range of interest rates (3%-10%) was explored and yielded neglegible impacts on the magnitude of the amortized costs (Table A.B.8-A.B.17).

RESULTS AND DISCUSSION

In this section, we present empirical evidence illustrating how drought and water conservation efforts by water agencies combine to reduce influent quality and flow thereby reducing the amount and quality of water available for reuse. We then identify, using the RWRM, a cost-effective treatment strategy that can mitigate the water quality impacts of drought and water conservation and return effluent water quality to levels consistent with pre-drought levels that also meet international irrigation recommendations. Finally, we relax the effluent quality constraints that were imposed to meet the pre-drought and international recommendations to illustrate how recognition of local conditions and end-use quality requirements can be used to further lower the treatment costs associated with drought response.

The role of drought and water conservation strategies on influent and effluent quality from 2011 to 2015

Changes in influent flow and quality arising from drought and water conservation can have deleterious effects on wastewater treatment plants and treated wastewater recipients. For IEUA – RP1, recipients of the disinfected, tertiary-treated effluent include industry, irrigators (e.g., for golf courses, schools, and parks), municipalities (e.g., for groundwater recharge), and the environment (e.g., discharge into a local creek)[31, 71]. As illustrated in Figure 3.3, IEUA-RPI influent volumes dropped 14% (from 27.8 to 24 MGD) between 2011 and 2015[65]. Such a decrease is not surprising given that in response to the drought that gripped California in 2013, its governor issued a Drought State of Emergency Proclamation in January 2014, asking all Californians to reduce water consumption by 20% and directing local water suppliers to implement local water shortage contingency plans[72]. As shown in Figure 3.3, there is a significant drop in influent flow following the January 2014 proclamation. In April 2015, the State announced conservation mandates that were implemented in June 2015 and required water suppliers to meet conservation reduction targets that varied between 4 to 36% relative to 2013 use. IEUA's lowest inflow level immediately followed the 28% regional conservation mandate announcement it was expected to meet beginning in June 2015[73]. From June to December 2015, IEUA was able to achieve monthly water use reductions of 27.5, 34, 28.5, 30.5, 26.5, 17.5, and 21%, respectively, relative to 2013.

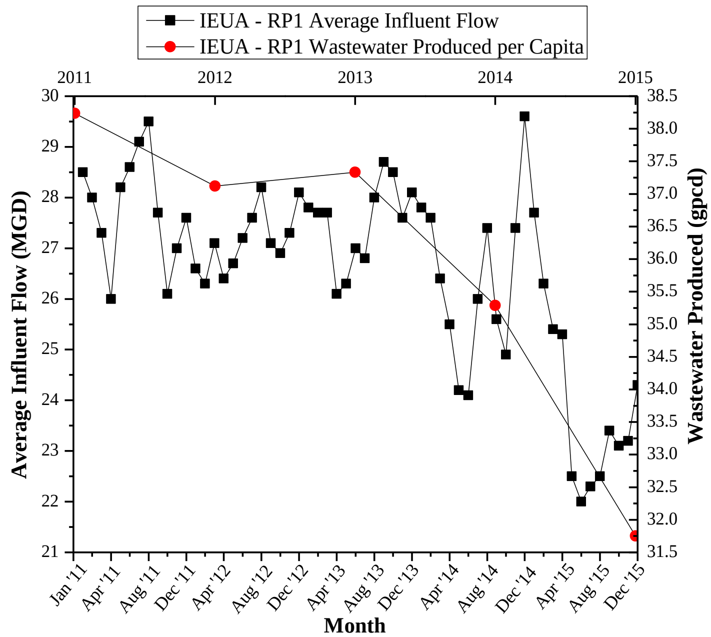

Figure 3.3. IEUA – RP1 inflows vs. wastewater produced per capita in the IEUA – RP1 service area in 2011-2015

While changes in influent flow mostly mirror changes in water conservation, other factors also matter. For instance, we speculate that California's storm of the decade in August 2014, and winter storms that occurred December 2014 to January 2015, and September to December 2015[74-78] increased influent flows via infiltration. Furthermore, freshwater usage within IEUA service area increased in July 2014 by 9% compared to the previous month which resulted in an 8% increase in inflow to the IEUA – RP1 plant[79]. Consequently, freshwater usage and seasonal storms disrupted the influent wastewater correlation with conservation.

In addition to influencing the quantity of influent, drought and conservation may impact the quality of the influent via two mechanisms. First, less water use as a result of conservation activities such as fewer or shorter showers, fewer or fuller loads of laundry, or fewer flushes of the toilet can lead to an increase in the concentrations of the constituents in the generated wastewater. Second, the water supplies that agencies use to provide potable water can become degraded. As shown in Figure 3.4, potable water within the IEUA service area, which is sourced from a mixture of ground and surface water supplies, exhibited a deterioration in quality over the 2011-2015 period in terms of TDS concentrations[80-103]. From 2011 to 2015, average TDS concentrations in the potable water supply increased from 276 mg/l to 330 mg/l, a 20% increase. The increased TDS concentrations in potable water contributed to the increased TDS concentrations entering the IEUA – RP1 plant, which rose from 432 mg/l to 510 mg/L between 2011 and 2015.

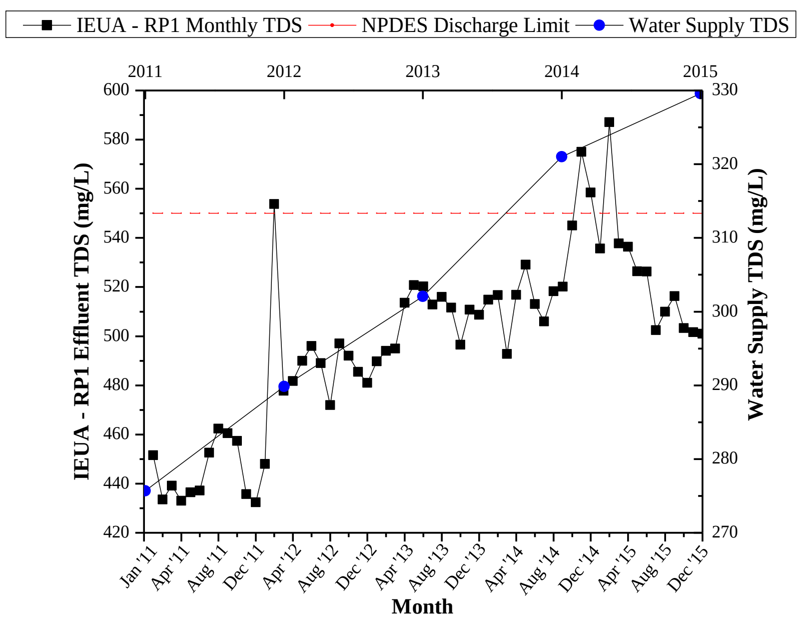

Figure 3.4. TDS concentrations in 2010-2015

The combination of poorer quality water supplies coupled with conservations activities resulted in an increase in pollution concentrations in the influent of IEUA – RP1 between 2011 and 2015 (Figure 3.5). As the worst years of drought were between 2013-2015, the effects of the drought and subsequent indoor water conservation are clearly observed through an increase in the concentrations of certain constituents, including: TDS, electrical conductivity (EC_w), ions (sodium (Na^+), chloride (Cl^-), calcium (Ca^{2+}), HCO_3^-), and nutrients (Inset Figure 5). In addition to the increase in TDS (18% increase), the concentrations of EC_w, Ca^{2+}, Na^+, Cl^-, and PO_4-P in the plant influent also showed an 8%, 13.6%, 16%, 6%, and 28% increase, respectively.

One consequence of higher constituent concentrations in the treatment plant influent is higher constituent concentrations in the effluent. As illustrated in Figure 3.6, between 2011 and 2015 there was an 8 to 16% increase in many of the constituent concentrations at IEUA – RP1, including EC_w (8%) (Inset Figure 3.6), TDS (7%), Ca^{2+} (16%), Na^+ (14%), HCO_3^- (11%), and Cl^- (13%). This is not surprising given that many standard wastewater treatment processes are not designed to remove these constituents effectively[40]. Another consequence is potential discharge violations and fines. Recalling Figure 3.4, during the drought period, there were four discreet events in which IEUA – RP1 effluent discharges violated the NPDES discharge limitations in term of TDS concentrations. These violations can lead to financial penalties, complicated correction processes, and even permit termination[63, 65].

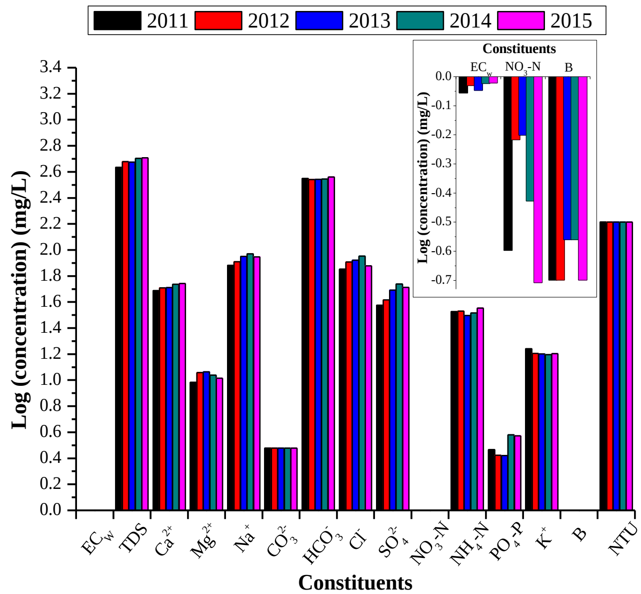

Figure 3.5. IEUA – RP1 influent quality with EC_w, NO_3-N, and B concentrations (inset) in 2011-2015

According to the NPDES permit, the monthly average concentration and loading rate for TDS shall not exceed 550 mg/L and 366,960 lb/day agency-wide, respectively. TDS concentrations exceeded these limits once in 2012, twice in 2014, and once in 2015. While concentrations exceeded permit limits throughout the drought, salt loadings were significantly below the NPDES limit highlighting the lower influent flows that give rise to higher constituent concentrations.

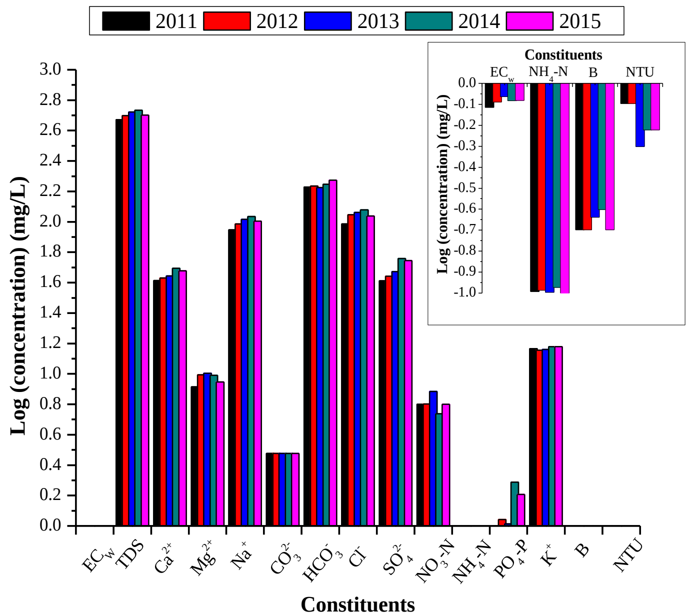

Figure 3.6. IEUA – RP1 effluent quality with EC_w, NH_4-N, B, and turbidity (inset) in 2011-2015

Overall, then, the drought and water conservation measures enacted because of the drought combined to both decrease the quantity and quality of wastewater inflow to IEUA – RP1. While conservation measures are critical to prolonging water supplies, indoor conservation measures may have adverse down-stream effects that reduce the role wastewater reuse may play in helping to mitigate the impacts of drought. Indeed, while average daily water reuse for irrigation volumes remained fairly steady within the IEUA service area from 2011 to 2015, the quality of the water declined. Furthermore, the lower overall treated volumes manifested themselves in lower discharges into surface waters by about 38%. While wastewater treatment plants are limited in mitigating the impacts of lower flows, there may be cost-effective opportunities to mitigate the increase in constituent concentrations that often accompany the lower flows, which we turn to next.

Cost-effective treatment to mitigate poorer quality influent during drought

What the above section has shown, then, is that at a time when the reuse of treated municipal wastewater may be most useful—during drought—the quantity and quality of water available for reuse may decline, thereby reducing its ability to help mitigate the impacts of drought. This section investigates the ability of wastewater treatment plants to address one of the issues—reduced influent quality. Specifically, we illustrate how our model can be used to identify a cost-effective treatment strategy that plants may adopt to mitigate the impacts of drought on the quality of treated municipal wastewater.

To illustrate how wastewater treatment plants might cost-effectively mitigate the poorer quality influent during times of drought, we use the actual water quality parameters IEUA – RP1 confronted in 2015 as a measure of the poorer quality influent and effluent that may result during drought. We then investigate and apply a cost-effective treatment strategy that achieves comparable water quality parameters representative of what might be observed in a non-drought year in which the effluent is suitable for crop irrigation and can be safely used for downstream users without deteriorating surface or ground water quality. For our case here, we choose the water quality parameters observed in the effluent of the IEUA – RP1 plant in 2011 as an example of such a situation. As shown in Table 3.1, the concentrations of many water constituents in 2015 IEUA – RP1 effluent exceeded the levels observed in 2011, including the "no restriction" levels listed in Table A.B.7. In particular, water quality parameters associated with salinity were sufficiently high in 2015 to warrant some additional treatment, including EC_w (0.828 vs. 0.7 dS/m), TDS (504 vs. 450 mg/L), Na^+ (4.42 vs. 3 me/L), NO_3-N (6.314 vs. 5 mg/L), Cl^- (3.08 vs. 3 me/L) and

HCO_3^- (3.08 vs. 1.5 me/L).

To improve 2015 effluent quality to 2011 levels, as well as to be in compliance with FAO guidelines and NPDES limits, it is necessary for the wastewater treatment plant to implement some degree of desalination to reduce salt concentrations in the plant effluent. Traditional wastewater treatment processes, like the IEUA – RP1 plant, were not designed to reduce salinity; consequently, the salinity of some wastewater effluent used for irrigation is above the maximum tolerance for certain crops, which can harm both the soil and plants[32, 66, 104, 105]. The FAO general irrigation water quality guidelines specify that the use of irrigation water with water quality parameters that exceed certain minimums should be "slightly to moderately" restricted, depending on the concentration (Table A.B.7)[66]. So while we constrain our model to meet the 2011 IEUA – RP1 effluent quality levels, our model also requires that both the FAO guidelines and NPDES limits be met. These additional requirements that go above and beyond the 2011 water quality parameters affect the EC_w, TDS, Na^+, HCO_3^-, Cl^-, and NH_4-N, as noted by the asterisk in Table 3.1.

Using the RWRM, we identify the most cost-effective wastewater treatment trains that would improve wastewater effluent quality to sufficient quality under three different scenarios. First, our (baseline) Scenario A involves achieving the water quality standards achieved in 2011 at IEUA – RP1 with the additional FAO and NPDES requirements. The results, shown in the bottom row of Table 3.1, identifies the most cost-effective solution to consist of blending effluent treated by MF with effluent treated under the MF-RO train. Specifically, the cost-effective solution consists of blending a mix of 47.1% MF effluent with 52.9% MF-RO effluent, which then undergoes disinfection. Because the model has to choose a treatment train for which no constituent concentrations exceed those concentrations in 2011 or that meet FAO guidelines, we see that the binding constituent is HCO_3^-, with lower concentrations for all the other constituents. For example, TDS concentrations were 44% lower than TDS values measured in 2011 (253.45 mg/L vs. 450 mg/L).

Table 3.1. Baseline, Drought, and Scenario Concentrations and Costs

	2015 IEUA – RP1 effluent quality (mg/l)	2011 IEUA – RP1 effluent quality, FAO guidelines, and NPDES discharge permit (mg/l)	Percent change from 2011 Values		
			Scenario A	Scenario B	Scenario C
Parameters					
EC (dS/m)	0.828	0.7*	-32%	-22%	-19%
TDS	504	450*	-44%	-35%	-33%
Ca^{2+}	47.58	41.16	-36%	-26%	-23%
Mg^{2+}	8.833	8.228	-40%	-30%	-27%
Na^+	101.08	69*	-38%	-28%	-25%
CO_3^{2-}	3	3	-52%	-45%	-42%
HCO_3^-	187.88	91.5*	0%	+16%	+20%
Cl^-	109.25	106.5*	-66%	-60%	-59%
SO_4^{2-}	55.5	40.92	-40%	-30%	-27%
NO_3-N	6.314	5*	-38%	-28%	-26%
NH_4-N	0.1	4.5**	-99%	-99%	-99%
PO_4-P	1.611	0.861	-14%	0%	+4%
K^+	15.08	14.64	-47%	-38%	-36%
B	0.2	0.2	-50%	-43%	-41%
TC (MPN/100 ml)	<2	<2	-100%	-100%	-100%
Turbidity (NTU)	0.6	0.8	-92%	-91%	-91%
% Blending					
MFCl	N/A	N/A	47.1%	55.0%	57.2%
RO-M	N/A	N/A	52.9%	45.0%	42.8%
Treatment cost ($/m^3)***	$0.69	$0.69	$0.775	$0.747	$0.740
*FAO guidelines **NPDES discharge limits ***Costs based on 2013 U.S. Dollar					

In this sense, the model chooses a solution that overtreats the influent in every dimension except HCO_3^- and thus produces a superior quality effluent. The bottom row of Table 1 illustrates that the cost associated with this treatment scenario is \$0.775/m^3, which is 12% higher than the current treatment cost incurred by the IEUA – RP1 plant.

The second scenario—scenario B—imposes a less restrictive constraint on HCO_3^- concentrations, which were the binding constraint from the baseline scenario. The initial HCO_3^- constraint was based on FAO recommendations. HCO_3^- limits, as stated in the FAO's guidelines, are important because elevated HCO_3^- concentrations can have negative effects on soil permeability, which impacts water infiltration particularly during periods of low humidity and high evaporation[66]. Carbonate species will combine with Ca^{2+} and Magnesium (Mg^{2+}) and precipitate as Ca-$MgCO_3$. This process removes Ca^{2+} and Mg^{2+} from the soil water and results in an increase in the proportion of Na^+ which, in turn, increases the sodium hazard in the soil[66]. The effect of soil permeability is evaluated by residual sodium carbonate (RSC), which is an index of the sodicity hazard associated with the irrigation water. Water containing less than 1.25 me/L of RSC is considered safe for irrigation[106]. The RSC is defined by the formula:

$$RSC\ (\frac{me}{L}) = (CO_3^{2-} + HCO_3^-) - (Ca^{2+} + Mg^{2+})$$

Elevated concentrations of certain irrigation water constituents (10-20% greater than FAO guidelines) can be tolerated when considered with other factors, such as sodium adsorption ratio (SAR), RSC, and EC_w, that affect crop yield[66]. Thus, relaxing HCO_3^- concentration constraints (scenario B) may be tolerable when HCO_3^- concentrations in irrigation water do not exceed 1.8 me/L or 109.8 mg/L (20% over the FAO limit), which corresponds to an RSC value of -0.84 me/L. The RSC value generated under scenario A was -0.18 me/L, which is far below the FAO's guidelines, again indicating *overtreatment* of the wastewater.

As shown in Table 3.1, the cost-effective strategy arising from relaxing the HCO_3^- constraint while still maintaining the 2011 concentration limits for the other constituents (scenario B) consists of a blend that is 55% MF effluent and 45% MF-RO effluent, again followed by chlorine disinfection. By relaxing the HCO_3^- constraint, the cost-effective solution relies less on the costlier MF-RO treatment than the strategy under scenario A, thereby lowering treatment costs by approximately 4%. Of course, costs are still higher compared to current 2015 costs, around 8%, but overall water quality improves for all of the remaining parameters except phosphorous (PO_4-P), which is now the binding constraint.

The last scenario considers relaxing the phosphorus constraint, a constraint defined by its 2011 value. While phosphorous is one of the essential nutrients for plant growth, excessive phosphorus can interfere with iron and zinc uptake by plant roots[107]. In addition, excess phosphorus from wastewater discharge and runoff can contribute to algal blooms and low dissolved oxygen levels in water bodies, which in turn can harm aquatic species[47, 108-110]. Currently, there is no limitation on the discharged total phosphorus in wastewater effluent for streams in California[111]. Therefore, phosphorus concentrations in treated wastewater could be above the threshold over which it impacts crop yield (2 mg/L PO_4-P)[66]. As Table 3.1 illustrates, PO_4-P concentration limits that achieve the 2011 levels are significantly less than what is identified by the FAO. The final scenario, then, identifies the cost-effective solution of treating the 2015 influent to 2011 effluent levels except that in addition to relaxing the HCO_3^- constraint as imposed in scenario B, the PO_4-P constraints are relaxed to FAO standards.

As the final column in Table 3.1 illustrates, relaxing both the HCO_3^- and PO_4-P relative to scenario A (the 2011 levels with strict FAO constraints on HCO_3^-) results in a further reduction in the use of RO for treatment. As shown, the proportion of influent treated by MF increased to over 57%, while the amount treated by RO dropped to 43%, for about a 5% reduction in costs relative to scenario A yet still a 7% increase from current treatment costs. While this treatment strategy leads to an increase in 15 of the 16 constituent concentrations relative to scenario A, the effluent would still be considered overtreated relative to the 2011 values.

CONCLUSIONS

In times of drought, which are expected to increase in both frequency and severity, the availability of freshwater resources decreases and its quality, particularly with respect to salinity, often declines. In response, water agencies are increasingly implementing conservation measures to reduce demand while investing in the reuse of treated municipal wastewater to augment supply, both with the intention to increase resilience. Our results show that both the water supply effects of drought *and* the conservation measures enacted in response to it combine to reduce the quantity and quality of influent available for treatment which, in turn, reduces the quantity and quality of treated municipal wastewater for reuse under conventional treatment processes. Our modeling results also illustrate that cost-effective blending strategies can be implemented to mitigate the water quality effects, increasing the value of the remaining effluent for reuse, whether it be for surface water augmentation, groundwater replenishment, or irrigation of crops, golf courses, or landscapes. We also highlight the benefits of treating the wastewater based on the characteristics of demand. In our case study, relaxing the constraints on both HCO_3^- and PO_4-P concentrations relative to the pre-drought levels lead to reduced treatment costs while still achieving an effluent quality superior to pre-drought levels for all the other water quality parameters.

From a management and policy perspective, three significant conclusions can be drawn from our analysis. First, municipalities, cities, and regions that rely on both conservation and reuse as a means to address drought and water scarcity need to recognize the potential dependence of the latter on the former in terms of its effect on the potential supply of treated municipal wastewater. To the extent possible, efforts to promote and advocate for outdoor water conservation rather than indoor conservation break this dependence. Second, drought and the conservation measures enacted in response can result in poorer quality water, particularly with respect to salinity. Given that conventional treatment processes are not designed to address these higher constituent loads, the value of the remaining effluent likely decreases relative to the downstream demands it serves. Consequently, recognizing this relationship should help the recipients of the treated municipal wastewater better plan for such outcomes and thus engage in cost-effective adaptation. Finally, while wastewater treatment plants themselves cannot mitigate the reduced flows that are the result of the drought and conservation measures, our modeling results illustrate that cost-effective treatment trains can be developed to mitigate the water quality effects of drought and conservation thereby increasing the value of the remaining effluent for reuse. We propose that by working together, recipients of treated municipal wastewater and the agencies themselves can identify cost-effective strategies in terms of the degree of treatment that provides the greatest benefit to society.

Our conclusions here are not simply fodder for the academic mill. In our particular case study, treated wastewater makes up a significant portion of flows in the Santa Ana watershed, as well as Southern California's waterways. It plays an even more significant role during drought conditions, when precipitation and snowmelt decrease. Thus, it is important to consider how deteriorating wastewater effluent quality, and in particular, elevated salinity levels, impacts downstream users. Our modeling results demonstrate that incorporating a desalination step into the wastewater treatment process can alleviate some of these downstream concerns at a cost that is within 8% (currently \$0.69/m^3 vs. \$0.74/m^3 under scenario C) of current treatment costs. The resulting effluents are composed of partially desalinated wastewater that is suitable for crop irrigation and stream augmentation at a quality that protects wastewater treatment agencies from discharge violations and prevents surface water quality from further deterioration.

Identification of such low-cost wastewater treatment strategies should be useful to municipalities, both in California and globally, as they continue to strive to improve their resilience to drought via demand side management strategies that include reducing indoor water use and supply augmentation strategies that include wastewater reuse[33]. The RWRM model used in this research can be easily adapted to other applications to identify low-cost strategies given its flexibility and replicability. The model requires unit cost and effectiveness parameters on commonly used treatment technologies, parameters that are regularly reported in the academic literature and/or industry reports. The parameters that represent regulatory, surface water, or crop threshold constraints on effluent quality also can readily attained from public documents. For instance, this model was previously used to identify the most cost-effective treatment solutions when the treated wastewater effluent is used for irrigating citrus and turfgrass in Southern California[32]. Consequently, the RWRM is a flexible and easily adaptable model that can assist water managers in their efforts to develop water portfolios that cost-effectively and reliably respond to drought and increasing water scarcity worldwide.

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Chapter 4

The Role of Existing and Emerging Water Resources in Managing Groundwater Aquifers in the Face of Climate Uncertainty

In preparation for *Nature of Climate Change*

Tran, Q., Jassby, D., Ajami, H., Franklin, B., and Schwabe, K.

ABSTRACT

Water resources are under increasing stress due to excessive population growth, climate change, and its impact on the frequency and intensity of drought. Decreases in surface water supply due to prolonged drought have increased reliance on groundwater. This in turn, impacted groundwater availability and/or have increased pumping costs due to deeper water levels. Consequently, opportunities to augment groundwater supplies through the reuse of wastewater and desalination are increasingly being explored and adopted. Substituting treated municipal wastewater for untreated or treated freshwater will allow the latter to be used to better cope with droughts and population growth. The objective of this paper is to develop a constrained optimization model that incorporates surface and groundwater interactions to both simulate and evaluate the impacts of changes in economic, biophysical, and institutional factors on municipal water management costs and groundwater sustainability. This framework also can be used to the impacts of water management alternatives on groundwater systems and identify cost-effective solutions to water agencies under different climate scenarios—as represented by different and stochastic natural recharge rates and evapotranspiration—to water agencies. With the costs of water management likely to increase in the future, and climate change further straining the availability of water resources, our framework can be a useful tool to managers and policymakers as they seek to better understand consequences of changes economic, biophysical, and institutional factors and develop cost-effective strategies in response to those changes.

INTRODUCTION

Southern California's water scarcity is compounded by its arid climate and fast growing population[67]. In California, the agricultural sector is the largest consumer of fresh water, which is particularly complicated for water managers in the southern part of the state, where the majority of the population resides and where natural rainfall is erratic[68]. As water becomes scarcer, agriculture that is on the urban fringe and which competes with municipal demands might have a difficult time avoiding an ever-increasing water cost due to the increased scarcity. In addition, Southern California's already low and erratic rainfall is expected to experience increased variability in rainfall intensity and timing (due to climate change), which will further complicate the management of water resources for large urban centers[69, 70]. Currently, Southern California residents rely on groundwater and imported water from Northern California, and the Colorado River, delivered via the state water project (SWP) and the Colorado River Aqueduct (CRA), respectively[69]. However, as the recent 2012-2016 drought has demonstrated, SWP allocations can vary significantly (between 5% - 85% of average allocations), which prompted emergency water conservation mandates by the state's governor, and resulted in dramatic increases in groundwater withdrawals by farmers[14-22]. The increased withdrawals resulted in rapid depletion of groundwater resources, and led to the passage of the landmark Sustainable Groundwater Management Act (SGMA)[48]. SGMA is created to raise awareness of the importance of groundwater resources and promote groundwater management to achieve the sustainable management of groundwater resources in California[46, 47]. Effective and successful groundwater management requires significant efforts, commitment, and collaboration from water managers, public water agencies, and communities to protect and sustain groundwater resources, especially under future water uncertainty[45, 47]. In the face of climate uncertainty and increased drought probability, there is a strong interest by local water agencies to decrease their reliance on imported water supplies, thereby increasing local long-term drought resilience and water resource sustainability[11, 37, 38, 40, 41].

Municipalities tend to use the least-cost water resources available, which typically include surface water or groundwater[2, 3]. Depending on scarcity, these resources can be augmented by other, more expensive supplies, including imported water, desalinated seawater and brackish water, and recycled wastewater. As traditional water sources dwindle, municipal agencies have been exploring methods to shore up their supplies. One method that has attracted significant attention is using recycled wastewater to augment local supplies[2, 24, 27, 28, 30-35, 50-53, 55, 57, 59, 60, 71-82]. The rationale for this attention is the reliability of this source, and recent advancements in treatment that have dramatically reduced the cost of wastewater recycling[83]. Depending on the level of treatment, treated wastewater can be used for a wide range of applications, including commercial irrigation, discharge to surface water for environmental purposes, or recharging groundwater basins[11, 12]. The volumes of wastewater available for recycling are dependent on indoor water usage. In Southern California, approximately 13 percent (670,000 acre-feet/year (AFY)) of wastewater is actively recycled, out of 5,000,000 AFY of wastewater being treated[6]. The balance of treated municipal wastewater is released into the environment. Therefore, recycled wastewater has the potential of substantially contributing to the water resources portfolio of municipalities in Southern California.

Currently, groundwater is the lowest-cost water source in Southern California, with the cost to produce this water dependent on its quality and the depth from which it is withdrawn[2, 44]. Nonetheless, there is limited collaboration on groundwater management between agencies that share an aquifer[45-47]. Information relating to groundwater extraction, groundwater use, managed and natural recharge throughout California is limited[46]. Unconstrained use of this source has led to groundwater table depletion, land subsidence, and impact of water quality[45]. Groundwater depletion and degradation of groundwater aquifers results from a lack of effective governance[47]. Moreover, climate change conditions have an immediate impact on the natural recharge in some regions. The major groundwater recharge source in mountain regions of the western U.S. is natural precipitation (snowmelt and rain)[40]. However, the amount of natural recharge in Southern California is expected to decrease as recharge shifts from snowmelt-dominated to rainfall-dominated pattern[40]. The coupling of climate change and a growing population presents a challenge to sustainable management of groundwater resources; as demand increases and recharge decreases groundwater levels drop, which results in increased production costs and potential depletion[40]. Thus, municipalities are exploring adding additional resources to their resource portfolios, with cost, quality, and reliability concerns serving as guidelines in this quest[7, 8, 11, 41, 84].

In addition, as the water scarcity progresses coupled with the impacts of climate change conditions on water supplies, desalination water become an attractive potable water source that can supplement the municipal water needs under future uncertainty[48, 49]. This source of water is abundant but comes with extremely expensive treatment costs[50, 51]. The effects of groundwater depletion and climate change may leave municipalities with no other choice except incorporating desalination water as a water supply source to meet their residential and commercial water demands.

In this manuscript, we describe a complex supply-demand model that evaluates the role of recharge from treated municipal wastewater on local water supply reliability, groundwater sustainability, and recommends cost-effective water management alternatives to water agencies. The model is a complex water balance that considers trade-offs between water supplies and demands, while taking into account both climate-change scenarios that affect local recharge rates and evapotranspiration rates, as well as changes in population, treatment costs, and regional supplies. The model predicts water availability, groundwater extraction, technological needs, and supplemental water sources designed to meet the demands of a municipality over a 100-year period to give water agencies the tools needed to make informed management decisions under climate uncertainty. The model uses the Bunker Hill groundwater basin, located in Southern California, as an example. However, the modularity and flexibility of the model allow it to be applied to any groundwater basin.

MATERIALS AND METHOD

Model Inputs

The model inputs include the cost of each treatment process (2015$ U.S.), aquifer parameters, salinity of each water source, population at the entity using the aquifer, groundwater level, imported water allocations, natural recharge of the aquifer, and evapotranspiration (ET) from irrigated lands (Table A.C.1). The treatment costs for each treatment process include construction and administrative costs and were amortized over 15-20 years[2]. All costs were adjusted to 2015 U.S. Dollars using the Consumer Price Index[85]. The basin is located at the top of the Santa Ana River watershed and covers approximately 92,000 acres[86]. The basin provides water to approximately 650,000 people in the cities of Redlands, Highland, San Bernardino, Loma Linda, Colton, Rialto, Bloomington, Fontana, Grand Terrace, and Riverside. The area of the groundwater basin in this research is assumed to be constant with regard to the basin depth. The land surface above the aquifer is 1,200 ft above the mean sea level (msl). The current groundwater level of the basin is 780 ft above msl (420 ft below land surface) and the bottom of the aquifer is 400 ft below msl[86]. To avoid a solution which includes the complete depletion of the aquifer, we placed a 100 ft buffer at the lower bound of the basin (i.e., groundwater levels cannot drop below 300 ft below msl). The initial TDS in the groundwater basin is 331 mg/L[87] and the upper bound of the TDS in the groundwater basin was set at 500 mg/L[88]. Additional data on groundwater basin parameters, salinity, and recharge can be found in Table A.C.1.

We investigated the level of the aquifer at two different time frames under different climate conditions to examine the sensitivity of the model. First, we assume the aquifer is full when the groundwater level is at 780 ft above msl and depleted with the groundwater level is at sea level. Our “full” assumption represents the historic condition of the basin over the 50-year period from 1950-1999 (baseline period), whereas the projected condition is the future condition of the basin from 2000-2099 (projected period). The climate and hydrological parameters including the representative concentration pathway (RCP), precipitation, natural recharge and ET are extracted from the Global Climate Model (GCM-CCSM4) and the Variable Infiltration Capacity (VIC) model. RCPs are scenarios that describe alternative trajectories for carbon dioxide emissions and the resulting atmospheric concentration[89]. The RCPs describe 4 different scenarios based on different assumptions about socio-economic change, energy consumption and land usage, and the emissions of greenhouse gases and air pollutants[89]. Here, we investigate the impacts of climate change on the groundwater basin's sustainability by investigating two climate scenarios: i) average conditions (based on RCP4.5 projections), and ii) extreme conditions (based on RCP8.5 projections)[39]. The RCP4.5 is a stabilization scenario and assumes that climate policies are involved to achieve the goal of limiting emissions and radiative forcing[90]. RCP8.5, on the other hand, takes into account the assumptions about high population, slow income growth, modest rates of technological change leading to high green-house gas emissions in the absence of climate change policies[91]. In this model, RCP8.5 is considered as the extreme climate condition while RCP4.5 is the average climate condition. The stochastic recharge and ET is calculated using the Monte Carlo simulations and running horizon methods. Monte Carlo simulations are used to develop confidence intervals and summary statistics for our choice and state variables based on multiple runs of the stochastic dynamic optimization model. The running horizon method is used to eliminate endpoint effects associated with our finite-horizon dynamic optimization routine. The probably distribution that represents stochastic recharge is based on fitting a normal distribution to 300 data points related to natural recharge in the region under different climate change scenarios[92].

Development of the Constrained Supply-Demand Optimization Model

The supply-demand water balance model involves implementing a constrained optimization groundwater model to evaluate management options under climate change uncertainty. Importantly, the model minimizes the cost of providing water to meet a utility's responsibility to provide water to its residential and commercial clients by choosing the most cost-effective combination among groundwater (GW), imported water (IW), surface water (SW), desalinated water (DW) and recycled wastewater (RW); the modular nature of the model allows these potential resources to be adjusted/removed and thus allows for a wide range of scenarios to be evaluated. The constraints imposed on this model ensure the supplied water meets regulatory (i.e., water quality parameters) and end-use demand at the least cost.

The alternative water sources, demands, and water pathways for a hypothetical municipal entity in Southern California are described in Figure 4.1. Demand is categorized by residential, agricultural, and commercial, industrial and institutional (CII), while potential supplies include groundwater, surface water, imported water, and desalinated seawater. "Used" water can either be directed to a wastewater treatment plant, or, if used for irrigation, some of it evapotranspires and the rest percolates to the groundwater. Wastewater can either be treated and discharged to the environment (secondary treatment and released into the Santa Ana River (SAR)), treated and returned to the groundwater via percolation basins, or used for irrigation (tertiary treatment and/or reverse osmosis (RO)). In this model, no direct potable reuse is permitted, although it would be possible to modify the model to accommodate this. In addition, groundwater is recharged via managed aquifer recharge (recycled water and/or imported water) and natural recharge (precipitation/infiltration, snowmelt, etc.). Water for irrigation can be obtained from groundwater, imported water, and/or recycled water.

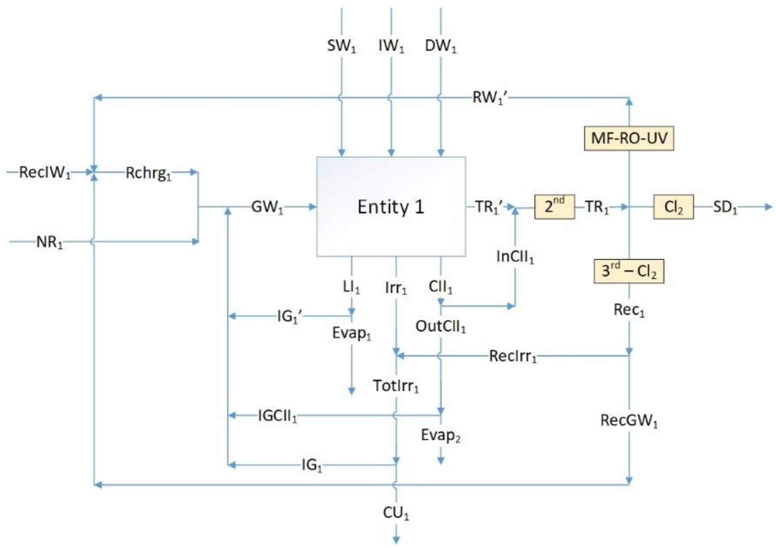


Figure 4.1. Schematic of Water Supplies and Demands by Entity 1

The water balance (i.e., water usage) at an entity is defined by Equation 1 (total supplies must equal total demand in acre-feet per year):

$$GW_1(t) + IW_1(t) + SW_1(t) + DW_1(t) = TR_1'(t) + Irr_1(t) + CII_1(t) + LI_1(t) \quad (1)$$

where, on the supply side $GW_1(t)$ is groundwater extraction, $IW_1(t)$ is import water, $SW_1(t)$ is surface water, $DW_1(t)$ is desalinated seawater; on the demand side, $TR_1'(t)$ is wastewater generated by residential indoor usage, $Irr_1(t)$ is groundwater for commercial irrigation, $CII_1(t)$ is commercial, industrial, and institutional (CII) water use, and $LI_1(t)$ is residential outdoor water usage. $LI_1(t)$ values range between 50-60%[93-97] of the total freshwater needed for residential usage, and residential water usage at entity 1 is estimated to be 98.4 gallons per capita per day ($GPCD_1$) in 2015[98].

The objective of this model is to minimize the total net present value (NPV) cost (C) of water supplied over a period of 100 years under various constraints, including meeting demand each year. The yearly cost of water supplied $SS(t)$ is defined by the total cost of groundwater extraction, imported water (e.g., SWP), surface water, desalinated water, wastewater treatment costs, and groundwater recharge costs with discount factors.

Details on equations used are described in the supporting information (Appendix C). The model will minimize the total cost of water supplied over 100 years subject to water availability conditions and demand constraints (Equation 2):

$$C = \sum_{t=1}^{t=100} SS(t) \tag{2}$$

Modeling Methodology

The supply-demand optimization water model was used to evaluate management options of a hypothetical groundwater basin in Southern California under different basin level and climate conditions (i.e., full aquifer vs. stressed aquifer, average (RCP4.5) vs. extreme (RCP8.5) climate conditions, baseline (historic) vs. projected conditions). Evapotranspiration and natural precipitation data were extracted from VIC and GCM-CCSM4 models. There are 6 different scenarios associated with each climate condition being investigated: 1) Baseline, full aquifer; 2) Baseline, stressed aquifer; 3) Projected, full aquifer; 4) Projected, stressed aquifer; 5) Projected, no agricultural irrigation; and 6) Projected, high salinity imported water from Colorado River (Table 4.1). These scenarios are examined to inform water agencies about groundwater/basin management options under future uncertainty.

Table 4.1. Descriptions of Different Scenarios

Scenario(s)	Description(s)	Time Frame	Groundwater level (ft)[(1)]
Scenario 1	Baseline, Full Aquifer	1950-1999	780
Scenario 2	Baseline, Stressed Aquifer	1950-1999	780
Scenario 3	Projected, Full Aquifer	2000-2099	0
Scenario 4	Projected, Stressed Aquifer	2000-2099	0
Scenario 5	Projected, No Irrigation Water[(2)]	2000-2099	0
Scenario 6	Projected, High Salinity Imported Water from Colorado River (TDS ~ 658.33 mg/L)[(3)]	2000-2099	0

[(1)]Groundwater level above mls; [(2)]Assume no irrigation water usage at entity 1; [(3)]Assume no SWP and the only source of imported water is from high salinity Colorado River with TDS approximately 658.33 mg/L

We applied the model to a groundwater basin in Southern California. The modelling framework is designed to represent demand for each major sector and the viable sources of water that can be used to meet each type of demand. By identifying the most cost-effective set of management decisions over a period of 100 years, the model suggests different combinations of groundwater extraction and artificial recharge rates. The model can choose various combinations of imported water and recycled wastewater to ensure water demands are met at least cost.

RESULTS AND DISCUSSION

Scenario 1 – Baseline, full aquifer (h=780 ft)

Under the baseline condition, when the basin is full (depth-to-water level is at 780 ft below land surface), the stochastic natural recharge each period is generated randomly based on draws from a probability distribution representing historic natural recharge. The stochastic recharge under the extreme climate condition RCP8.5 is better than the average condition RCP4.5 (Figure 4.2)[99], which is consistent with the recent findings from Allen and Luptowitz (2017). The warming of sea surface temperature in the tropical Pacific region has caused an increase in precipitation in California under RCP8.5. Thus, under different climate conditions, some areas experience less natural recharge while others will experience more natural recharge. The effect of climate change has a more definitive impact on reference evapotranspiration (ET) due to temperature increases under the extreme climate condition as seen in Figure 4.2[15, 71]. The stochastic ET is generated randomly using the mean and standard deviations of the historic data. The effect of ET is more severe under extreme climates due to high atmospheric temperatures. However, the differences between stochastic natural recharge and evapotranspiration are less significant during the baseline timeframe. The natural recharge and evapotranspiration under RCP8.5 are within 3% compared to those under RCP4.5 condition.

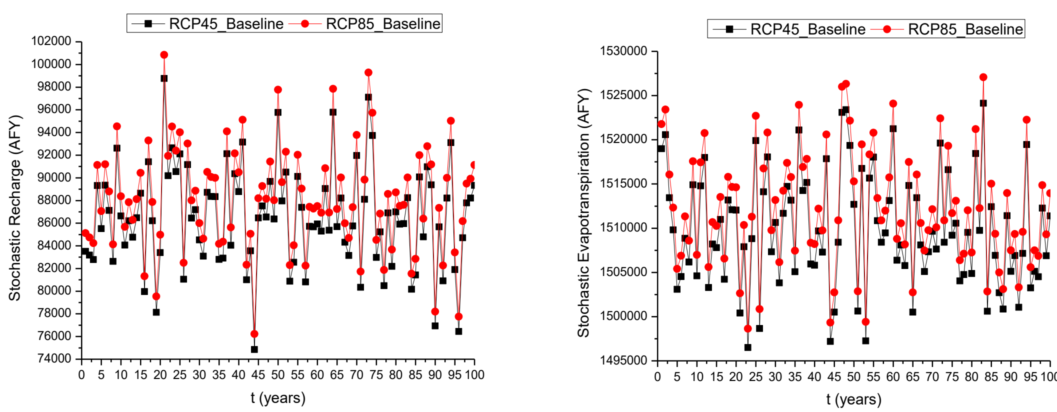
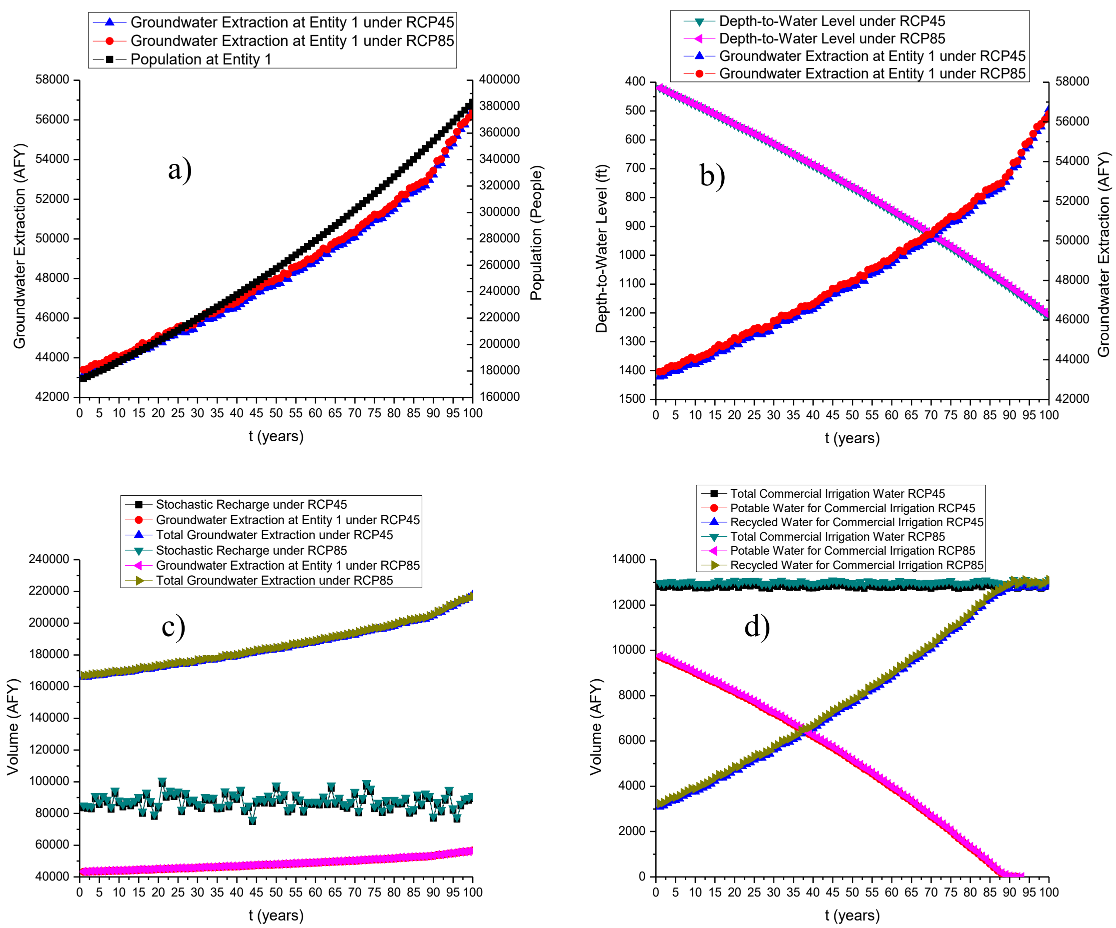


Figure 4.2. Baseline Stochastic Recharge and Evapotranspiration (1950-1999)

The population at entity 1 in 2000 was approximately 174,000 people and increased with the rate of 0.8% annually. At the end of year 2099, the population reaches slightly over 383,000 people. The groundwater extraction at entity 1 to supply residential and commercial water needs increases from approximately 43,000 to 57,000 AFY after 100 years due to population growth (Figure 4.3a). Under the baseline scenario, the groundwater extractions are nearly identical under the different RCP conditions since the natural recharge and ET are relatively similar for both cases; thus, the depth-to-water levels are expected to be the same. The groundwater level drops approximately 800 ft from 420 ft to 1200 ft below land surface in the next 100 years when groundwater is the only water supply source (Figure 4.3b). The total groundwater extraction of the aquifer (from all entities using the same basin) is always higher than the stochastic recharge; thus, the water table drops continually until it reaches the msl (1200 ft below land surface) at year 100 as seen in Figure 4.3b and 4.3c. The total agricultural irrigation water is a combination of disinfected, tertiary treated recycled water and potable water from groundwater (Figure 4.3d). In this paper, the total agricultural irrigation volume is about 13,000 AFY (on average) and varies with respect to stochastic ET. As recycled water is dependent on the residential indoor water usage, and indoor water use increases over time due to population growth, more recycled water is produced from municipal wastewater treatment plants and delivered for agricultural irrigation. At the same time, the cost of potable water use from groundwater increases as the water table drops over time. Therefore, the recycled water becomes a larger fraction of irrigated water use than the potable water source. As shown, after year 94, 100 percent of agricultural irrigation water is from recycled water due to sufficient recycled water production (Figure 4.3d). The most cost-effective way for water agencies to deal with recycled water is either discharging it to the environment (via surface water) or delivering it to crops (via agricultural irrigation). It is more costly for managed aquifer recharge because the entity needs to account for the recharge cost and the groundwater extraction cost to pull that water up for usage. The environmental flow to SAR is disinfected, secondary treated recycled water and is maintained at least 16,000 AFY. The total recycled water produced by entity 1 (TR_1) can be used for environmental flow (SD_1) and/or irrigation ($RecIrr_1$) and managed aquifer recharge ($RecGW_1$) purposes (Figure 4.3e-f).



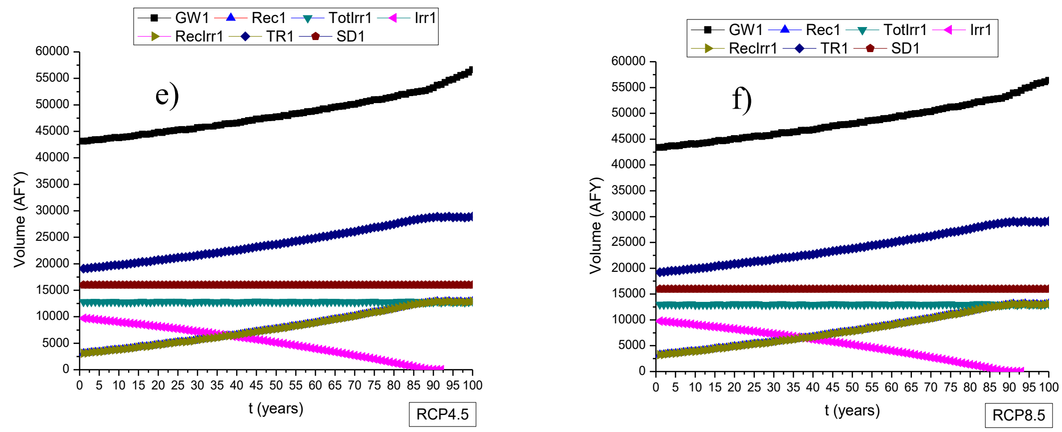


Figure 4.3. Comparison under Different Climate Conditions (Baseline/Full Aquifer)

a) Groundwater Extraction vs. Population
b) Groundwater Extraction vs. Depth-to-Water Level
c) Groundwater Extraction vs. Stochastic Recharge
d) Agricultural Irrigation
e) Water Sources under RCP4.5
f) Water Sources under RCP8.5

The total dissolved solids (TDS) in the groundwater basin is monitored using the TDS of incoming recharge sources to the basin, such as natural recharge, groundwater infiltration from irrigation and managed aquifer recharge using either imported water or recycled water. The current TDS level in the groundwater basin is 331.27 mg/L[70]. Since the only recharge source of the basin under the baseline condition is from natural recharge (TDS ~300 mg/L), the TDS in the basin reaches equilibrium at 300 mg/L. Under this scenario, the effect of climate change is negligible, and the model produced similar results for both RCP4.5 and RCP8.5 conditions (Figure 4.4).

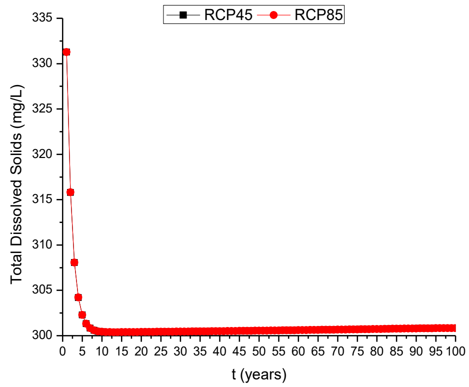


Figure 4.4. Total Dissolved Solids in Groundwater Basin (Baseline/Full Aquifer)

The groundwater extraction cost is the cost to lift groundwater to land surface. As the basin level decreases as seen in Figure 4.3(b), the cost to bring water to land surface also increases. In this scenario, the groundwater basin is full at year 1 and water agencies rely solely on groundwater in the next 100 years as the main water supply source. The cost of groundwater extraction increases from \$5.5 million per year (year 1) to almost \$17 million per year (year 100) (Figure 4.5). In 100 years, the net present value (NPV) of water supply cost for entity 1 is approximately \$525 million under either of the climate scenarios evaluated (Figure 4.5).

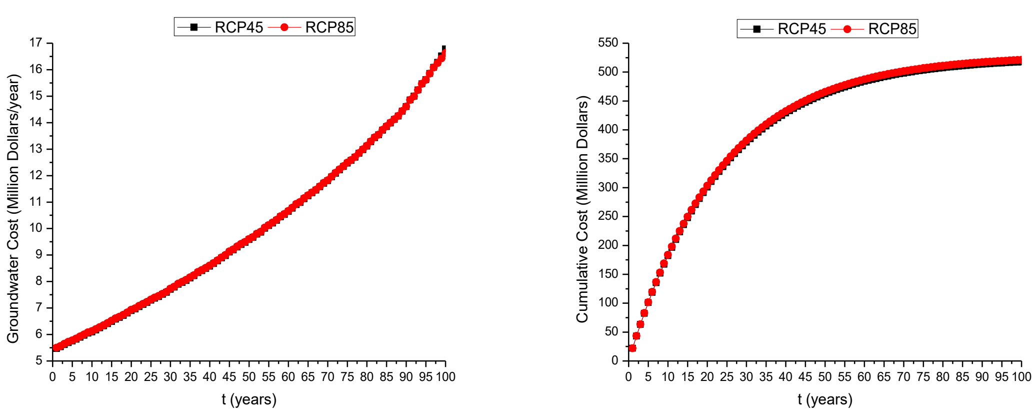


Figure 4.5. Water Cost under Different Climate Conditions (Baseline/Full Aquifer)

Scenario 2 – Baseline, stressed aquifer (h=0ft)

This scenario examines the baseline condition when the aquifer is stressed. The current depth-to-water level in this scenario is at 1200 ft below land surface which is also msl. Groundwater extraction begins to drop around year 35 due to the rising pumping costs associated with the lower water table (Figure 4.6a). At year 48, the depth-to-water level reaches the buffer level (100 ft above the basin bottom) and a hard constraint was put at this level to prevent further groundwater extraction (Figure 4.6b). At this time, the natural recharge begins to recharge the basin gradually (Figure 4.6c). However, since groundwater is still the least expensive water source, largely due to low per unit energy costs, groundwater recharged in the previous year will be used up the next year. There is no long-term accumulation of groundwater recharge in this scenario. The total agricultural irrigation is similar to the previous scenario, with recycled water completely replacing potable water at year 94 (Figure 4.6d). Under this scenario, the effect of climate change is slight. The extraction under RCP8.5 is higher than that of RCP4.5 after year 45, largely due to the higher natural recharge rate observed under RCP8.5. This will be explained further in the later part.

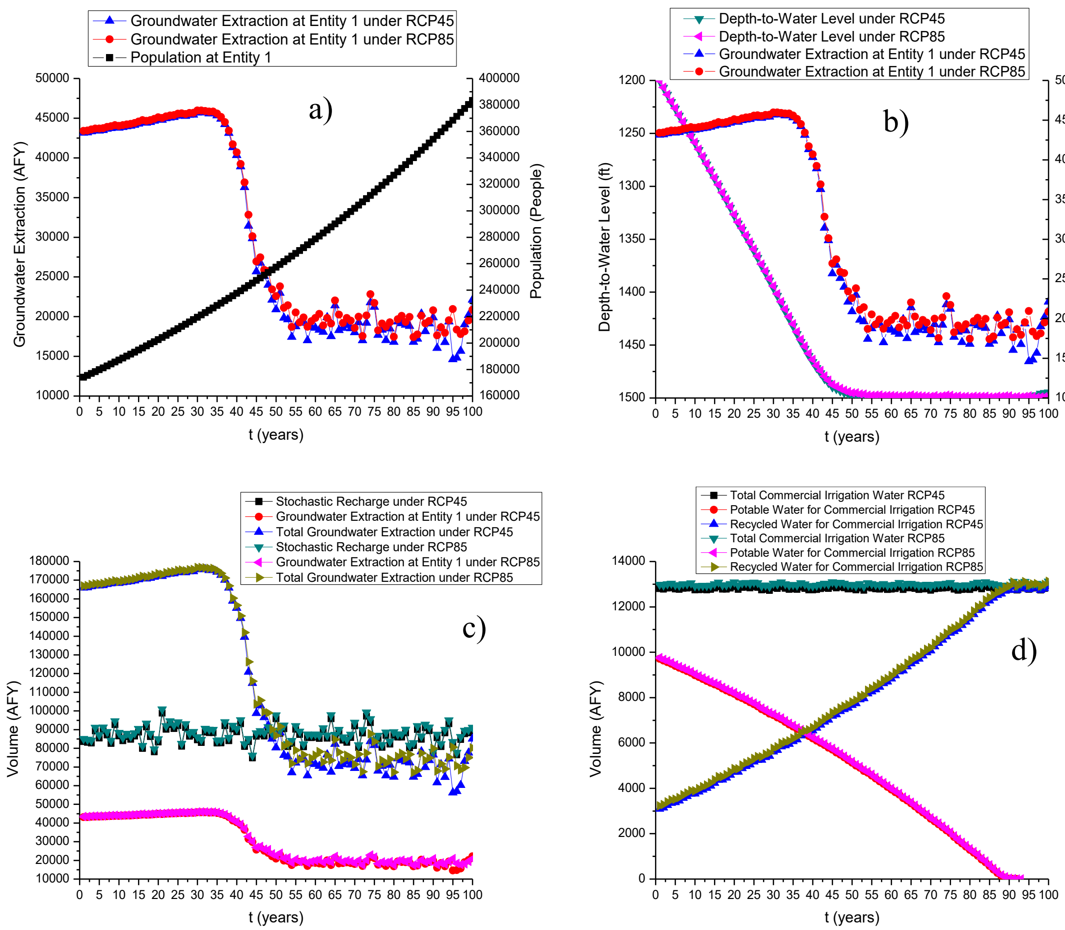


Figure 4.6. Comparison under Different Climate Conditions (Baseline/Stressed Aquifer)
a) Groundwater Extraction vs. Population
b) Groundwater Extraction vs. Depth-to-Water Level
c) Groundwater Extraction vs. Stochastic Recharge
d) Agricultural Irrigation

The cost-effective water supply distribution clearly is different due to the stressed condition of the aquifer and the different climate conditions. For this scenario, managed aquifer recharge using imported water and recycled water is implemented under RCP8.5. The total imported water allocation for entity 1 is 102,600 AFY. The available allocations under different climate condition RCP4.5 and RCP8.5 are 60 and 10 percent, respectively. As mentioned earlier, at year 35, groundwater extraction starts dropping due to groundwater availability. Imported water and desalination water are then purchased as additional water supplies to meet residential and commercial water demand. However, because of different imported water allocations under RCP4.5 and 8.5, there are different water sources that contribute to the water supplies (Figure 4.7).

Under RCP4.5, the available imported water allocation at entity 1 (San Bernardino) is 61,560 AFY. Entity 1 never uses up its available imported water allocation under this climate condition and, therefore, the combination of groundwater and imported water are sufficient to meet the entity's water demand. Natural recharge starts recharging the basin and groundwater extraction depends solely on the rate of natural recharge.

On the other hand, the imported water allocation at entity 1 is only 10,260 AFY under RCP8.5. The combination of groundwater and imported water are not enough to meet demand. Therefore, groundwater is being further utilized (more extractions) to reduce water cost and additional desalination water is purchased with imported water starting in year 35 to meet the water demand. The cost of desalination water in this model is approximately $2,100/AF which is the most expensive potable water source. Thus, the total water cost is expected to increase in proportion with the amount of desalination water used. Total agricultural irrigation water is met by purchasing potable and recycled water. After year 90, recycled water produced is used to provide 100 percent for irrigation water and additional managed aquifer recharge.

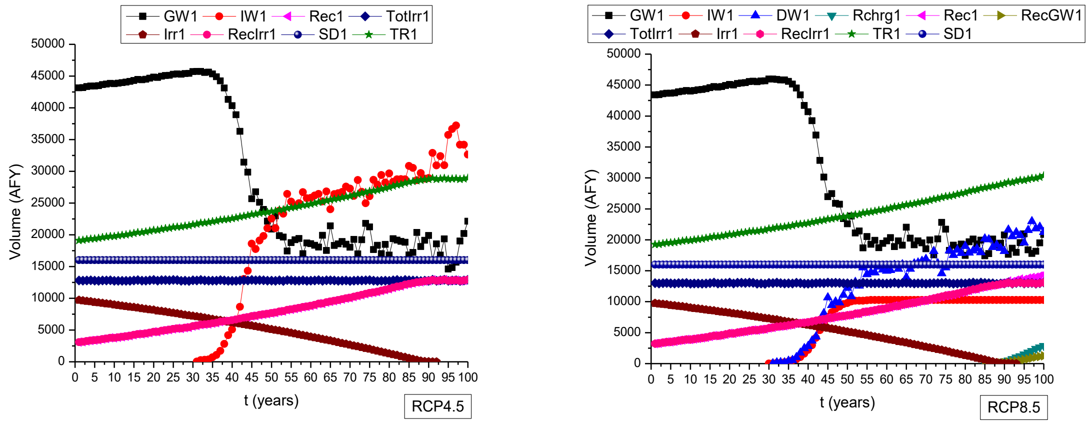


Figure 4.7. Water Sources under Different Climate Conditions (Baseline/Stressed Aquifer)

Similarly, the TDS in the groundwater basin is maintained at 300 mg/L for both RCP4.5 and RCP8.5. After year 90, there is managed aquifer recharge using recycled water with TDS of about 500 mg/L under RCP8.5; thus slightly increase the TDS in the aquifer (Figure 4.8).

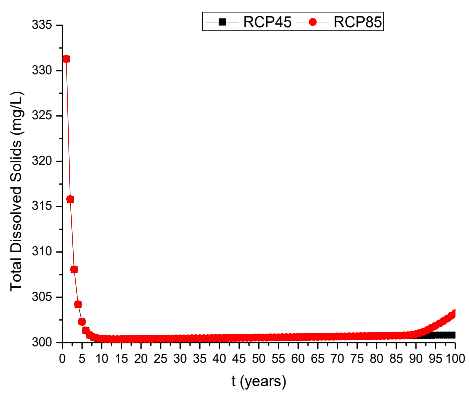


Figure 4.8. Total Dissolved Solids in Groundwater Basin
(Baseline/Stressed Aquifer)

The cost of groundwater extraction decreases as the groundwater volume decreases. As explained earlier, the cost of groundwater extraction for RCP8.5 is slightly higher than that of RCP4.5 due to higher extraction resulting from higher natural recharge. The cost of groundwater extraction is nearly $12.5 million annually at year 1, yet falls to $8 million per year at year 100 (Figure 4.9). The total water cost after the 100-year period under RCP8.5 condition is higher than RCP4.5 due to the additional amount of desalination water purchased (Figure 4.9). The effect of climate condition amplifies the stressed condition of the basin, thus resulting in water shortages and higher water supply costs for entity 1.

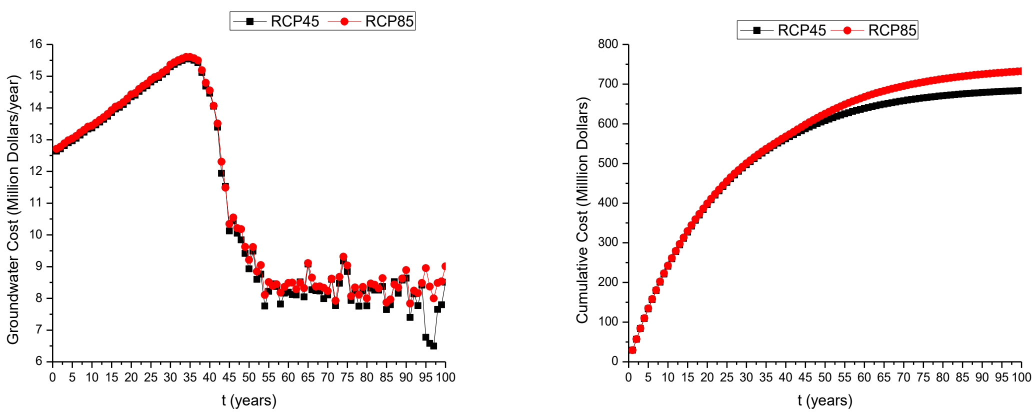
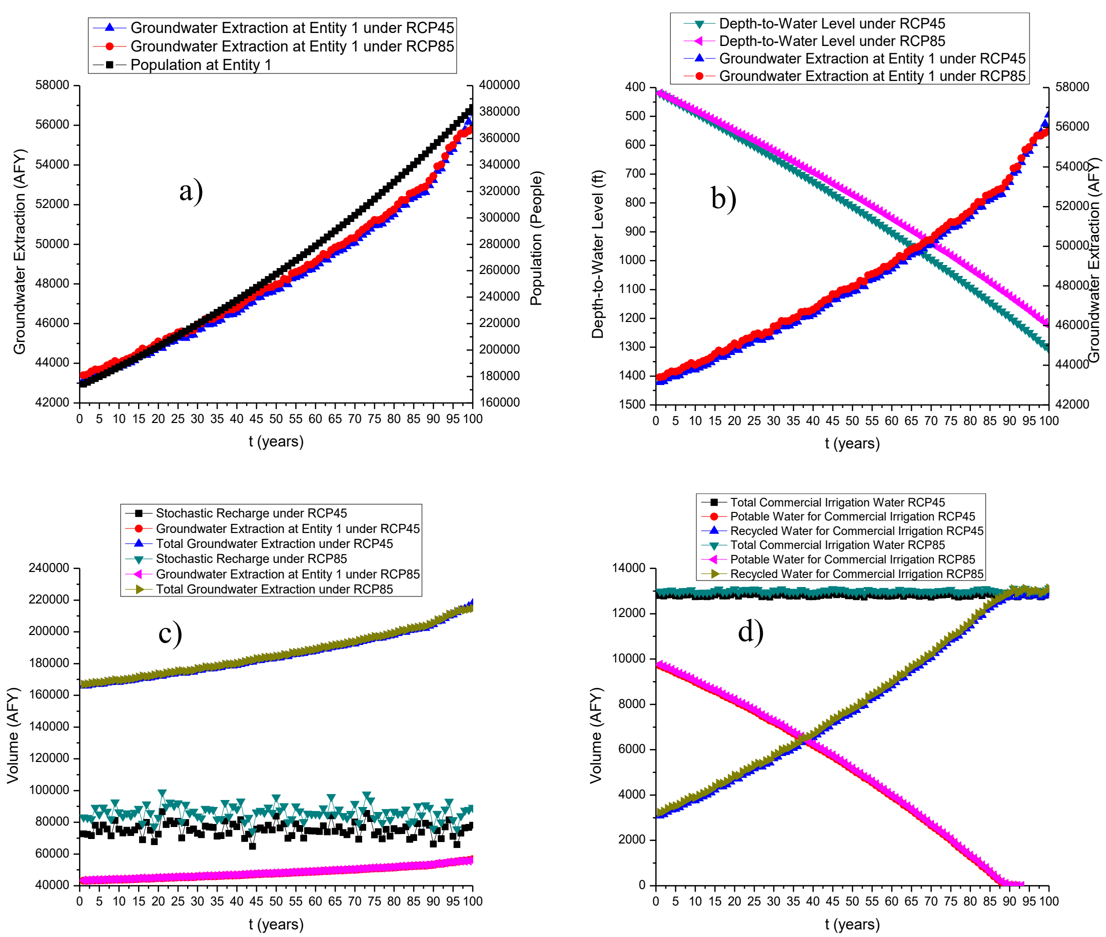


Figure 4.9. Water Cost under Different Climate Conditions (Baseline/Stressed Aquifer)

Scenario 3 – Projected, full aquifer (h=780 ft)

This scenario examines the projected condition of the aquifer in 2000-2099. The starting depth-to-water level is at 780 ft below land surface. The natural recharge and ET show significant differences compared to the baseline conditions in scenarios 1 and 2. The differences in projected average natural recharge rate and ET under RCP4.5 are 16% and 6.7 % and under RCP8.5 are 3.7% and 8.2% compared to the baseline conditions, respectively. The groundwater extraction rates are nearly identical under both climate scenarios as in scenario 1. The groundwater availability is abundant with the full aquifer;

thus, given the low cost to pumping a full aquifer, all demand is met by groundwater extraction Figure 4.10a). However, due to higher natural recharge under RCP8.5 (14%) compared to RCP4.5 as seen in Figure 10c, distance to the water table under RCP8.5 is less than RCP4.5 even though the groundwater extraction volumes are almost the same (Figure 4.10b). The difference in depth-to-water level under RCP8.5 is about 60 ft. The distribution of water sources to meet total agricultural irrigation water are, again, similar to the above cases (Figure 4.10d-f).



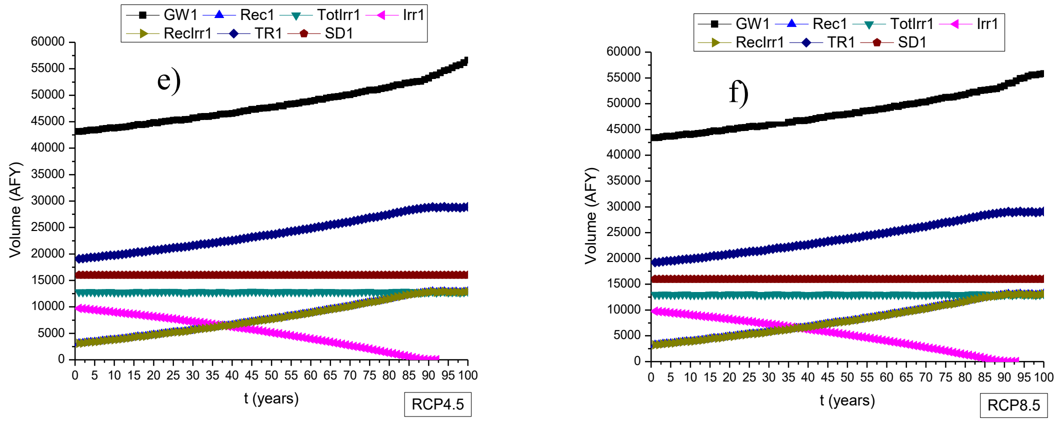


Figure 4.10. Comparison under Different Climate Conditions (Projected/Full Aquifer)
a) Groundwater Extraction vs. Population
b) Groundwater Extraction vs. Depth-to-Water Level
c) Groundwater Extraction vs. Stochastic Recharge
d) Agricultural Irrigation
e) Water Sources under RCP4.5
f) Water Sources under RCP8.5

The groundwater basin is only recharged by natural precipitation under both climate conditions. Therefore, TDS in the basin is maintained at 300 mg/L as scenario 1 (Figure 4.11).

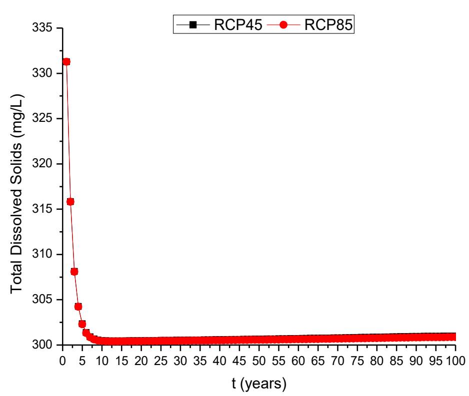


Figure 4.11. Total Dissolved Solids in Groundwater Basin
(Projected/Full Aquifer)

Groundwater extraction cost under RCP8.5 is lower than that of RCP4.5 due to the smaller depth-to-water level caused by higher natural recharge under this climate condition. At year 100, the groundwater extraction costs under RCP8.5 and RCP4.5 are $17 and $18 million per year, respectively (Figure 4.12). This difference in cost is negligible when the NPV is used to calculate the cumulative cost that the entity has to pay for 100 years, which is around $550 million (Figure 4.12).

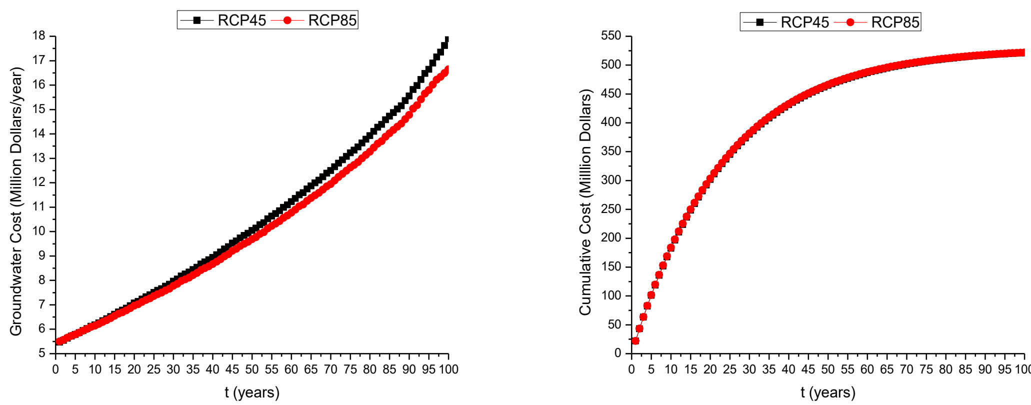


Figure 4.12. Water Cost under Different Climate Conditions (Projected/Stressed Aquifer)

Scenario 4 – Projected, stressed aquifer (h=0 ft)

Under this projected condition, the groundwater level begins at the msl which we considered a stressed aquifer. The difference in stochastic recharge is more distinguished under the different climate conditions. Surprisingly, extreme climate condition (RCP8.5) projects wetter years as seen in Figure 4.13. The stochastic recharge under RCP8.5 varies from 74,000 to 100,000 AFY whereas it only varies from 64,000 to 86,000 AFY under RCP4.5. The difference is about 10,000 – 15,000 AFY. This is also supported in a recent study on climate change by Allen and Luptowitz (2017)[99]. Similarly, the effect of climate condition also plays an important part in predicting the evapotranspiration rate. The relationship between the atmospheric temperature and ET is clearly observed here. As the climate becomes dryer due to higher temperature, ET is also increasing (Figure 4.13)[72, 73]. The effect of climate change on ET under the projected conditions is more severe than under the baseline conditions because of the differences in atmospheric temperatures in these periods. The average global temperatures are projected to increase by between 3°F to 12°F by 2100[74].

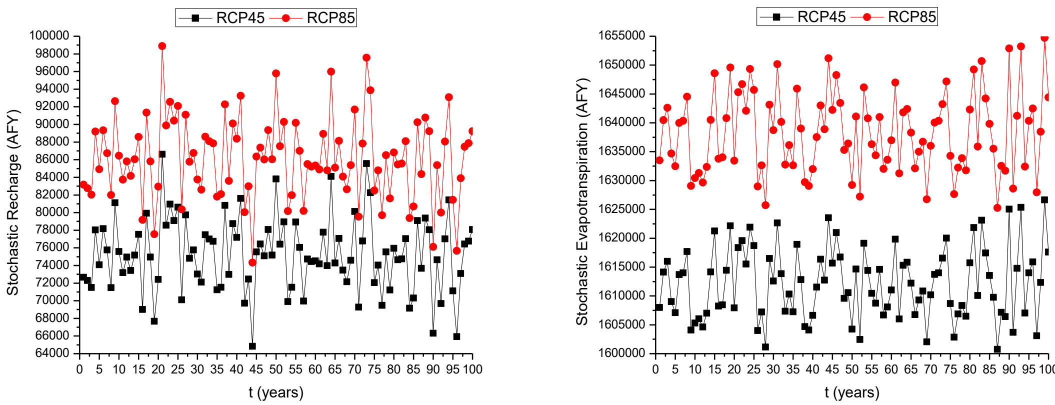
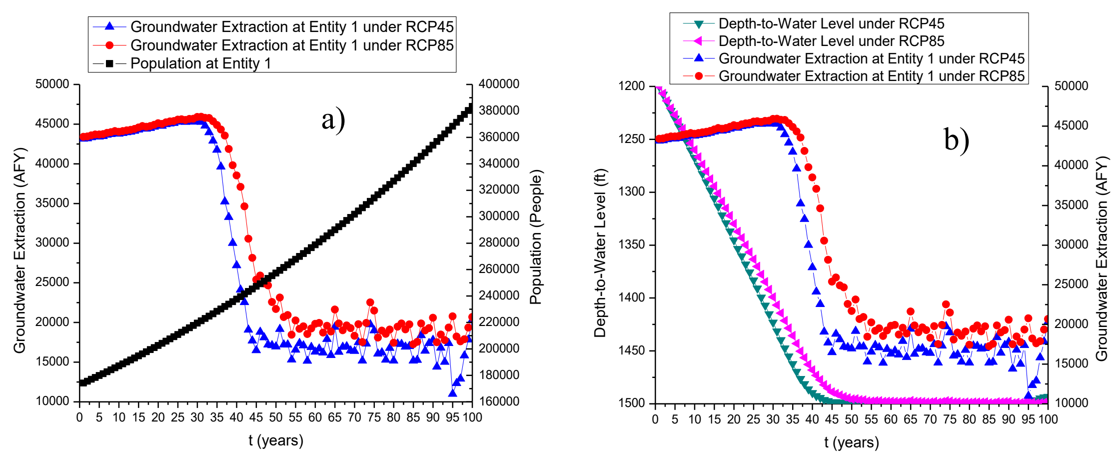


Figure 4.13. Projected Stochastic Recharge and Evapotranspiration (2000-2099)

More natural recharge under RCP8.5 also affects the groundwater basin and the extraction rates. More groundwater extractions were observed under RCP8.5 after year 35 (Figure 4.14a and 4.14b). However, due to significant difference in natural recharge, the basin reaches the buffer constraint faster under RCP4.5 compared to RCP8.5 (year 45 vs. year 50). The difference in groundwater extraction under different climate conditions is about 5,000 – 10,000 AFY. The higher natural recharge gives RCP8.5 more available groundwater for extraction. However, due to limited imported water (10,260 AFY) and the stressed condition of the aquifer, the entity has to purchase additional desalination water in order to meet its residential and commercial water needs at a higher cost under RCP8.5. The imported water reaches its maximum allocation after year 45. The model decides to start incorporating desalination water after year 30 (Figure 4.14f). Here, the model decides to extract as much groundwater as possible to reduce the overall water costs resulting from additional imported and desalination water. The depth-to-water level under RCP8.5 starts to hit the buffer layer after year 50 instead of year 45 under RCP4.5 (Figure 4.14b) due to differences in natural recharge. From there, water supplies rely on imported water and desalination water while the groundwater extraction relies solely on the natural recharge rates.

On the other hand, even though the natural recharge under RCP4.5 is slightly worse than RCP8.5 (Figure 4.14d), the available imported water is significantly higher than that of RCP8.5 (61,560 AFY vs. 10,260 AFY). Therefore, groundwater and imported water are sufficient to supply the entity's water demand (Figure 4.14e).



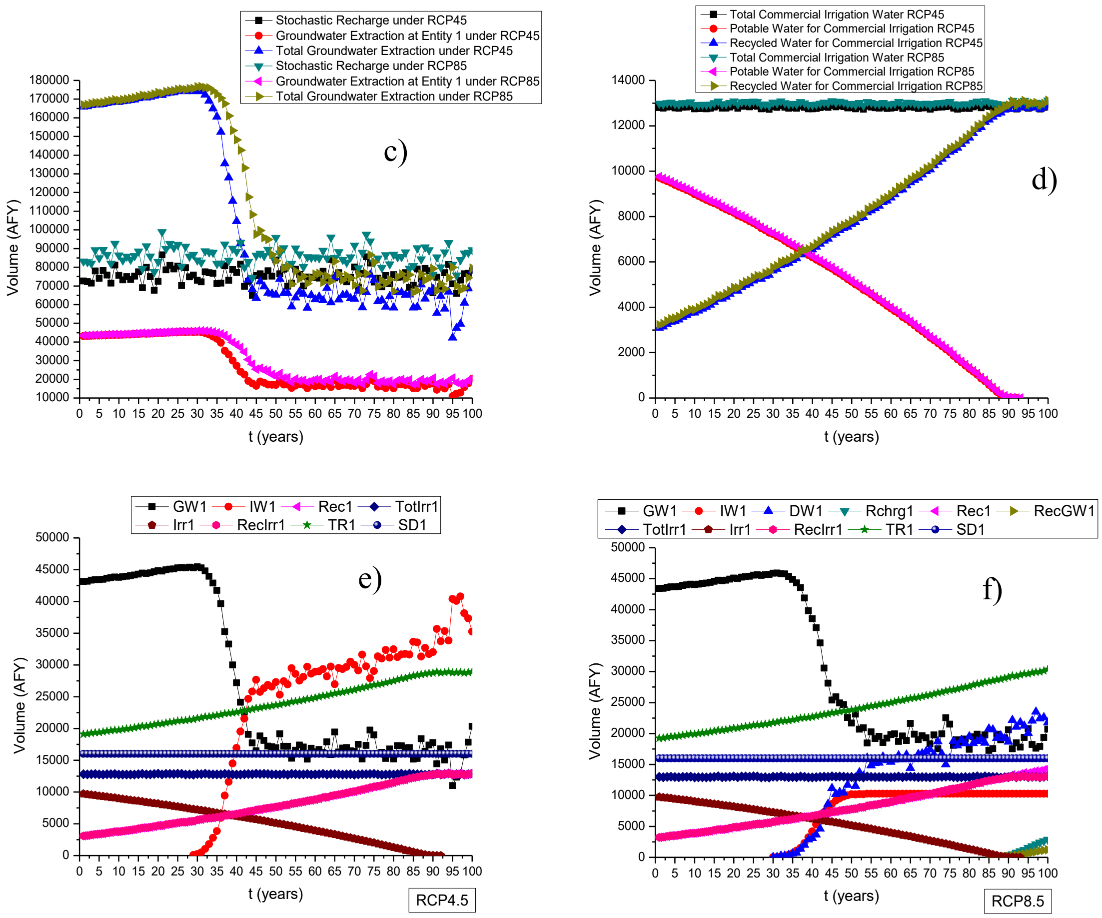


Figure 4.14. Comparison under Different Climate Conditions (Projected/Stressed Aquifer)

a) Groundwater Extraction vs. Population

b) Groundwater Extraction vs. Depth-to-Water Level

c) Groundwater Extraction vs. Stochastic Recharge

d) Agricultural Irrigation

e) Water Sources under RCP4.5

f) Water Sources under RCP8.5

The TDS in this scenario is similar to scenario 2 where the basin is stressed during the baseline period. The groundwater extraction cost under RCP8.5 is higher than that of RCP4.5 after year 30 (Figure 4.15). This is because of lower groundwater extraction volumes after year 30 under RCP4.5 as explained above. The total water cost under RCP8.5 is higher than RCP4.5 after 100 years (736 vs. 690 million dollars) due to the amount of groundwater extractions and desalination water supplied to meet the entity's demands (Figure 4.15). The difference in total cost becomes noticeable after year 45 when desalination water is adopted as part of the supply.

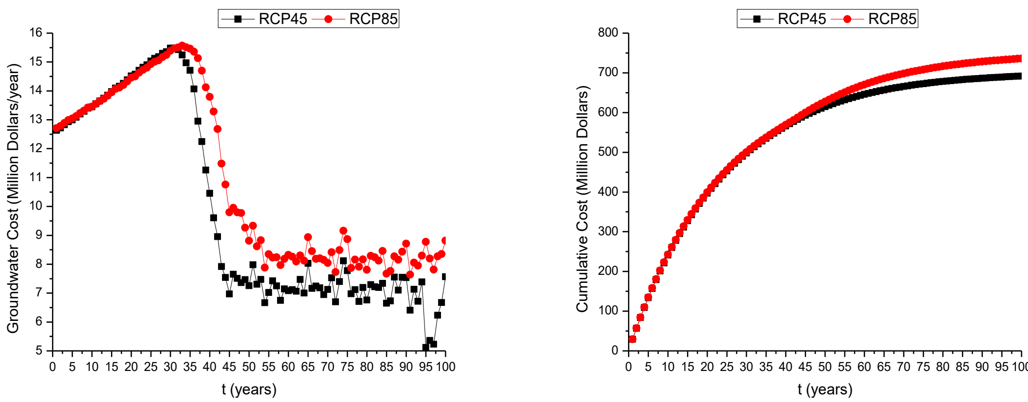


Figure 4.15. Water Cost under Different Climate Conditions (Projected/Stressed Aquifer)

Scenario 5 – Projected, Stressed Aquifer, No Agricultural Irrigation

The above scenarios discuss how to meet the entity's standard residential and commercial water demands. In this scenario, the model will predict how to meet the entity's water needs without agricultural irrigation. As seen in Figure 4.16a, the entity's water need is reduced significantly (10,000 AFY less) if the need for agricultural irrigation is omitted. The groundwater basin will reach the buffer layer after year 60 and 75 under RCP4.5 and RCP8.5, respectively, approximately 15 and 25 years after they are reached under the scenarios that include irrigated agriculture (Figure 4.16b). The groundwater extraction under RCP4.5 is, again, lower than RCP8.5 after year 45. This is because the groundwater basin under RCP4.5 reaches the buffer layer faster than RCP8.5 due to lower natural recharge; thus, imported water under RCP4.5 becomes a significant water supply source earlier than under RCP8.5. After reaching the buffer layers (year 60 and 75 under

RCP4.5 and RCP8.5 respectively), the amount of groundwater extracted thereafter relies solely on the natural recharge (Figure 4.16c).

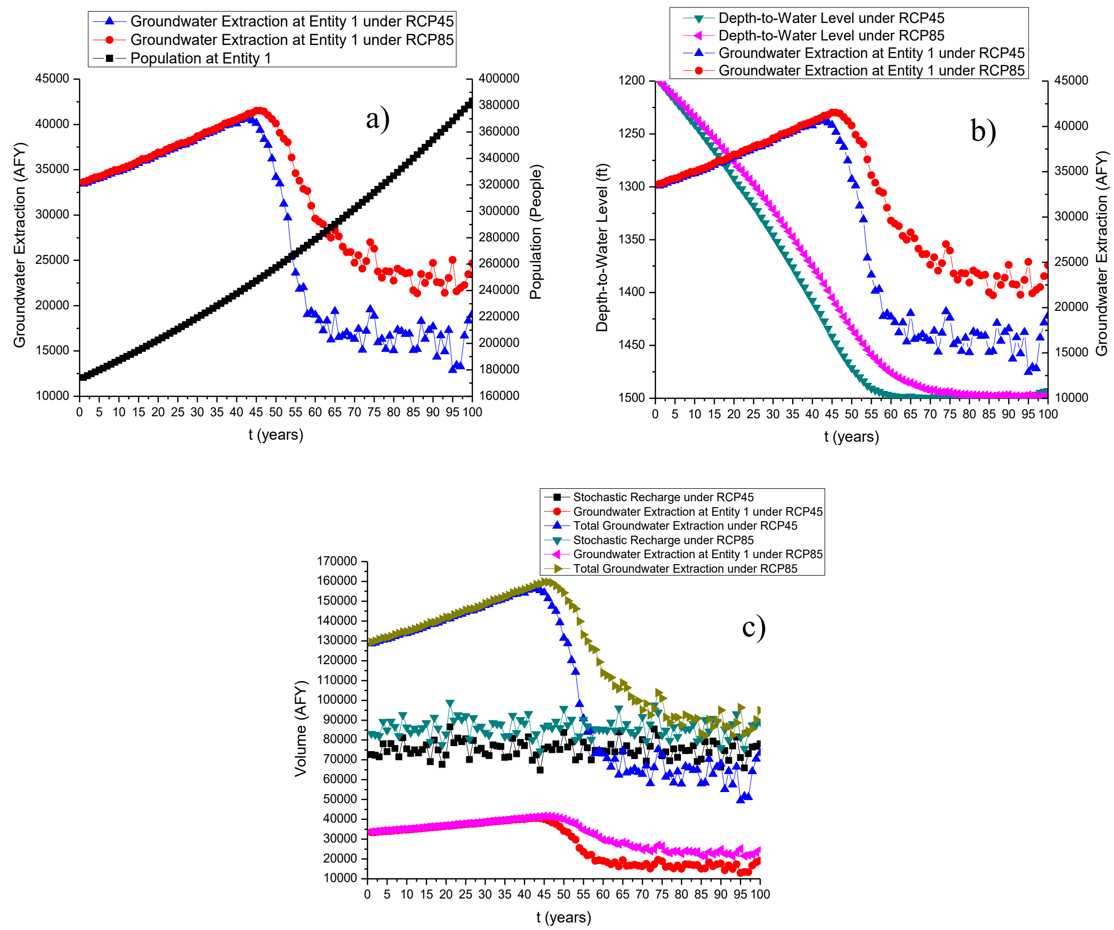


Figure 4.16. Comparison under Different Climate Conditions (Projected/Stressed Aquifer/No Agricultural Irrigation)
a) Groundwater Extraction vs. Population
b) Groundwater Extraction vs. Depth-to-Water Level
c) Groundwater Extraction vs. Stochastic Recharge

Under this scenario, even though the natural recharge is lower for RCP4.5, the available imported water is, again, greater than RCP8.5 (Figure 4.16c). Imported water starts to supplement the entity's water need after year 40 and becomes a major water supply source after year 60 when the groundwater basin reaches the buffer layer (Figure 4.17 and

Figure 4.16). All the recycled water is discharge to the surface water since this is the most cost-effective solution. On the other hand, due to limited imported water under RCP8.5, the entity has to, again, purchase additional desalination water to supply its water needs. The imported water is adopted in year 40 and reaches its full allocation after year 60. Also, around year 55, recycled water starts to recharge the basin to provide more groundwater availability and to reduce the total water cost. By year 75, the groundwater basin reaches its buffer layer and desalination water becomes the major water supply source under RCP8.5 (Figure 4.17).

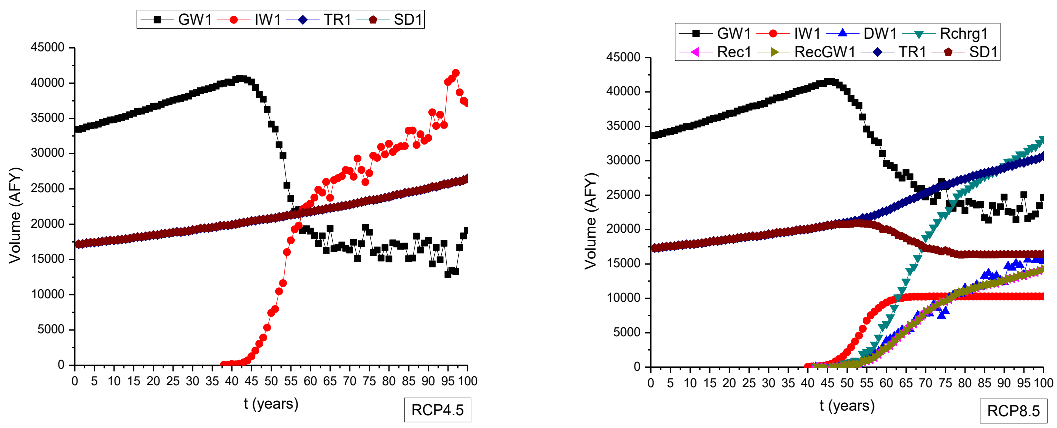


Figure 4.17. Water Sources under Different Climate Condition (Projected/Stressed Aquifer/No Agricultural Irrigation)

On the quality side of the ledger, after year 55, TDS in the groundwater basin starts to increase due to the amount of recycled water used for managed aquifer recharge (Figure 4.18). The TDS in the recycled water is approximately 500 mg/L whereas the TDS in the groundwater basin at year 1 is around 331.27 mg/L. The basin is recharged from 3 different sources: natural precipitation, groundwater infiltration from irrigation, and recycled water, with TDS concentrations varying by source from 300 mg/L (natural recharge) to 500 mg/L (recycled water). Therefore, the only factor that causes TDS concentration in the groundwater basin to increase significantly is recycled water. However, this increase is still lower that the amount TDS regulated by U.S. EPA (500 mg/L) for secondary standard in drinking water.[66] Thus, this is still acceptable as a water supply source.

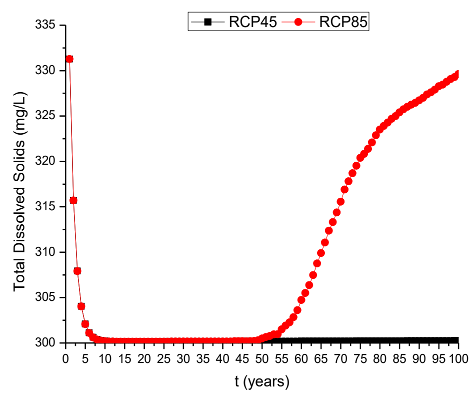


Figure 4.18. Total Dissolved Solids in Groundwater Basin (Projected/Stressed Aquifer/No Agricultural Irrigation)

The groundwater cost for RCP8.5 is higher than RCP4.5 due to higher groundwater volume extraction as mentioned above. After year 45 for RCP4.5 and year 50 for RCP8.5, the groundwater costs decrease because of reduced groundwater extractions after reaching its buffer layers. At this point, imported water and desalination water start to comprise a larger faction of overall water use and costs. The need for potable water in this case is lower than the above cases due to neglecting the total agricultural irrigation, and recycled water also contributes in recharging the groundwater basin under RCP8.5. The total desalination water purchased is also lower than the previous case (15,000 AFY vs. 23,000 AFY). Therefore, the costs of water supply from different climate conditions are not significantly different. After 100 years, the total water cost under RCP4.5 and RCP8.5 are

$523 vs $534 million, respectively (Figure 4.19). Relative to the previous scenarios, the costs to supply water are less here given the lower demand to due to the absence of irrigated acreage.

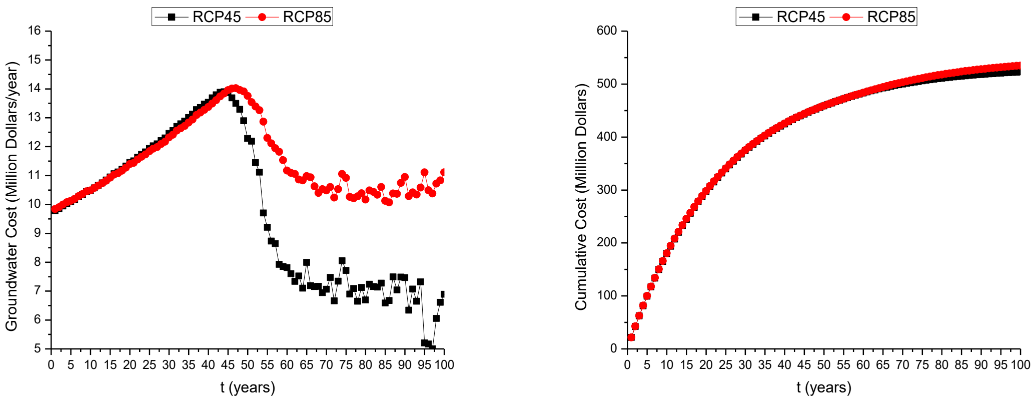


Figure 4.19. Water Cost under Different Climate Condition (Projected/Stressed Aquifer/No Agricultural Irrigation)

Scenario 6 – Projected, stressed aquifer, high salinity imported water from Colorado River (TDS = 658.33 mg/L)

As mentioned earlier, imported water is a combination of SWP and Colorado River water. The TDS of water from Colorado River was approximately 658.33 mg/L in 2015[75]. This scenario demonstrates the extreme situation where SWP is no longer available and the only source of imported water is high salinity water from Colorado River whose cost is approximately $332.39/AF in 2015 U.S. Dollar[100]. Similar to scenario 4 and 5, the groundwater extraction is higher under the RCP8.5 due to limited amount of imported water and extremely expensive desalination water (Figure 4.20a). The model predicts that the groundwater basin under RCP8.5 will reach the buffer layer at year 80 regardless of the higher natural recharge under RCP8.5 compared to RCP4.5 (Figure 4.20b). Under RCP8.5, the entity reaches is imported water allocation after year 15, which is the reason the entity has to utilize groundwater as much as possible thereafter. Therefore, the groundwater basin reaches the buffer layer faster and the entity has to rely on expensive desalination water as a major water source after year 80. In order to mitigate the impact of water shortage and minimize the cost of water supplies, a portion of tertiary-treated recycled water is used for managed aquifer recharge after year 90, which causes an increase in the groundwater salinity (Figure 4.20f). On the other hand, the depth-to-water level under RCP4.5 does not reach the lower bound constraint on the aquifer because of the availability of imported water allocation. The maximum imported water allocation under RCP4.5 and RCP8.5 are 61,560 AFY and 10,260 AFY, respectively. Water from Colorado River comes with lower cost compared to SWP. The model utilizes this source of water in addition with groundwater and reduces the groundwater extraction rates under RCP4.5 (Figure 4.20e). Therefore, groundwater basin under this scenario only depletes by approximately 125 ft after 100 years (Figure 4.20b).

The total agricultural irrigation is also a combination of potable water and recycled water. After year 94, the total agricultural irrigation water is 100% recycled water (Figure 4.20d). Additional recycled water is used to recharge the groundwater basin after year 90 under RCP8.5, which causes an increase in TDS in the groundwater basin from year 90 to 100 (Figure 4.21).

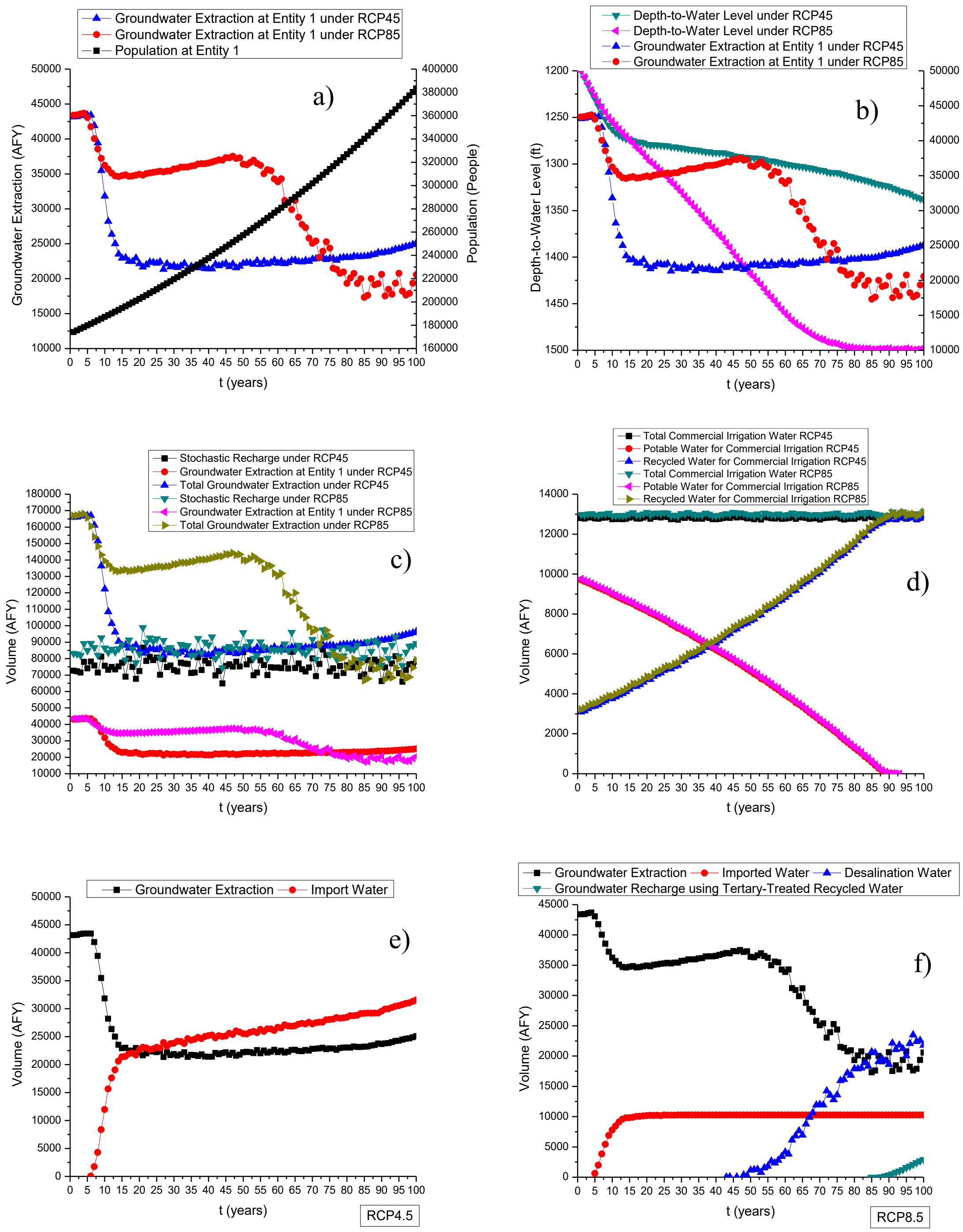


Figure 4.20. Comparison under Different Climate Conditions (Projected/Stressed Aquifer/Colorado River Water)

a) Groundwater Extraction vs. Population

b) Groundwater Extraction vs. Depth-to-Water Level

c) Groundwater Extraction vs. Stochastic Recharge

d) Agricultural Irrigation

e) Water Sources under RCP4.5
f) Water Sources under RCP8.5

The entity starts purchasing imported water in year 5 and reaches its maximum allocation in year 15 under RCP8.5. The TDS in the water supply starts to increase after year 10 as a result. After this period, desalination water plays an important part in supplementing water to the entity, which brings the TDS in the water supply down to 320 mg/L at year 100.

The water supply under RCP4.5 only relies on groundwater and imported water. Imported water is used in addition with groundwater source after year 5 (Figure 4.21). After year 15, high salinity imported water becomes a major water supply source which brings the TDS in water supplies up to 500 mg/L after year 84. According the secondary drinking standards established by U.S. EPA, potable water supplies with TDS of less than 500 mg/L is considered safe for drinking purpose. Therefore, the water supplies under this scenario still meet regularly requirements regarding TDS limits for drinking water.

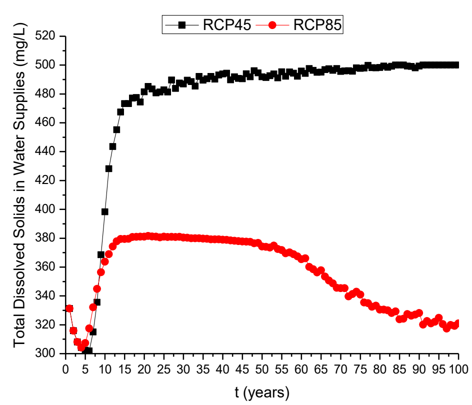


Figure 4.21. Total Dissolved Solids (Projected/Stressed Aquifer/Colorado River Water)

The cost of groundwater extraction under RCP4.5 is lower than RCP8.5 due to the volume extracted. The total water costs are similar to the previous cases, with RCP4.5's cost is lower than RCP8.5's (665 vs. 690 million dollars) at year 100 as in Figure 4.22.

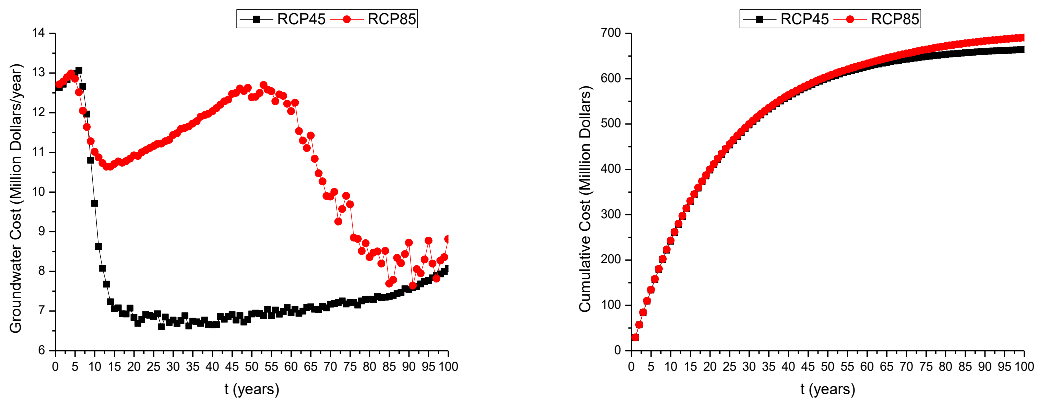


Figure 4.22. Water Cost under Different Climate Conditions (Projected/Stressed Aquifer/Colorado River Water)

Summary of Comparisons

Overall, the effect of climate change amplifies the model's predictions in all cases. In Figure 4.23, the natural recharge is greatest under RCP8.5 during 1950-1999 period and least under RCP4.5 during 2000-2099 period. This also explains why the depth-to-water level under RCP4.5 is always higher than RCP8.5, and the 1950-1999 period is always better than 2000-2099 period in term of natural recharge and groundwater table levels (Figure 4.23). Regarding the depth-to-water level after 100-year period, the best scenario is when imported water is only comprised of high salinity water from Colorado River since the groundwater level does not reach the buffer layer constraint. The unit price of Colorado River is more comparable with the unit cost of groundwater; thus, making it compete with groundwater supply to meet the entity's demands. By doing so, the groundwater basin reaches the buffer layer in later years compared to other scenarios. The only drawback of this scenario is the high salinity water from Colorado River. However, it is proven than the TDS in the drinking water supply still complies with the secondary standards for drinking water. On the other hand, regarding the total water cost, the most cost-effective scenario is when there is no agricultural irrigation since the total water supplies is minimized and when the basin is full (Figure 4.24a). The worst scenario is the case where the basin is stressed under RCP8.5 during the projected period (Figure 4.24a). Even though there is more recharge under RCP8.5, the total water cost is always more expensive at the end of the term due to the cost of imported and desalination water.

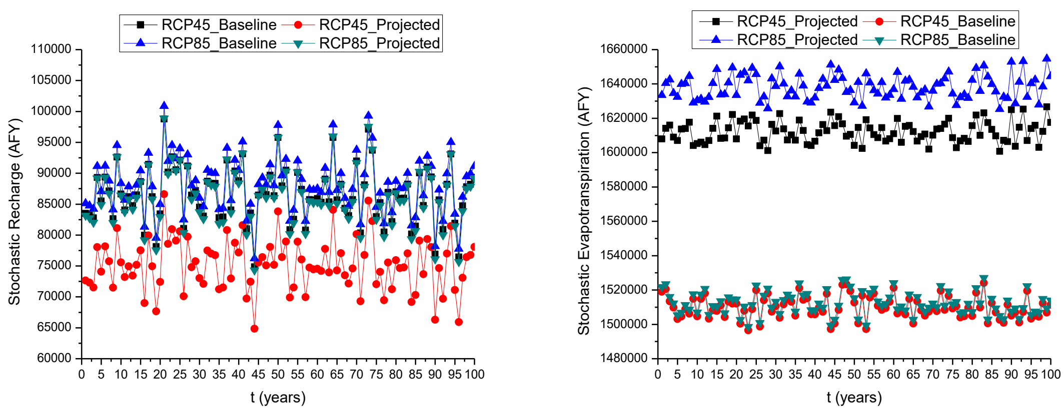


Figure 4.23. Stochastic Recharge and Evapotranspiration Comparisons

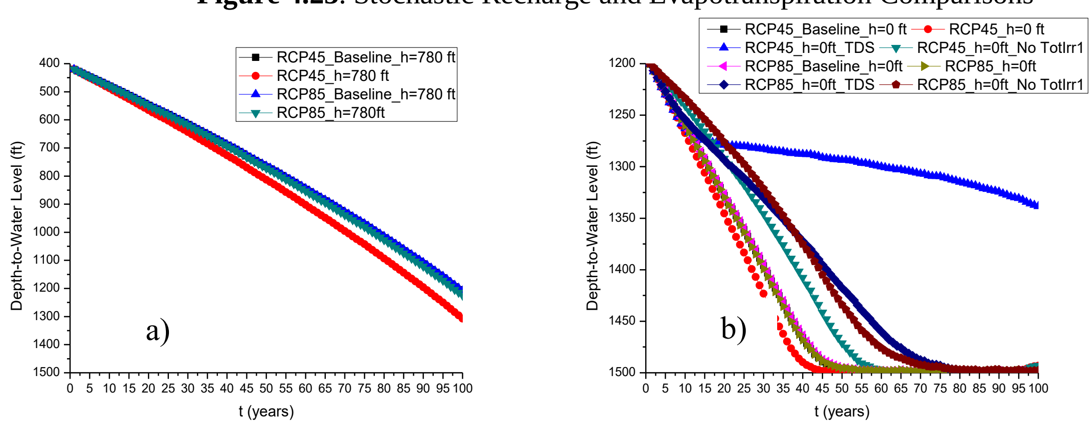


Figure 4.24. Depth-to-Water Level Comparisons
a) Full Aquifer
b) Stressed Aquifer

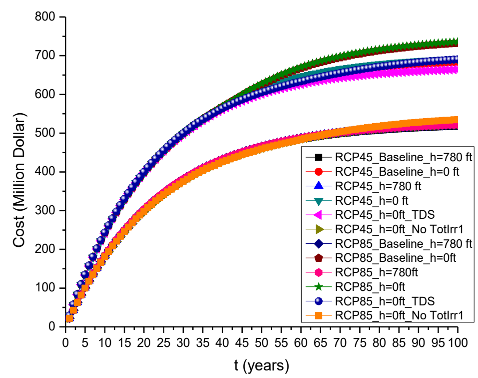


Figure 4.25. Total Water Cost in 100-Year Period Comparisons

CONCLUSIONS

The supply-demand water balance model studied a specific groundwater aquifer in Southern California to investigate projected groundwater availability and water supplies under different climate conditions. The studied entity uses different water sources such as groundwater, imported water, recycled water, and desalination water to meet its water demands at the most cost-effective solutions. The project also demonstrated the effects of different climate conditions (RCP4.5 vs. RCP8.5) on water availability and natural recharge during baseline period (1950-1999) and projected period (2000-2099) by simulating different scenarios. In addition, the model also simulated the extreme case where water from Colorado River (with TDS approximately 700 mg/L) is the only source that made up the imported water. The key element of this project is to offer water agencies tools to make informed and cost-effective water management decisions that acknowledge, and have implications for, future water supply and demand conditions while taking into account both climate-change scenarios that affect local recharge rates and evapotranspiration rates, as well as changes in population, treatment costs, and regional supplies.

Since groundwater is the least expensive water source relative to the other sources we evaluate, agencies are increasing extracting water for their residential and commercial water needs as predicted by the model. That explains the effort to drawdown aquifers until the water table reaches its buffer layer. Every scenario here involves groundwater usage to minimize the overall cost of water supplies. The volumes of groundwater extracted after reaching the buffer layer solely depends on the natural recharge in the previous period. When the basin is full, there is enough groundwater to meet water demands; thus, the total water cost is relative low since there is no need to purchase additional water from other sources. However, when the groundwater basin is stressed, municipality agencies need to explore other water sources such as imported water from SWP and desalination water to meet their water demands, both of which are more costly sources. Therefore, the total cost of water supply also increases accordingly.

The effect of climate conditions on water shortages was amplified relative to the baseline conditions, especially when the groundwater basin was stressed. The natural recharge decreases as the pattern of precipitation is shifted from snowmelt-dominated to rainfall-dominated events. Moreover, the evapotranspiration rates increase as a result of increase in atmospheric temperatures. The SWP allocations during drought period are significantly reduced. During the 2011-2017 period, the SWP allocations varied between 5% to 85%, which drove Southern California water agencies to consider alternativewater sources that are more locally reliable. The resulted mitigation possibility for this scenario was to incorporate desalination water at earlier years to meet residential and commercial water demands. Currently, desalination water is the most expensive water source among other sources, which results in extremely expensive costs of water supply. Consequently, as the future of groundwater is not certain due to ineffective groundwater management especially under projected climate conditions, significantly expensive desalination water may become the answer to water shortages under future uncertainty.

In addition, treated wastewater is commonly discharged to surface water as the most cost-effective water management strategy. As the water becomes more scarce, recycled water becomes a big portion of commercial water usage. However, treated municipal wastewater is often used as the last fallback for managed aquifer recharge. The model takes into account both the cost to treat wastewater for aquifer recharge and the cost to bring it up for usage. This cost is higher than the cost of imported water alone. Moreover, recycled water used for managed aquifer recharge from conventional wastewater treatment processes can negatively impact the groundwater quality. As the TDS concentration in the treated wastewater for recharge is approximately 500 mg/L, the TDS in groundwater aquifer also increases as a result. Therefore, the model almost always incorporates imported water as a water supply source before considering managed aquifer recharge option.

As water becomes scarcer, especially in and around urban environments, the use of water for irrigated agriculture is likely to be reconsidered, especially since agricultural irrigation often uses the most water. As such, agriculture on the urban fringe and which competes with municipal demands might have a difficult time avoiding an ever-increasing water cost due to the increased scarcity. Strategies include reducing acreage, increasing irrigation efficiency, and/or changing crop type to more drought-tolerant crops are recommended to mitigate such impacts.

The model developed in this research offers water agencies a tool to make cost-effective water decisions under future uncertainty, especially with respect to climate change. Demand for groundwater use will continue to increase in lockstep with population growth, and further problems will likely be caused by climate change conditions. Agencies will and always utilize groundwater resource as the main water supply source until its cost rise significantly and/or regulations limit overdraft. In term of the cost of water supply and the groundwater table level, the least sustainable scenarios are when the basin is stressed whereas the most sustainable cases are when the basin is full and agricultural irrigation water is reduced. Therefore, water agencies need collaboration to implement sustainable groundwater management practices including restriction of groundwater extraction, increase groundwater recharge, etc. in the face of water scarcity and demands.

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Chapter 5

Summary and Conclusions

The overall goal of this research was to explore water supply alternatives, improve local water supply reliability and provide greater resilience to drought and climate change conditions. These goals were achieved by developing the RWRM model for cost-effective alternative agricultural irrigation source. The RWRM was modified to address the impacts of drought on water/wastewater quality and the mitigation efforts to reduce such impacts. Furthermore, the Water Balance was developed to investigate the trade-offs between water supplies and demands, while taking into account both climate-change scenarios that affect local recharge rates and evapotranspiration rates, as well as changes in population, treatment costs, and regional supplies. The model predicts water availability, groundwater extraction, technological needs, and supplemental water sources designed to meet the demands of a municipality over a 100-year period to give water agencies the tools needed to make informed management decisions under climate uncertainty.

Wastewater Reuse for Agriculture: A Development of a Regional Water Reuse Decision-Support Model (RWRM) for Cost-Effective Irrigation Sources

In Chapter 2, the RWRM was developed to produce irrigation water suitable for a wide range of crops without causing negative impacts on crops and soil. Parameters such as salinity, heavy metals, and pathogens were incorporated in the model and minimized to comply with existing regulations (U.S. EPA and California Title 22 for water reuse) and safe agriculture practices. The mechanism of the RWRM is the blending technique using effluents from different wastewater treatment processes within a particular treatment train so that the final irrigation water will meet crop requirements at minimum costs. In all cases, a certain fraction of desalination water was used to reduce salinity in the blended product. For crops that are more sensitive to any of the representative parameters, more stringent constraints/restrictions were put in place and the model responded with larger portions of RO water in the blended products; thus, resulting in significantly higher treatment costs. On the other hand, when the level of restriction in crop requirements is reduced, the model also reduced the RO portions of the blended products and resulted in the lower treatment costs. By utilizing wastewater for agricultural irrigation, large portion of freshwater resources will become available for more essential water demands under extreme water scarcity conditions.

In addition to providing alternative irrigation source to farmers, reusing wastewater for agricultural irrigation also offers cost savings on synthetic fertilizers because of nutrient availability in wastewater streams. Depending on different types of crops, the nutrient demands are also different. Sensitive crops often require larger portion of RO in the blended product, thus reducing the secondary and tertiary effluent portions. Therefore, the nutrients available in the final blended products are also reduced and farmers need to supply additional synthetic fertilizers to meet crop's nutrient demands. Consequently, farmers need to pay higher treatment cost for their irrigation water due to larger RO portion and additional cost for synthetic fertilizers. Vice versa, crops with more tolerant thresholds require less RO portions and more secondary/tertiary effluent portions. Thus, the available nutrients in the final blended product becomes more significant which, in turn, reducing the cost of synthetic fertilizers and the overall treatment cost.

While there were a number of simplifying assumptions in the model presented above, two in particular are worth discussing. First, we assumed a cost-minimization framework that identified the least-cost solution to produce treated wastewater given particular water quality constraints. This framework overlooks possible solutions in which the users of the water may be willing to accept poorer quality water in certain dimensions (along with the consequent yield reductions) for a lower price. To represent this alternative, crop-water production functions could be incorporated into the model along with their associated prices and costs so that the model could identify efficient solutions in which net benefits are maximized, where net benefits are defined by the profits to the water user less the costs to the wastewater treatment plant operator. Crop-water-salinity production functions developed and utilized in Letey et al. (1985), Kan et al. (2002), and Schwabe et al. (2006) are being further developed for such an extension[58-60]. Second, our model focuses on the water quality requirements associated with two separate crops in isolation. In reality, demand for the treated municipal wastewater nor the current distribution will likely allow wastewater treatment managers to produce water for a single type of crop or product. There are a number of ways the model could be adjusted to reflect this, including incorporating the net benefits framework mentioned above. Alternatively, the current cost-minimization framework could be modified to generate solutions subject to constraints imposed on the most sensitive parameters across the array of crops. As this model is intended to be flexible and allow for the evaluation of a wide variety of treatment processes and output scenarios, such explorations are easily incorporated into the current framework.

Using the RWRM, wastewater treatment trains can be optimized to produce irrigation water suitable for a wide range of crops with varying salinity tolerance, reducing the impact on soil and crop quality that is currently experienced by irrigators using conventionally treated wastewater. Salinity, heavy metals, and pathogens were minimized to comply with existing regulations and safe agriculture practices. By utilizing this blending technique as an alternative irrigation source for agriculture, freshwater resources would be reserved to cope with drought-induced extreme water scarcity.

The Implications of Drought and Water Conservation on the Reuse of Municipal Wastewater: Recognizing Impacts and Identifying Mitigation Possibilities

Chapter 3 investigated the impacts of drought on water/wastewater quality and freshwater availability with respect to salinity. This work also demonstrated the flexibility and replicability of the RWRM developed in previous work. In times of drought, which are expected to increase in both frequency and severity, the availability of freshwater resources decreases and its quality, particularly with respect to salinity, often declines. In response, water agencies are increasingly implementing conservation measures to reduce demand while investing in the reuse of treated municipal wastewater to augment supply, both with the intention to increase resilience. Our results show that both the water supply effects of drought *and* the conservation measures enacted in response to it combine to reduce the quantity and quality of influent available for treatment which, in turn, reduces the quantity and quality of treated municipal wastewater for reuse under conventional treatment processes. Our modeling results also illustrate that cost-effective blending strategies can be implemented to mitigate the water quality effects, increasing the value of the remaining effluent for reuse, whether it be for surface water augmentation, groundwater replenishment, or irrigation of crops, golf courses, or landscapes. We also highlight the benefits of treating the wastewater based on the characteristics of demand. In our case study, relaxing the constraints on both HCO_3^- and PO_4-P concentrations relative to the pre-drought levels lead to reduced treatment costs while still achieving an effluent quality superior to pre-drought levels for all the other water quality parameters.

From a management and policy perspective, three significant conclusions can be drawn from our analysis. First, municipalities, cities, and regions that rely on both conservation and reuse as a means to address drought and water scarcity need to recognize the potential dependence of the latter on the former in terms of its effect on the potential supply of treated municipal wastewater. To the extent possible, efforts to promote and advocate for outdoor water conservation rather than indoor conservation break this dependence. Second, drought and the conservation measures enacted in response can result in poorer quality water, particularly with respect to salinity. Given that conventional treatment processes are not designed to address these higher constituent loads, the value of the remaining effluent likely decreases relative to the downstream demands it serves. Consequently, recognizing this relationship should help the recipients of the treated municipal wastewater better plan for such outcomes and thus engage in cost-effective adaptation. Finally, while wastewater treatment plants themselves cannot mitigate the reduced flows that are the result of the drought and conservation measures, our modeling results illustrate that cost-effective treatment trains can be developed to mitigate the water quality effects of drought and conservation thereby increasing the value of the remaining effluent for reuse. We propose that by working together, recipients of treated municipal wastewater and the agencies themselves can identify cost-effective strategies in terms of the degree of treatment that provides the greatest benefit to society.

Our conclusions here are not simply fodder for the academic mill. In our particular case study, treated wastewater makes up a significant portion of flows in the Santa Ana watershed, as well as Southern California's waterways. It plays an even more significant role during drought conditions, when precipitation and snowmelt decrease. Thus, it is important to consider how deteriorating wastewater effluent quality, and in particular, elevated salinity levels, impacts downstream users. Our modeling results demonstrate that incorporating a desalination step into the wastewater treatment process can alleviate some of these downstream concerns at a cost that is within 8% (currently \$0.69/m^3 vs. \$0.74/m^3 under scenario C) of current treatment costs. The resulting effluents are composed of partially desalinated wastewater that is suitable for crop irrigation and stream augmentation at a quality that protects wastewater treatment agencies from discharge violations and prevents surface water quality from further deterioration.

Identification of such low-cost wastewater treatment strategies should be useful to municipalities, both in California and globally, as they continue to strive to improve their resilience to drought via demand side management strategies that include reducing indoor water use and supply augmentation strategies that include wastewater reuse[33]. The RWRM model used in this research can be easily adapted to other applications to identify low-cost strategies given its flexibility and replicability. The model requires unit cost and effectiveness parameters on commonly used treatment technologies, parameters that are regularly reported in the academic literature and/or industry reports. The parameters that represent regulatory, surface water, or crop threshold constraints on effluent quality also can readily attained from public documents. For instance, this model was previously used to identify the most cost-effective treatment solutions when the treated wastewater effluent is used for irrigating citrus and turfgrass in Southern California[32]. Consequently, the RWRM is a flexible and easily adaptable model that can assist water managers in their efforts to develop water portfolios that cost-effectively and reliably respond to drought and increasing water scarcity worldwide.

The Role of Existing and Emerging Water Resources in Managing Groundwater Aquifer in the Face of Climate Uncertainty

A supply-demand optimization water balance model of a specific groundwater aquifer in Southern California to investigate projected groundwater availability and water supplies under different climate conditions was developed in Chapter 4. The studied entity uses different water sources such as groundwater, imported water, recycled water, and desalination water to meet its water demands at the most cost-effective solutions. The project also demonstrated the effects of different climate conditions (RCP4.5 vs. RCP8.5) on water availability and natural recharge during baseline period (1950-1999) and projected period (2000-2099) by simulating different scenarios. In addition, the model also simulated the extreme case where water from Colorado River (with TDS approximately 700 mg/L) is the only source that made up the imported water. The key element of this project is to offer water agencies tools to make informed and cost-effective water management decisions that acknowledge, and have implications for, future water supply and demand conditions while taking into account both climate-change scenarios that affect local recharge rates and evapotranspiration rates, as well as changes in population, treatment costs, and regional supplies.

Since groundwater is the least expensive water source relative to the other sources we evaluate, agencies are increasing extracting water for their residential and commercial water needs as predicted by the model. That explains the effort to drawdown aquifers until the water table reaches its buffer layer. Every scenario here involves groundwater usage to minimize the overall cost of water supplies. The volumes of groundwater extracted after reaching the buffer layer solely depends on the natural recharge in the previous period. When the basin is full, there is enough groundwater to meet water demands; thus, the total water cost is relative low since there is no need to purchase additional water from other sources. However, when the groundwater basin is stressed, municipality agencies need to explore other water sources such as imported water from SWP and desalination water to meet their water demands, both of which are more costly sources. Therefore, the total cost of water supply also increases accordingly.

The effect of climate conditions on water shortages was amplified relative to the baseline conditions, especially when the groundwater basin was stressed. The natural recharge decreases as the pattern of precipitation is shifted from snowmelt-dominated to rainfall-dominated events. Moreover, the evapotranspiration rates increase as a result of increase in atmospheric temperatures. The SWP allocations during drought period are significantly reduced. During the 2011-2017 period, the SWP allocations varied between 5% to 85%, which drove Southern California water agencies to consider alternative water sources that are more locally reliable. The resulted mitigation possibility for this scenario was to incorporate desalination water at earlier years to meet residential and commercial water demands. Currently, desalination water is the most expensive water source among other sources, which results in extremely expensive costs of water supply. Consequently, as the future of groundwater is not certain due to ineffective groundwater management especially under projected climate conditions, significantly expensive desalination water may become the answer to water shortages under future uncertainty.

In addition, treated wastewater is commonly discharged to surface water as the most cost-effective water management strategy. As the water becomes more scarce, recycled water becomes a big portion of commercial water usage. However, treated municipal wastewater is often used as the last fallback for managed aquifer recharge. The model takes into account both the cost to treat wastewater for aquifer recharge and the cost to bring it up for usage. This cost is higher than the cost of imported water alone. Moreover, recycled water used for managed aquifer recharge from conventional wastewater treatment processes can negatively impact the groundwater quality. As the TDS concentration in the treated wastewater for recharge is approximately 500 mg/L, the TDS in groundwater aquifer also increases as a result. Therefore, the model almost always incorporates imported water as a water supply source before considering managed aquifer recharge option.

As water becomes scarcer, especially in and around urban environments, the use of water for irrigated agriculture is likely to be reconsidered, especially since agricultural irrigation often uses the most water. As such, agriculture on the urban fringe and which competes with municipal demands might have a difficult time avoiding an ever-increasing water cost due to the increased scarcity. Strategies include reducing acreage, increasing irrigation efficiency, and/or changing crop type to more drought-tolerant crops are recommended to mitigate such impacts.

The model developed in this research offers water agencies a tool to make cost-effective water decisions under future uncertainty, especially with respect to climate change. Demand for groundwater use will continue to increase in lockstep with population growth, and further problems will likely be caused by climate change conditions. Agencies will and always utilize groundwater resource as the main water supply source until its cost rise significantly and/or regulations limit overdraft. In term of the cost of water supply and the groundwater table level, the least sustainable scenarios are when the basin is stressed whereas the most sustainable cases are when the basin is full and agricultural irrigation water is reduced. Therefore, water agencies need collaboration to implement sustainable groundwater management practices including restriction of groundwater extraction, increase groundwater recharge, etc. in the face of water scarcity and demands.

Appendix A:

Supporting Information for Chapter 2

Table A.A.1. Average removal rates (%) of contaminants by each treatment process relative to Plant Influent

Water parameter	Symbol	Primary Effluent	Secondary Effluent	Granular Filtration	Microfiltration	Ultrafiltration	Nanofiltration	Reverse Osmosis
				SALINITY				
Salt Content								
Electrical Conductivity	EC_W	0	0	17.57	0	2.4	32	98
(or)								
Total Dissolved Solids	TDS	0	5.07	-3.16	5.07	25.17	59.85	98.10
Cations and Anions								
Calcium	Ca^{++}	0	0	11.87	0	4	56.7	99
Magnesium	Mg^{++}	0	0	5.98	0	21.08	81	99
Sodium	Na^{+}	0	0	-12.62	0	0	4.5	98
Carbonate	CO_3^{--}	0	0	0.00	0	0	99.05	97
Bicarbonate	HCO_3^{-}	0	0	54.55	0	0	6.1	98
Chloride	Cl^{-}	0	-12.37	-18.29	-12.37	-12.37	-12.37	97.75
Sulfate	SO_4^{--}	4.84	4.84	-17.65	4.84	4.84	99.05	99.05
				NUTRIENTS				
Nitrate-Nitrogen	NO_3-N	-100	-2450	-2500	-2450	-2450	-2373.5	-282.5
Ammonium-Nitrogen	NH_4-N	12.47	99.50	97.51	99.54	99.50	99.50	99.99
Phosphate-Phosphorus	PO_4-P	23.71	13.40	31.96	13.40	13.40	13.40	99.13
Potassium	K^{+}	0	0	5.66	0	7	10	98
				OTHER				
Boron	B	0	0	0	0	0	0	95
Total Coliforms	TC	99	99.90	99.9999998	100	100	100	100
TSS					100	100	100	100

Table A.A.2. Summary of treated municipal wastewater quality after each treatment process* [1-8]

Water parameter	Symbol	Unit	Plant Influent [1, 9]	Primary effluent [9]	Secondary Effluent [9]	Deep Filtration [7, 9]	MBR permeate [8]	MF permeate [2]	UF permeate [5, 6]	NF permeate [4]	RO permeate [2]	UV/AOP Product [3]
SALINITY												
Salt Content												
Electrical Conductivity	EC_W	dS/m	1.0336	*1.0336*	*1.0336*	0.852	*1.0088*	*1.0336*	1.009	0.7038	*0.062*	0.065
(or)												
Total Dissolved Solids	TDS	mg/L	522.5	*522.5*	496	539	*522.5*	496	390.997	209.808	29.76	33
Cations and Anions												
Calcium	Ca^{++}	mg/L	55.6	*55.6*	*55.6*	49	*53.376*	*55.6*	53.376	24.075	*0.556*	<0.5
Magnesium	Mg^{++}	mg/L	11.7	*11.7*	*11.7*	11	*9.234*	*11.7*	9.234	2.223	*0.117*	<0.5
Sodium	Na^{+}	mg/L	95.9	*95.9*	*95.9*	108	*95.9*	*95.9*	*95.9*	91.584	*2.877*	10.1
Carbonate	CO_3^{--}	mg/L	0	*0*	*0*	0	*0*	*0*	0	0	*0*	
Bicarbonate	HCO_3^{-}	mg/L	292.6	*292.6*	*292.6*	133	*292.6*	*292.6*	*292.6*	274.751	*8.778*	7.5
Chloride	Cl^{-}	mg/L	109.9	*109.9*	123.5	130	109.396	123.5	*123.5*	123.5	3.705	7.9
Sulfate	SO_4^{--}	mg/L	57.8	55	*55*	68	56.818	55	*55*	0.55	0.55	0.1
NUTRIENTS												
Nitrate-Nitrogen	NO_3-N	mg/L	0.2	0.4	5.1	5.2	3.8	5.1	5.1	4.947	0.204	1.67
Ammonium-Nitrogen	NH_4-N	mg/L	40.1	35.1	0.2	<1	0.1903	0.186	*0.2*	0.2	0.008	N/A
Phosphate-Phosphorus	PO_4-P	mg/L	9.7	7.4	8.4	6.6	6.51	8.4	*8.4*	8.4	0.084	N/A
Potassium	K^{+}	mg/L	15.9	*15.9*	*15.9*	15	*14.787*	*15.9*	14.787	14.31	*0.477*	0.8
OTHER												
Boron	B	mg/L	0.3	*0.3*	*0.3*	0.3	*0.3*	*0.3*	*0.3*	*0.3*	*0.015*	0.27
Acid/Basicity	pH	1-14	NR	NR	NR	7	NR	NR	NR	8.2	NR	5.9
Sodium Adsorption Ratio	SAR	meq/L	3.0	*3.0*	*3.0*	3.6	3.2	3.0	3.2	4.8	0.9	<2.4
Total Coliforms	TC	MPN/100ml	*1.00E+09*	*1.00E+07*	*1.00E+06*	<2	0	0	0	<2	RD	<1
Total Suspended Solid	TSS	mg/L	343.1	85.4	4	<2	0	0	0	0	0	N/A
Turbidity	NTU	NTU	164	104	2.62	0.83	0.15	0.105	0.45	0.35	0.03	0.07

***** Values in Table A.A.2 can vary based on geographical location, time of year, and activities that generate the wastewater. The values reported here are average values generated from the above mentioned references, and can vary significantly.

Table A.A.3. Guidelines of Water Quality for Irrigation Water of Specific Crops

Water parameter	Symbol	Usual range in irrigation water[10]	Unit	Citrus (mg/L)[11]	Turfgrass (mg/L)[2, 12, 13]
SALINITY					
Salt Content					
Electrical Conductivity	EC_W	0 – 3	dS/m	0.72	< 1.2
(or)					
Total Dissolved Solids	TDS	0 – 2000	mg/L	<500	< 832
Cations and Anions					
Calcium	Ca^{++}	0-400	mg/L	42	<100
Magnesium	Mg^{++}	0-60	mg/L	8.5	<40
Sodium	Na^{+}	0-920	mg/L	50 – 70	< 70
Carbonate	CO_3^{--}	0-30	mg/L	ND	<15
Bicarbonate	HCO_3^{-}	0-610	mg/L	105	< 120
Chloride	Cl^{-}	0-1065	mg/L	75 – 81	< 70
Sulfate	SO_4^{--}	0-960	mg/L	29 – 55	<90
NUTRIENTS					
Nitrate-Nitrogen	NO_3-N	0 – 10	mg/L	6.1 – 7	<10
Ammonium-Nitrogen	NH_4-N	0 – 5	mg/L	ND	<5
Phosphate-Phosphorus	PO_4-P	0 – 2	mg/L	1.1	<2
Potassium	K^{+}	0 – 2	mg/L	11.5	<20
OTHER					
Boron	B	0 – 2	mg/L	< 0.25	<0.5
Acid/Basicity	pH	6.0 – 8.5	unit	7.1 - 7.2	6.5 - 8.4
Sodium Adsorption Ratio*	SAR	0-15	meq/L	< 3 meq/L	<1.5 meq/L
Total Coliforms	TC	ND	MPN/100ml	10^6	23 or 2.2

$$^{*}\ SAR\left(\frac{meq}{L}\right) = \frac{Na^{+}}{\sqrt{\frac{Ca^{2+}+Mg^{2+}}{2}}}$$

Table A.A.4. Average individual treatment costs of different wastewater processes adjusted to 2013* U.S. Dollars[3, 8, 14-26]

Treatment Process	2013 Unit Cost	
	Small-Medium Plants (≤ 20000 m^3/d)	Large Plants (> 20000 m^3/d)
Activated Sludge	\$0.42 ± 0.09	\$0.25 ± 0.04
MBR	\$0.89 ± 0.33	\$0.61 ± 0.36
Granular Filtration	\$1.24 ± 0.21	\$0.19 ± 0.06
MF	\$0.38 ± 0.27	\$0.23 ± 0.09
UF	\$0.50 ± 0.27	\$0.28 ± 0.15
NF	\$0.78 ± 0.45	\$0.39 ± 0.09
RO	\$1.14 ± 0.54	\$0.37 ± 0.15
O_3	\$0.04 ± 0.02	\$0.03 ± 0.01
Cl_2	\$0.01 ± 0.01	\$0.02 ± 0.01
UV	\$0.01 ± 0.01	\$0.02 ± 0.02

*All values have been adjusted to 2013 U.S. Dollars using ENR CCI = 9547 and CPI = 697.836

Table A.A.5. Fertilizers supplied using blended irrigation sources[27-38]

	Citrus				Turfgrass			
	Nutrients supplied via…							
	Synthetic Fertilizer Alone (kg/ha-yr)	Blended wastewater under…			Synthetic Fertilizer Alone (kg/ha-yr)	Blended wastewater under…		
		Scenario A (kg/ha-yr)	Scenario B (kg/ha-yr)	Scenario C (kg/ha-yr)		Scenario A (kg/ha-yr)	Scenario B (kg/ha-yr)	Scenario C (kg/ha-yr)
N	100-400	0.88	1.32	2.00	98-195	1.37	2.20	3.12
P_2O_5	0-228*	33.1	87.2**	172**	49+	120**	201**	317**
K_2O	135-224	37.5	90.8	174	146	128	208*	642*

*Phosphorous application rate depends on phosphorous concentrations in soil and leaves, as well as age of trees

**Exceeds demand under certain situations

+Average value of many strains of turf grass

Table A.A.6. Costs and Cost Savings on synthetic fertilizer using blended irrigation sources ($/ha-yr)

	Citrus				Turfgrass			
	Costs from Synthetic Fertilizer Alone ($/ha-yr)	Synthetic fertilizer cost savings under…			Costs from Synthetic Fertilizer Alone ($/ha-yr)	Synthetic fertilizer cost savings under…		
		Scenario A ($/ha-yr)	Scenario B ($/ha-yr)	Scenario C ($/ha-yr)		Scenario A ($/ha-yr)	Scenario B ($/ha-yr)	Scenario C ($/ha-yr)
N	343.97	0.76	1.14	1.73	168.75	1.18	1.90	2.70
P_2O_5	270.88*	39.48	104.14**	204.96**	58.29	143.35**	239.67**	378.25**
K_2O	163.09	26.91	65.13	124.65	105.04+	91.96	148.82**	230.62**
Total Value (Cost) per ha-yr	(777.94)	67.15	170.41	331.34	(332.08)	236.49	390.39	611.57

*Phosphorous application rate depends on phosphorous concentrations in soil and leaves, as well as age of tree
**Exceeds demand under certain situations
+Average value of many strains of turf grass

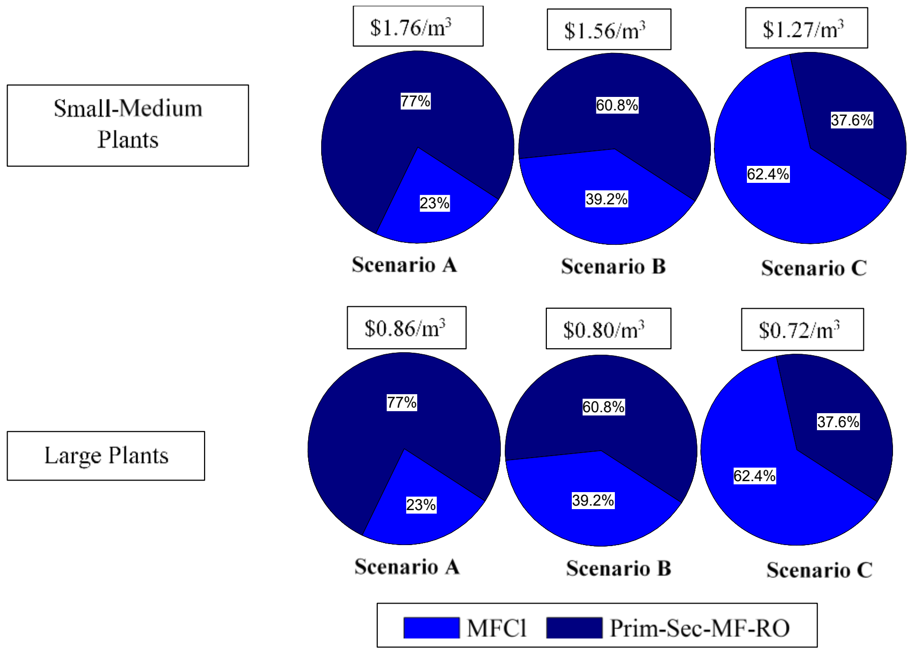

Figure A.A.F1. Optimized blending ratios for unrestricted access turfgrass irrigation from MF-RO treatment train for small-medium and large treatment facilities; due to TDS restrictions, all model solutions require some degree of desalination (RO). Three scenarios were investigated: (A) with crop nutrient and bicarbonate constraints (baseline); (B) without crop nutrient constraints; and (C) without crop nutrient and bicarbonate constraints.

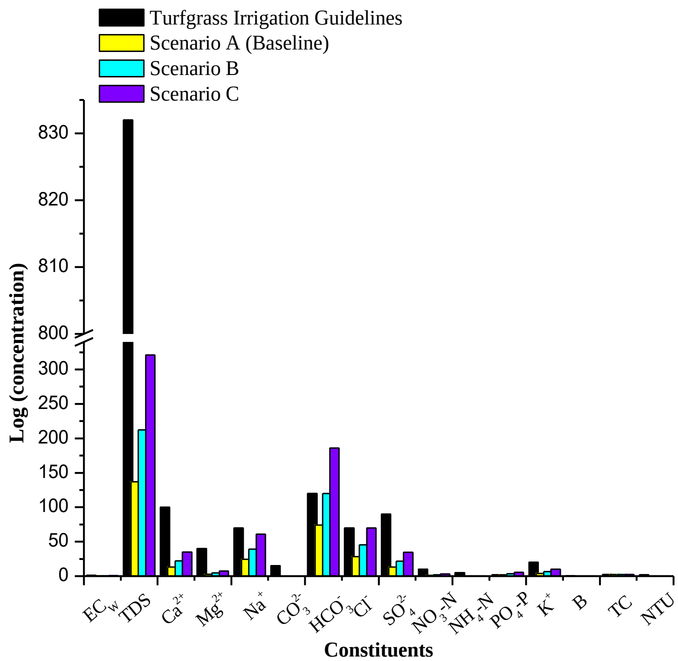

Figure A.A.F2. Comparison of irrigation guidelines for unrestricted access turfgrass (black bars) with the water quality parameters of the different blending ratios from the MF-RO treatment train under the three different constraints: (yellow bars) with crop nutrient constraints, i.e., baseline (RSC = 0.32 meq/L); (blue bars) without crop nutrient constraints (RSC = 0.47 meq/L); and (purple bars) without nutrient and bicarbonate constraints (RSC = 0.69 meq/L).

Sensitivity Analysis

Treatment Cost

Citrus

Table A.A.7. Sensitivity analysis on treatment cost for citrus

Scenarios	Lower-Range Treatment Costs						Higher-Range Treatment Costs					
	Small-Medium plants (≤20000 m^3/d)			Large plants (>20000 m^3/d)			Small-Medium plants (≤20000 m^3/d)			Large plants (>20000 m^3/d)		
	A	B	C	A	B	C	A	B	C	A	B	C
MBRCl				12	33.9							
SecEff	12.1	33.8	67.7			67.7	12.1	33.8	67.7	12.1	33.8	67.7
RO	87.9	66.2	32.2	88	66.1	32.3	87.9	66.2	32.2	87.9	66.2	32.2
Treatment Cost	$1.09	$0.90	$0.61	$0.49	$0.44	$0.35	$2.75	$2.23	$1.41	$1.38	$1.18	$0.87

Table A.A.8. Sensitivity analysis on removal performances for citrus

Scenarios	Lower Removal Performances						Higher Removal Performances					
	Small-Medium plants (≤20000 m^3/d)			Large plants (>20000 m^3/d)			Small-Medium plants (≤20000 m^3/d)			Large plants (>20000 m^3/d)		
	A	B	C	A	B	C	A	B	C	A	B	C
SecEff	8.4	28.8	66.3	8.4	28.8	66.3	12.1	34.5	69	12.1	34.5	69
RO-M	91.6	71.2	33.7	91.6	71.2	33.7	87.9	65.5	31	87.9	65.5	31
Treatment Cost	$1.89	$1.58	$1.01	$0.89	$0.77	$0.54	$1.83	$1.47	$0.97	$0.87	$0.73	$0.53

Table A.A.9. Sensitivity analysis on cost of disinfection of citrus irrigation water

Scenarios	Small-Medium plants (≤20000 m^3/d)			Large plants (>20000 m^3/d)		
	A	B	C	A	B	C
Non-disinfection	$1.83	$1.50	$0.99	$0.87	$0.74	$0.53
Chlorine Disinfection	$1.88	$1.55	$1.04	$0.92	$0.79	$0.58

Restricted Access Turfgrass

Table A.A.10. Sensitivity analysis on treatment cost for turfgrass with restricted access

Scenarios	Lower-Range Treatment Costs						Higher-Range Treatment Costs					
	Small-Medium plants (≤20000 m^3/d)			Large plants (>20000 m^3/d)			Small-Medium plants (≤20000 m^3/d)			Large plants (>20000 m^3/d)		
	A	B	C	A	B	C	A	B	C	A	B	C
MBRCl				22.7	39.2							
SecEffCl	23	39.2	62.4			62.4	23	39.2	62.4	23	39.2	62.4
RO	77	60.8	37.6	77.3	60.8	37.6	77	60.8	37.6	77	60.8	37.6
Treatment Cost	$0.99	$0.86	$0.66	$0.47	$0.43	$0.37	$2.49	$2.11	$1.55	$1.29	$1.15	$0.94

Table A.A.11. Sensitivity analysis on removal performances for turfgrass with restricted access

Scenarios	Lower Removal Performances						Higher Removal Performances					
	Small-Medium plants (≤20000 m^3/d)			Large plants (>20000 m^3/d)			Small-Medium plants (≤20000 m^3/d)			Large plants (>20000 m^3/d)		
	A	B	C	A	B	C	A	B	C	A	B	C
SecEffCl	19.8	34.5	59.1	19.8	34.5	59.1	23	39.8	62.9	23	39.8	62.9
RO-M	80.2	65.5	40.9	80.2	65.5	40.9	77	60.2	37.1	77	60.2	37.1
Treatment Cost	$1.72	$1.50	$1.13	$0.83	$0.74	$0.60	$1.67	$1.42	$1.07	$0.81	$0.71	$0.58

Table A.A.12. Sensitivity analysis on cost of disinfection of turfgrass irrigation water (Restricted)

Scenarios	Small-Medium plants (≤20000 m^3/d)			Large plants (>20000 m^3/d)		
	A	B	C	A	B	C
Non-disinfection	$1.65	$1.41	$1.06	$0.79	$0.69	$0.56
Chlorine Disinfection	$1.67	$1.43	$1.08	$0.81	$0.71	$0.58

Unrestricted Access Turfgrass

Table A.A.13. Sensitivity analysis on treatment cost for turfgrass with unrestricted access

Scenarios	Lower-Range Treatment Costs						Higher-Range Treatment Costs					
	Small-Medium plants (≤20000 m^3/d)			Large plants (>20000 m^3/d)			Small-Medium plants (≤20000 m^3/d)			Large plants (>20000 m^3/d)		
	A	B	C	A	B	C	A	B	C	A	B	C
MBRCl	22.7	39.2	62.7	22.7	39.2	62.7						
MFCl							23	39.2	62.4	23	39.2	62.4
RO	77.3	60.8	37.3	77.3	60.8	37.3	77	60.8	37.6	77	60.8	37.6
Treatment Cost	$1.04	$0.93	$0.79	$0.47	$0.43	$0.38	$2.65	$2.38	$1.99	$1.37	$1.28	$1.16

Table A.A.14. Sensitivity analysis on removal performances for turfgrass with unrestricted access

Scenarios	Lower Removal Performances						Higher Removal Performances					
	Small-Medium plants (≤20000 m^3/d)			Large plants (>20000 m^3/d)			Small-Medium plants (≤20000 m^3/d)			Large plants (>20000 m^3/d)		
	A	B	C	A	B	C	A	B	C	A	B	C
MFCl	19.8	34.5	59.1	19.8	34.5	59.1	23	39.8	62.9	23	39.8	62.9
RO-M	80.2	65.5	40.9	80.2	65.5	40.9	77	60.2	37.1	77	60.2	37.1
Treatment Cost	$1.80	$1.63	$1.35	$0.87	$0.82	$0.73	$1.76	$1.57	$1.31	$0.86	$0.80	$0.72

Table A.A.15. Sensitivity analysis on cost of disinfection of turfgrass irrigation water (Unrestricted)

Scenarios	Small-Medium plants (≤20000 m^3/d)			Large plants (>20000 m^3/d)		
	A	B	C	A	B	C
Non-disinfection	$1.74	$1.54	$1.25	$0.84	$0.78	$0.70
Chlorine Disinfection	$1.76	$1.56	$1.27	$0.86	$0.80	$0.72

Treatment Cost Analysis

Table A.A.16. Parameters and cost of a full scale NF process [17]

Drinking water production capacity (m3/d)	100000
WRR (%)	90
Pressure (Pa)	6.00E+05
Total membrane area (m2)	159094
Pump efficiency (%)	70
Power demand (kWh/m3)	0.54
Membrane lifetime (years)	5
Cost of membrane (E/m2)	19
Membrane surface area (m2)	0.216
Energy Costs (Euro/kWh)	0.1
Amortization period (years)	15
Interest rate (%)	8
Capital cost (Euro)	
Pipes and valves	900607
Instruments and control	3530480
Tanks and frames	1633436
Miscellaneous	6536824
Pumps	236559
Membranes	2863686
Pressure vessels	1909124
Total capital costs	17610716
Operating costs (Euro/m3)	
Amortization[1]	0.067
Membrane replacement	0.017
Energy	0.048
Maintenance	0.01
Pretreatment	0.023
Chemicals	0.01
Concentrate disposal	0.037
Total treatment cost (Euro/m3)	0.214
Conversion 1Euro = $1.10 (2003)	$1.10
Total treatment cost ($/m3)	$0.24

[1] $A = P\frac{r(1+r)^n}{(1+r)^n - 1}$ where A is the payment amount per period, P is the initial Principal (loan amount), r is the interest rate per period, and n is the total number of payments or periods.

Table A.A.17. Parameters and cost of a full scale MF process [3, 17]

Drinking water production capacity (m3/d)	265000
WRR (%)	90
Pressure (Pa)	6.00E+05
Total membrane area (m2)	626092.2
Pump efficiency (%)	70
Power demand (kWh/m3)	0.12225
Membrane lifetime (years)	5
Cost of membrane (E/m2)	13.74272
Energy Costs (Euro/kWh)	0.1
Amortization period (years)	15
Interest rate (%)	8
Capital cost (Euro)	
Pipes and valves	900607
Instruments and control	3530480
Tanks and frames	1633436
Miscellaneous	6536824
Pumps	236559
Membranes	8604206.501
Pressure vessels	1909124
Total capital costs	23351236.5
Operating costs (Euro/m3)	
Amortization	0.031338725
Membrane replacement	0.019767856
Energy	0.012225
Maintenance	0.01
Pretreatment	0.023
Chemicals	0.01
Concentrate disposal	0.037
Total treatment cost (Euro/m3)	0.143331581
Conversion 1Euro = $1.25520 (2009)	$1.26
Total treatment cost ($/m3)	$0.18

Table A.A.18. Parameters and cost of a full scale MBR process [8]

Drinking water production capacity (m3/d)	3785.4
WRR (%)	90
Membrane surface area (m2)	0.93
Amortization period (years)	15
Interest rate (%)	8
Capital cost ($K)	
Headworks	834
MBR Process Costs	1750
MBR Tank	180
Operation-laboratory building	368
Maintenance Building	154
Subtotal ($K)	3286
Site Development, 15%	492.9
Installation of MF/MBR, 30%	525
Process Piping, 15%	492.9
Instrumentation, 2%	65.72
Electrical distribution and control, 16%	525.76
Electrical Service, 5%	164.3
Subtotal ($K)	5552.58
Contigency, 10%, $K	555.258
Total capital costs, $K	6107.838
Operating costs ($K/yr))	
Amortization ($/m3)	0.57384294
Personnel	85
Supervision-administration	31
Power	115
Spare Parts-replacement	6
Sludge Handling and Disposal	84
MBR chemicals	1
Maintenance Clean	2
Membrane Replacement	26
Total O&M cost in 1st year ($K/yr)	350
Total Estimated O&M cost, $K	$2,995.82
O&M cost ($/m3)	$0.28
Total treatment cost ($/m3)	$0.86

Table A.A.19. Parameters and cost of a full scale UF process [18, 20]

Drinking water production capacity (m3/d)	100000
WRR (%)	90
Power demand (kWh/m3)	0.31
Amortization period (years)	15
Interest rate (%)	8
Capital cost ($)	
Total capital costs	$5,520,000.00
Operating costs ($/m3)	
Amortization	$0.019631631
O&M	$0.17
Total treatment cost ($/m3) (2002)	$0.189

Table A.A.20. Parameters and cost of a full scale RO process [15, 17]

Drinking water production capacity (m3/d)	265000
WRR (%)	90
Pressure (Pa)	6.00E+05
Pump efficiency (%)	70
Power demand (kWh/m3)	0.75
Membrane lifetime (years)	5
Energy Costs (Euro/kWh)	0.1
Amortization period (years)	15
Interest rate (%)	8
Capital cost (Euro)	
Pipes and valves	900607
Instruments and control	3530480
Tanks and frames	1633436
Miscellaneous	6536824
Pumps	236559
Membranes	5736137.67
Pressure vessels	1909124
Total capital costs	20483167.67
Operating costs (Euro/m3)	
Amortization	0.027489609
Membrane replacement	0.013178571
Energy	0.075
Maintenance	0.01
Pretreatment	0.023
Chemicals	0.01
Concentrate disposal	0.037
Total treatment cost (Euro/m3)	0.195668179
Conversion 1Euro = $1.25520 (2009)	$1.26
Total treatment cost ($/m3)	$0.246

Table A.A.21. Treatment cost of a full scale Ozonation process [19]

Drinking water production capacity (m3/d)	378540
Amortization period (years)	15
Interest rate (%)	8
Capital cost ($)	
Total capital costs	$18,000,000.00
Operating costs ($/m3)	
Amortization	$0.016911341
O&M	$0.01
Total treatment cost ($/m3)	$0.028

Table A.A.22. Treatment cost of a full scale UV process [15]

Drinking water production capacity (m3/d)	94635
Amortization period (years)	15
Interest rate (%)	8
Capital cost ($)	
Total capital costs	$6,421,000.00
Total O&M costs	$315000
Operating costs ($/m3)	
Amortization	0.024130604
O&M	0.010132658
Total treatment cost ($/m3)	$0.034

Table A.A.23. Treatment cost of a full scale Chlorine disinfection process (20 mg/L) [25]

Drinking water production capacity (m3/d)	75708
WRR (%)	90
Amortization period (years)	15
Interest rate (%)	8
Capital cost ($)	
Total capital costs	$3,949,000.00
Total O&M costs	$379100
Operating costs ($/m3)	
Amortization	0.018550801
O&M	0.015243217
Total treatment cost ($/m3)	$0.034

Table A.A.24. Treatment costs of different wastewater treatment processes

Treatment Process	Plant capacity	Unit	Capital Unit Cost ($/m^3)	O&M Unit Cost ($/m^3)	Unit cost ($/m^3)	Year	CCI	CPI	PPI	References
Activated sludge	3800	m^3/d			$0.40	2004	7115	565.8	148.5	[19]
Activated sludge	6000	m^3/d			$0.34	2001	6342	530.4	140.7	[2]
Activated sludge	18927.05	m^3/d	$0.25	$0.14	$0.38	1996	5622	469.9		[39]
Activated sludge	19000	m^3/d			$0.25	2004	7115	565.8	148.5	[19]
Activated sludge	37854.1	m^3/d	$0.21	$0.12	$0.34	1996	5622	469.9		[39]
Activated sludge	38000	m^3/d			$0.20	2004	7115	565.8	148.5	[19]
Activated sludge	76000	m^3/d			$0.17	2004	7115	565.8	148.5	[19]
Activated sludge	2500	Mm^3/yr			$0.22	2003	6695	551.1	143.3	[14]
MBR	1500	m^3/d	$0.42	$0.17	$0.59	2000	6221	515.8		[40]
MBR	3000	m^3/d	$1.24	$0.13	$1.37	2015		709.998		[41]
MBR	3785.4	m^3/d			$0.54	2004	7115	565.8	148.5	[42]
MBR	3785.4	m^3/d	$0.57	$0.28	$0.86	2001	6342	530.4	140.7	[8]
MBR	3785.4	m^3/d			$0.61	2006	7751	603.9		[43]
MBR	3800	m^3/d			$0.42	2004	7115	565.8	148.5	[19]
MBR	19000	m^3/d	$0.94	$0.02	$0.96	2012	9299	687.761		[42]
MBR	20000	m^3/d			$0.50	2006	7751	603.9		[43]
MBR	38000	m^3/d			$0.22	2004	7115	565.8	148.5	[19]
Tertiary	456	m^3/d			$1.29	2008	8310	644.951		[44]
Tertiary	3044	m^3/d			$1.00	2008	8310	644.951		[44]
Tertiary	75000	m^3/d	$0.25	$0.38	$0.62	2005	7446	585		[45]
Tertiary	378541	m^3/d			$0.10	1994	5408	444		[39]
Tertiary	378541	m^3/d	$0.03	$0.06	$0.09	1991	4835	408		[39]
Tertiary	6000	Mm^3/yr			$0.20	2003	6695	551.1	143.3	[14]

MF	3785.41	m^3/d			$0.60	2006	7751	603.9		46
MF	3800	m^3/d			$0.15	1996	5622	469.9		24
MF	5000	m^3/d	$0.03	$0.13	$0.16	2000	6221	515.8	138	16
MF	23000	m^3/d			$0.09	1996	5622	469.9		24
MF	37854.1	m^3/d			$0.30	2006	7751	603.9		46
MF	265000	m^3/d	$0.04	$0.14	$0.18	2009	8570	642.658	172.5	3, 17
MF	378541	m^3/d			$0.20	2006	7751	603.9		46
UF	378	m^3/d			$0.45	1994	5408	444		47
UF	3780	m^3/d			$0.25	1994	5408	444		47
UF	3785.41	m^3/d			$0.75	2006	7751	603.9		46
UF	3800	m^3/d			$0.20	1996	5622	469.9		24
UF	20000	m^3/d	$0.05	$0.17	$0.22	2007	7967	621.106	166.6	18
UF	23000	m^3/d			$0.13	1996	5622	469.9		24
UF	37800	m^3/d			$0.20	1994	5408	444		47
UF	37854.1	m^3/d			$0.48	2006	7751	603.9		46
UF	100000	m^3/d	$0.02	$0.17	$0.19	2002	7751	603.9	138.9	18, 20
UF	378541	m^3/d			$0.25	2006	7751	603.9		46
UF	378541	m^3/d			$0.09	1994	5408	444		39
NF	3000	m^3/d	$1.03	$0.57	$1.60	2015		709.998		41
NF	3785.41	m^3/d	$0.20	$0.53	$0.73	1995	5471	456.5	127.9	48
NF	10200	m^3/d	$0.20	$0.26	$0.47	1993	5210	432.7		49
NF	16300	m^3/d	$0.20	$0.26	$0.46	1996	5622	469.9		49
NF	18000	m^3/d	$0.19	$0.24	$0.44	2010	8799	653.198		49
NF	18927.05	m^3/d	$0.11	$0.18	$0.28	1995	5471	456.5	127.9	48
NF	20000	m^3/d			$0.27	2002	7751	603.9	138.9	21
NF	37854.1	m^3/d	$0.08	$0.14	$0.22	1995	5471	456.5	127.9	48

NF	50000	m^3/d	$0.15	$0.21	$0.36	2008	8310	644.951		[49]
NF	53000	m^3/d			$0.25	2003	6695	551.1	143.3	[17]
NF	65830	m^3/d	$0.15	$0.21	$0.36	1996	5622	469.9		[49]
NF	82650	m^3/d	$0.14	$0.21	$0.35	1996	5622	469.9		[49]
NF	94625	m^3/d	$0.13	$0.21	$0.34	1996	5622	469.9		[49]
NF	100000	m^3/d	$0.13	$0.20	$0.32	2006	7751	603.9		[49]
NF	100000	m^3/d	$0.07	$0.17	$0.24	2003	6695	551.1	143.3	[17]
NF	123000	m^3/d	$0.13	$0.19	$0.32	2008	8310	644.951		[49]
NF	132650	m^3/d	0.12	$0.18	$0.31	2006	7751	603.9		[49]
NF	150000	m^3/d	$0.12	$0.17	$0.30	2008	8310	644.951		[49]
NF	171300	m^3/d	$0.12	$0.17	$0.29	2005	7446	585		[49]
NF	193750	m^3/d	$0.10	$0.16	$0.26	2005	7446	585		[49]
NF	567851.15	m^3/d	$0.08	$0.13	$0.20	1995	5471	456.5	127.9	[48]
RO	90	m^3/d			$0.65	2015		709.998		[50]
RO	91.2	m^3/d			$1.09	2015		709.998		[50]
RO	3000	m^3/d	$1.02	$0.72	$1.74	2015		709.998		[41]
RO	38000	m^3/d	$0.07	$0.21	$0.28	2004	7115	565.8	148.5	[19]
RO	265000	m^3/d	$0.07	$0.21	$0.25	2009	8570	642.658	172.5	[15, 17]
RO	378541	m^3/d			$0.15	1994	5408	444		[39]
RO	378541	m^3/d	$0.12	$0.19	$0.31	1991	4835	408		[39]
O3	1500	m^3/d			$0.04	1995	5471	456.5	127.9	[22]
O3	6000	m^3/d			$0.02	1995	5471	456.5	127.9	[22]
O3	15000	m^3/d			$0.02	1995	5471	456.5	127.9	[22]
O3	30000	m^3/d			$0.01	1995	5471	456.5	127.9	[22]
O3	30000	m^3/d			$0.02	1995	5471	456.5	127.9	[22]
O3	378540	m^3/d	$0.02	$0.01	$0.03	2007	7967	621.106	166.6	[15, 19]

O3	378541	m^3/d			$0.02	1994	5408	444		39
O3	378541	m^3/d	$0.00	$0.02	$0.02	1991	4835	408		39
Cl2	1500	m^3/d			$0.01	1995	5471	456.5	127.9	22
Cl2	6000	m^3/d			$0.01	1995	5471	456.5	127.9	22
Cl2	15000	m^3/d			$0.01	1995	5471	456.5	127.9	22
Cl2	30000	m^3/d			$0.00	1995	5471	456.5	127.9	22
Cl2 10mg/L	75708	m^3/d	$0.02	$0.01	$0.03	1999	6060	499	133	25
Cl2 10mg/L	94635	m^3/d	$0.01	$0.01	$0.02	2007	7967	621.106	166.6	15
Cl2 gas 10 mg/L	94635	m^3/d	$0.01	$0.004	$0.01	2007	7967	621.106	166.6	15
Cl2 20mg/L	75708	m^3/d	$0.02	$0.015	$0.035	1999	6060	499	133	25
UV	1500	m^3/d			$0.01	1995	5471	456.5	127.9	22
UV	6000	m^3/d			$0.01	1995	5471	456.5	127.9	22
UV	15000	m^3/d			$0.01	1995	5471	456.5	127.9	22
UV	30000	m^3/d			$0.01	1995	5471	456.5	127.9	22
UV	94635	m^3/d	$0.02	$0.01	$0.03	2007	7967	621.106	166.6	15
UV - 80 mJ/cm2	113562	m^3/d			$0.01	1999	6060	499	133	26
UV	265000	m^3/d	$0.001	$0.03	$0.03	2009	8570	642.658	172.5	3
UV-100 mJ/cm2	2400	m^3/d			$0.02	2002	7751	603.9	138.9	51
UV-160 mJ/cm2	2400	m^3/d			$0.04	2002	7751	603.9	138.9	51
Groundwater (without subsidies)	265000	m^3/d	$0.37	$0.34	$0.71	2010	8799	653.198	179.8	23

Table A.A.25. Average individual treatment costs of different wastewater processes with time adjustment (2013)

Treatment Process	Plant capacity	Unit	2013 Capital Cost	2013 O&M Cost	2013 Capital Unit Cost[a]	2013 O&M Unit Cost[b]	2013 Unit cost	Average 2013 Unit cost
Activated sludge	3800	m^3/d					$0.49	$0.48
Activated sludge	6000	m^3/d					$0.45	
Activated sludge	18927.05	m^3/d			$0.42	$0.20	$0.62	
Activated sludge	19000	m^3/d					$0.31	
Activated sludge	37854.1	m^3/d			$0.36	$0.19	$0.55	$0.32
Activated sludge	38000	m^3/d					$0.25	
Activated sludge	76000	m^3/d					$0.21	
Activated sludge	2500	Mm^3/yr					$0.28	
MBR	1500	m^3/d			$0.64	$0.23	$0.87	$0.89
MBR	3000	m^3/d					$1.35	
MBR	3785.4	m^3/d					$0.66	
MBR	3785.4	m^3/d	$9,194,501.64	$3,941,533.43	$0.86	$0.37	$1.23	
MBR	3785.4	m^3/d					$0.70	
MBR	3800	m^3/d					$0.52	
MBR	19000	m^3/d			$0.96	$0.02	$0.98	$0.61
MBR	20000	m^3/d					$0.58	
MBR	38000	m^3/d					$0.27	
Tertiary	456	m^3/d					$1.39	$1.24
Tertiary	3044	m^3/d					$1.09	
Tertiary	75000	m^3/d			$0.32	$0.45	$0.77	$0.33
Tertiary	378541	m^3/d					$0.15	
Tertiary	378541	m^3/d			$0.06	$0.09	$0.16	
Tertiary	6000	Mm^3/yr					$0.25	
MF	3785.41	m^3/d					$0.69	$0.38

MF	3800	m^3/d					$0.22	
MF	5000	m^3/d			$0.05	$0.17	$0.22	
MF	23000	m^3/d					$0.13	$0.23
MF	37854.1	m^3/d					$0.35	
MF	265000	m^3/d	$32,651,934.27	$13,287,948.95	$0.04	$0.15	$0.20	
MF	378541	m^3/d					$0.23	
UF	378	m^3/d					$0.71	$0.50
UF	3780	m^3/d					$0.39	
UF	3785.41	m^3/d					$0.87	
UF	3800	m^3/d					$0.30	
UF	20000	m^3/d	$32,651,934.27	$1,251,284.12	$0.06	$0.19	$0.25	
UF	23000	m^3/d					$0.19	$0.28
UF	37800	m^3/d					$0.31	
UF	37854.1	m^3/d					$0.55	
UF	100000	m^3/d	$6,799,050.45	$6,434,675.25	$0.02	$0.20	$0.22	
UF	378541	m^3/d					$0.29	
UF	378541	m^3/d					$0.14	
NF	3000	m^3/d					$1.57	$0.78
NF	3785.41	m^3/d			$0.35	$0.81	$1.16	
NF	10200	m^3/d			$0.37	$0.43	$0.80	
NF	16300	m^3/d			$0.34	$0.38	$0.73	
NF	18000	m^3/d			$0.21	$0.26	$0.47	
NF	18927.05	m^3/d			$0.18	$0.27	$0.45	
NF	20000	m^3/d					$0.31	
NF	37854.1	m^3/d			$0.14	$0.21	$0.35	$0.39
NF	50000	m^3/d			$0.17	$0.23	$0.40	
NF	53000	m^3/d					$0.32	

NF	65830	m^3/d			$0.25	$0.31	$0.56	
NF	82650	m^3/d			$0.24	$0.31	$0.55	
NF	94625	m^3/d			$0.23	$0.30	$0.53	
NF	100000	m^3/d			$0.15	$0.23	$0.38	
NF	100000	m^3/d	$27,873,529.00	$6,973,425.28	$0.10	$0.21	$0.31	
NF	123000	m^3/d			$0.14	$0.21	$0.35	
NF	132650	m^3/d			$0.15	$0.21	$0.36	
NF	150000	m^3/d			$0.14	$0.19	$0.33	
NF	171300	m^3/d			$0.15	$0.20	$0.35	
NF	193750	m^3/d			$0.13	$0.19	$0.32	
NF	567851.15	m^3/d			$0.13	$0.19	$0.32	
RO	90	m^3/d					$0.64	
RO	91.2	m^3/d					$1.07	$1.14
RO	3000	m^3/d					$1.71	
RO	38000	m^3/d	$24,576,648.21	$3,233,171.13	$0.09	$0.26	$0.35	
RO	265000	m^3/d	$28,641,525.87	$19,954,382.27	$0.08	$0.23	$0.31	$0.37
RO	378541	m^3/d					$0.23	
RO	378541	m^3/d			$0.24	$0.33	$0.57	
O3	1500	m^3/d					$0.06	
O3	6000	m^3/d					$0.04	$0.04
O3	15000	m^3/d					$0.03	
O3	30000	m^3/d					$0.02	
O3	30000	m^3/d					$0.02	
O3	378540	m^3/d	$21,569,725.12	$1,491,087.17	$0.02	$0.01	$0.03	$0.03
O3	378541	m^3/d					$0.04	
O3	378541	m^3/d			$0.00	$0.03	$0.04	
Cl2	1500	m^3/d					$0.02	$0.01

Cl2	6000	m^3/d					$0.01	
Cl2	15000	m^3/d					$0.01	
Cl2	30000	m^3/d					$0.01	$0.02
Cl2 10mg/L	75708	m^3/d	$5,632,099.83	$317,032.91	$0.03	$0.01	$0.04	
Cl2 10mg/L	94635	m^3/d	$1,743,552.78	$371,890.98	$0.01	$0.01	$0.02	
Cl2 gas 10 mg/L	94635	m^3/d	$2,547,624.20	$123,589.15	$0.01	$0.004	$0.01	
Cl2 20mg/L	75708	m^3/d	$6,221,304.13	$530,159.57	$0.03	$0.02	$0.05	$0.05
UV	1500	m^3/d					$0.02	$0.01
UV	6000	m^3/d					$0.01	
UV	15000	m^3/d					$0.01	
UV	30000	m^3/d					$0.01	$0.02
UV	94635	m^3/d	$7,694,400.28	$353,914.37	$0.03	$0.01	$0.04	
UV - 80 mJ/cm^2	113562	m^3/d					$0.01	
UV	265000	m^3/d	$445,600.93	$2,594,333.68	$0.001	$0.03	$0.03	
UV-100 mJ/cm2	2400	m^3/d					$0.02	$0.03
UV-160 mJ/cm2	2400	m^3/d					$0.04	
Groundwater (without subsidies)	265000	m^3/d			$0.40	$0.37	$0.77	$0.77

a: All values have been adjusted to 2013 values using ENR CCI = 9547

b: All values have been adjusted to 2013 values using CPI = 697.836

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Appendix B:

Supporting Information for Chapter 3

Descriptions of the 11 treatment configurations:

Municipal wastewater is treated at municipal wastewater treatment plants, which typically use primary, secondary, tertiary, and disinfection processes to meet state and federal regulations. The primary treatment stage includes the mechanical screening of large and coarse objects, followed by a clarifier where inorganic particles settle out. Water from the clarifier then continues to the secondary treatment stage that consists of biological reactors (such as activated sludge (AS) or membrane bioreactors (MBRs)) coupled to another clarification stage; in MBRs, clarification is achieved through the incorporation of membranes directly into the reactors. The purpose of the secondary treatment is to break down nutrients and organic matters biologically, typically through the use of activated sludge. Effluent from the secondary clarifier is either disinfected and released to surface water or further treated with tertiary treatment such as granular filtration (GF), membrane separation (microfiltration (MF), ultrafiltration (UF), nanofiltration (NF), reverse osmosis (RO)) and/or advanced oxidation processes (AOP), followed by disinfection (Cl_2, UV, O_3) (Tchobanoglous et al., 2003).

(1) TERT-MF-NF: This treatment train includes plant influent, primary screening, activated sludge, granular filtration, microfiltration, nanofiltration, and disinfection.
(2) TERT-MF-RO: This treatment train includes plant influent, primary screening, activated sludge, granular filtration, microfiltration, reverse osmosis, and disinfection.
(3) TERT-UF-NF: This treatment train includes plant influent, primary screening, activated sludge, granular filtration, ultrafiltration, nanofiltration, and disinfection.
(4) TERT-UF-RO: This treatment train includes plant influent, primary screening, activated sludge, granular filtration, ultrafiltration, reverse osmosis, and disinfection.
(5) TERT: This treatment train includes plant influent, primary screening, activated sludge, granular filtration, and disinfection.
(6) MF-NF: This treatment train includes plant influent, primary screening, activated sludge, microfiltration, nanofiltration, and disinfection.
(7) MF-RO: This treatment train includes plant influent, primary screening, activated sludge, microfiltration, reverse osmosis, and disinfection.
(8) UF-NF: This treatment train includes plant influent, primary screening, activated sludge, ultrafiltration, nanofiltration, and disinfection.
(9) UF-RO: This treatment train includes plant influent, primary screening, activated sludge, ultrafiltration, reverse osmosis, and disinfection.
(10) MBR-NF: This treatment train includes plant influent, primary screening, membrane bioreactors, nanofiltration, and disinfection.
(11) MBR-RO: This treatment train includes plant influent, primary screening, membrane bioreactors, reverse osmosis, and disinfection.

Table A.B.1. Average removal rates (%) of contaminants by each treatment process relative to Plant Influent in 2011 at IEUA – RP1

(Adham and Trussell, 2001; U.S. EPA, 2004; Xia et al., 2004; García-Figueruelo et al., 2009; Norouzbahari et al., 2009; Groundwater Replenishment System (GWRS), 2014; Inland Empire Utilities Agency (IEUA), 2013;2014; Price, 2016)

Water parameter	Symbol	Primary Effluent	Secondary Effluent	Granular Filtration	Microfiltration	Ultrafiltration	Nanofiltration	Reverse Osmosis
SALINITY								
Salt Content								
Electrical Conductivity	EC_W	*0*	*0*	12.58	*0*	2.40	32.00	94.00
(or)								
Total Dissolved Solids	TDS	*0*	*0*	-8.89	-8.89	14.16	53.94	93.47
Cations and Anions								
Calcium	Ca^{++}	*0*	*0*	15.27	0	4.00	56.70	99.00
Magnesium	Mg^{++}	*0*	*0*	14.51	*0*	21.08	81.00	99.00
Sodium	Na^{+}	*0*	*0*	-16.48	0	0.00	4.50	97.00
Carbonate	CO_3^{--}	*0*	*0*	0.00	*0*	21.62	99.00	99.00
Bicarbonate	HCO_3^{-}	*0*	*52.11*	52.11	*52.11*	*52.11*	55.03	98.56
Chloride	Cl^{-}	*0*	*0*	-35.97	0	0.00	0.00	97.00
Sulphate	SO_4^{--}	*0*	*0*	-8.89	0	0.00	99.00	99.00
NUTRIENTS								
Nitrate-Nitrogen	NO_3-N	*0*	*-2394.47*	-2394.47	-2394.47	-2394.47	-2319.63	0.22
Ammonium-Nitrogen	NH_4-N	*0*	*99.70*	99.70	99.72	99.70	99.70	99.99
Phosphate-Phosphorus	PO_4-P	*51.56*	*58.26*	*70.54*	58.26	*58.26*	58.26	99.58
Potassium	K^{+}	*0*	*0*	15.96	*0*	7.00	10.00	97.00
OTHER								
Boron	B	*0*	*0*	0.00	*0*	0	*0*	95.00

Total Coliforms	TC	*99*	*99.90*	99.9999998	100	100	100	100
Total Suspended Solids	TSS	0	97.4831	99.6324	100	100	100	100
Turbidity	NTU	36.59	98.4024	99.5122	99.95732	99.72561	99.786585	99.9817

Table A.B.2. Average removal rates (%) of contaminants by each treatment process relative to Plant Influent in 2015 at IEUA – RP1

(Adham and Trussell, 2001; García-Figueruelo et al., 2009; Norouzbahari et al., 2009; GWRS, 2014; IEUA, 2013;2014; Price, 2016)

Water parameter	Symbol	Primary Effluent	Secondary Effluent	Granular Filtration	Microfiltration	Ultrafiltration	Nanofiltration	Reverse Osmosis
SALINITY								
Salt Content								
Electrical Conductivity	EC_W	*0*	*0*	12.94	*0*	2.40	32.00	94.00
(or)								
Total Dissolved Solids	TDS	*0*	*1.18*	1.18	1.18	22.10	58.20	94.07
Cations and Anions								
Calcium	Ca^{++}	*0*	*0*	13.76	0	4.00	56.70	99.00
Magnesium	Mg^{++}	*0*	*0*	14.74	*0*	21.08	81.00	99.00
Sodium	Na^{+}	*0*	*0*	-14.32	0	0	4.50	97.00
Carbonate	CO_3^{--}	*0*	*0*	0	*0*	21.62	99	99.00
Bicarbonate	HCO_3^{-}	*0*	*48.35*	48.35	*48.35*	*48.35*	51.50	98.45
Chloride	Cl^{-}	*0*	*0*	-44.86	0	0	0	97
Sulphate	SO_4^{--}	*0*	*0*	-7.41	0	*0*	99	99
NUTRIENTS								
Nitrate-Nitrogen	NO_3-N	*0*	*-3121.43*	-3121.43	-3121.43	-3121.43	-3024.79	-28.86
Ammonium-Nitrogen	NH_4-N	*0*	*99.72*	99.72	99.74	99.72	99.72	99.99
Phosphate-Phosphorus	PO_4-P	*34.54*	*55.48*	*56.87*	55.48	55.48	55.48	99.55
Potassium	K^{+}	*0*	*0*	5.75	*0*	7.00	10.00	97.00
OTHER								
Boron	B	*0*	*0*	0	*0*	0	*0*	95.00
Total Coliforms	TC	*99*	*99.90*	99.9999998	100.0	100	100	100

Total Suspended Solids	TSS	0	*98.9967*	99.5566	100	100	100	100
Turbidity	NTU	36.59	98.4024	99.6341	99.9360	99.7256	99.7866	99.9817

Table A.B.3. 2011 IEUA – RP1 – Summary of treated municipal wastewater quality after each treatment process (Adham and Trussell, 2001; U.S. EPA, 2004; Xia et al., 2004; García-Figueruelo et al., 2009; Norouzbahari et al., 2009; GWRS, 2014; IEUA, 2013;2014; Price, 2016)

Water parameter	Symbol	Unit	Plant Influent	Primary effluent	Secondary Effluent	Deep Filtration	MBR permeate	MF permeate	UF permeate	NF permeate	RO permeate
SALINITY											
Salt Content											
Electrical Conductivity	EC_W	dS/m	0.8785	*0.8785*	*0.8785*	0.768	0.857416	*0.8785*	0.857416	0.59738	0.05271
(or)											
Total Dissolved Solids	TDS	mg/L	431.61	*431.61*	*470*	470	431.61	470	370.501	198.81	28.2
Cations and Anions											
Calcium	Ca^{++}	mg/L	48.58	*48.58*	*48.58*	41.16	46.6368	48.58	46.6368	21.03514	0.4858
Magnesium	Mg^{++}	mg/L	9.625	*9.625*	*9.625*	8.228	7.59605	*9.625*	7.59605	1.82875	0.09625
Sodium	Na^{+}	mg/L	76.17	*76.17*	*76.17*	88.72	76.17	76.17	76.17	72.74235	2.2851
Carbonate	CO_3^{--}	mg/L	3	*3*	*3*	3	2.3514	*3*	*2.3514*	3	0.03
Bicarbonate	HCO_3^{-}	mg/L	354.0074	*354.0074*	*169.5312*	169.5312	354.0074	*169.5312*	169.5312	159.1898	5.0859
Chloride	Cl^{-}	mg/L	71.25	*71.25*	*71.25*	96.88	70.9232	71.25	71.25	71.25	2.1375
Sulfate	SO_4^{--}	mg/L	37.58	*37.58*	*37.58*	40.92	38.8223	37.58	*37.58*	0.3758	0.3758
NUTRIENTS											
Nitrate-Nitrogen	NO_3-N	mg/L	0.253	*0.253*	*6.311*	6.311	2.4035	6.311	6.311	6.12167	0.25244
Ammonium-Nitrogen	NH_4-N	mg/L	33.602	*33.602*	*0.1016*	0.1016	0.1822	0.0945	0.1016	0.1016	0.0041
Phosphate-Phosphorus	PO_4-P	mg/L	*2.923*	*1.416*	*1.22*	*0.861*	1.2457	1.22	*1.22*	1.22	0.0122
Potassium	K^{+}	mg/L	17.42	*17.42*	*17.42*	14.64	16.2006	*17.42*	16.2006	15.678	0.5226
OTHER											
Boron	B	mg/L	*0.2*	*0.2*	*0.2*	0.2	0.2	*0.2*	*0.2*	*0.2*	0.01
Sodium Adsorption Ratio	SAR	meq/L	2.61	*2.61*	*2.61*	3.291	2.721	2.61	2.72	4.08	0.78

Total Coliforms	TC	MPN/100ml	1.00E+09	1.00E+07	1.00E+06	<2 (7-day median)	0	1	0	<2	<2
Total Suspended Solid	TSS	mg/L	321.82	N/A	8.1	<2	0.8525	0	0	0	0
Turbidity	NTU	NTU	164	104	2.62	0.80	0.15	0.105	0.45	0.35	0.03

Table A.B.4. 2015 IEUA – RP1 – Summary of treated municipal wastewater quality after each treatment process (Adham and Trussell, 2001; U.S. EPA, 2004; Xia et al., 2004; García-Figueruelo et al., 2009; Norouzbahari et al., 2009; GWRS, 2014; IEUA, 2013;2014; Price, 2016)

Water parameter	Symbol	Unit	Plant Influent	Primary effluent	Secondary Effluent	Deep Filtration	MBR permeate	MF permeate	UF permeate	NF permeate	RO permeate
SALINITY											
Salt Content											
Electrical Conductivity	EC_W	dS/m	0.9511	*0.9511*	*0.9511*	0.828	*0.928274*	*0.9511*	0.928274	0.646748	*0.057066*
(or)											
Total Dissolved Solids	TDS	mg/L	510.04	*510.04*	*504*	504	510.04	504	397.3032	213.192	*30.24*
Cations and Anions											
Calcium	Ca^{++}	mg/L	55.17	*55.17*	*55.17*	47.58	*52.9632*	55.17	52.9632	23.88861	*0.5517*
Magnesium	Mg^{++}	mg/L	10.36	*10.36*	*10.36*	8.833	*8.176112*	*10.36*	8.176112	1.9684	*0.1036*
Sodium	Na^{+}	mg/L	88.42	*88.42*	*88.42*	101.08	*88.42*	88.42	88.42	84.4411	*2.6526*
Carbonate	CO_3^{--}	mg/L	3	*3*	*3*	3	*2.3514*	*3*	2.3514		*0.03*
Bicarbonate	HCO_3^{-}	mg/L	363.7674	*363.7674*	*187.88*	187.88	*363.7674*	*187.88*	*187.88*	176.4193	*5.6364*
Chloride	Cl^{-}	mg/L	75.42	*75.42*	*75.42*	109.25	75.07404	75.42	75.42	75.42	2.2626
Sulfate	SO_4^{--}	mg/L	51.67	*51.67*	*51.67*	55.5	53.3781	51.67	51.67	0.5167	0.5167
NUTRIENTS											
Nitrate-Nitrogen	NO_3-N	mg/L	0.196	*0.196*	*6.314*	6.314	1.862	6.314	6.314	6.12458	0.25256
Ammonium-Nitrogen	NH_4-N	mg/L	35.77	*35.77*	*0.1*	0.1	0.193934	0.093	0.1	0.1	0.004
Phosphate-Phosphorus	PO_4-P	mg/L	*3.735*	*2.445*	*1.663*	*1.611*	2.150981	1.663	*1.663*	1.663	0.01663
Potassium	K^{+}	mg/L	16	*16*	*16*	15.08	*14.88*	*16*	14.88	14.4	*0.48*
OTHER											
Boron	B	mg/L	*0.2*	*0.2*	*0.2*	0.2	*0.2*	*0.2*	0.2	*0.2*	*0.01*
Sodium Adsorption Ratio	SAR	meq/L	2.86	2.86	2.86	3.526	2.98	2.86	2.98	4.45	0.86

Total Coliforms	TC	MPN/100ml	1.00E+09	1.00E+07	1.00E+06	<2 (7-day median)	0	1	0	<2	<2
Total Suspended Solid	TSS	mg/L	451.02	*0*	N/A	<2	0	0	0	0	0
Turbidity	NTU	NTU	164	*104*	2.62	0.6	0.15	0.105	0.45	0.35	0.03

Table A.B.5. Average individual treatment costs of different wastewater (US$2013)*
(Owen et al., 1995; Adham et al., 1996; U.S. EPA, 1999a;b; Soller et al., 2002; Gorenflo et al., 2003; Côté et al., 2004; Costa and de Pinho, 2006;Gómez et al., 2007; Leong et al., 2008; Al-Sahali et al., 2008; GWRS, 2010, 2014; Hernandez-Sancho et al., 2011)

Treatment Process	Large Plants (> 20000 m^3/d)
Activated Sludge	$0.25 ± 0.04
MBR	$0.61 ± 0.36
Granular Filtration	$0.19 ± 0.06
MF	$0.23 ± 0.09
UF	$0.28 ± 0.15
NF	$0.39 ± 0.09
RO	$0.37 ± 0.15
O_3	$0.03 ± 0.01
Cl_2	$0.02 ± 0.01
UV	$0.02 ± 0.02

*All values have been adjusted to 2013 U.S. Dollars using ENR CCI = 9547 and CPI = 697.836

Table A.B.6. IUEA 2011 and 2015 Data

Water parameter	Symbol	Unit	Plant Influent (IEUA) (mg/L)		Primary effluent (IEUA) (mg/L)		Secondary Effluent (IEUA) (mg/L)		Deep Filtration (IEUA) (mg/L)	
			2011	2015	2011	2015	2011	2015	2011	2015
SALINITY										
Salt Content										
Electrical Conductivity	EC_w	dS/m	0.8785	0.9511	0.8785	*0.9511*	0.8785	*0.9511*	0.768	0.828
(or)										
Total Dissolved Solids	TDS	mg/L	431.61	510.04	431.61	*510.04*	470	*504*	470	504
Cations and Anions										
Calcium	Ca^{++}	mg/L	48.58	55.17	48.58	*55.17*	48.58	*55.17*	41.16	47.58
Magnesium	Mg^{++}	mg/L	9.625	10.36	9.625	*10.36*	9.625	*10.36*	8.228	8.833
Sodium	Na^+	mg/L	76.17	88.42	76.17	*88.42*	76.17	*88.42*	88.72	101.08
Carbonate	CO_3^{--}	mg/L	3	3	3	*3*	3	*3*	3	3
Bicarbonate	HCO_3^-	mg/L	354.007	363.767	354.007	*363.767*	169.5312	*187.88*	169.5312	187.88
Chloride	Cl^-	mg/L	71.25	75.42	71.25	*75.42*	71.25	*75.42*	96.88	109.25
Sulphate	SO_4^{--}	mg/L	37.58	51.67	37.58	*51.67*	37.58	*51.67*	40.92	55.5
NUTRIENTS										
Nitrate-Nitrogen	NO_3-N	mg/L	0.253	0.196	0.253	*0.196*	6.311	*6.314*	6.311	6.314
Ammonium-Nitrogen	NH_4-N	mg/L	33.602	35.77	33.602	*35.77*	0.1016	*0.1*	0.1016	0.1
Phosphate-Phosphorus	PO_4-P	mg/L	2.923	*3.735*	1.416	*2.445*	1.22	*1.663*	0.861	*1.611*
Potassium	K^+	mg/L	17.42	16	17.42	*16*	17.42	*16*	14.64	15.08
MISCELLANEOUS										
Boron	B	mg/L	0.2	*0.2*	0.2	*0.2*	0.2	*0.2*	0.2	0.2
Sodium Adsorption Ratio	SAR*		2.61	2.86	2.61	*2.86*	2.61	*2.86*	3.29	3.52
Total Coliforms	TC	MPN/100ml	1E+09	1E+09	1E+07	1E+07	1E+06	1E+06	<2 (7-day median)	<2 (7-day median)
TSS	TSS	mg/L	544.09	451.02	N/A	*0*	N/A	N/A	<2	<2
Turbidity	NTU	NTU	164	164	104	*104*	2.62	2.62	0.8	0.6

$*SAR = \frac{Na}{\sqrt{(Ca+Mg)/2}}$ where Na, Ca, Mg are expressed in meq/L

Table A.B.7. FAO irrigation water quality guidelines
(Ayers and Westcot, 1985)

Potential Irrigation Problem				Unit	Degree of Restriction on Use		
					None	**Slight to Moderate**	**Severe**
Salinity (affects crop water availability)							
	EC_w			dS/m	< 0.7	0.7 – 3.0	> 3.0
	(or)						
	TDS			mg/l	< 450	450 – 2000	> 2000
Infiltration (affects infiltration rate of water into the soil. Evaluate using EC_w and SAR together)							
SAR*	= 0 – 3	**and EC_w**	=		> 0.7	0.7 – 0.2	< 0.2
	= 3 – 6		=		> 1.2	1.2 – 0.3	< 0.3
	= 6 – 12		=		> 1.9	1.9 – 0.5	< 0.5
	= 12 – 20		=		> 2.9	2.9 – 1.3	< 1.3
	= 20 – 40		=		> 5.0	5.0 – 2.9	< 2.9
Specific Ion Toxicity (affects sensitive crops)							
	Sodium (Na)						
	surface irrigation			me/l	< 3	3 – 9	> 9
	sprinkler irrigation			me/l	< 3	> 3	
	Chloride (Cl)						
	surface irrigation			me/l	< 4	4 – 10	> 10
	sprinkler irrigation			me/l	< 3	> 3	
	Boron (B)			mg/l	< 0.7	0.7 – 3.0	> 3.0
	Trace Elements						
Miscellaneous Effects (affects susceptible crops)							
	Nitrogen (NO_3-N)			mg/l	< 5	5 – 30	> 30
	Bicarbonate (HCO_3)						
	(overhead sprinkling only)			me/l	< 1.5	1.5 – 8.5	> 8.5
	pH				**Normal Range 6.5 – 8.4**		

$*SAR = \frac{Na}{\sqrt{(Ca+Mg)/2}}$ where Na, Ca, Mg are expressed in me/L

Table A.B.8. Parameters and cost of a full scale NF process (Costa and de Pinho, 2006)

Drinking water production capacity (m3/d)	100000
WRR (%)	90
Pressure (Pa)	6.00E+05
Total membrane area (m2)	159094
Pump efficiency (%)	70
Power demand (kWh/m3)	0.54
Membrane lifetime (years)	5
Cost of membrane (E/m2)	19
Membrane surface area (m2)	0.216
Energy Costs (Euro/kWh)	0.1
Amortization period (years)	15
Interest rate (%)	8
Capital cost (Euro)	
Pipes and valves	900607
Instruments and control	3530480
Tanks and frames	1633436
Miscellaneous	6536824
Pumps	236559
Membranes	2863686
Pressure vessels	1909124
Total capital costs	17610716
Operating costs (Euro/m3)	
Amortization[2]	0.067
Membrane replacement	0.017
Energy	0.048
Maintenance	0.01
Pretreatment	0.023
Chemicals	0.01
Concentrate disposal	0.037
Total treatment cost (Euro/m3)	0.214
Conversion 1Euro = $1.10 (2003)	$1.10
Total treatment cost ($/m3)	$0.24

[2] $A = P\frac{r(1+r)^n}{(1+r)^n-1}$ where A is the payment amount per period, P is the initial Principal (loan amount), r is the interest rate per period, and n is the total number of payments or periods.

Table A.B.9. Parameters and cost of a full scale MF process (Costa and de Pinho, 2006; GWRS, 2014)

Drinking water production capacity (m3/d)	265000
WRR (%)	90
Pressure (Pa)	6.00E+05
Total membrane area (m2)	626092.2
Pump efficiency (%)	70
Power demand (kWh/m3)	0.12225
Membrane lifetime (years)	5
Cost of membrane (E/m2)	13.74272
Energy Costs (Euro/kWh)	0.1
Amortization period (years)	15
Interest rate (%)	8
Capital cost (Euro)	
Pipes and valves	900607
Instruments and control	3530480
Tanks and frames	1633436
Miscellaneous	6536824
Pumps	236559
Membranes	8604206.501
Pressure vessels	1909124
Total capital costs	23351236.5
Operating costs (Euro/m3)	
Amortization	0.031338725
Membrane replacement	0.019767856
Energy	0.012225
Maintenance	0.01
Pretreatment	0.023
Chemicals	0.01
Concentrate disposal	0.037
Total treatment cost (Euro/m3)	0.143331581
Conversion 1Euro = $1.25520 (2009)	$1.26
Total treatment cost ($/m3)	$0.18

Table A.B.10. Parameters and cost of a full scale MBR process (Adham and Trussell, 2001)

Drinking water production capacity (m3/d)	3785.4
WRR (%)	90
Membrane surface area (m2)	0.93
Amortization period (years)	15
Interest rate (%)	8
Capital cost ($K)	
Headworks	834
MBR Process Costs	1750
MBR Tank	180
Operation-laboratory building	368
Maintenance Building	154
Subtotal ($K)	3286
Site Development, 15%	492.9
Installation of MF/MBR, 30%	525
Process Piping, 15%	492.9
Instrumentation, 2%	65.72
Electrical distribution and control, 16%	525.76
Electrical Service, 5%	164.3
Subtotal ($K)	5552.58
Contigency, 10%, $K	555.258
Total capital costs, $K	6107.838
Operating costs ($K/yr))	
Amortization ($/m3)	0.57384294
Personnel	85
Supervision-administration	31
Power	115
Spare Parts-replacement	6
Sludge Handling and Disposal	84
MBR chemicals	1
Maintenance Clean	2
Membrane Replacement	26
Total O&M cost in 1st year ($K/yr)	350
Total Estimated O&M cost, $K	$2,995.82
O&M cost ($/m3)	$0.28
Total treatment cost ($/m3)	$0.86

Table A.B.11. Parameters and cost of a full scale UF process (Al-Sahali et al., 2008; Gómez et al., 2007)

Drinking water production capacity (m3/d)	100000
WRR (%)	90
Power demand (kWh/m3)	0.31
Amortization period (years)	15
Interest rate (%)	0.08
Capital cost ($)	
Total capital costs	$5,520,000.00
Operating costs ($/m3)	
Amortization	$0.019631631
O&M	$0.17
Total treatment cost ($/m3) (2002)	$0.189

Table A.B.12. Parameters and cost of a full scale RO process (Costa and de Pinho, 2006;Leong et al., 2008)

Drinking water production capacity (m3/d)	265000
WRR (%)	90
Pressure (Pa)	6.00E+05
Pump efficiency (%)	70
Power demand (kWh/m3)	0.75
Membrane lifetime (years)	5
Energy Costs (Euro/kWh)	0.1
Amortization period (years)	15
Interest rate (%)	8
Capital cost (Euro)	
Pipes and valves	900607
Instruments and control	3530480
Tanks and frames	1633436
Miscellaneous	6536824
Pumps	236559
Membranes	5736137.67
Pressure vessels	1909124
Total capital costs	20483167.67
Operating costs (Euro/m3)	
Amortization	0.027489609
Membrane replacement	0.013178571
Energy	0.075
Maintenance	0.01
Pretreatment	0.023
Chemicals	0.01
Concentrate disposal	0.037
Total treatment cost (Euro/m3)	0.195668179
Conversion 1Euro = $1.25520 (2009)	$1.26
Total treatment cost ($/m3)	$0.246

Table A.B.13. Treatment cost of a full scale Ozonation process (Côté et al., 2004)

Drinking water production capacity (m3/d)	378540
Amortization period (years)	15
Interest rate (%)	8
Capital cost ($)	
Total capital costs	$18,000,000.00
Operating costs ($/m3)	
Amortization	$0.016911341
O&M	$0.01
Total treatment cost ($/m3)	$0.028

Table A.B.14. Treatment cost of a full scale UV process (Leong et al., 2008)

Drinking water production capacity (m3/d)	94635
Amortization period (years)	15
Interest rate (%)	8
Capital cost ($)	
Total capital costs	$6,421,000.00
Total O&M costs	$315000
Operating costs ($/m3)	
Amortization	0.024130604
O&M	0.010132658
Total treatment cost ($/m3)	$0.034

Table A.B.15. Treatment cost of a full scale Chlorine disinfection process (20 mg/L) (U.S. EPA, 1999a)

Drinking water production capacity (m3/d)	75708
WRR (%)	90
Amortization period (years)	15
Interest rate (%)	8
Capital cost ($)	
Total capital costs	$3,949,000.00
Total O&M costs	$379100
Operating costs ($/m3)	
Amortization	0.018550801
O&M	0.015243217
Total treatment cost ($/m3)	$0.034

Table A.B.16. Treatment costs of different wastewater treatment processes

Treatment Process	Plant capacity	Unit	Capital Unit Cost ($/m^3)	O&M Unit Cost ($/m^3)	Unit cost ($/m^3)	Year	CCI	CPI	PPI	References
Activated sludge	3800	m^3/d			$0.40	2004	7115	565.8	148.5	(Côté et al., 2004)
Activated sludge	6000	m^3/d			$0.34	2001	6342	530.4	140.7	(U.S. EPA, 2004)
Activated sludge	18927.05	m^3/d	$0.25	$0.14	$0.38	1996	5622	469.9		(Asano, 1998)
Activated sludge	19000	m^3/d			$0.25	2004	7115	565.8	148.5	(Côté et al., 2004)
Activated sludge	37854.1	m^3/d	$0.21	$0.12	$0.34	1996	5622	469.9		(Asano, 1998)
Activated sludge	38000	m^3/d			$0.20	2004	7115	565.8	148.5	(Côté et al., 2004)
Activated sludge	76000	m^3/d			$0.17	2004	7115	565.8	148.5	(Côté et al., 2004)
Activated sludge	2500	Mm^3/yr			$0.22	2003	6695	551.1	143.3	(Hernandez-Sancho et al., 2011)
MBR	1500	m^3/d	$0.42	$0.17	$0.59	2000	6221	515.8		(Visvanathan et al., 2000)
MBR	3000	m^3/d	$1.24	$0.13	$1.37	2015		709.998		(Taheran et al., 2016)
MBR	3785.4	m^3/d			$0.54	2004	7115	565.8	148.5	(Guo et al., 2014)
MBR	3785.4	m^3/d	$0.57	$0.28	$0.86	2001	6342	530.4	140.7	(2001)
MBR	3785.4	m^3/d			$0.61	2006	7751	603.9		(DeCarolis et al., 2007)
MBR	3800	m^3/d			$0.42	2004	7115	565.8	148.5	(Côté et al., 2004)
MBR	19000	m^3/d	$0.94	$0.02	$0.96	2012	9299	687.761		(Guo et al., 2014)
MBR	20000	m^3/d			$0.50	2006	7751	603.9		(DeCarolis et al., 2007)
MBR	38000	m^3/d			$0.22	2004	7115	565.8	148.5	(Côté et al., 2004)
Tertiary	456	m^3/d			$1.29	2008	8310	644.951		(Berbeka et al., 2012)
Tertiary	3044	m^3/d			$1.00	2008	8310	644.951		(Berbeka et al.,

										2012)
Tertiary	75000	m^3/d	$0.25	$0.38	$0.62	2005	7446	585		(Côté et al., 2005)
Tertiary	378541	m^3/d			$0.10	1994	5408	444		(Asano, 1998)
Tertiary	378541	m^3/d	$0.03	$0.06	$0.09	1991	4835	408		(Asano, 1998)
Tertiary	6000	Mm^3/yr			$0.20	2003	6695	551.1	143.3	(Hernandez-Sancho et al., 2011)
MF	3785.41	m^3/d			$0.60	2006	7751	603.9		(Messalem, 2006)
MF	3800	m^3/d			$0.15	1996	5622	469.9		(Adham et al., 1996)
MF	5000	m^3/d	$0.03	$0.13	$0.16	2000	6221	515.8	138	(Soller et al., 2002)
MF	23000	m^3/d			$0.09	1996	5622	469.9		(Adham et al., 1996)
MF	37854.1	m^3/d			$0.30	2006	7751	603.9		(Messalem, 2006)
MF	265000	m^3/d	$0.04	$0.14	$0.18	2009	8570	642.658	172.5	(Costa and de Pinho, 2006; GWRS, 2014)
MF	378541	m^3/d			$0.20	2006	7751	603.9		(Messalem, 2006)
UF	378	m^3/d			$0.45	1994	5408	444		(Wiesner et al., 1994)
UF	3780	m^3/d			$0.25	1994	5408	444		(Wiesner et al., 1994)
UF	3785.41	m^3/d			$0.75	2006	7751	603.9		(Messalem, 2006)
UF	3800	m^3/d			$0.20	1996	5622	469.9		(Adham et al., 1996)
UF	20000	m^3/d	$0.05	$0.17	$0.22	2007	7967	621.106	166.6	(Gómez et al., 2007)
UF	23000	m^3/d			$0.13	1996	5622	469.9		(Adham et al., 1996)
UF	37800	m^3/d			$0.20	1994	5408	444		(Wiesner et al., 1994)
UF	37854.1	m^3/d			$0.48	2006	7751	603.9		(Messalem, 2006)

UF	100000	m^3/d	$0.02	$0.17	$0.19	2002	7751	603.9	138.9	(Al-Sahali et al., 2008; Gómez et al., 2007)
UF	378541	m^3/d			$0.25	2006	7751	603.9		(Messalem, 2006)
UF	378541	m^3/d			$0.09	1994	5408	444		(Asano, 1998)
NF	3000	m^3/d	$1.03	$0.57	$1.60	2015		709.998		(Taheran et al., 2016)
NF	3785.41	m^3/d	$0.20	$0.53	$0.73	1995	5471	456.5	127.9	(Bergman, 1996)
NF	10200	m^3/d	$0.20	$0.26	$0.47	1993	5210	432.7		(Shaalan et al., 2014)
NF	16300	m^3/d	$0.20	$0.26	$0.46	1996	5622	469.9		(Shaalan et al., 2014)
NF	18000	m^3/d	$0.19	$0.24	$0.44	2010	8799	653.198		(Shaalan et al., 2014)
NF	18927.05	m^3/d	$0.11	$0.18	$0.28	1995	5471	456.5	127.9	(Bergman, 1996)
NF	20000	m^3/d			$0.27	2002	7751	603.9	138.9	(Gorenflo et al., 2003)
NF	37854.1	m^3/d	$0.08	$0.14	$0.22	1995	5471	456.5	127.9	(Bergman, 1996)
NF	50000	m^3/d	$0.15	$0.21	$0.36	2008	8310	644.951		(Shaalan et al., 2014)
NF	53000	m^3/d			$0.25	2003	6695	551.1	143.3	(Costa and de Pinho, 2006)
NF	65830	m^3/d	$0.15	$0.21	$0.36	1996	5622	469.9		(Shaalan et al., 2014)
NF	82650	m^3/d	$0.14	$0.21	$0.35	1996	5622	469.9		(Shaalan et al., 2014)
NF	94625	m^3/d	$0.13	$0.21	$0.34	1996	5622	469.9		(Shaalan et al., 2014)
NF	100000	m^3/d	$0.13	$0.20	$0.32	2006	7751	603.9		(Shaalan et al., 2014)
NF	100000	m^3/d	$0.07	$0.17	$0.24	2003	6695	551.1	143.3	(Costa and de Pinho, 2006)
NF	123000	m^3/d	$0.13	$0.19	$0.32	2008	8310	644.951		(Shaalan et al., 2014)
NF	132650	m^3/d	0.12	$0.18	$0.31	2006	7751	603.9		(Shaalan et al., 2014)

NF	150000	m^3/d	\$0.12	\$0.17	\$0.30	2008	8310	644.951		(Shaalan et al., 2014)
NF	171300	m^3/d	\$0.12	\$0.17	\$0.29	2005	7446	585		(Shaalan et al., 2014)
NF	193750	m^3/d	\$0.10	\$0.16	\$0.26	2005	7446	585		(Shaalan et al., 2014)
NF	567851.15	m^3/d	\$0.08	\$0.13	\$0.20	1995	5471	456.5	127.9	(Bergman, 1996)
RO	90	m^3/d			\$0.65	2015		709.998		(Piemonte et al., 2015)
RO	91.2	m^3/d			\$1.09	2015		709.998		(Piemonte et al., 2015)
RO	3000	m^3/d	\$1.02	\$0.72	\$1.74	2015		709.998		(Taheran et al., 2016)
RO	38000	m^3/d	\$0.07	\$0.21	\$0.28	2004	7115	565.8	148.5	(Côté et al., 2004)
RO	265000	m^3/d	\$0.07	\$0.21	\$0.25	2009	8570	642.658	172.5	(Costa and de Pinho, 2006;Leong et al., 2008)
RO	378541	m^3/d			\$0.15	1994	5408	444		(Asano, 1998)
RO	378541	m^3/d	\$0.12	\$0.19	\$0.31	1991	4835	408		(Asano, 1998)
O3	1500	m^3/d			\$0.04	1995	5471	456.5	127.9	(Owen et al., 1995)
O3	6000	m^3/d			\$0.02	1995	5471	456.5	127.9	(Owen et al., 1995)
O3	15000	m^3/d			\$0.02	1995	5471	456.5	127.9	(Owen et al., 1995)
O3	30000	m^3/d			\$0.01	1995	5471	456.5	127.9	(Owen et al., 1995)
O3	30000	m^3/d			\$0.02	1995	5471	456.5	127.9	(Owen et al., 1995)
O3	378540	m^3/d	\$0.02	\$0.01	\$0.03	2007	7967	621.106	166.6	(Côté et al., 2004;Leong et al., 2008)
O3	378541	m^3/d			\$0.02	1994	5408	444		(Asano, 1998)
O3	378541	m^3/d	\$0.00	\$0.02	\$0.02	1991	4835	408		(Asano, 1998)
Cl2	1500	m^3/d			\$0.01	1995	5471	456.5	127.9	(Owen et al.,

										1995)
Cl2	6000	m^3/d			\$0.01	1995	5471	456.5	127.9	(Owen et al., 1995)
Cl2	15000	m^3/d			\$0.01	1995	5471	456.5	127.9	(Owen et al., 1995)
Cl2	30000	m^3/d			\$0.00	1995	5471	456.5	127.9	(Owen et al., 1995)
Cl2 10mg/L	75708	m^3/d	\$0.02	\$0.01	\$0.03	1999	6060	499	133	(U.S. EPA, 1999a)
Cl2 10mg/L	94635	m^3/d	\$0.01	\$0.01	\$0.02	2007	7967	621.106	166.6	(Leong et al., 2008)
Cl2 gas 10 mg/L	94635	m^3/d	\$0.01	\$0.004	\$0.01	2007	7967	621.106	166.6	(Leong et al., 2008)
Cl2 20mg/L	75708	m^3/d	\$0.02	\$0.015	\$0.035	1999	6060	499	133	(U.S. EPA, 1999a)
UV	1500	m^3/d			\$0.01	1995	5471	456.5	127.9	(Owen et al., 1995)
UV	6000	m^3/d			\$0.01	1995	5471	456.5	127.9	(Owen et al., 1995)
UV	15000	m^3/d			\$0.01	1995	5471	456.5	127.9	(Owen et al., 1995)
UV	30000	m^3/d			\$0.01	1995	5471	456.5	127.9	(Owen et al., 1995)
UV	94635	m^3/d	\$0.02	\$0.01	\$0.03	2007	7967	621.106	166.6	(Leong et al., 2008)
UV - 80 mJ/cm2	113562	m^3/d			\$0.01	1999	6060	499	133	(U.S. EPA, 1999b)
UV	265000	m^3/d	\$0.001	\$0.03	\$0.03	2009	8570	642.658	172.5	(GWRS, 2014)
UV-100 mJ/cm2	2400	m^3/d			\$0.02	2002	7751	603.9	138.9	(Liberti et al., 2003)
UV-160 mJ/cm2	2400	m^3/d			\$0.04	2002	7751	603.9	138.9	(Liberti et al., 2003)
Groundwater (without subsidies)	265000	m^3/d	\$0.37	\$0.34	\$0.71	2010	8799	653.198	179.8	(GWRS, 2010)

Table A.B.17. Average individual treatment costs of different wastewater processes with time adjustment (2013 unit cost)

Treatment Process	Plant capacity	Unit	2013 Capital Cost	2013 O&M Cost	2013 Capital Unit Cost[a]	2013 O&M Unit Cost[b]	2013 Unit cost	Average 2013 Unit cost
Activated sludge	3800	m^3/d					$0.49	$0.48
Activated sludge	6000	m^3/d					$0.45	
Activated sludge	18927.05	m^3/d			$0.42	$0.20	$0.62	
Activated sludge	19000	m^3/d					$0.31	
Activated sludge	37854.1	m^3/d			$0.36	$0.19	$0.55	$0.32
Activated sludge	38000	m^3/d					$0.25	
Activated sludge	76000	m^3/d					$0.21	
Activated sludge	2500	Mm^3/yr					$0.28	
MBR	1500	m^3/d			$0.64	$0.23	$0.87	$0.89
MBR	3000	m^3/d					$1.35	
MBR	3785.4	m^3/d					$0.66	
MBR	3785.4	m^3/d	$9,194,501.64	$3,941,533.43	$0.86	$0.37	$1.23	
MBR	3785.4	m^3/d					$0.70	
MBR	3800	m^3/d					$0.52	
MBR	19000	m^3/d			$0.96	$0.02	$0.98	$0.61
MBR	20000	m^3/d					$0.58	
MBR	38000	m^3/d					$0.27	
Tertiary	456	m^3/d					$1.39	$1.24
Tertiary	3044	m^3/d					$1.09	
Tertiary	75000	m^3/d			$0.32	$0.45	$0.77	$0.33
Tertiary	378541	m^3/d					$0.15	
Tertiary	378541	m^3/d			$0.06	$0.09	$0.16	
Tertiary	6000	Mm^3/yr					$0.25	
MF	3785.41	m^3/d					$0.69	$0.38

MF	3800	m^3/d					$0.22	
MF	5000	m^3/d			$0.05	$0.17	$0.22	
MF	23000	m^3/d					$0.13	$0.23
MF	37854.1	m^3/d					$0.35	
MF	265000	m^3/d	$32,651,934.27	$13,287,948.95	$0.04	$0.15	$0.20	
MF	378541	m^3/d					$0.23	
UF	378	m^3/d					$0.71	$0.50
UF	3780	m^3/d					$0.39	
UF	3785.41	m^3/d					$0.87	
UF	3800	m^3/d					$0.30	
UF	20000	m^3/d	$32,651,934.27	$1,251,284.12	$0.06	$0.19	$0.25	
UF	23000	m^3/d					$0.19	$0.28
UF	37800	m^3/d					$0.31	
UF	37854.1	m^3/d					$0.55	
UF	100000	m^3/d	$6,799,050.45	$6,434,675.25	$0.02	$0.20	$0.22	
UF	378541	m^3/d					$0.29	
UF	378541	m^3/d					$0.14	
NF	3000	m^3/d					$1.57	$0.78
NF	3785.41	m^3/d			$0.35	$0.81	$1.16	
NF	10200	m^3/d			$0.37	$0.43	$0.80	
NF	16300	m^3/d			$0.34	$0.38	$0.73	
NF	18000	m^3/d			$0.21	$0.26	$0.47	
NF	18927.05	m^3/d			$0.18	$0.27	$0.45	
NF	20000	m^3/d					$0.31	
NF	37854.1	m^3/d			$0.14	$0.21	$0.35	$0.39
NF	50000	m^3/d			$0.17	$0.23	$0.40	

NF	53000	m^3/d					$0.32	
NF	65830	m^3/d			$0.25	$0.31	$0.56	
NF	82650	m^3/d			$0.24	$0.31	$0.55	
NF	94625	m^3/d			$0.23	$0.30	$0.53	
NF	100000	m^3/d			$0.15	$0.23	$0.38	
NF	100000	m^3/d	$27,873,529.00	$6,973,425.28	$0.10	$0.21	$0.31	
NF	123000	m^3/d			$0.14	$0.21	$0.35	
NF	132650	m^3/d			$0.15	$0.21	$0.36	
NF	150000	m^3/d			$0.14	$0.19	$0.33	
NF	171300	m^3/d			$0.15	$0.20	$0.35	
NF	193750	m^3/d			$0.13	$0.19	$0.32	
NF	567851.15	m^3/d			$0.13	$0.19	$0.32	
RO	90	m^3/d					$0.64	
RO	91.2	m^3/d					$1.07	$1.14
RO	3000	m^3/d					$1.71	
RO	38000	m^3/d	$24,576,648.21	$3,233,171.13	$0.09	$0.26	$0.35	
RO	265000	m^3/d	$28,641,525.87	$19,954,382.27	$0.08	$0.23	$0.31	$0.37
RO	378541	m^3/d					$0.23	
RO	378541	m^3/d			$0.24	$0.33	$0.57	
O3	1500	m^3/d					$0.06	
O3	6000	m^3/d					$0.04	$0.04
O3	15000	m^3/d					$0.03	
O3	30000	m^3/d					$0.02	
O3	30000	m^3/d					$0.02	$0.03
O3	378540	m^3/d	$21,569,725.12	$1,491,087.17	$0.02	$0.01	$0.03	

O3	378541	m^3/d					$0.04	
O3	378541	m^3/d			$0.00	$0.03	$0.04	
Cl2	1500	m^3/d					$0.02	
Cl2	6000	m^3/d					$0.01	$0.01
Cl2	15000	m^3/d					$0.01	
Cl2	30000	m^3/d					$0.01	
Cl2 10mg/L	75708	m^3/d	$5,632,099.83	$317,032.91	$0.03	$0.01	$0.04	$0.02
Cl2 10mg/L	94635	m^3/d	$1,743,552.78	$371,890.98	$0.01	$0.01	$0.02	
Cl2 gas 10 mg/L	94635	m^3/d	$2,547,624.20	$123,589.15	$0.01	$0.004	$0.01	
Cl2 20mg/L	75708	m^3/d	$6,221,304.13	$530,159.57	$0.03	$0.02	$0.05	$0.05
UV	1500	m^3/d					$0.02	
UV	6000	m^3/d					$0.01	$0.01
UV	15000	m^3/d					$0.01	
UV	30000	m^3/d					$0.01	
UV	94635	m^3/d	$7,694,400.28	$353,914.37	$0.03	$0.01	$0.04	
UV - 80 mJ/cm^2	113562	m^3/d					$0.01	$0.02
UV	265000	m^3/d	$445,600.93	$2,594,333.68	$0.001	$0.03	$0.03	
UV-100 mJ/cm2	2400	m^3/d					$0.02	$0.03
UV-160 mJ/cm2	2400	m^3/d					$0.04	
Groundwater (without subsidies)	265000	m^3/d			$0.40	$0.37	$0.77	$0.77

a: All values have been adjusted to 2013 values using ENR CCI = 9547

b: All values have been adjusted to 2013 values using CPI = 697.836

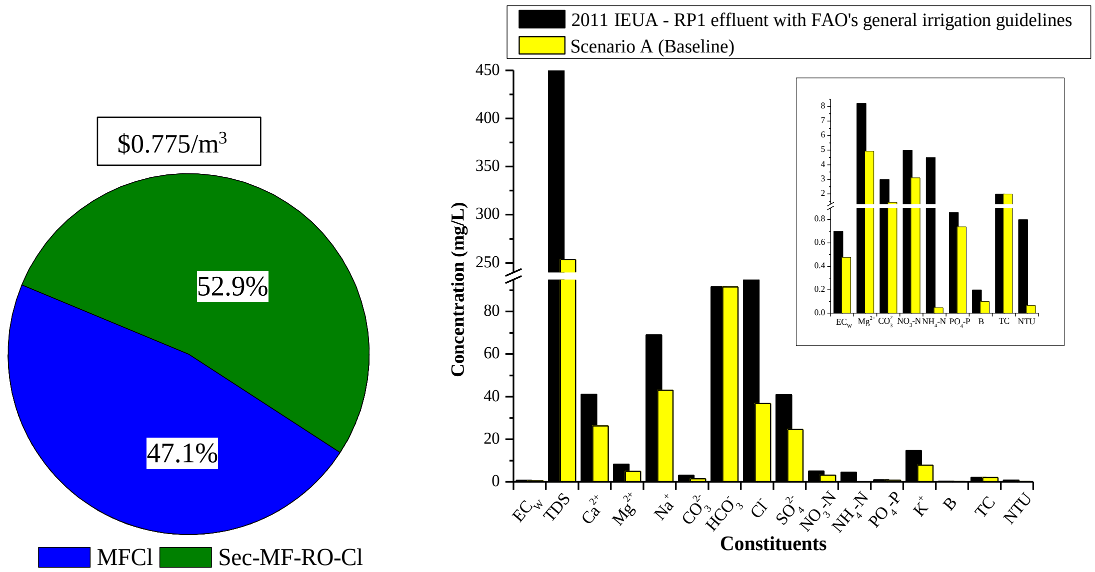

Figure A.B.F1. Cost-effective solution for scenario A

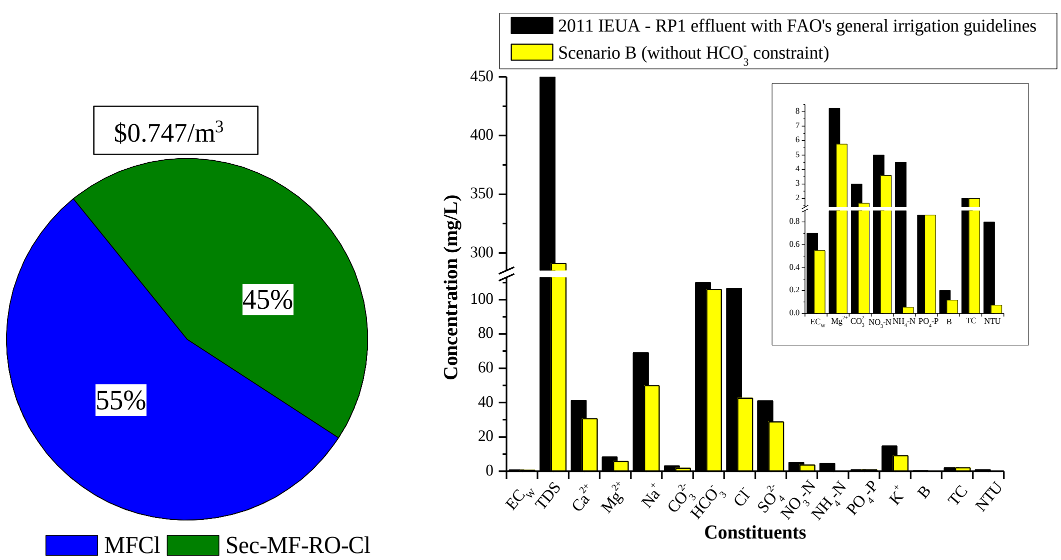

Figure A.B.F2. Cost-effective solution for scenario B (without HCO3- constraint)

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Appendix C:

Supporting Information for Chapter 4

Development of the Model

The water balance (i.e., water usage) at an entity is defined by Equation 1 (total supplies must equal total demand in acre-feet per year):

$$GW_1(t) + IW_1(t) + SW_1(t) + DW_1(t) = TR_1'(t) + Irr_1(t) + CII_1(t) + LI_1(t) \quad (3)$$

where, on the supply side $GW_1(t)$ is groundwater extraction, $IW_1(t)$ is import water, $SW_1(t)$ is surface water, $DW_1(t)$ is desalinated seawater; on the demand side, $TR_1'(t)$ is wastewater generated by residential indoor usage, $Irr_1(t)$ is groundwater for commercial irrigation, $CII_1(t)$ is commercial, industrial, and institutional (CII) water use, and $LI_1(t)$ is residential outdoor water usage. $LI_1(t)$ values range between 50-60%[93-97] of the total freshwater needed for residential usage, and residential water usage is estimated to be 98.4 gallons per capita per day ($GPCD_1$) in 2015[98]. The total residential indoor and outdoor water use is based on the total population ($EstPOP_1(t)$) and average residential water usage for a given entity (Equation 2):

$$TR_1'(t) + LI_1(t) = GPCD_1 \times EstPOP_1(t) \quad (4)$$

Commercial irrigation ($TotIrr_1(t)$) is separated from the $CII_1(t)$ for a simpler calculations[98]. Total commercial irrigation water can come from 2 different water sources – groundwater and disinfected tertiary treated recycled water ($RecIrr_1(t)$) (Equation 3):

$$TotIrr_1(t) = Irr_1(t) + RecIrr_1(t) \quad (5)$$

$CII_1(t)$ is comprised of both indoor ($InCII_1(t)$) and outdoor ($OutCII_1(t)$) water usage (Equation 4), in which $InCII_1(t)$ is approximately two third of the total $CII_1(t)$ [84] (Equation 5):

$$CII_1(t) = InCII_1(t) + OutCII_1(t) \quad (6)$$

$$InCII_1(t) = \frac{2}{3} \times CII_1(t) \tag{7}$$

A part of the residential and commercial irrigation water infiltrates into the groundwater system ($IG_1'(t)$) (where it recharges the basin), where the percent of infiltrated water (X_1) is the deep infiltration rate (estimated at 2%[101]), while another part is lost to evaporation (Equation 6). The residual commercial irrigation water ($IG_1(t)$) is used to irrigate crops (Equation 7):

$$IG_1'(t) = \frac{X_1}{100} \times LI_1(t) \tag{8}$$

$$IG_1(t) = \frac{X_1}{100} \times TotIrr_1(t) \tag{9}$$

$$IGCII_1(t) = \frac{X_1}{100} \times OutCII_1(t) \tag{10}$$

$$Irr_1(t) + RecIrr_1(t) = IG_1(t) + CU_1(t) \tag{11}$$

where $IG_1'(t)$ is the outdoor residential irrigation infiltration, $IG_1(t)$ is the agricultural irrigation infiltration, and $IGCII_1(t)$ is the outdoor CII sector infiltration at entity 1 at time $t. X_1$% of the outdoor commercial, industrial and institution water supply ($OutCII_1(t)$) infiltrates into the ground to recharge the groundwater $IGCII_1(t\ )$ (Equation 8). The potable water and recycled water supplies make up the total irrigation water for the entity. Most of this source is delivered to crop ($CU_1(t)$) while a small portion infiltrates into the ground for recharge $IG_1(t)$ (Equation 9).

Wastewater from indoor residential water use ($TR_1'(t)$) is combined with indoor water use from the CII sector ($InCII_1(t)$), for a total wastewater volume ($TR_1(t)$) (Equation 10). This wastewater is treated using a secondary treatment process (i.e., activated sludge). Here, treated wastewater is either disinfected and discharged to a receiving water body ($SD_1(t)$) (SAR in the case of the Bunker Hill Basin) or further treated using a tertiary process ($Rec_1(t)$) or an advanced treatment process (Microfiltration (MF) – Reverse Osmosis (RO) – Ultraviolet Disinfection (UV)) to recharge the groundwater basin via artificial groundwater recharge ($RW_1'(t)$) (Equation 11). In this model, wastewater that underwent a desalination ($RW_1'(t)$) is distinguished from tertiary treated wastewater ($Rec_1(t)$) because of its lower salt load. Tertiary-treated water can be used either for commercial irrigation or for recharging the groundwater basin ($RecGW_1(t)$) (Equation 12).

$$TR_1(t) = TR_1'(t) + InCII_1(t) \tag{12}$$

$$TR_1(t) = RW_1'(t) + SD_1(t) + Rec_1(t) \tag{13}$$

where $TR_1'(t)$ and $InCII_1(t)$ are the wastewater generated from residential indoor water usage and CII indoor usage at entity 1, respectively. $TR_1(t)$ is the total secondary treated wastewater from the total indoor water usage. $RW_1'(t)$ is the advanced treated wastewater for recharge via MF, RO and disinfected with UV. $SD_1(t)$ is the environmental flow discharge to SAR and $Rec_1(t)$ is the disinfected tertiary treated wastewater for reuse ($RecIrr_1(t)$ – recycled water for commercial irrigation) and recharge ($RecGW_1(t)$ – recycled water for artificial groundwater recharge) purpose:

$$Rec_1(t) = RecIrr_1(t) + RecGW_1(t) \tag{14}$$

The ratio of population of entity 1 over the population of other groundwater basin pumpers along the aquifer is estimated based on population. For example, the population of entity 1 is 199,634 people and is about 26% of the total population sharing the same basin (777,926 people). This value is designated as $SanB$ ratio.

Artificial groundwater recharge ($Rchrg_1(t)$) is comprised of recycled water

($RecGW_1(t)$) and imported water ($RecIW_1(t)$). Typically, there are multiple entities that extract water from a given aquifer. However, these extracting entities are not necessarily those that treat the wastewater. In fact, due to economies of scale, wastewater treatment plants often treat the wastewater generated by water provided from several extraction entities. Ultimately, the volume of recycled water generated by a treatment plant is determined by the size and habits of the population it serves. To map treated wastewater available for recharge to the needs of a particular extracting entity (which may only need a fraction of the recycled water meet a smaller population's demand), the total volume available for recharge to an extraction entity is normalized by the % volume of the total wastewater from a given aquifer treated by a particular treatment plant (Equation 13).

produced is subjected to the population within the area. For instance, the wastewater treatment plant treating the wastewater from this entity 1 also treats wastewater coming from 2 other entities. The population of entity 1 is estimated at 199,634 people whereas it is 23,751 and 101,733 people at entity 2 and 3, respectively. Therefore, the ratio of the wastewater generated from entity 1 compare to entity 2 and 3 is 61% based on population. The total wastewater being treated at the wastewater treatment plant is approximately 33 MGD. There are 2 other wastewater treatment plants treating wastewater from other entities along this aquifer. The total wastewater treated from these 3 wastewater treatment plants is 47 MGD. Thus, the total wastewater produced by entity 1 is about 43% of the total wastewater treated. This number is designated by $RSanB$ ratio. Since most of the variables are designated for entity 1, the total artificial recharge is supposed to scale to a regional level (all entities sharing the same groundwater basin):

$$Rchrg_1(t) = \frac{(RecGW_1 + RW_1'(t))}{RSanB} + RecIW_1(t) \quad (15)$$

where $RecIW_1(t)$ is the total untreated SWP used to recharge the groundwater basin. The cost of this source is also based on the population ratio. Entity with higher population will extract more groundwater to supply its residential and commercial needs; thus, contribute to a bigger portion of this recharge cost. The total cost of recharge at each entity ($Rcost_1(t)$) equals the cost of a portion of untreated SWP and recycled water for recharge purpose:

$$Rcost_1(t) = SanB \times sUIW_1 \times RecIW_1(t) + sRecGW_1 \times RecGW_1(t) + sRW_1' \times RW_1'(t) \quad (16)$$

The cost of groundwater extraction ($sGW_1(t)$) is defined as (Equation 15):

$$\begin{aligned} sGW_1(t) = SanB \times (AGWC \times BHtotGWall_1(t) \\ + PELEC \times \left(h_{land} - h(t)\right) \times BHtotGWall_1(t) \\ + \frac{PELEC}{gwarea \times spyld} \times \frac{(BHtotGWall_1)^2}{2} \end{aligned} \quad (17)$$

where $AGWC$ is the average pumping cost coefficient in U.S. Dollars related to equipment use, $BHtotGWall_1$ is the volume of extracted groundwater, $PELEC$ is the electricity cost (per unit of elevation (including drawdown), per unit of volume), h_{land} is the height of the surface (above sea level), and $h(t)$ is the height of the groundwater aquifer (relative to sea level) at time t. The height if the aquifer is bounded by the lower (h_{min}) and upper (h_{max}) bounds as determined by the aquifer geometry. $gwarea$ is the aquifer surface area, and $spyld$ is the specific yield of the aquifer[102]. Well and pump equipment costs and drawdown were obtained from the city of Riverside[103] and the energy costs were obtained from the city of San Bernardino's pumping costs[3]. All costs were adjusted to 2015 U.S. Dollars.

The unit cost of irrigation water ($sIrr_1(t)$) and CII sector ($sCII_1(t)$) using groundwater source were obtained from the city of San Bernardino water rates[104], and were calculated by averaging the cost of supplied water, where sDW_1 is the unit cost of desalination water (Equation 16). Total irrigation water is generated from recycled water and groundwater; therefore, its unit cost ($sTotIrr_1(t)$) is determined by the volume-averaged cost of tertiary treated recycled water and groundwater. The city must treat wastewater coming into the plant before discharging it and must account for the cost of treatment. Therefore, the cost of recharging treated recycled water back to the groundwater basin and recycled water for irrigation is only the cost after secondary treatment process.

$$sIrr_1(t) = \frac{sGW_1(t) + GW_1(t) \times sTreat_1 + sIW_1 \times IW_1(t) + (sSW_1 + sTreat_1) \times SW_1(t) + sDW_1 \times DW_1(t)}{GW_1(t) + IW_1(t) + SW_1(t) + DW_1(t)} \quad (18)$$

The unit cost of extracted groundwater ($sGWunit_1(t)$) includes the energy cost to bring the water to the land surface and the treatment cost ($sTreat_1$) to deliver the water to its residential and commercial demands at entity 1:

$$sGWunit_1(t) = \frac{sGW_1(t) + GW_1(t) \times sTreat_1}{GW_1(t)} \quad (19)$$

$$sCII_1(t) = sIrr_1(t) \quad (20)$$

$$sTotIrr_1(t) = \frac{sIrr_1(t) \times Irr_1(t) + sRecIrr_1(t) \times RecIrr_1(t)}{TotIrr_1(t)} \quad (21)$$

where $sGWunit_1(t)$ is the unit cost of groundwater, $sTreat_1$ is the unit cost to treat groundwater/surface water/SWP to drinking standards, $sIrr_1(t)$ is the unit cost of commercial irrigation water using freshwater, $sRecIrr_1(t)$ is the unit cost of disinfected tertiary treated recycled water used for commercial irrigation, sIW_1 is the unit cost of treated SWP, and sSW_1 is the unit cost of surface water.

The unit cost of total commercial irrigation water ($sCU_1(t)$) is the same as the unit cost of water delivered to crops (Equation 20):

$$sCU_1(t) = sTotIrr_1(t) \tag{22}$$

The population size relying on the groundwater is subject to a linear increase with time with a rate ($POPrate_1$) of 0.8% (Equation 21).

$$EstPOP_1(t+1) = EstPOP_1(t) \times (1 + \frac{POPrate_1}{100}) \tag{23}$$

$$ResBHtotGW_1(t+1) = ResBHtotGW_1(t) \times (1 + \frac{POPrate_1}{100}) \tag{24}$$

where $EstPOP_1(t)$ is the estimated population at time t and $ResBHtotGW_1(t)$ is the total groundwater extraction for residential usage at time t.

Groundwater withdrawn from the basin can be used for municipal or agricultural purposes. In this paper, the residential water usage is subject to change as a function of population size. The groundwater extraction is directly proportional with the population in the service area. In other words, the ratio of groundwater extraction from entity 1 compared to the groundwater extractions from the entire basin is $SanB$:

$$BHtotGWall_1(t) = (\frac{1}{SanB}) \times GW_1(t) \tag{25}$$

Each entity in Southern California is entitled to an annual water allocation from the SWP, which depends on the availability of water. The total SWP (both treated and untreated) has to be less than the allocation for that year (Equation 24):

$$RecIW_1(t) + IW_1(t) \leq \frac{PercAlloSWP}{100} \times SWPAlloc \tag{26}$$

where $PercAlloSWP$ is the percent SWP allocation and $SWPAlloc$ is the total SWP that the entities are entitled to.

The salinity of the groundwater basin is an important issue that is constantly being monitored. The salinity of a groundwater can increase due to recharge of non-desalinated wastewater (wastewater typically has higher salinity), recharge with high salinity imported water (such as water from the CRA), or infiltration of irrigation water (Equation 25):

$$TDSRchrg_1(t+1) = (TDSRchrg_1(t) + \frac{TDSRecGW_1(t) \times RecGW_1(t) + TDSRW_1'(t) \times RW_1'(t) + TDSRecIW_1 \times RecIW_1(t) + TDSnrchg * nrchg + TDSIG_1 \times IG_1(t) + TDSIG_1'(t) \times IG_1'(t) + TDSIGCII_1(t) * IGCII_1(t)}{RecIW_1(t) + RecGW_1(t) + IG_1(t) + IG_1'(t) + IGCII_1(t) + RW_1' + nrchg})/2 \tag{27}$$

$$\frac{TDSRchrg_1(t) \times GW_1(t) + TDSRecIW_1(t) \times IW_1(t) + TDSSW_1(t) \times SW_1(t) + TDSDW_1(t) \times DW_1(t)}{GW_1(t) + IW_1(t) + SW_1(t) + DW_1(t)} \leq 500 \tag{28}$$

Where $TDSRchrg_1(t)$ is the total dissolved solids (TDS) in the groundwater basin at time t, $TDSRecGW_1(t)$ is the TDS of disinfected tertiary treated recycled water for used for recharge, $TDSRW_1'(t)$ is the TDS of RO treated wastewater for recharge, $TDSRecIW_1(t)$ is the TDS of untreated SWP used for recharge, $TDSnrchg(t)$ is the TDS of natural recharge groundwater, $nrchg$ is the annual 100-year average natural recharge from (based on predictions for 2000-2099), $TDSIG_1'(t)$ is the TDS of percolated groundwater from outdoor residential irrigation, $TDSIGCII_1(t)$ is the TDS of percolated groundwater from the CII sector, $TDSSW_1(t)$ is the TDS in the surface water, and $TDSDW_1(t)$ is the TDS in desalinated seawater. The TDS of the municipal supply is calculated by the average of TDS in the groundwater, import water, surface water and desalination water and has the threshold of 500 mg/L according to U.S.EPA for secondary drinking standards (Equation 26)[88].

The $TDSIG_1(t)$ is the TDS in the commercial irrigation water infiltration and is calculated based on its components:

$$TDSIG_1(t) = \frac{TDSIrr_1 \times Irr_1(t) + TDSRecIrr_1 \times RecIrr_1(t)}{TotIrr_1(t)} \tag{28}$$

The objective of this model is to minimize the total net present value (NPV) cost (C) of water supplied over a period of 100 years. The yearly cost of water supplied $SS(t)$ is defined by the total cost of groundwater extraction, imported water (e.g., SWP), surface water, desalinated water, wastewater treatment costs, and groundwater recharge costs with discount factors (Equation 28):

$$\begin{aligned} SS(t) = (&sGW_1(t) + GW_1(t) \times sTreat_1 + sIW_1 \times IW_1(t) \\ &+ (sSW_1 + sTreat_1) \times SW_1(t) + sDW_1 \times DW_1(t) + Rcost_1(t) \\ &+ sRecIrr_1 \times RecIrr_1(t) + sSD_1 \times SD_1(t)) \times df(t) \end{aligned} \tag{29}$$

The model will minimize the total cost of water supplied over 100 years (Equation 29):

$$C = \sum_{t=1}^{t=100} SS(t) \tag{30}$$

Model Inputs

Table A.C.1. Parameter Values[14-22, 39, 86, 87, 93-98, 104, 105]

Parameter	Description	Value
CAoutdr	Residential outdoor water use	60%
PercoEff	Efficiency of percolation ponds	90%
BHtotGW1	Total groundwater extraction from the basin at year 1	158028 AFY
Gwarea	Aquifer area	92488.23 acres
PELEC	Electricity cost for pumping groundwater	$0.213/AF-ft
AGWC	Pumping cost coefficient	$35.75
Hland	Elevation of surface	1200 ft
Hmax	Maximum water table height vs. msl	1080 ft
Hmin	Bottom of the aquifer vs. msl	-400 ft
Spyld	Specific yield of the basin	0.13
Swic	Surface water infiltration coefficient	0.4
R	Interest rate	0.05%
POPrate1	Population increase rate	0.8%
GPCD1	Water demand per person per day	98.42 GPCD
SWPAlloc	Imported water allocation	102600 AFY
SD1	Minimum discharge to SAR (to maintain flow in river)	16000 AFY
CII1o	Baseline commercial-industrial-institutional water demand	13342.49 AFY
TotIrr1o	Baseline total commercial irrigation water	12000 AFY
Nrchg45	Average natural recharge under RCP 4.5 over 100 years	68327.9 AFY
Stdev45	Standard deviation RCP 4.5 in 100 years	73646.4
Nrchgb45	Baseline natural recharge under RCP 4.5 in 50 years	81165.5 AFY

Stdevb45	Standard deviation of natural recharge under RCP 4.5 in 50 years	75391.5
ET45	Average ET rate of alfalfa under RCP 4.5 over 100 years	1611504.3 AFY
stdevET45	Standard deviation of ET for alfalfa under RCP 4.5 over 100 years	104419.9
ETo45	Baseline average ET rate of alfalfa under RCP 4.5 over 100 years	1510604.4 AFY
PercAlloSWP45	Average imported water allocation % under RCP 4.5	60%
Nrchg85	Average natural recharge Under RCP 8.5 over 100 years	79069.5 AFY
Stdev85	Standard deviation of natural recharge under RCP 8.5 over 100 years	81438.9
Nrchgb85	Baseline natural recharge under RCP 8.5 over 50 years	82129.9 AFY
Stdevb85	Baseline standard deviation of natural recharge under RCP8.5 over 50 years	79094.9
ET85	Average ET rate for alfalfa under RCP 8.5 over 100 years	1637490.3 AFY
stdevET85	Standard deviation for ET of alfalfa under RCP8.5 over 100 years	118738.7
ETo85	Baseline average ET rate of alfalfa under RCP8.5 in 50 years	1513142.5 AFY
PercAlloSWP85	Imported water allocation (%) under RCP 8.5	10%
TDSRecGW1	TDS of disinfected tertiary treated wastewater for groundwater recharge	500 mg/L
TDSRW1prime	TDS of advanced treated	64 mg/L

	wastewater for groundwater recharge	
TDSRecIW1	TDS of untreated imported water for groundwater recharge	294 mg/L
TDSDW1	TDS of desalinated seawater	182 mg/L
TDSnrchg	TDS of natural recharge infiltrating into the groundwater	300 mg/L
TDSIrr1	TDS of infiltrated water from commercial irrigation	331.27 mg/L
sIW1	Cost of treated imported water	$923/AF
sUIW1	Cost of untreated imported water	$582/AF
sDW1	Cost of desalinated seawater	$2,100/AF
sTreat1	Cost of treating groundwater and surface water to drinking water standards	$341/AF
sIGCII1	Cost of freshwater from outdoor CII infiltrate into groundwater	$0
GWRS1	cost of MF-RO-UV to treat secondary effluent	$510/AF
AS	cost of activated sludge process	$401/AF
GAC	Cost of granular filtration	$414/AF
Cl	Cot of chlorination disinfection	$25.10/AF

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