THESIS

ASSESSING TRADOFFS OF URBAN WATER DEMAND REDUCTION STRATEGIES

Submitted by

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In partial fulfillment of the requirements

For the Degree of Master of Science

Colorado State University

Fort Collins, Colorado

Fall 2019

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ABSTRACT

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In many cities across the World, traditional sources of potable water supply can become susceptible to shortage due to increased water demands from rapid urbanization and more frequent and extreme drought conditions. Understanding impacts of city-scale conservation and water reuse is important for water managers to implement cost effective water saving strategies and develop resilient municipal water systems. Innovative water reuse systems are becoming more cost effective, technologically viable and socially accepted. However, there is still a need for comparative assessment of alternative sources; graywater, stormwater and wastewater use along with indoor and outdoor conservation, implemented at the municipal scale.

This study applies the Integrated Urban Water Model (IUWM) to three U.S. cities; Denver, CO; Miami, FL; and Tucson, AZ. We assess the tradeoffs between cost and water savings for a range of solutions composed of up to three strategies; to understand interactions between strategies and their performance under the influence of local precipitation, population density and land cover. A global sensitivity analysis method was used to fit and test model parameters to historical water use in each city. Alternative source and conservation strategies available in IUWM were simulated to quantify annual water savings. Alternative source strategies simulate collection of graywater, stormwater and wastewater to supplement demands for toilet flushing, landscape irrigation and potable supply. A non-dominated sorting function was applied that minimizes annual demand and total annualized cost to identify optimal strategies.

Results show discrete strategy performance in demand reduction between cities influenced by local climate conditions, land cover and population density. Strategies that include use of stormwater can achieve highest demand reduction in Miami, where precipitation and impervious area is large resulting in larger generation of stormwater compared to other study cities. Indoor conservation was frequently part of

optimal solutions in Tucson, where indoor water use is higher per capita compared to other study cities. The top performing strategies overall in terms of water savings and total cost were found to be *efficient irrigation systems* and *stormwater for irrigation.* While use of stormwater achieves large demand reduction relative to other strategies, it only occurred in non-dominated solutions that were characterized by higher cost. This strategy can be very effective for demand reduction, but is also costly. On the contrary, *efficient irrigation systems* are frequently part of low-cost solutions across all three study cities.

Overall, this study introduces a framework for assessing cost and efficacy of water conservation and reuse strategies across regions. Results identify optimal strategies that can meet a range of demand reduction targets and stay within financial constraints.

ACKNOWLEDGEMENTS

Looking back, this MS program and research project has helped me develop personally and professionally and has increased my capacity to overcome challenges and maintain positivity in all that I do. I feel very fortunate to have studied in this department at Colorado State University, a world-renowned water resources program. I have yet to unearth the full value of this experience.

I am very grateful to my advisors; Mazdak Arabi and Sybil Sharvelle for giving me the opportunity to work on such a novel project as part of Urban Water Innovation Network consortium. Sybil, thank you for your constant support, feedback, and time as we solved problems and advanced this project together. You helped me remain positive and confident in my abilities to complete this work. We pushed the IUWM to uncharted territory and I am very proud of it. Mazdak, thank you for your guidance, good humor and friendship. Your feedback and support were critical for the advancements in this work. Thank you for being available to explain a difficult concept or just to chat. It's been a great time participating in UWIN together with such an incredible group of researchers and academics. I am very grateful to have had the opportunity to know and work with Andre Dozier. He helped me get this project going and it really wouldn't have been possible without his attention and generosity. He is missed dearly. I would like to thank Dr. Chris Goemans for participating in my committee. Thank you, Chris, for sharing your economic expertise and providing thorough review of my work. Thank you to my colleagues in the office.

Of most importance in my life is my family. I would like to thank my Mom, Dad and sister Julia for their love and support throughout this time. Dad, thank you for your feedback on my work and solid advice. Also, I am grateful to my Grandma, Christine for her love and support of my education. I want to thank all my friends for the good times, love and support throughout this long and bumpy road. I see a beach ahead, let's go.

This work was supported by the Agriculture and Food Research Initiative of the USDA National Institute of Food and Agriculture (NIFA) grant number #2012-67003-1990 and the National Science Foundation (NSF) SRN: Urban Water Innovation Network (U-WIN): Transitioning Towards Sustainable Urban Water Systems, grant number #1444758.

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CHAPTER 1: BACKGROUND AND MOTIVATION

1.1 Research Motivation

The freshwater supplies of many cities across the world are becoming susceptible to shortages under rapid urbanization and an increasingly variable climate. As cities grow into the 21^{st} century, higher potable water demands will increase stress on existing water infrastructure and supplies. In addition, periodic and extreme droughts exacerbate the problem in many regions. Infrastructure in the US is ageing and continuously needing replacement and renewal. The U.S. EPA's sixth national assessment of public water system infrastructure needs, shows a capital improvement investment estimate of $473 billion over the next 20 years, required for distribution infrastructure, treatment plants, and storage facilities (EPA, 2010). Much of the water infrastructure in developed countries is reaching the end of its design life and it is apparent that these systems won't be able to meet future challenges (Hering, Waite, & Luthy, 2013).

As our infrastructure is modernized, it is critical that systems are designed to be resilient to growing demand and water shortages, while preserving the health of the natural and urban environment. Traditionally, cities have met growing demands by securing additional supply with the construction of new storage infrastructure, transporting water over large distances and expanding centralized conveyance systems. The conventional approach of water supply and treatment through large centralized systems can be inflexible when supplies run low and costly to repair and replace. This forces cities to make complex and costly decisions on how to supply additional freshwater.

Even with growing populations, municipal water consumption in the United States has declined by 5% over the last decade (EPA, 2010). In the U.S. and other developed nations, per capita water use is leveling out thanks to advancements in water conservation through installment of high efficiency appliances, toilets and fixtures. Water conservation campaigns; tiered rates and incentives have influenced people to use less water as well. Despite these trends, sheer population growth in urban areas can very well push available supplies to the limit in the future. On top of this, climate change and its uncertain impacts on precipitation, drought intensities and water supply will likely exacerbate the problem for many cities located in arid areas.

In light of these challenges, it is important for cities to think strategically and creatively about how to maintain a reliable supply under their respective climatic, growth and economic conditions. A wealth of urban water solutions are being considered under the new approach of integrated urban water management. Cities can conserve potable water sources by implementing decentralized water reuse systems from household to neighborhood scales to harness graywater, stormwater, or wastewater for beneficial uses. There is not a one solution fits all approach, it is important to consider a diverse portfolio of strategies that complement local climate conditions as well as the unique characteristics of the city. In conjunction with continued improvements in water conservation and efficient water use indoors and outdoors, it is important cities maximize the beneficial use of water at all stages of the urban cycle to help maintain supplies well into the future. There is still work to be done to understand the best viable combinations of strategies that a city should implement to meet water savings goals at appropriate cost.

New technologies and methods of water reuse and conservation are becoming cost effective, better understood and implemented in cities across the world. Water reuse from graywater, stormwater and treated wastewater is becoming an attractive solution for cities to mitigate shortage by maximizing the beneficial use of freshwater sources. The solutions emerging include the reuse of stormwater, graywater and wastewater for non-potable as well as potable use. Recent guidance on non-potable water sources has fostered regulatory processes that enable use of alternate water sources (NBRC, 2018; Sharvelle et al., 2017). Although these strategies are validated in practice and regulation, their cost effectiveness and water savings potential remain unclear at municipal scale implementations. Integrated demand modeling using the Integrated Urban Water Model (IUWM) and the cost assessment of this study is one way to understand city-wide impacts of reuse and conservation strategies.

1.2 Background

1.2.1 Understanding Decentralized Non-Potable Water Systems

Decision making regarding water reuse systems is complex and there are many different infrastructural scales, source waters and end use combinations to consider. Available water sources, temporal conditions and water demands vary by city and these factors influence the effectiveness of water reuse systems. Solutions need to account for the interdependent factors that influence water systems, these being the local hydrologic & environmental conditions, economic limits, social acceptability, energy constraints and land use (Wilcox & Nasiri, 2016). In addition, we must understand the city landscape, existing infrastructure and present water use behaviors to focus demand reduction strategies on sectors of highest use.

Decentralized Non-Potable Water (DNW) Systems collect and treat locally generated stormwater, roof runoff, wastewater, or graywater and distribute these waters to beneficial localized end uses. These systems minimize long distance import and export of water through a city and they can be particularly useful to implement in high density developments or where existing treatment systems are near capacity (Sharvelle et al., 2017). Decentralized water reuse can refer to systems that serve many scales from individual household and multi-residential to a whole neighborhood or district. These systems are also referred to as onsite-nonpotable water systems (ONWS) and are becoming more common in new sustainably driven developments as they can achieve high levels of green building certification by maximizing the social, environmental and economic benefits of a project (NBRC, 2018). They maximize indoor and outdoor water conservation and can reduce impacts of stormwater runoff from their sites through green infrastructure or collection systems.

Decentralized collection and reuse systems can increase flexibility or security in times of shortage and lower the cost of infrastructure replacement (Hering et al., 2013). In addition, they can reduce point source pollution, help maintain natural flows, and contribute to the wellbeing of a city (Moglia, Alexander, & Sharma, 2011).

Implementing integrated water systems with water reuse from alternative sources is certainly a change from the norm and it will take continued efforts in assessment and test cases to garner wide spread adoption. Not all cities will necessarily need widespread implementation of alternative source strategies however many will benefit in the long run out of future necessity to maximize their water resources. Conservation and efficiency strategies are always beneficial and important to assess to what degree they should be pushed to achieve savings goals.

The identified barriers to wide adoption of these strategies, include uncertainties of total system cost and long-term performance, system reliability, and monitoring of water quality especially in household or multi-residential systems (Hering et al., 2013). Another limitation is the expense of building separate piping when supplying non-potable reuse water. Direct potable water reuse addresses this issue and has emerged as a viable technology in water scarce areas such as Singapore, California, Texas and New Mexico (Hering et al., 2013). There is still a need for more empirical information quantifying system success and failure and studies concerning the implementation of decentralized systems at a full system scale (Burn, Maheepala, & Sharma, 2012; Wilcox & Nasiri, 2016). There are however several studies that assess reuse systems at development/neighborhood scale in terms of triple bottom line objectives and community acceptance (Burn et al., 2012). Determining the most adequate integrated water systems for a city requires an understanding of potential savings, costs and environmental impacts.

1.2.2 Comparative Assessment Studies

Several studies have compared the feasibility, costs and impacts of water demand reduction strategies. A comprehensive study assessing the viability of graywater and stormwater reuse systems was conducted by the National Academies of Science Committee on *The Beneficial Use of Graywater and Stormwater; An Assessment of Risks, Costs, and Benefits* (Luthy, Atwater, Daigger, & Drewes, 2016). The study addresses the potential water savings and suitability of stormwater and graywater systems for non-potable use in terms of water quantity and quality, financial cost, treatment and storage, at multiple scales. The study included an analysis of water demand reduction potential using graywater and stormwater for irrigation and toilet flushing for a simulated neighborhood in six U.S. cities. The committee concluded that household stormwater or roof runoff collection for toilet flushing and/or irrigation is dependent on storage capacity and timing of precipitation events. In arid climates, there is less savings potential with stormwater due to the mismatch between irrigation demands and seasonal precipitation patterns. Graywater is a more reliable source of water in arid regions and can significantly offset potable demands for irrigation when used to irrigate low water use landscapes (NAP 2016). It was deduced that stormwater and graywater reuse at a neighborhood or regional scale would significantly reduce potable demands, however would require substantial investment in infrastructure such as storage, treatment, building modification and dual-distribution systems. Cities in arid regions can benefit from sparse but high intensity rainfall events with the use of large storage systems or groundwater recharge, as is done widely in Los Angeles, CA (LADWP, 2015; Luthy et al., 2016).

Australian entities have been leaders in promoting, implementing and studying decentralized reuse systems and they have become increasingly affordable and commonplace in the country (Moglia et al., 2011; Wilcox & Nasiri, 2016). An Australian study of 15 integrated management projects across the country demonstrated successful reductions in potable water supply, positive community acceptance and lower water bills (Mitchell, 2006). The sites were small neighborhood to regional scale and included many innovative systems of stormwater, graywater and wastewater for reuse. Sites with combinations of reuse systems and conservation saw potable water savings of 40% to 80% (Mitchell 2006). An example of a large system is Rouse Hill Water Recycling Plant in Sydney. It has the capacity to treat five million gallons of wastewater per day and redistribute for non-potable uses to 15,000 homes connected to a dual-reticulation system, saving an estimated 20% of potable water (Mitchell, 2006). Research in Australia suggests the 'optimum scale of integrated water recycling systems is in the range of 1,000 to 10,000 connections. The study found no limitations to implementing systems due to weather factors, and they are well applicable across climate zones. However, it was found that there is 'a lack of a robust assessment tool to evaluate the merits of proposed alternative water servicing options, against environmental, social and economic criteria, considering long term time horizons' (Mitchell, 2006). Lastly, the authors emphasized the importance of integrating all components of the urban water cycle and a focus on implementing total system solutions versus isolated systems.

Life cycle assessments (LCA) of decentralized graywater and wastewater reuse systems have considered a variety of system scales, types and resulting impacts in comparison to the conventional, centralized system. One study assessed low impact development (LID) technologies consisting of roof runoff for irrigation and toilet flushing, xeriscaping, and stormwater collection in bio-retention can supply non-potable water (Jeong, Broesicke, Drew, Li, & Crittenden, 2016). These were assessed in five residential zones of increasing population density. Impacts to water consumption, human health, and the environment were quantified using TRACI 2.1 LSA metrics. TRACI 2.1 is an U.S. EPA tool for reduction and assessment of chemicals and other environmental impacts. Results showed that stormwater practices studied, including roof runoff reduces potable water demand by 50% in single-family zones and 25% in multi-family zones. Savings are negligible in very high density zones due to lack of appropriate area to implement the studied practices (Jeong et al., 2016).

Another LCA study by Jeong et al. (2018) assessed potable water savings of small-scale hybrid graywater systems that work in conjunction with the centralized system and their life cycle impacts to electricity consumption, the ecosystem and human health. The simulated graywater systems reduced non-potable water demands further in single-family zones (17-49%) than in multi-family zones (6-32%), due to higher irrigation demands in single-family zones. The benefit of combining graywater reclamation with stormwater retaining LID is reported to be greater in single-family zones with the potential to reduce non-potable water demand by 44% - 82% (Jeong, Broesicke, Drew, & Crittenden, 2018).

Stormwater capture for direct beneficial use as well as groundwater recharge has been shown to be an effective strategy to conserve water supply in semi-arid regions (SCWC, 2018). Existing stormwater systems that divert to a stream can be retrofitted to supply large storage tanks or supply aquifer recharge (Luthy et al., 2016). The Los Angeles Department of Water Planning *Stormwater Capture Master Plan* provides key insights on the cities' extensive array of stormwater capture projects for water supply and integrated planning process. Future implementation potential is determined by quantifying costs and collection capacities of current projects. A wide variety of system types are analyzed for future implementation including sub regional direct use and aquifer infiltration as well as on site direct use and infiltration, and on site direct use. A total or life-cycle cost and performance framework was developed to compare future scenarios of the wide array of potential systems. Total lifecycle cost per acre-ft of captured stormwater was compared for each system type. Direct use projects come at a higher cost than aquifer recharge due to treatment and distribution costs, but an "economy of scale" is possible with sub regional collection. It was found that Los Angeles could increase water supply by 68,000 to 114,000 acre-feet per year within 20 years (LADWP, 2015).

The Urban Water Optioneering Tool (UWOT) is a decision support tool developed by Makropoulos and Butler (2010). It uses mass balance and optimization to compare scenarios of stormwater collected at regional level and graywater reuse at the household and development scale. Also included are high efficiency indoor fixtures and appliances and outdoor use efficiency at the household scale. The tool can identify tradeoffs across sustainability indicators (water demand, wastewater, energy, land use), from combinations of water savings strategies. Overall performance of the water cycle can be optimized. When potable demand is minimized, there are counter tradeoffs in operational cost, energy and land use. A study applying the model reported stormwater harvesting and greywater reuse can reduce potable water demand by 27% in new developments (Makropoulos & Butler, 2010).

Residential indoor water conservation is one of the more cost-effective means to attain overall water savings in any city. The Residential End Use Study Version 2 (REUSv2) provides a thorough assessment of water use in single-family households across the United States (W. DeOreo & Mayer, 2016). The study collected and analyzed residential flow meter data of 1,000 single family homes across the country and developed models to forecast residential demand. The study also evaluated conservation potential and factors influencing residential water use. Since their prior end use study (DeOreo, 2011) they found 'average indoor water use decreased by 15.4% from 69 gpcd to 58.5 gpcd from REUS1999 to REUS2016 (W. DeOreo & Mayer, 2016). This has been seen across the country and is a result of more homes using high efficiency appliances and efficient toilets and fixtures. The potential for additional indoor conservation still exists in the coming years with the replacement of old toilets and clothes washers (W. DeOreo & Mayer, 2016). The study also assessed outdoor irrigation use in single-family homes and found that conservation programs can be most impactful if they target over-irrigators and reduction in irrigated area can save more water than efficiency measures (W. DeOreo & Mayer, 2016).

A principal consideration in deciding how to implement demand reduction strategies, as with any water infrastructure, is economic feasibility. There is uncertainty in the lifecycle costs of water reuse systems as they are still novel, and vary considerably by system scale, treatment capacity, and treatment requirements. Despite the known benefits of using alternative water sources for reuse, water utilities are still hesitant to implement large scale reuse systems in their long term plans because of the lack of documentation of costs, performance, and associated risks (Luthy et al., 2016).

The Pacific Institute conducted a life-cycle cost analysis of several ‘alternative water supply’ and conservation strategies implemented in California (Cooley & Phurisamban, 2016). Systems considered include small and large stormwater capture projects, non-potable and potable reuse of wastewater, indoor fixture efficiency, irrigation efficiency and conversion to xeriscape. The study determines cost per acre-foot of conserved water though efficiency measures or volume processed in reuse systems, accounting for full capital and operating costs of a project or conservation device over its lifetime. To note, wastewater recycling for non-potable reuse was found to be less expensive than indirect potable reuse because of lower treatment requirements, even with added expense of dual distribution piping. Many of the efficiency measures and conversion to xeriscape had a “negative” life-cycle cost, meaning water savings cost over the lifetime of the measure are greater than the cost to implement.

1.2.3 Summary

While the literature is replete with life-cycle assessment studies that focus on a variety of environmental, energy and technological factors in addition to demand reduction of particular reuse systems and scales (Jeong et al., 2018, 2016), there remains a lack of studies that seek to assess total system solutions by identifying strategies that achieve the most water demand reduction at the lowest cost for a given city or region. Studies that assess use of alternate water sources typically don’t include comparisons to end use efficiency practices. Many of the studies focus either on building scale or centralized and do not include comparisons from the household to municipal scale or city-wide adoption of practices. Studies have explored the costs of water demand reduction strategies (Cooley & Phurisamban, 2016; LADWP, 2015; Luthy et al., 2016; Trussell et al., 2012) but there is a lack of comparative assessment of all strategies discussed in this review applied at a full city scale in consideration of cost and demand reduction.

Implementing a variety of alternative reuse systems at wide adoption across a city has the potential to be less costly and more sustainable than expanding centralized wastewater treatment capacities. In addition, these strategies can mitigate the need to procure additional fresh water sources, which can be very costly or impossible in cities reaching the limits of their available supplies. Understanding how and why certain strategies work in a particular city can influence city managers to consider them more. The National Academies Committee on Graywater and Stormwater use stated that 'Multi-criteria decision analysis or broadly defined benefit-cost analysis are two important tools that may be useful for evaluating future management strategies that create such a broad spectrum of valuable outcomes' (Luthy et al., 2016).

1.3 Research Objectives

This study seeks to assess water demand reduction and associated costs of city-wide adoption of water conservation and reuse strategies through integrated modeling. The objectives of the study are to assess the cost to potable water demand reduction benefit tradeoffs between water conservation and reuse strategies. In addition, understand the local effects of precipitation, population density and land cover on strategy performance. The Integrated Urban Water Model (IUWM) (Sharvelle, Dozier, Arabi, & Reichel, 2017) was calibrated to three U.S. cities to model residential and all outdoor water demands, followed by running a suite of potable water demand reduction strategies. A non-dominated solution ranking procedure was used to assess tradeoffs between demand reduction and total cost. The study cities are; Denver CO, Miami, FL and Tucson, AZ.

Demand reduction strategies modeled include indoor conservation, outdoor irrigation efficiency and use of alternative water supplies to supplement indoor and outdoor demands, including potable use. Optimal combinations of demand reduction strategies are identified that span a range of total costs. The most effective strategies in each city are identified. In addition, the most cost-effective solutions that minimize wastewater outflow are identified. This study presents a framework to assess tradeoff of water demand reduction strategies at the municipal scale across diverse regions.

CHAPTER 2: METHODS & MATERIALS

2.1 Study Overview

The methodology taken to meet the objectives of this study consisted of the following steps; First, three study cities were selected and observed data of water use was obtained to calibrate and test the IUWM. A sensitivity analysis and calibration procedure was applied to determine best fit parameters that match observed use. The calibrated models for each city represented the baseline (current) condition. Next, solutions were generated by running combinations of IUWM's water demand reduction strategies using the same implementation levels for the three cities. Annualized life-cycle cost of each strategy was found in the literature or estimated based on system type and level of adoption. An optimization of solutions was conducted using a non-dominated ranking procedure that minimizes model outputs of annual potable water demand and annualized total cost. Non-dominated solutions and the strategies they are composed of were assessed with frequency analysis.

2.2 City Characteristics

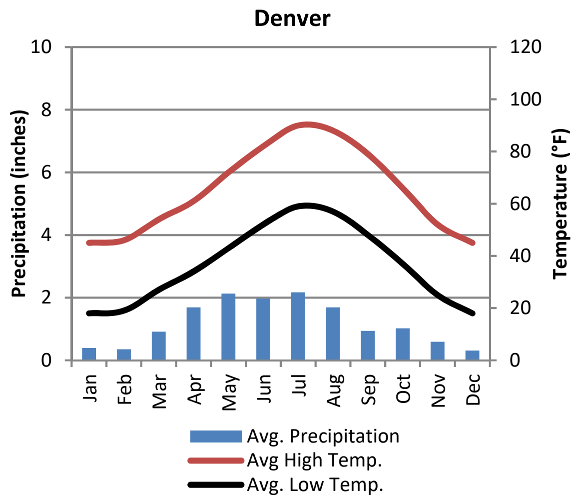
The three cities analyzed in this study are Denver, Colorado; Miami, Florida; and Tucson, Arizona. These cities were selected for their distinct climatic and urban land use characteristics (Table 1) in consideration of the objective to assess how local factors affect performance of water demand reduction strategies.

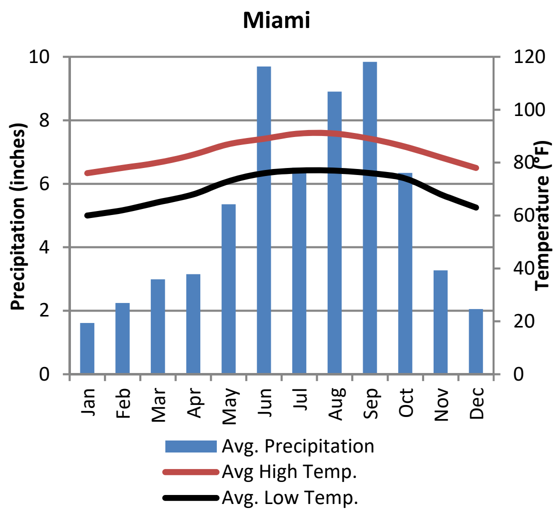
Table 1: Modeled area descriptive statistics

City	Population	Households	Avg. Household Size	Area (mi^2)	Population Density (people/mi^2)	% Impervious	Annual Precip. (in)
Denver	904,504	387,745	2.3	216.4	4,179	39%	14
Miami	1,292,905	436,166	3.0	242.0	5,342	31%	62
Tucson	530,513	216,634	2.4	200.4	2,647	29%	12

Note: these values do not reflect city boundaries or the water service area but are based on modelled areas where extensive data on water use were available

The three cities have distinct climates (Figure 1). According to the Koppen climate classification system, Denver lies in a semi-arid, continental climate zone. Miami lies in a tropical monsoon climate zone. Tucson lies in a mid-latitude steppe and desert climate zone.





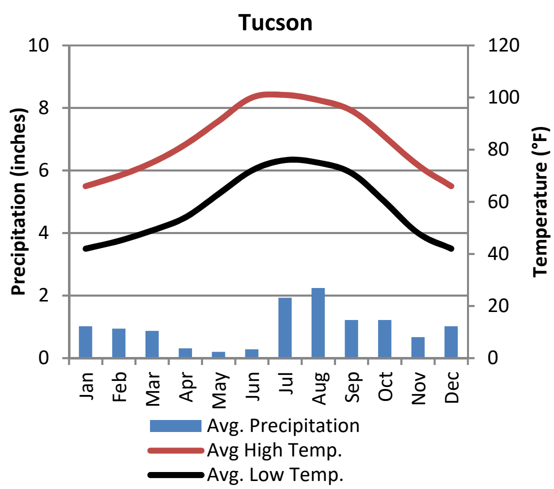
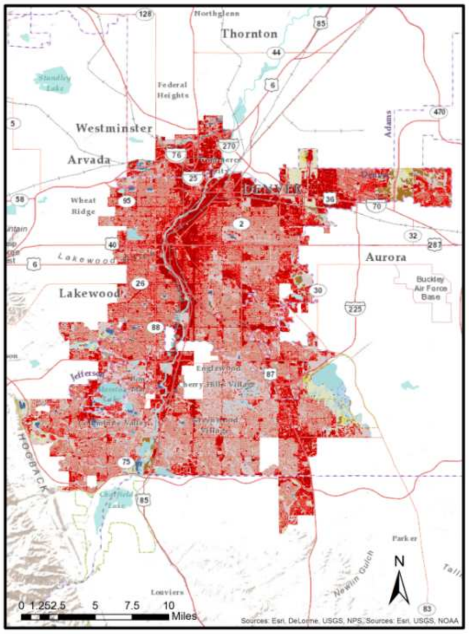
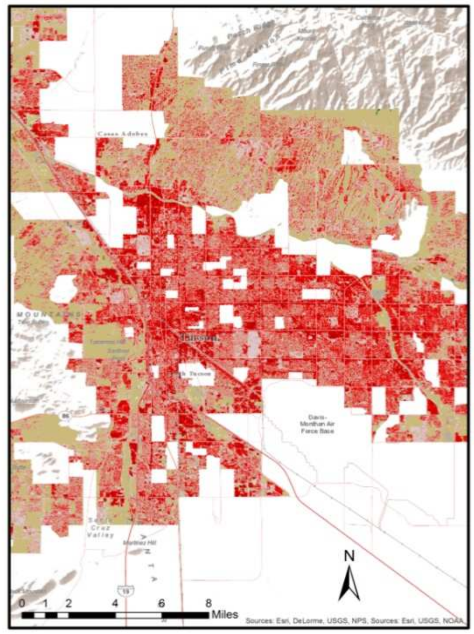


Figure 1: Average monthly precipitation and temperature (1981-2010). Weather stations: Denver – Stapleton; Miami – International Airport; Tucson - 85707

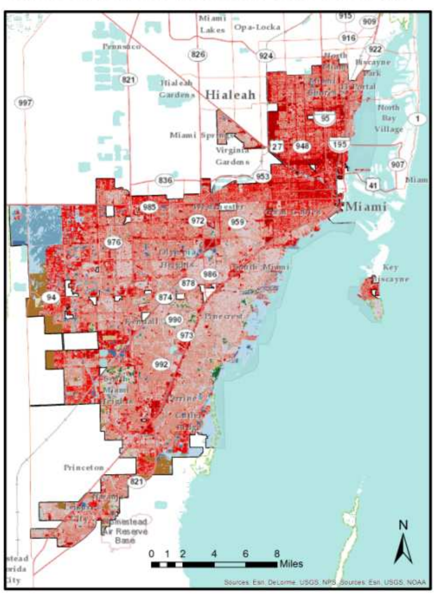
Denver Modeled Area - NLCD Land Cover Classification



Tucson Modeled Area - NLCD Land Cover Classification



Miami Modeled Area - NLCD Land Cover Classification



NLCD Developed Area Classes		Impervious Area
21	Open Space	0 - 20 %
22	Low Intensity	20 - 49%
23	Medium Intensity	50 - 79%
24	High Intensity	80 - 100%

Figure 2: National Land Cover Database developed area classification maps of modeled service areas

2.3 The Integrated Urban Water Model (IUWM)

IUWM is a mass balance municipal water use and forecasting tool that quantifies residential, commercial and outdoor irrigation demands. In addition, IUWM simulates indoor and outdoor conservation strategies, and has explicit capacities to evaluate the potential for use of alternate water sources (i.e. graywater, wastewater, stormwater runoff and roof runoff) a range of scales (single building to municipal) (Sharvelle et al., 2017).

2.3.1 Indoor Residential Water Demand

Indoor residential water demand is modeled in IUWM using a demand profile function that relates daily household use to estimated household size (W. DeOreo & Mayer, 2016). Total daily water use $q^{res,in}$ represented by a power function considering average household size in the spatial subunit (s_i = population/households) and parameters α and β(Sharvelle et al., 2017). The spatial subunit of this study is U.S. Census block group and total indoor demand is an aggregate of each block group using their respective number of households n^{hsd} and populations. The indoor household demand profile function is defined as:

$$\boldsymbol{q^{res,in} = \alpha_i * s_i^{\beta} * n^{hsd}} \quad \textbf{Eq. 1}$$

Figure 3 displays the demand profile functions that model daily household water use in gallons per household per day (GPHD). The 'user-defined' function seen in the figure uses Denver's calibrated α and β parameters (see Section 2.2) and average household size s_i value of 2.3 people per household.

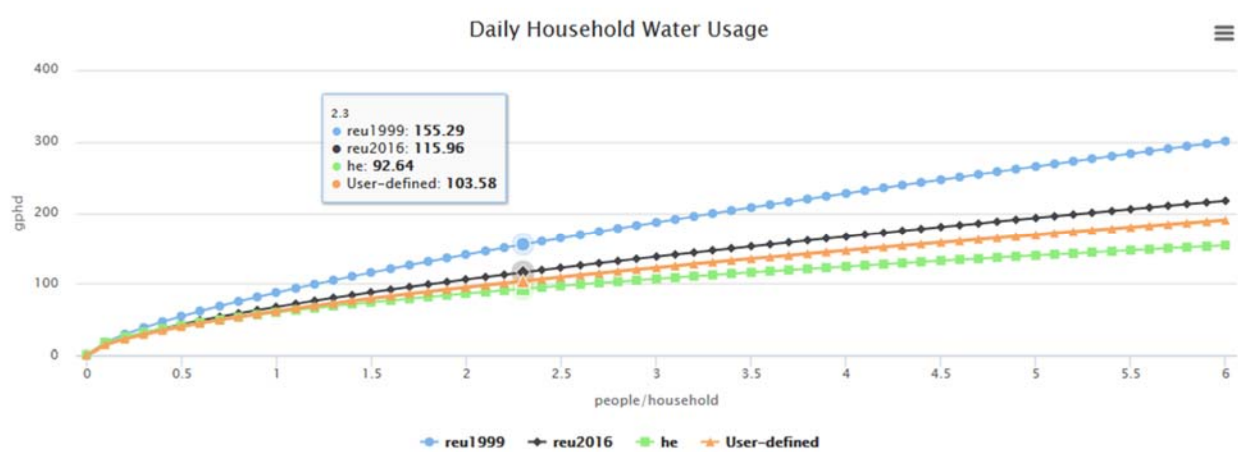


Figure 3: Demand Profile Function Chart. User-defined function is that of Denver's calibrated parameters and average household size s = 2.3. GPHD = Gallons per household per day

2.3.2 Outdoor Demand

Total outdoor demand is estimated with IUWM by calculating a depth of irrigation applied across an estimated irrigated area based on NLCD (National Land Cover Database, MRLC) developed land cover categories. For this reason, outdoor demand parameters are calibrated to total outdoor use data; which includes residential and non-residential outdoor use. Daily irrigation requirements are estimated as a fraction of reference evapotranspiration (ET_0) using the Penmen-Monteith equation available within IUWM. The net irrigation requirement (*NIR*) replicates the actual irrigation applied by taking a fraction of plant water requirements determined by ET_0, plant factor, fraction of precipitation events responded, irrigation efficiency and daily precipitation (Sharvelle et. al, 2017). The daily irrigation depth equation is written as follows, for a spatial subunit *i* and timestep *t*:

$$\boldsymbol{q_{i,t}^{irr} = NIR * \left(\frac{k_i^{pf} * ET_0 - k_i^{pcp} * r_{i,t}^{pcp}}{k_i^{eff}}\right)} \qquad \textbf{Eq. 2}$$

NIR is the net irrigation requirement fraction, to model actual irrigation applied, k_i^{pf} is plant factor or crop coefficient, ET_0 is reference evapotranspiration (inches), k_i^{pcp} is the fraction of precipitation events responded to by the irrigator, k_i^{pcp} is daily precipitation (inches), and k_i^{eff} represents irrigation application efficiency.

2.4 Model Calibration

A consistent methodology was conducted to calibrate and test the model in the three study cities. For each city, four years of water use data were aggregated monthly and spatially by U.S. census block group. IUWM is a spatial model, therefore to ensure quality and accuracy it is critical that the calibrated area is accompanied with water use data that encompasses the modeled area in its entirety. This analysis modeled all residential indoor use and all outdoor water use from residential and non-residential use for irrigation. Outdoor use of commercial and industrial (CII) service includes irrigation of parks and open spaces. Irrigation of all service types is included in calibration since IUWM models outdoor demand using land cover classification and does not distinguish residential from non-residential areas.

Calibrated block groups were validated for full coverage of water use data over space and time. Water meter data obtained from cities was sometimes incomplete; having gaps in time or did not include all water users in a block group. Each city required different approaches to ensure full coverage and each city differed slightly in the outdoor parameters that needed to be calibrated (see sections 2.3.2, 2.4.2, and 2.5.1). The number of block groups that included full coverage were randomly divided into calibration and testing sets, 80:20 respectively, using the *subset* tool in ArcGIS. This ensured spatial variation of block groups selected for each calibration and testing. The calibrated parameters for the complete data set were used to assess city wide adoption of water demand reduction strategies, while individual block group results enabled spatial assessment of water use and model performance across the service area.

Calibration was conducted using the Sobol Global Sensitivity Analysis technique (Sobol, 2001) to assess statistical performance of parameter sets in comparison with the observed data series (Sharvelle et al., 2017). To select the 'best' performing set, a max log likelihood function was applied which assesses the most frequent occurring parameters in sets that produce demand estimates most closely matching observed data. At the city scales of this study, 800-1200 runs were sufficient to converge on good fitting parameters. Estimates of the log-likelihood function assume an auto-regressive transformation of errors after transforming data by the natural logarithm (Tasdighi, Arabi, Harmel, & Line, 2018). Log-likelihood was determined for each model run corresponding to time series of observed and modeled water use. This was performed at each block group individually across the whole set of block groups included. The maximum likelihood parameters for the sum of training and testing sets of block groups were ultimately used to set baseline conditions and run scenarios of strategies.

Model performance in training and testing is quantified by the following error statistics; mean relative error (MRE), bias fraction (BIAS), and Nash-Sutcliffe Coefficient of Efficiency (NSCE) (Sharvelle et al., 2017).

2.5 Denver

2.5.1 Denver Water Use Data

Denver Water provided monthly water meter data from 2011 to 2016 across its service area in the Denver metro area. Meter data were categorized by service agreement type and their respective location identified by US census blocks. The service agreement types were combined into two categories to be implemented with IUWM; residential and commercial, institutional & industrial (CII). Monthly meter data were aggregated into 'residential' and 'CII' categories and summed by census block group for analysis. Both categories were divided into indoor and outdoor use by considering the average consumption during non-irrigation winter months of November –February as a proxy for indoor use throughout the year. CII use in summer months was obtained after removing the winter average use and added to residential outdoor use to represent total outdoor water use.

Denver Water provided a water supply service area shapefile. The modeled service area encompassed 704 block groups in the Denver metro area. To prevent disclosure of individual customer use, Denver Water excluded blocks with five or fewer meters from the dataset. The data in the removed blocks consisted of just two percent of all Denver Water meters included in the dataset. Block groups containing these blocks were not used in calibration and testing. Several additional block groups were excluded in cases where they were located beyond the service area boundary. These consisted of solely industrial land use areas and block groups bordering Lakewood that included master meters serving unknown areas. After exclusion, a remaining 393 block groups were used for calibration and testing (Figure 4).

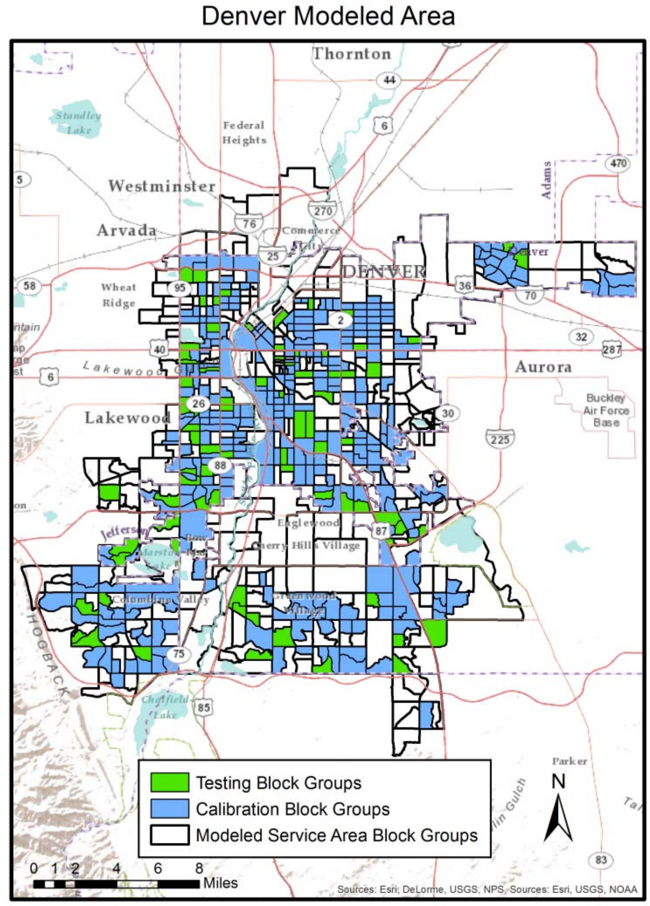


Figure 4: Map of Denver modeled service area and respective calibration and testing block groups

2.5.2 Denver Calibration Procedure

One thousand parameter sets were run in the sensitivity analysis for the outdoor parameters ($\mathrm{k_i^{pf}}$, k_i^{pcp}, $\mathrm{A_{i,c}}$ for each NLCD class) and three hundred for the two indoor parameters α and β. These parameters were varied within realistic ranges (Table 2). *NIR*, k_i^{pf} and the minimum threshold temperature for which irrigation is applied T_i^{irr} (°C), were assumed values and were not varied in calibration; $\mathrm{k_i^{pf}}$ was fixed at 0.8, a vegetation coefficient estimated for cool season grass (ANSI/ASABE, 2007). A $\mathrm{k_i^{pf}}$ of 0.8 was also fitting for Fort Collins in the IUWM demonstration paper (Sharvelle et al., 2017). *NIR* was fixed at 45% of the theoretical irrigation requirement, the national average (W. DeOreo & Mayer, 2016) and also applied to Fort Collins, CO in the IUWM demonstration paper. T_i^{irr} (°C) was held at 13°C, having been calibrated for Fort Collins, CO (Sharvelle et al., 2017). The IUWM default irrigation efficiency k_i^{eff} of 0.71 was held constant; it represents the fraction of water that is used by the plant. Indoor parameters α and β were calibrated to monthly estimated residential indoor use using the winter average baseline as discussed prior in section 2.3.1.

Table 2: Denver calibrated and applied model parameter values

IUWM Parameter	Description	Calibration Range	Calibrated and Assumed Values*
α	Indoor DPF	40 - 100	61.24
β	Indoor DPF	0.5 - 0.99	0.631
NIR (%)	Net irrigation requirement met	20 - 100	45%*
k_i^{pcp} (%)	Precipitation events responded to	20 - 80	76%
k_i^{eff}	Irrigation application efficiency	-	0.71*
T_i^{irr} (°C)	Threshold temperature	-	13°C*
$\mathrm{k_i^{pf}}$	Plant factor	0.5 - 0.9	0.8*
$A_{i,c}$ (%), c = open	Open space area irrigated	30 - 90	53%
$A_{i,c}$ (%), c = low	Low density area irrigated	30 - 90	30%
$A_{i,c}$ (%), c = medium	Medium density area irrigated	10 - 70	59%
$A_{i,c}$ (%), c = high	High density area irrigated	2 - 30	11%

****Assumed parameter values. DPF = demand profile function***

A second outdoor calibration was performed to check the parameters held constant in the first run; plant factor and net irrigation requirement k_i^{met} (%). The top performing run indicated an *NIR* value of 43%, very close to the REUS national average of 45% held constant in the first calibration. Plant factor $\mathrm{k}_\mathrm{i}^\mathrm{pf}$ resulted in 0.7, while a $\mathrm{k}_\mathrm{i}^\mathrm{pf}$ of 0.8 was used in the first calibration run. The percent irrigated areas were comparable. The second calibration did not perform significantly better in training and testing, therefore the first calibration results were used in the municipal scale strategy runs.

The calibrated parameter values (Table 2) were used to compare model predictions to observed data (Figure 5). It can be noted that observed water use was higher in 2012 than in 2013-2015 (Figure 5). The summer of 2012 was particularly hot and dry due to drought at that time. It is likely that more landscape irrigation water was applied with the reduction in rainfall and higher temperatures. The model accounted for these climatic factors and matched 2012 demands quite well. The following years were more climatically consistent, and the model tended to overestimate outdoor demand in those years. The high use in 2012 likely had an impact in the overestimation of the following years due to the model's use of averaged observed monthly values in the calibration. It is appropriate to calibrate to years with variabilities in climate since the region will likely continue to see drier than average years in the future.

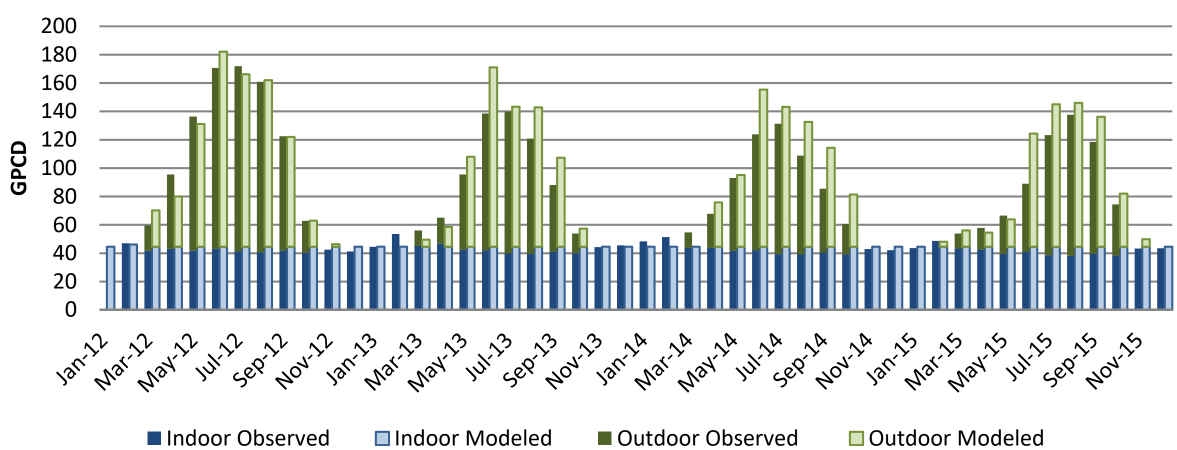


Figure 5: Denver monthly testing results of observed use and modeled indoor and outdoor demand in gallons per capita per day (GPCD)

Calibration performed well achieving a mean relative error (MRE) of -0.55% for indoor use and -1.42% for outdoor use. Negative MRE indicates model overestimation. Testing MRE for outdoor was within 10% of observed, a limit deemed acceptable in this study (Table 3), considering it performed slightly better than the testing MRE for Fort Collins, CO in (Sharvelle et al., 2017)

Table 3: Denver calibration and testing statistics

Use	**Calibration 2012-2015**			**Testing 2012-2015**		
	NSCE	MRE	BIAS	NSCE	MRE	BIAS
Residential Indoor	-	-0.55%	-0.01%	-	-2.80%	-2.42%
Outdoor	0.922	-1.42%	-0.51%	0.926	-9.82%	-9.80%

Mean Relative Error (MRE), Bias Fraction (BIAS), Nash-Sutcliffe Coefficient of Efficiency (NSCE)

2.6 Miami

2.6.1 Miami Water Use Data

Individual water meter data were provided by the Miami-Dade Water Utility for the years of 2012-2016 and half of 2017. Raw data were quarterly billed use amounts and were divided evenly across the number of days in each billing period, then aggregated monthly. The service area is composed of 502,288 meters identified by a premise (location) ID. The raw data were composed of 247 premise types, covering residential and commercial, institutional and industrial categories. This analysis aggregated all residential, residential sprinkler, and commercial sprinkler meters and is referred hereon as total use for the calibration process. The analysis set consisted of 300,000 single family homes, 140,000 apartments, duplexes, and townhouses, 4,631 commercial sprinkler and park meters, and 3,969 residential sprinkler meters. Residential use was aggregated by block group, the unit area used by IUWM in this study.

To ensure full coverage of residential water use data for training and testing the model, block groups with partial coverage of point data in residential areas were excluded from calibration. This was done by visually identifying and excluding those blocks where residential points did not appear to cover all households or residential areas within the block group. Of the 848 block groups in the modeled service area, 387 block groups were used for calibration and testing the model. The ArcGIS Subset tool was used to divide the data set 80:20 into training and testing sets respectively (Figure 6)

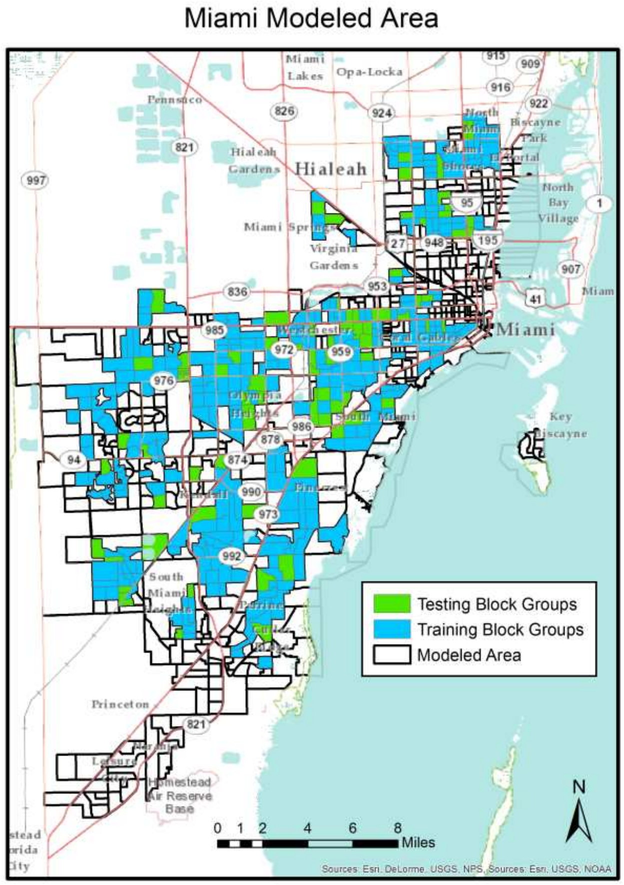


Figure 6: Map of Miami modeled service area and respective calibration and testing block groups

2.6.2 Miami Calibration Procedure

There is no simple way to separate indoor and outdoor water use in Miami, since outdoor use occurs year-round. An analysis was conducted to understand residential indoor use in the Miami service area. Unfortunately, the analysis did not provide an adequate estimation of indoor α and β parameters to be used for Miami (see Appendix A.1). In turn, the REUSv2 national average indoor demand profile function parameters, α and β were held constant during calibration and ultimately in strategy runs to estimate indoor water use.

Calibration was conducted to total water use data from February 2012 through June 2017. The parameters calibrated for Miami were; $A_{i,c}$ (%) for open, low, medium and high-density areas, $\mathrm{k_i^{pf}}$, *NIR* (%), k_i^{pcp}(%), (**Error! Reference source not found.**). IUWM has the option to use daily, monthly average or annual average reference evapotranspiration (ET_0) in calibration. It was found that outdoor water use is not responsive to plant requirements as determined by daily nor monthly ET_0. Using annual average ET_0 was found to best fit the model to observed total use. When calibrating to total use with set indoor α and β parameters, the model attributes indoor demand from the parameters and number of households and calibrates the outdoor parameters to the remaining total use.

Table 4: Miami calibrated and applied parameter values

IUWM Parameter	Description	Calibration Range	Calibrated and Assumed Values*
α	Indoor DPF	-	67.5*
β	Indoor DPF	-	0.62*
NIR (%)	Net irrigation requirement met	20 - 90	21%
k_i^{pcp}(%)	Precipitation events responded to	20 - 90	68%
k_i^{eff}	Irrigation application efficiency	-	0.71*
T_i^{irr}(°C)	Threshold temperature	-	13°C*
$\mathrm{k_i^{pf}}$	Plant factor	0.5 - 0.85	0.68
$A_{i,c}$ (%), *c* = open	Open space area irrigated	30 - 90	69%
$A_{i,c}$ (%), *c* = low	Low density area irrigated	20 - 90	79%
$A_{i,c}$ (%), *c* = medium	Medium density area irrigated	10 - 90	43%
$A_{i,c}$ (%), *c* = high	High density area irrigated	5 - 60	16%

Assumed parameter values. DPF = demand profile function

The calibrated parameters represent average water use over the 5-year period. Miami training MRE was within 1% and a testing MRE of 10% (**Error! Reference source not found.**). Testing results show a consistent underestimation of observed water demand. The exact cause of this could not be determined; however, a potential cause is the presence of individual users with high irrigation use within testing block groups.

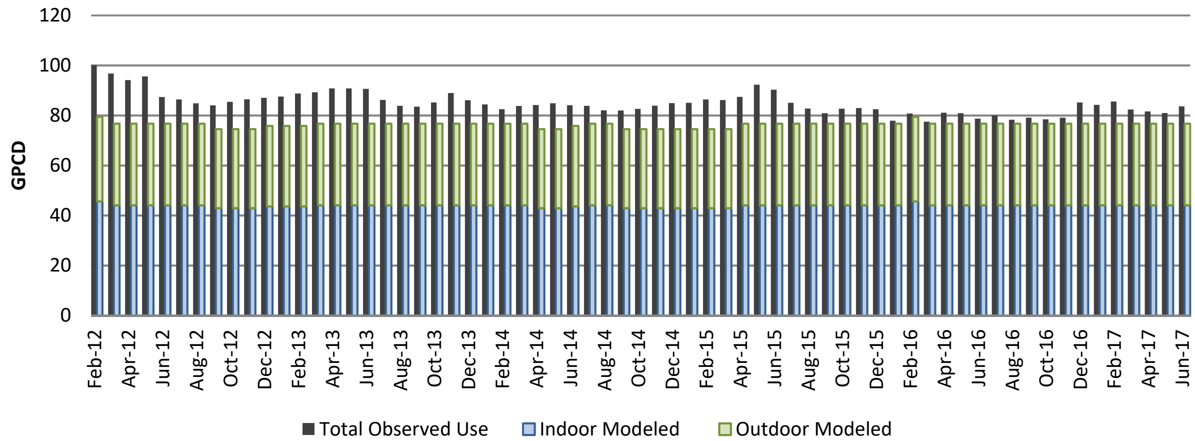


Figure 7: Miami monthly testing set results of observed use vs modeled indoor and outdoor demand in gallons per capita per day

Table 5: Miami calibration and testing statistics

Use	Calibration: Jan. 2012 - June 2015			Testing: Jan. 2012 - June 2015		
	NSCE	MRE	BIAS	NSCE	MRE	BIAS
Total	0.22	-0.7%	-0.4%	-2.97	10.0%	10.3%

Mean Relative Error (MRE), Bias Fraction (BIAS), Nash-Sutcliffe Coefficient of Efficiency (NSCE)

2.7 Tucson

2.7.1 Tucson Water Use Data & Calibration Procedure

Tucson City water utility provided monthly water use data of their service area within the Tucson metro area for 2012-2017. Raw data were provided by block group and service type. Irrigation service types indicated water use during the winter months. Therefore, it was assumed that some irrigation occurs through the winter months and must be accounted for in the regular residential data. Another indication was that the sum of residential use was consistently higher in winter months than the indoor model estimation. For this, it was not possible to use any winter months to estimate baseline indoor use for rest of the year. Service types encompassing residential, irrigation and reclaimed water for irrigation for residential and commercial were aggregated to *total use* for model calibration. Indoor α and β values used for Tucson are those found in the REUS v2 study for Scottsdale, AZ (W. DeOreo & Mayer, 2016). The threshold temperature was changed from 13°C to 5 °C, to allow the model to allocate outdoor demand when average daily temperature drops within 5-13 °C. Block groups that did not contain residential meter data were not used in the analysis. 332 block groups were used to test the model and 83 for training (Figure 8).

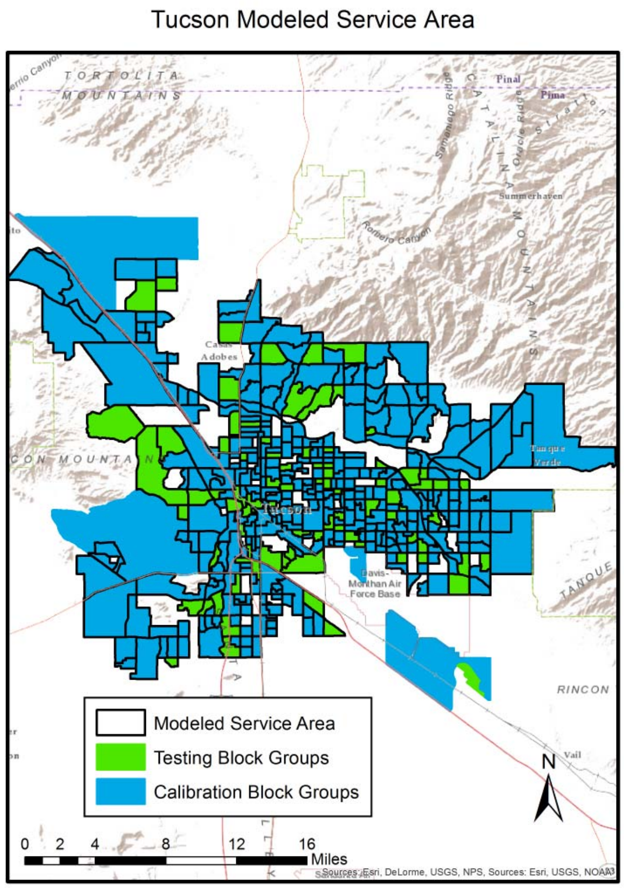


Figure 8: Map of Tucson Modeled Service Area and respective calibration and testing block groups

Table 6: Tucson calibrated and applied model parameter values

IUWM Parameter	Description	Calibration Range	Calibrated and Assumed Values*
α	Indoor DPF	-	65.0*
β	Indoor DPF	-	0.75*
NIR (%)	Net irrigation requirement met	20 - 100	23%
k_i^{pcp}(%)	Precipitation events responded to	20 - 90	83%
k_i^{eff}	Irrigation application efficiency	-	0.71*
T_i^{irr}(°C)	Threshold temperature	-	5°C*
$\mathrm{k_i^{pf}}$	Plant factor	0.5 - 0.8	0.65
$A_{i,c}$ (%), c = open	Open space area irrigated	10 - 80	58%
$A_{i,c}$ (%), c = low	Low density area irrigated	10 - 70	28%
$A_{i,c}$ (%), c = medium	Medium density area irrigated	10 - 70	34%
$A_{i,c}$ (%), c = high	High density area irrigated	5 - 60	7%

Assumed parameter values. DPF = demand profile function

Overall, the results were very good with a testing MRE of -2.1%, a slight overestimation. There appeared to be a tendency for overestimating outdoor water use in the spring months (Figure 9) and annually, as indicated by an MRE of -2.1% in testing (**Error! Reference source not found.**).

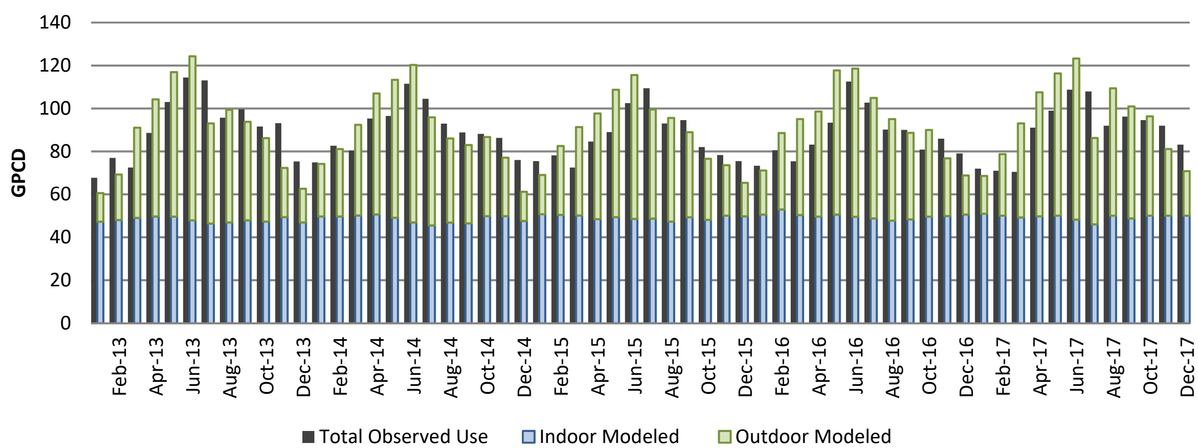


Figure 9: Tucson monthly testing set results of observed use vs modeled indoor and outdoor demand in gallons per capita per day

Table 7: Tucson calibration and testing statistics

Use	Calibration: 2013-2017			Testing: 2013-2017		
	NSCE	MRE	BIAS	NSCE	MRE	BIAS
Total	-0.001	-1.5%	-1.2%	0.02	-2.10%	-2.06%

Mean Relative Error (MRE), Bias Fraction (BIAS), Nash-Sutcliffe Coefficient of Efficiency (NSCE)

2.8 IUWM Water Conservation and Reuse Strategies

For each city, the calibrated parameters obtained through training were applied across the modeled area to set the baseline or current scenario. IUWM was used to simulate the use of four alternative water sources, namely *stormwater, graywater*, *roof runoff*, and *wastewater* to supply four end uses: residential *toilet flushing, irrigation,* and two combined end use strategies; *toilet flushing & irrigation* and *potable & irrigation*. The scale of the reuse strategies is modified with IUWM parameters that set the level of adoption or collection and storage capacities (**Error! Reference source not found.**). The parameters allow the user to best approximate the scale and type of system being modeled.

Strategy parameters were set to levels in best interest of comparability, where each strategy simulates an ambitious but pragmatic implementation of its respective system (Table 8). A *medium* adoption level was used in the combination of strategies described shortly in Section 2.7. A set of 20 single strategies applied at the *aggressive* level of adoption was also run to assess comparability with the combinations. The mass balance equations that allocate these water reuse sources to end uses are described in Sharvelle et al. (2017).

For *indoor conservation, roof runoff and graywater use*, $\mathrm{k}^{\mathrm{adopt}}(\%)$ is the fraction of households in the modeled area that adopt the practice. In the case of *stormwater collection and use,* the fraction of total runoff that can be collected is chosen and applied in a daily mass balance using the estimated storage capacity of 3,000 gallons/household. This water offsets the end use demands on a daily time step until storage has been depleted. Likewise, with wastewater reuse, 25% of available wastewater offsets daily end use demand.

Outdoor conservation is modeled through three strategies: conversion to *xeriscape*, use of *efficient irrigation systems* and *advanced irrigation systems*. *Xeriscape* is modeled by reducing the plant factor to 0.5, simulating a balance between significant low water use landscapes along with irrigated turfgrass across the modeled area. Xeriscape landscape is estimated to have a plant factor of 0.3. The plant factor of 0.5 represents approximately 50% of irrigated area converted to xeriscape.

Alternative source strategies serve four end use categories, these being; irrigation, toilet flushing, combined irrigation and toilet flushing, and combined potable and irrigation. Potable is combined with irrigation in consideration that source water treated to potable quality would enter the existing distribution system that supplies water for both purposes. The mass balance equations that allocate alternative sources to end uses are described in section *2.3 Use of alternative water sources* of the principal IUWM paper (Sybil Sharvelle et al., 2017).

Table 8: IUWM strategy parameter values related to adoption

Conservation and Alternate Source Water Strategies	Parameter	Medium Adoption	Aggressive Adoption
1. Indoor Conservation[1]	Adopting households (k_{ic}^{adopt})	50%	90%
2. Advanced Irrigation Systems	Irrigation demand reduction	10%	30%
3. Xeriscape	Scaled plant factor (k_{i}^{pf})	0.5	0.3
4. Efficient Irrigation Systems	Irrigation efficiency (k_{i}^{pf})	0.85	0.98
5. Graywater Use[2]	Adopting households (k_{gw}^{adopt})	30%	60%
6. Roof Runoff collection and use[3]	Adopting households (k_{rr}^{adopt})	30%	60%
7. Stormwater collection and use[4]	Available stormwater for capture	40%	80%
8. Wastewater Recycling	Available wastewater for end use	25%	50%

[1]Indoor conservation: high efficiency homes α = 59.6 and β = 0.53, (W. B. DeOreo, 2011)
[2]Graywater storage = 200 gal/household
[3]Roof runoff: relative storage = 200 gal/household. Percent impervious area collected = 20%
[4]Stormwater relative storage = 3,000 gal/household

2.9 Cost Assessment of Demand Reduction Strategies

Total annualized costs for each water demand reduction strategy were directly referenced from literature or developed through estimation based on literature values for system components and number of systems consistent with assumed parameters (Table 8). Total cost per unit refers to the lifetime cost of a system or strategy considering capital, operations & maintenance (O&M) and distribution when applicable. Depending on the strategy, unit costs can be per thousand gallons, per household, or per unit area (Table 9). Strategies' total cost is attained by spanning these costs across the modeled area using respective adoption parameters (Table 8), and represent the scale and system type the parameters simulate. Detailed methodology for cost estimates and lifespan assumptions is to follow in the coming sections. For cost estimation, a pragmatic scale of adoption needed to be assumed for each strategy. In brief, strategy total costs arising from the single-family household level include; *indoor conservation*, *advanced irrigation systems, roof runoff for irrigation & toilet flushing* and *graywater for irrigation*. The cost of *stormwater* for all end uses is that of a neighborhood or sub-regional collection system with dual-piping, reflective of typical applications of stormwater use (Luthy et al., 2016). While graywater irrigation systems are common at the single residence scale (Luthy et al., 2016), *graywater use for toilet flushing* requires further treatment rendering systems more complex with more practical application at the multi-residential or neighborhood scale (Luthy et al., 2016). The cost of all strategies modeling an alternative source for potable reuse combined with irrigation reflects that of the sum of centralized wastewater treatment and secondary direct potable reuse (DPR). Costs of *wastewater for toilet flushing & irrigation* (single and combined), reflect centralized non-potable reuse facilities and separate distribution for use, the most common scale that water recycling projected are implemented (Trussell et al., 2012).

Alternative source strategies with costs per unit volume were referenced from studies that compiled and compared the life-cycle or annualized lifetime costs (Cooley & Phurisamban, 2016; LADWP, 2015; Luthy et al., 2016; Raucher & Tchobanoglous, 2014). Total cost by volume was determined by the product of each strategy's unit volume cost and annual water reused or processed through the respective strategy.

Table 9: Strategy unit cost, system type and lifespan

Strategy	Total Unit Cost	Unit	System Type	Lifespan
Conservation Strategies:				
Indoor Conservation	$ 1,584.00	/hsd	HE appliances & fixtures	30 year
Xeriscape	$ 1.55	/ft^2	Landscape Conversion	50 year
Efficient Irrigation Systems	$ 5.12	/sprinkler	High Efficiency Sprinklers	15 year
Advanced Irrigation Systems	$ 255.00	/home	Irrigation Controllers	15 year
Graywater Reuse for:				
Irrigation	$ 2,300	/hsd	Single Family Residence	30 year
Toilet Flushing	$ 4.62	/kgal	Multi-Res/Neighborhood	30 year
Toilet Flushing & Irrigation	$ 4.62	/kgal	Multi-Res/Neighborhood	30 year
Potable & Irrigation	$ 10.48	/kgal	Centralized DPR	30 year
Roof Runoff for:				
Irrigation	$ 1,500	/hsd	Single Family Residence	20 year
Toilet Flushing	$ 1,600	/hsd	Single Family Residence	20 year
Toilet Flushing & Irrigation	$ 1,600	/hsd	Single Family Residence	20 year
Potable & Irrigation	$ 10.88	/kgal	Centralized DPR	30 year
Stormwater Reuse for:				
Irrigation	$ 3.28	/kgal	Neighborhood-Subregional	30 year
Toilet Flushing	$ 18.41	/kgal	Neighborhood-Subregional	100 year
Toilet Flushing & Irrigation	$ 18.41	/kgal	Neighborhood-Subregional	100 year
Potable & Irrigation	$ 10.48	/kgal	Centralized DPR	30 year
Wastewater Reuse for:				
Irrigation	$ 4.73	/kgal	Centralized Non-Potable	NA
Toilet Flushing	$ 5.02	/kgal	Centralized Non-Potable	NA
Toilet Flushing & Irrigation	$ 5.02	/kgal	Centralized Non-Potable	NA
Potable & Irrigation	$ 6.14	/kgal	Centralized DPR	30 year

DPR = direct potable reuse; hsd = household; O&M = operations & maintenance; kgal = thousand gallons. NA indicates lifespan used to calculate annualized cost was not reported

Total annualized cost is attained using the uniform series net present value function, where total cost refers to present value cost (PVC) as it is named in the referenced literature (EPA & CEE, 2010). This function is applied to strategies with estimated total costs, but not for those where literature values were available for cost per volume processed. All annualized total costs were calculated using a 5% discount factor. To test comparative sensitivity to the discount rate, all strategy annualized costs were also computed at a 6% and 7% discount factor. The relative magnitude between strategy costs did not change with either of the higher rates, see Table 17 in the appendix for comparison of Denver's annualized costs. The uniform series, net present value function is the following:

$$TAC = TC * \frac{i(1+r)^n}{(1+r)^n - 1} \qquad \text{Eq. 3}$$

Where *TAC* is total annualized cost, *TC* is total cost, *r* is the discount rate, and *n* is lifespan in years. In general, the costs lie on society as whole as the actual payer of each strategy can vary from individual households to utilities to state or federal entities. Explanation of each strategy system modeled and the applied cost is to follow.

2.9.1 Indoor Conservation

Indoor conservation (IC) is simulated in IUWM by applying the predefined household profile function for high efficiency new homes (HENH), outlined in the latest residential end use study (W. DeOreo & Mayer, 2016). Originally demand profile α and β for HENH was found by DeOreo in an analysis of new single family homes with high efficiency fixtures and appliances that meet or exceed EPA WaterSense specifications (DeOreo, 2011).

The level of medium adoption for *indoor conservation* was set at 50%, simulating the installation of high efficiency appliances and fixtures in half of all households in the modeled area, representing an ambitious, yet pragmatic adoption consistent with other practices. The adoption parameter of 50% was estimated considering the likelihood that half of households in the modeled area have older fixtures that could be replaced with more efficient fixtures.

The cost of installing a high efficiency clothes washer, dishwasher, two toilets and three faucet aerators was referenced from the EPA's 2005 Combined Retrofit Report (EPA, 2005). The cost per household was summed to \$1,584 and subsequently multiplied by the number of households adopting based on assumed by k_{ic}^{adopt} to attain a total cost of implementing the strategy (Table 10 and Eq. 4). This approach assumes that all appliances in a household converting to high efficiency need to be replaced and does not account for existing high efficiency fixtures. In the absence of knowledge of existing indoor water fixtures in residences, it is not possible to estimate on a fixture by fixture basis, and this approach applies a conservative cost estimate to *indoor conservation*.

Table 10: Indoor conservation unit costs per household

Fixture	No.	Unit Cost		Total Cost	
Toilets	2	$	363	$	726
Clothes Washers	1	$	818	$	818
Showerheads	2	$	13	$	25
Faucet Aerators	3	$	5	$	15
Total Cost per Household:				**$**	**1,584**

$$\boldsymbol{TC_{IC} = TC^{hsd} * 0.5 * n_i^{hsd}} \qquad \textbf{Eq. 4}$$

Where; TC_{IC} is total cost ($/year) of *indoor conservation*, TC^{hsd} is total cost per household ($/household/year), and n_i^{hsd} is the number of modeled households across spatial subunit *i*. Total cost was then annualized over a 15-year lifespan and 5% discount rate using the uniform series capital recovery factor equation (Eq. 3).

2.9.2 Efficient and Advanced Irrigation Systems

The baseline calibrated model estimated an outdoor demand accounting for 57%, 40%, and 50% of total annual demand in Denver, Miami, and Tucson respectively. Decreasing outdoor use through sprinkler efficiency improvements and the use of smart meters is one of the first strategies cities take to reduce potable demand. Reducing outdoor irrigation use can be accomplished through the installation of high efficiency sprinkler heads and more awareness of plant water requirements by the irrigator.

Efficient irrigation systems (EIS) simulates the use of high efficiency sprinkler heads and is modelled in IUWM by increasing the application efficiency k_i^{eff} in the daily irrigation depth calculation (Eq. 2) from 0.7 to 0.85. The total cost of *EIS* is an estimate of the cost of installing new high efficiency sprinkler heads to cover modeled irrigated area. The modeled irrigated area is the sum of each NLCD category area (open, low, medium, high density) multiplied by their respective calibrated percent irrigated area ($A_{i,c}$; section 2.3 – 2.5). Modeled irrigated area was divided by sprinkler spray area to determine number of heads to replace.

$$\boldsymbol{n^{heads} = \frac{\sum \textbf{irrigated area}}{\pi(sprinkler\ radius)^2}} \qquad \textbf{Eq. 5}$$

The average cost of a high efficiency spray sprinkler head is estimated at \$5.12, a value used in the 2016 EPA WaterSense specification report on spray sprinkler bodies (US EPA, 2016). This cost was multiplied by 1.5 to account for installation cost bringing the cost per head to \$7.68. Aggressive adoption of EIS considers the higher end of the line at \$7.00/head, plus same installation cost as prior. Average irrigated radius per spray sprinkler head was set at 12 ft. For medium adoption:

$$\boldsymbol{TC_{EIS} = \left(n^{heads}\right) * \left(\frac{\$5.12}{head} * 1.5\right)} \qquad \textbf{Eq. 6}$$

Total annual cost was attained by annualizing total capital over a 15-year lifespan and 5% discount rate using uniform series capital recovery factor equation (Eq. 3).

The *Advanced Irrigation Systems (AIS)* strategy simulates the installation of weather-based irrigation controllers in households with existing sprinkler systems. IUWM users select the AIS option and select a decreased percentage of demand resulting from the practice to model (see Table 8 for values assumed in this study). An average cost of these controllers was found to be \$155, considering the sale price of 10 EPA WaterSense certified systems gathered in a product search online. EPA estimates WaterSense irrigation controllers can save a typical home around 8,800 gallons of water per year (EPA WaterSense, 2017). To determine number of households to adopt this strategy, the estimated savings from AIS as a percentage of annual outdoor water demand was divided by the EPA estimated household savings of using irrigation controllers.

$$\boldsymbol{AIS\ k^{adopt} = \frac{AIS\ annual\ savings\ [gal]}{EPA\ annual\ savings\ per\ household\ [gal]}} \qquad \textbf{Eq. 7}$$

An estimated installation cost of \$100 per system was added to arrive at a cost per household of \$255. This figure was multiplied by the number of adopting households to arrive at the total cost. The following equations describe how the total cost of *AIS* is attained:

$$\boldsymbol{TC_{AIS} = \frac{\$255}{hsd} * k_{AIS}^{adopt}} \qquad \textbf{Eq. 8}$$

Total cost was annualized over a 15-year estimated lifetime of these systems at 5% discount rate using uniform series capital recovery factor equation (Equation 3).

2.9.3 Xeriscape Conversion

Xeriscape is modeled in IUWM by reducing the calibrated evapotranspiration plant factor of each city. The plant factor k_i^{pf} of xeriscape has been determined to approximate 0.33 of ET_0 for turf grass (Sovocool, 2005). A reduced k_i^{pf} applied across the modeled area simulates a combination of turf grass and xeriscape with lower plant water requirements, here called the *combined* plant factor; $k_{i,comb}^{pf}$ (Eq. 3). The fraction of irrigated area x_f converted to xeriscape for each city was determined by relating the scale between calibrated plant factor $k_{i,calib}^{pf}$ to xeriscape plant factor $k_{i,xeri}^{pf}$ shown in Eq. 6.

$$\boldsymbol{k_{i,comb}^{pf} = k_{i,calib}^{pf}(1 - x_f) + k_{i,xeri}^{pf}\,(x_f)} \qquad \textbf{Eq. 9}$$

The average cost of landscape conversion to *xeriscape* was estimated to be \$1.55 per square foot (Sovocool, 2005). The area estimated to be converted to *xeriscape* was attained as a function of the calibrated irrigated area for each NLCD urban class. Total cost of *xeriscape* strategy is outlined in the following equation:

$$\boldsymbol{TC_{xeri} = \frac{\$2.10}{sqft}(x_f * A_{i,c})} \qquad \textbf{Eq. 10}$$

Xeriscape total cost was annualized over a 50-year lifetime at a 5% discount rate using the uniform series capital recovery factor equation (Eq. 3).

2.9.4 Stormwater Use

Stormwater for toilet flushing and *irrigation & toilet flushing* strategy assumes neighborhood or sub-regional systems, where stormwater is collected over a small to medium urban watershed and stored in a large tank or detention basin before being treated and distributed (Table 11). Considering that stormwater for toilet use would need separate piping to reach households, *stormwater for toilet flushing* would be most feasible in a new development or redevelopment neighborhood. The cost of sub-regional systems was investigated in the *Stormwater Capture Master Plan* for the Los Angeles Department of Water and Power (LADWP, 2015). The report compiles the costs of stormwater collection and reuse systems in Los Angeles. Sub-regional or neighborhood-scale tank storage and reuse project total costs ranged from \$1,200 to just over \$6,000/AF captured depending on the complexity of the end use. Direct use total cost includes storage, pumps, and piping costs. Projects that met toilet flushing or other indoor uses were categorized as ‘complex direct use’ (LADWP, 2015). Therefore, the higher total cost attributed to ‘complex direct use’ projects of $6,000/AF ($18.41/kgal) was applied to the *stormwater toilet flushing stormwater* and *stormwater for irrigation & toilet flushing* strategies.

The total cost estimate for *stormwater for irrigation* strategy was referenced from a 2018 Southern California Water Coalition (SCWC) report. The SCWC analyzed 32 stormwater capture projects in southern California and calculated the total cost per acre-foot for each system. Capital costs were annualized at 5% over 30 years (SCWC, 2018). Project types included decentralized, centralized and retrofit systems. In the SCWC report, 25 of the 32 projects analyzed were retrofits of existing treatment plants. Retrofitting existing treatment plants to handle stormwater is a viable and attractive option as it is significantly less expensive than new decentralized treatment plants. To consider the implementation of both neighborhood regional and retrofit of existing plants, the reported median total cost of these projects was used; $1,070/AF ($3.28/kgal) to simulate *stormwater for irrigation*.

Stormwater for potable & irrigation was assigned the cost of centralized wastewater treatment in Denver ($3.28/kgal) plus high end cost of direct potable reuse (DPR) including conveyance cost ($6.14/kgal) (Raucher & Tchobanoglous, 2014); arriving at a total cost of $9.42/kgal (Table 11).

2.9.5 Roof Runoff Use

Roof runoff strategies in IUWM work by taking 20% of total stormwater runoff from impervious area for end-use allocation. The storage parameter in this study was set to 200 gal/household/day. The total cost for *roof runoff* for both combined and single end uses of *toilet flushing & irrigation* comes from the cost at the household level multiplied by the number of households adopting the practice shown previously in (Table 8). The cost of *roof runoff for irrigation* replicates a direct roof to landscape system. These systems include 50-gallon barrels connected to gutters and are generally easier to install to supplement solely outdoor demand. Reported cost of these systems run from $1,500-1,600 per household for both toilet flushing and irrigation (Luthy et al., 2016). A cost of $1,500 per household was applied to *roof runoff for irrigation.* A full household system with ultra violet (UV) treatment that includes toilet flushing as an end use has a reported cost of $1,600 and was applied to toilet flushing and dual irrigation and toilet flushing (TF&I) end uses (Luthy et al., 2016). An O&M cost per adopting household of $100 is added to account for replacing UV system bulb ($80) and additional energy and maintenance ($20) for roof runoff strategies with toilet flushing end use. Total cost was annualized at 5% over a 20-year life. Total cost was annualized over a 20-year lifetime at 5% discount rate using uniform series capital recovery factor equation (Eq. 3).

$$TC_{RR} = k_{rr}^{adopt} * TC^{hsd} \qquad \text{Eq. 11}$$

The combined *roof runoff for potable and irrigation* strategy was treated as a stormwater collection system. The cost of this strategy is the average of regional and sub-regional stormwater use systems plus the cost of direct potable reuse, including conveyance.

2.9.6 Graywater Reuse

IUWM simulates whole household graywater systems that incorporate bath, laundry, and non-kitchen sink graywater for end uses; toilet flushing, irrigation and potable. The cost applied for *Graywater for Irrigation* is that of a single household graywater system. Simple laundry-to-landscape systems require less treatment and are less expensive than a dual distribution system. However, the moderate and aggressive adoption levels considered for this study includes all household graywater for a specified end use. Graywater systems designed to collect all household graywater cost between $5,000 and $15,000 depending on treatment type and size of the home (Luthy et al., 2016). The NAP report also cites an example of a self-installed, whole-house, graywater collection and irrigation system with dual plumbing professionally installed during construction to cost $2,300. The cost of $2,300 was used and multiplied by the number of adopting households to attain a total capital cost. Total capital was annualized using the uniform series capital recovery factor equation (Eq. 3). In addition, total O&M cost of $100 per adopting households was included in the total annualized cost.

Graywater for toilet flushing, as well as the combined *toilet flushing & irrigation* end use was assigned a cost approximating that of a system serving multiple households or apartments. Cost estimates of larger scale graywater projects is sparse, however efficiencies of scale would be associated with graywater toilet flushing systems in large, new multi-residential developments (Luthy et al., 2016). Such systems would require separate distribution to supply graywater to a neighborhood or multi-residential units. Constructing these in new developments would be much more cost effective than retrofitting existing neighborhoods or multi-residential buildings (Luthy et al., 2016). Therefore, this study assumes costs associated with installation of graywater for toilet flushing in new or redevelopment neighborhoods and/or multi-residential buildings.

A prototype graywater system for toilet flushing and irrigation end uses was installed in Aspen Hall dormitory at Colorado State University in 2008. Hodgson estimated the total or life-cycle cost of graywater treatment systems of different treatment types and sizes ranging from 50-5,000 GPD, to simulate use of a range of multi-residential scales (Hodgson, 2012). Using the spreadsheet developed by Hodgson, a 5,000 GPD sand filter system total cost including capital, energy and O&M was calculated to be \$4.49/kgal, annualized at 5% over a 30-year life. The Aspen Hall pilot system study reported a plumbing, collection and distribution piping' to be 30% of the total capital cost (Cordery-cotter, Sharvelle, & Ph, 2014). This percentage of sand filter capital cost was annualized and added, arriving at a total annualized cost of \$4.62/kgal.

For the *graywater for potable* & *irrigation* strategy, total cost was assumed to be that of centralized wastewater treatment with the addition of direct potable reuse (DPR) treatment. Graywater would need to be treated similar to wastewater if used for potable end uses (Hill, Owen, & Trussell, 2017). Wastewater treatment cost is the wastewater billing rate for each city, gathered from their respective utility websites. Wastewater billing rates per thousand gallons of each city are; \$4.32 for Denver, \$4.47 for Miami and \$4.76 for Tucson. Total cost of DPR includes complete advanced treatment (CAT), conveyance, and brine management runs between \$820/AF and \$2000/AF (Raucher & Tchobanoglous, 2014). The high end of DPR of \$2,000/AF (\$6.14/kgal), plus the respective wastewater treatment cost in each city was applied to water processed through *graywater for potable* & *irrigation*.

2.9.7 Wastewater Reuse

The strategies of; *wastewater for toilet flushing, irrigation* and the combined *toilet flushing & irrigation* end use, simulate centralized non-potable reuse facilities and separate distribution for use. Wastewater reuse system costs are referenced from the 2016 Pacific Institute report; *The Cost of Alternative Water Supply and Efficiency Options in California* (Cooley & Phurisamban, 2016). The total cost of seven non-potable reuse projects including distribution with capacities less than 10,000 acre-ft per year are reported to have a median cost of $4.73/kgal of processed wastewater. Wastewater reuse strategies in this study process 25% of total wastewater generated across the modeled area. This annual volume is less than 10,000 acre-ft in Denver and Tucson and about 14,000 acre-ft in Miami, theoretically requiring just one to two small-scale reuse systems in practice. Here, use of cost for 'small' reuse projects of capacity less than 10,000 acre-ft per year was found to be appropriate.

Distribution cost is reported to be $950/AF or $2.92/kgal. An additional 10% of this distribution cost was included for toilet flushing and combined toilet flushing and irrigation end uses, to account for additional cost of distribution to reach individual households. Resulting total costs applied are $4.73/kgal for irrigation end use and $5.02 for combined toilet flushing and irrigation end use.

Wastewater for potable & irrigation reuse strategy is replicating the practice of direct potable reuse (DPR) from a complete advanced treatment (CAT) facility. Once treated, this water is generally introduced into water supply upstream of a water treatment plant. Total cost of DPR including CAT, treatment, conveyance, and brine management may run between $820/AF and $2000/AF (Raucher & Tchobanoglous, 2014). The upper end cost of $2,000 AF ($6.14/kgal) was used in analysis to account for pipeline construction and other requirements to ensure safe delivery of water with reliability

2.9.8 Total Annualized Costs

The total annualized costs for each strategy are presented in Table 12. These values were applied in the optimization procedure along with total annual demand (gallons).

Table 11: Strategies' total annualized costs by city, rounded to nearest thousand dollars

	Total Annualized Cost		
Strategy	**Denver**	**Miami**	**Tucson**
Conservation Strategies:			
Indoor Conservation	$ 19,977,000	$ 22,472,000	$ 11,161,000
Xeriscape	$ 109,012,000	$ 136,844,000	$ 46,686,000
Efficient Irrigation Systems	$ 3,500,000	$ 5,609,000	$ 2,091,000
Advanced Irrigation Systems	$ 5,457,000	$ 3,921,000	$ 2,771,000
Graywater Reuse for:			
Irrigation	$ 18,161,000	$ 20,429,000	$ 10,146,000
Toilet Flushing	$ 4,893,000	$ 6,985,000	$ 3,339,000
Toilet Flushing & Irrigation	$ 4,830,000	$ 9,934,000	$ 4,879,000
Potable & Irrigation	$ 13,814,000	$ 25,625,000	$ 12,684,000
Roof Runoff for:			
Irrigation	$ 920,000	$ 3,232,000	$ 988,000
Toilet Flushing	$ 1,746,000	$ 4,096,000	$ 1,252,000
Toilet Flushing & Irrigation	$ 1,746,000	$ 4,096,000	$ 1,252,000
Potable & Irrigation	$ 3,361,000	$ 18,056,000	$ 1,419,000
Stormwater Reuse for:			
Irrigation	$ 19,351,000	$ 37,608,000	$ 10,325,000
Toilet Flushing	$ 62,397,000	$ 91,823,000	$ 37,316,000
Toilet Flushing & Irrigation	$ 127,621,000	$ 267,197,000	$ 61,130,000
Potable & Irrigation	$ 75,227,000	$ 168,396,000	$ 36,846,000
Wastewater Reuse for:			
Irrigation	$ 6,218,000	$ 21,072,000	$ 9,727,000
Toilet Flushing	$ 16,573,000	$ 23,659,000	$ 11,307,000
Toilet Flushing & Irrigation	$ 16,573,000	$ 23,659,000	$ 11,307,000
Potable & Irrigation	$ 18,010,000	$ 26,249,000	$ 12,515,000

2.10 Multi-Objective Optimization

A set of 374 combinations of up to three single water demand reduction strategies were run in IUWM at the medium level of adoption described in Table 8. The combinations consist of all single strategies viable in conjunction with each other. The analysis was conducted to investigate the water demand reduction benefits of combined strategies, the tradeoffs between total cost and reduced demand for traditional water sources. A set of the 20 single strategies with a more aggressive adoption was modeled to compare with the medium adoption combinations.

A multi-objective optimization was conducted on the model solutions by applying a non-dominated ranking algorithm that minimizes total annual water demand and total annual cost. This produced a set of solutions in a ranking succession that offer tradeoffs between water demand and cost. The first ranking solutions are non-dominated and can be considered as the optimal front. In succession, 2nd ranking solutions are beat by at least one 1st ranking in both demand and cost. 3rd ranking solutions are beat by at least two other solutions, and so forth.

The frequency of single strategies within solutions ranking 1-3 was calculated. Solutions up to 3rd ranking were determined to be good solutions for their proximity to the non-dominated front (Figures 10-12). This allows us to calculate the relative frequency of strategies taking part in a larger set of top performing solutions for each city, rather than considering solely the non-dominated solutions.

Total annual water demand for traditional sources is defined as the average annual water demand met by the traditional water supply sources for a municipality. Use of alternative sources does not necessarily reduce water demand but replaces traditional water sources with other water sources (i.e., wastewater, stormwater, roof runoff, or graywater), treated to either non-potable or potable quality. In this paper, use of the term water demand reduction refers to demand for traditional water supplies.

2.10.1 Three-Variable Optimization

A three-variable optimization was conducted on the same set of solutions for which the non-dominated ranking procedure was applied with the goal of minimizing wastewater outflow in addition to annual cost and annual water demand. To be discussed fully in section 3.3.

CHAPTER 3: RESULTS AND DISCUSSION

3.1 Strategy Demand-Reduction Results

The demand reduction potential of the twenty single strategies varies considerably between one another and variabilities of the same strategy occur between cities as well (Table 12). We can compare the demand reduction potential of conservation to alternative source strategies and begin to assess the localized factors that influence performance between cities (Table 12). We note the large demand reduction potential of *stormwater for end uses* in Miami and the large potential of *xeriscape* in Denver. Similarities exist between Denver and Tucson in strategies' performance.

Table 12: Annual total demand reduction of single strategies at medium adoption

Conservation Strategies	Denver	Miami	Tucson
Xeriscape	22%	12%	12%
Efficient Irrigation Systems	9%	7%	8%
Advanced Irrigation Systems	6%	4%	5%
Indoor Conservation	2%	6%	6%
Alternative Source Strategies			
SW for I	17%	33%	16%
SW for TF	10%	14%	10%
SW for TF&I	20%	41%	17%
SW for P&I	21%	45%	17%
WW for I	4%	13%	10%
WW for TF	10%	13%	11%
WW for TF&I	10%	13%	11%
WW for P&I	9%	12%	10%
GW for I	3%	7%	6%
GW for TF	3%	4%	4%
GW for TF&I	3%	6%	5%
GW for P&I	4%	7%	6%
RR for I	0%	3%	0%
RR for TF	1%	2%	4%
RR for TF&I	1%	6%	6%
RR for P&I	1%	7%	5%

Percent annual demand reduction from total = sum of residential indoor and total outdoor demand

Demand reduction potential is not the full picture however, inclusion of cost in the ranking procedure of this study allows one to more adequately assess the feasibility and value of these strategies.

3.2 Solutions by City

The following sections will show the optimization results and successive ranking sets for each city. A set of non-dominated solutions selected for a closer look, referred hereafter as *standout solutions*, are presented in Tables 11 - 13. *Standout solutions* are those that provide considerable demand reduction without a steep rise in cost from the previous solution or are located just before a steep rise in cost without a considerable reduction in demand. Also included is the annual cost of traditional wastewater treatment applied to the wastewater volume outflow under the baseline condition (Figures 10 – 12). This cost is approximated using wastewater billing rate per thousand gallons. The baseline cost of wastewater treatment serves as a point of reference to compare relative costs of implementing the strategy combinations to the cost of treating wastewater with no strategies implemented.

3.2.1 Denver

Figure 10 displays all solutions for Denver in terms of annual demand in gallons per capita per day (GPCD) and annualized cost per capita. The modeled baseline annual demand for Denver is 104 GPCD and the cost of wastewater treatment in Denver is $4.34/kgal as of 2018, (denvergov.org). Comparing total cost of solutions to wastewater treatment cost at baseline, allows us to see that large savings can come at a cost not entirely prohibitive. Non-dominated solutions saving up to 20% of annual demand cost no more than a third of regular wastewater treatment (Figure 10).

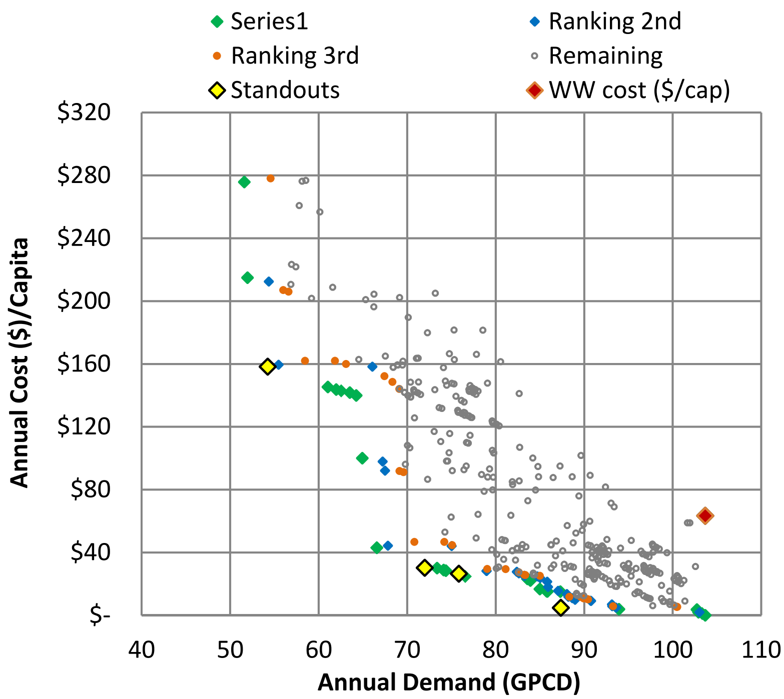


Figure 10: Denver plot of all solutions by annual cost per capita and annual demand reduction (%). Top performing solutions separated by color into non-dominated, 2nd, 3rd ranking solutions.

Under $20/capita, *efficient and advanced irrigation systems* and *graywater for toilet flushing* appear in standout solutions. *Aggressive EIS* on its own is a noteworthy standout solution. Above $20/capita, we see combinations of *xeriscape*, *wastewater for toilet flushing & irrigation* and *stormwater for irrigation* (Table 13).

Table 13: Denver's standout non-dominated solutions and baseline

Standouts	Annual Demand GPCD	Annual Cost	Annual Cost/Capita	Annual Demand Reduction %
SW for I + XS + WW for TF&I	54	$ 143,098,482	$ 158.21	48%
AIS + EIS +SW for I	72	$ 27,386,772	$ 30.28	31%
RR for TF&I + EIS + SW for I	76	$ 24,018,300	$ 15.31	17%
Aggressive EIS	87	$ 4,356,701	$ 4.82	16%
Baseline (WW cost $/cap)	104	$ 57,338,916	$ 63.39	-

AIS = advanced irrigation systems. EIS = efficient irrigation systems; XS = xeriscape; IC = indoor conservation; GW = graywater; RR = roof runoff; SW = stormwater; WW = wastewater; End uses: TF = toilet flushing; I = irrigation; P = potable

3.2.2 Miami

The standout solutions identified in Miami's solutions plot (Figure 11) are composed of the following strategies; under \$20/capita, the combination of; *graywater for toilet flushing*, *wastewater for irrigation*, and *roof runoff for irrigation* (Table 14). Greater than \$20/capita, we see solutions with combinations of *stormwater for irrigation, wastewater for toilet flushing & irrigation* and *efficient irrigation systems*. With abundant rainfall throughout the year in Miami and high impervious area (31%), stormwater reuse has a large potential for water demand reduction. However, the high volume available and in turn collected in this analysis translates to a high cost (Table 11).

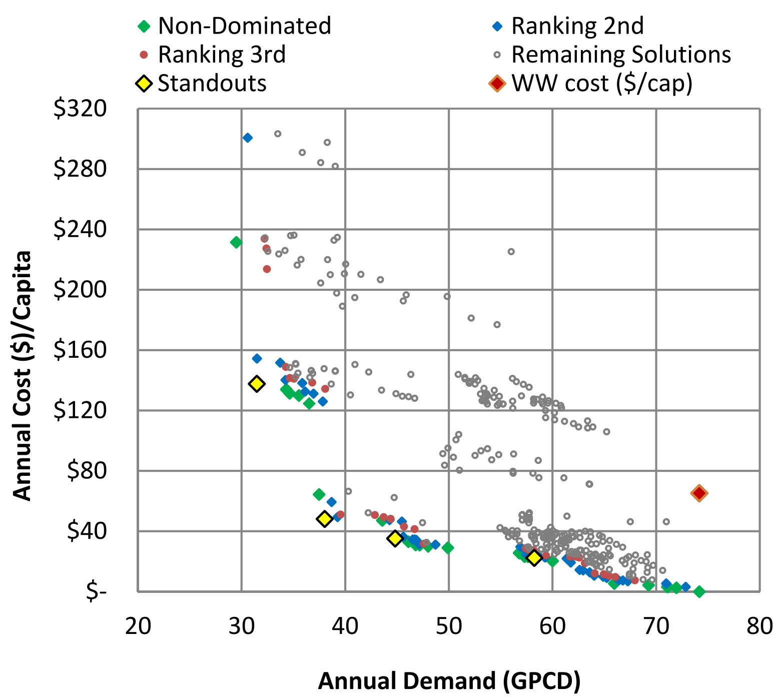


Figure 11: Miami plot of all solutions by annual cost per capita and annual demand reduction (%). Top performing solutions separated by color into non-dominated, 2nd, 3rd ranking solutions.

Table 14: Miami's standout non-dominated solutions and baseline

Standouts	Annual Demand GPCD	Total Annual Cost	Annual Cost/Capita	Annual Demand Reduction %
SW for P&I + EIS + WW for TF	31	$ 178,026,716	$ 137.70	58%
SW for I + EIS + WW for TF&I	38	$ 62,204,009	$ 48.11	49%
GW for TF + EIS + SW for I	45	$ 45,530,263	$ 35.22	40%
RR for I + EIS + WW for I	58	$ 28,969,082	$ 22.41	21%
Baseline (WW cost $/cap)	74	$ 84,308,048	$ 65.21	-

AIS = advanced irrigation systems. EIS = efficient irrigation systems; XS = xeriscape; IC = indoor conservation; GW = graywater; RR = roof runoff; SW = stormwater; WW = wastewater; End uses: TF = toilet flushing; I = irrigation; P = potable

3.2.3 Tucson

Four standout solutions of interest were identified in Tucson's non-dominated solutions plot (Figure 12), for costs under $20/capita: *efficient* and *advanced irrigation systems, indoor conservation*, and *roof runoff* for *irrigation.* Greater than $20/capita, we find combinations of; *efficient irrigation systems*, *indoor conservation* and *stormwater for irrigation* (Table 15).

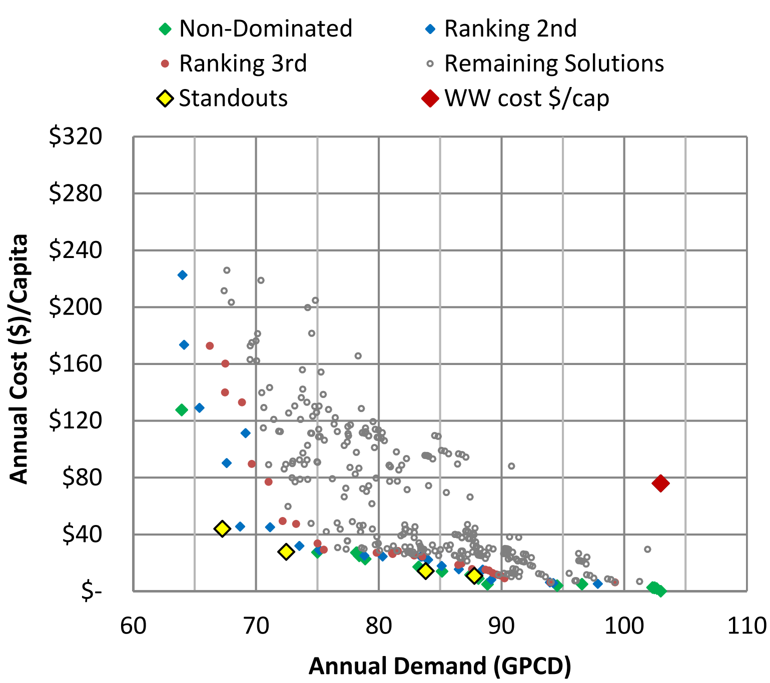


Figure 12: Tucson plot of all solutions by annual cost per capita and annual demand reduction (%). Top performing solutions separated by color into non-dominated, 2nd, 3rd ranking solutions.

Table 15: Tucson's standout non-dominated solutions and baseline

Standouts	Annual Demand GPCD	Total Annual Cost	Annual Cost/Capita	Annual Demand Reduction %
SW for I + EIS + WW for TF	67	$ 23,344,338	$ 44.00	35%
EIS + IC + SW for I	72	$ 14,712,160	$ 27.73	30%
AIS + IC + EIS	84	$ 7,537,650	$ 14.21	19%
EIS + IC + RR for I	88	$ 5,754,700	$ 10.85	15%
Baseline (WW cost $/cap)	103	$ 42,907,150	$ 75.95	-

AIS = advanced irrigation systems. EIS = efficient irrigation systems; XS = xeriscape; IC = indoor conservation; GW = graywater; RR = roof runoff; SW = stormwater; WW = wastewater; End uses: TF = toilet flushing; I = irrigation; P = potable

3.3 Demand Reduction Potential Across Study Cities

3.3.1 City Non-Dominated Fronts

The non-dominated solution fronts for each city are plotted together by annual cost per capita and percent annual demand reduction (Figure 13). Miami's non-dominated solutions offer a greater percent demand reduction for the same cost as the non-dominated solutions for Denver and Tucson. For Tucson, the highest solution achieves 38% annual demand reduction while Denver and Miami see solutions achieving up to 50% and 60%, respectfully. Wastewater reuse has greater impact in a more densely populated area such as Miami. In addition, stormwater reuse has great potential in Miami due to its high annual rainfall.

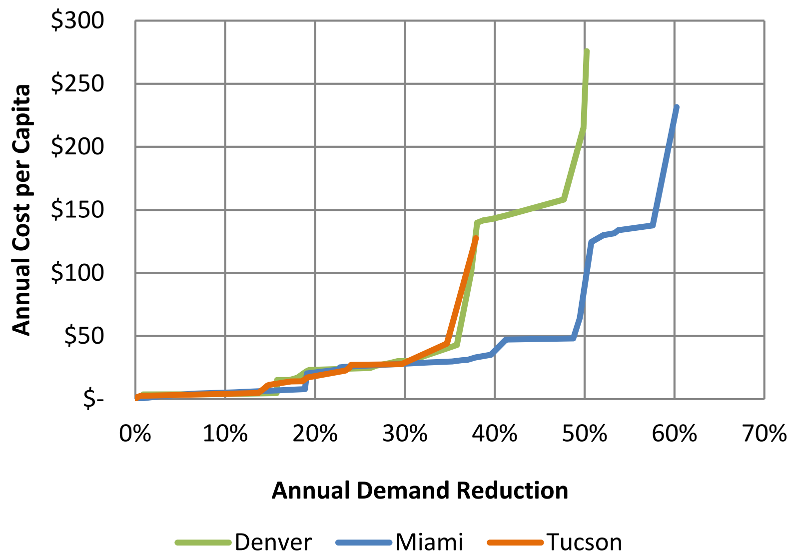


Figure 13: City comparison of non-dominated solution fronts by annual cost per capita and annual demand reduction (%)

Understanding how indoor and outdoor demands are allocated (Table 16) can inform why certain strategies reduce more demand in one city than another as we have seen (Table 12). Miami has a higher population density overall with an average household size of three people per household. It also has a much lower baseline demand of 74 gallons per capita per day (GPCD) and a lower outdoor annual demand of 40% of total. Denver has the highest outdoor demand at 57% of total and a low average household size of 2.3 people per household. Tucson has very similar characteristics to Denver, with an estimated annual outdoor demand of 50% and baseline per capita demand of 103 GPCD. The difference between baseline indoor demand and high efficiency indoor is the savings resulting from the *indoor conservation* strategy (Table 16).

Table 16: Explanatory baseline values by city

	Denver	Miami	Tucson
Outdoor demand fraction	57%	40%	50%
Average household size	2.3	3.0	2.4
Total per capita demand (GPCD)	104	74	103
Baseline indoor demand (GPHD)	104	132	127
High efficiency indoor (GPHD)[1]	93	107	95

[1]High efficiency indoor is calculated using average household size for each city. GPCD = annual gallons per capita per day, GPHD = annual gallons per household per day

3.3.2 Demand Reduction Strategy Frequency in Optimal Solutions

The relative frequency metric highlights strategies most prevalent among solutions ranking 1-3 i.e. the top performing solutions in terms of cost and demand reduction (Figure 14). Of note, the aggressive adoption set of strategies was not included in this analysis. *Efficient irrigation systems* and *stormwater for irrigation* are the most cost-effective strategies across the three cities (Figure 14). The variation in frequency of strategies between cities can be noted (Figure 14). For example, *xeriscape* is more prominent in Denver however *indoor conservation* is less impactful in Denver than Miami and Tucson. *Wastewater for irrigation* is more effective in Miami than other study cities, likely a result of high population density (Table 1) and thus large production of domestic wastewater. Reuse strategies with sole end use *toilet flushing* were removed from representation (Figure 14), as they performed approximately as well as the combined *toilet flushing & irrigation*. Other strategies occurring in less than 4% of combinations in all three cities were removed from the representation for better visualization.

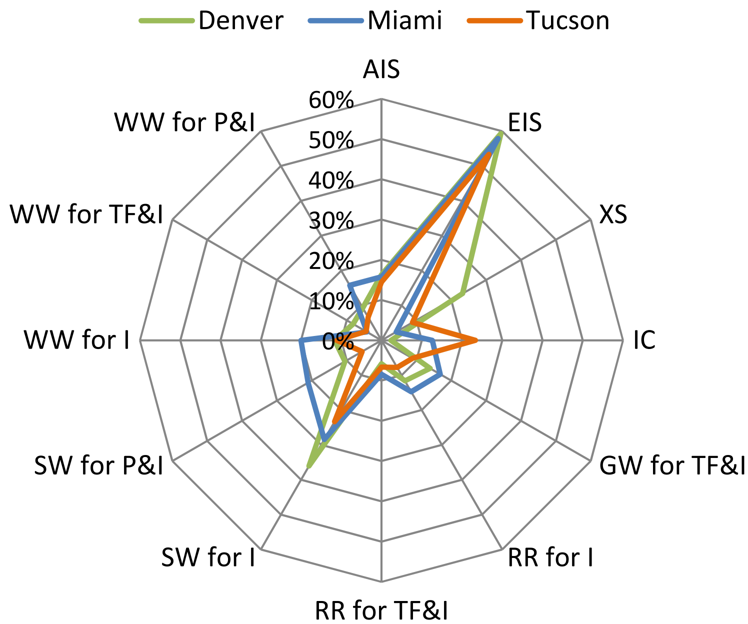


Figure 14: Relative frequency of single strategies in top ranking combinations of the 2-variable optimization. AIS = advanced irrigation systems. EIS = efficient irrigation systems; XS = xeriscape; IC = indoor conservation; GW = graywater; RR = roof runoff; SW = stormwater; WW = wastewater; End Uses: TF = toilet flushing; I = irrigation; P = potable.

The optimal solutions ranking 1-3 for each city were divided into three cost ranges or brackets: (1) less than \$20, (2) \$20 – \$40 and (3) greater than \$40 per capita per year. The relative frequency strategies within each cost category is shown stacked together for each city (Figure 15). Light, medium and dark color shading for each city color represents the low, medium and high cost brackets, respectfully. This demonstrates the cost ranges of solutions the strategies tend to be part of. We also see which strategies are most prevalent overall. *Efficient irrigation systems* is the most frequent strategy occurring in optimal solutions and appears in low, medium and high cost solution scenarios (Figure 15). *Stormwater for irrigation* is also frequently part of optimal solutions across the study cities. However, this solution does not appear in low cost solutions (Figure 15). Indoor conservation appears more frequently in optimal solutions in Tucson compared to Denver and Miami and is often part of low-cost solution scenarios. This is due to the relatively large potential for demand reduction via indoor conservation in Tucson compared to other cities (Table 14).

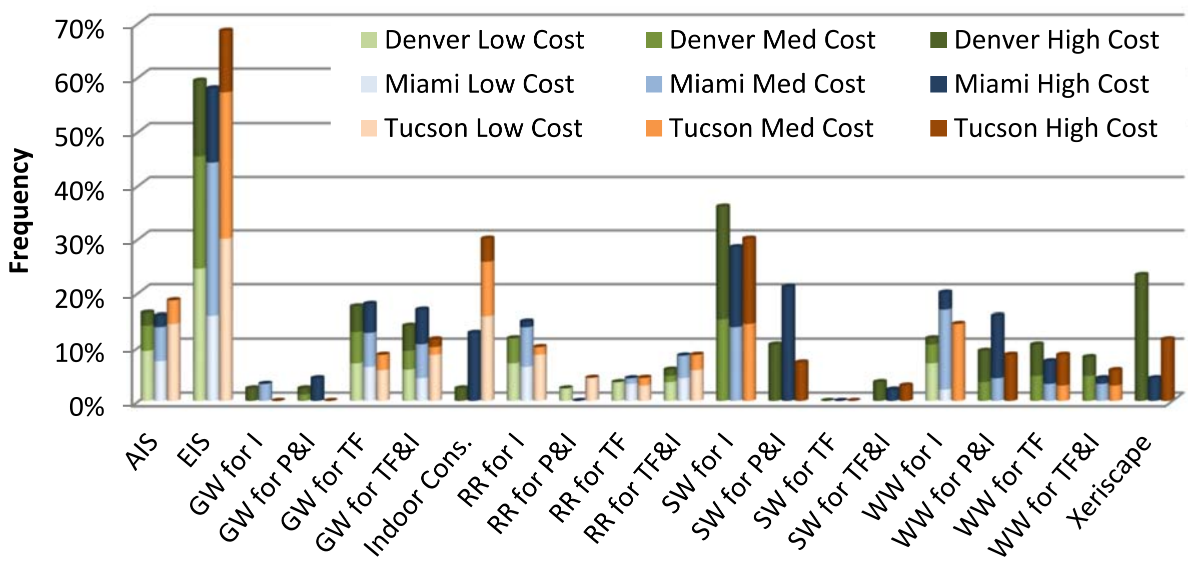
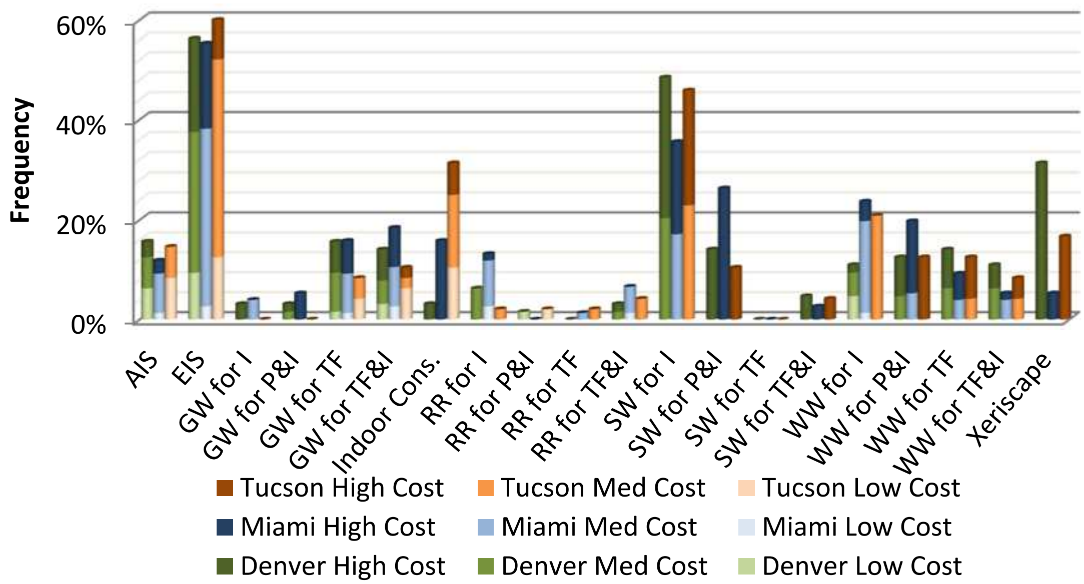


Figure 15: Frequency of single strategies within top ranking solutions separated into three cost tiers. Cost tiers; Low: <$20.00/capita, Medium: $20.00-$40.00/capita, High: >$40.00/capita. Total cost of solutions is represented by light to dark shading for each city's color. AIS = advanced irrigation systems. EIS = efficient irrigation systems; XS = xeriscape; IC = indoor conservation; GW = graywater; RR = roof runoff; SW = stormwater; WW = wastewater; End Uses: TF = toilet flushing; I = irrigation; P = potable.

Next, the frequencies of each strategy within the defined cost brackets were constrained by minimum demand reduction targets of the solutions they take part in. Solutions were sorted by those with greater than 15% and those greater than 30% annual demand reduction (Figure 16). Strategy frequencies or ratios in both sets are taken from the number of solutions greater than 15% demand reduction. While solutions that achieve greater than 15% demand reduction do include some low-cost solution scenarios, most are medium and high cost (Figure 16A). Results enable identification of strategies that are part of solutions achieving 30% or greater demand reduction and stay within the medium brackets. For example, in Miami's solutions achieving greater than 30% demand reduction, 17% of them contain *stormwater for irrigation* and are within the medium cost bracket (Figure 16B). For Tucson and Denver, the same strategy is most often in the high cost bracket. This likely due to the higher irrigation demands relative to annual precipitation) in Denver and Tucson (Table 1).

(A) Greater than 15% demand reduction



(B) Greater than 30% demand reduction

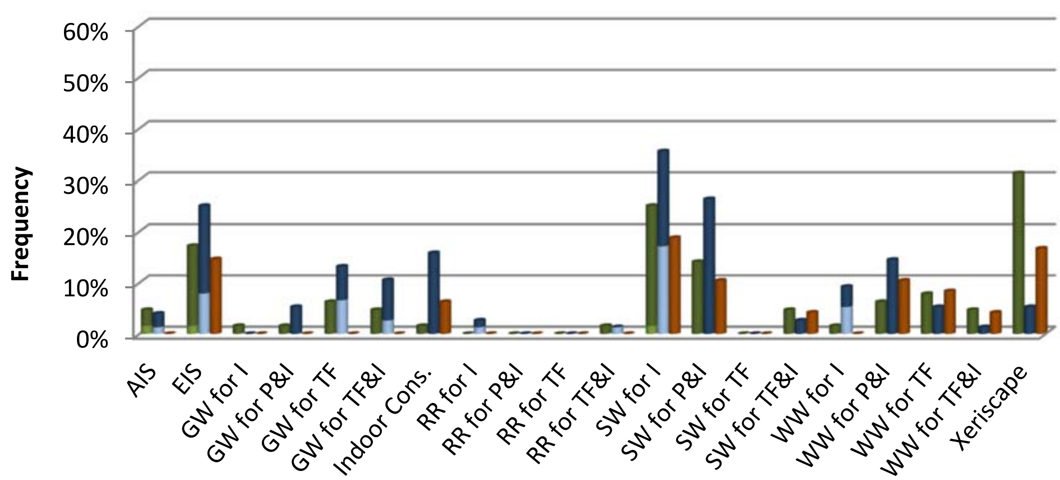


Figure 16: (A) Frequency of single strategies in solutions with 15% or greater annual demand reduction. (B) Frequency of single strategies in solutions with 30% or greater annual demand reduction. (A) & (B) separated into three cost tiers; Low: <$20.00/capita, Medium: $20.00-$40.00/capita, High: >$40.00/capita. Total cost of solutions is represented by light to dark shading for each city's color. AIS = advanced irrigation systems. EIS = efficient irrigation systems; XS = xeriscape; IC = indoor conservation; GW = graywater; RR = roof runoff; SW = stormwater; WW = wastewater; End Uses: TF = toilet flushing; I = irrigation; P = potable.

3.4 Discussion

Outdoor Water Use Efficiency Strategies

Efficient irrigation systems are shown to be the most cost-effective strategy for reducing outdoor demand. On its own, such systems can reduce demand by between 7-9% in Denver, Miami and Tucson, under medium adoption (Table 12). In comparison between cities, demand reduction achieved from *efficient irrigation systems* is higher in Denver because outdoor demand accounts for 57% of total, a higher fraction than Miami (40%) and Tucson (50%) (Table 16). *Xeriscape* is more impactful in Denver due again to the higher fraction of outdoor demand to total and prominence of irrigated turf grass landscape, as indicated by a calibrated k_i^{pf} of 0.8. Water requirements of turf grass are high in Denver throughout its hot, arid summer. In Tucson, the landscape consists of native desert vegetation with some turf grass, influencing a calibrated k_i^{pf} of 0.65. In comparison, calibrated k_i^{pf} was 0.68 for Miami. *Xeriscape* achieves high demand reductions of 22% in Denver and about 12% in both Miami and Tucson, however the strategy is expensive (Table 11) and we see that it takes part in only high-cost solutions (Figure 16B).

Indoor Efficiency Strategies

Indoor conservation (IC) reduces overall demand by 6% in both Miami and Tucson and 2% in Denver. The strategy is less impactful in Denver because baseline indoor use per household is already low on average (Table 16) and closer to the level of when high efficiency fixtures are widely adopted. The total cost of *IC* was the sum cost of installing high efficiency appliances in half of modeled households. Additional work could be done to vary the number of households adopting *IC* to optimize the total cost and demand reduction. *IC* complements the effectiveness of alternative source strategies that supplement toilet flushing and potable end uses, reducing potable water demand even further. In Denver, *IC* occurs less frequently in top solutions because the calibrated demand profile function reflects a low modeled indoor use of 104 GPHD at its average household size, just slightly higher than its modeled use for high efficiency fixture homes, (Table 15). *Indoor conservation* is more effective in Miami and Tucson, where baseline or current indoor use per household is higher, and there is more potential for indoor conservation.

Stormwater and Wastewater Reuse

Of the four alternative source strategies, stormwater use has the greatest potential to offset potable water demand in all three cities. A reflection of the large volume for reuse it can generate during precipitation events over a city. In the scenarios run in this study, solely stormwater reuse reduced potable demand by between 10-20% in Denver, 14-45% in Miami, and 10-17% in Tucson, depending on the end use (Table 12). There are policy issues that exist in Denver with respect to water law that prohibit widescale use of stormwater to meet water demand. Because use of stormwater can be highly effective for water demand reduction in Denver, policy changes that allow use of stormwater may be beneficial.

Stormwater strategies' performance varies considerably by end use considering the differences in cost between system scales as outlined in section 2.7.4. *Stormwater for irrigation* is the best performing end use for stormwater because of a lower cost reflecting its lower treatment requirements and greater feasibility. However, it is important to note that while *stormwater for irrigation* is frequently part of optimal solutions across all study cities, it does not appear in low cost solution scenarios (Figure 15). *Stormwater for toilet flushing* is more complex and expensive as it requires dual distribution piping. Alternative source performance is largely stipulated by the end use and if it can benefit from an economy of scale.

Precipitation increases stormwater and roof runoff potential for all end uses. In Denver and Tucson, most precipitation events do not necessarily occur during summer months when irrigation demand is high. Despite this the rainfall that does occur, usually in the spring and early summer in Denver and late summer monsoon season in Tucson, can substantially reduce freshwater demands in the successive days after an event with the storage capacities simulated (3,000 gal/hsd).

Wastewater reuse strategy simulates the reuse of 25% of generated wastewater in each city. It is most frequent in high demand reduction, high cost solutions. The total cost applied discussed in section 2.8.7, represents that of a centralized non-potable reuse facility including distribution costs.

Graywater and Roof Runoff Reuse

Graywater reuse strategies present a relatively low performance in terms of cost and demand reduction. There is a high total cost when spanned across a city at the scale of adoption set, relative to the resulting demand reduction. Graywater reuse demand reduction varies little between end uses but cost benefit performance is highly sensitive to the differences in cost related to each end use. Miami and Tucson have a higher population density or average household size and saw graywater demand reduction potential between 4-6% between end uses while Denver saw just 3-4% savings (Table 16). Strategies at the single residence scale do not benefit from economies of scale like stormwater can. However, graywater systems can offer payback periods of as low as 5 years (Luthy et al., 2016). *Graywater reuse for irrigation* does not provide enough volume to meet water requirements of homes with turf grass landscapes. However, it can be impactful in arid regions such as Tucson to supplement low water use landscapes (Luthy et al., 2016). This is demonstrated in the results; 11% demand reduction was observed in Tucson for use of *graywater for irrigation* with *xeriscape* versus solely *xeriscape*, almost twice that of the same comparison in Denver and Miami (Table 17).

Table 17: Total annual demand reduction from combined graywater for irrigation and xeriscape

	Xeriscape	**GW for I + XS**	**Added Demand Reduction**
Denver	39%	44%	4%
Miami	12%	18%	6%
Tucson	24%	35%	11%

Roof runoff collection and use systems have a longer payback period of 5-26 years depending on storage volume and end-use. Very low demand reduction potential was observed for roof runoff collection and use across all study cities (Table 12). *Roof runoff for irrigation* systems are less effective in locations with distinct wet and dry seasons (Luthy et al., 2016), which is the case for Denver and Tucson especially in the arid west. However, these cities can still benefit from spring storms and monsoons to supplement landscape demands. While frequency that roof runoff collection and use is low compared to other strategies (Figure 14), it is most often part of low-cost solutions (Figure 15).

Comparing strategies between cities, we see higher savings when a strategy targets the greater fraction of indoor or outdoor demand to total. For example, Denver's outdoor demand accounts for 57% of total for this irrigation conservation and xeriscape strategies save slightly more than in Miami and Tucson.

3.5 Three Variable Optimization: Cost, Demand, and Wastewater Output

Cities with combined sewer and stormwater or those with wastewater treatment plants at risk of reaching capacity may look to solutions to reduce wastewater volumes entering their treatment systems. Many cities in the Northeast and Great Lakes region have combined sewer systems and face the problem of combined sewer overflows (CSOs) (U.S. EPA, 2004). CSOs occur when collected stormwater exceeds capacity of the receiving infrastructure or wastewater treatment plant, causing excess wastewater to flood streets or to be discharged into nearby water bodies, (U.S. EPA, 2004). In cities with combined sewers, reducing immediate wastewater volumes through the reuse of graywater and can defer investments to expand wastewater treatment facilities (Luthy et al., 2016). IUWM does not model stormwater intake into the wastewater stream as would be the case with combined sewer systems. However, it does quantify the reductions in wastewater output through indoor conservation, graywater reuse and alternative wastewater treatment for non-potable reuse.

IUWM strategies of *indoor conservation* and *graywater reuse* can offset or reduce the volume of wastewater reaching the treatment facility. Wastewater for non-potable reuse can do the same if it is a decentralized treatment system. In many cases, centralized wastewater treatment plants are retrofitted to treat and redistribute wastewater for non-potable reuse (Cisneros, 2014).

The relative frequency of strategies within the non-dominated solutions of the 3-variable optimization is compared to the relative frequency of 1-3 ranking solutions of the 2-variable optimization shown prior, for each city (Figure 17).

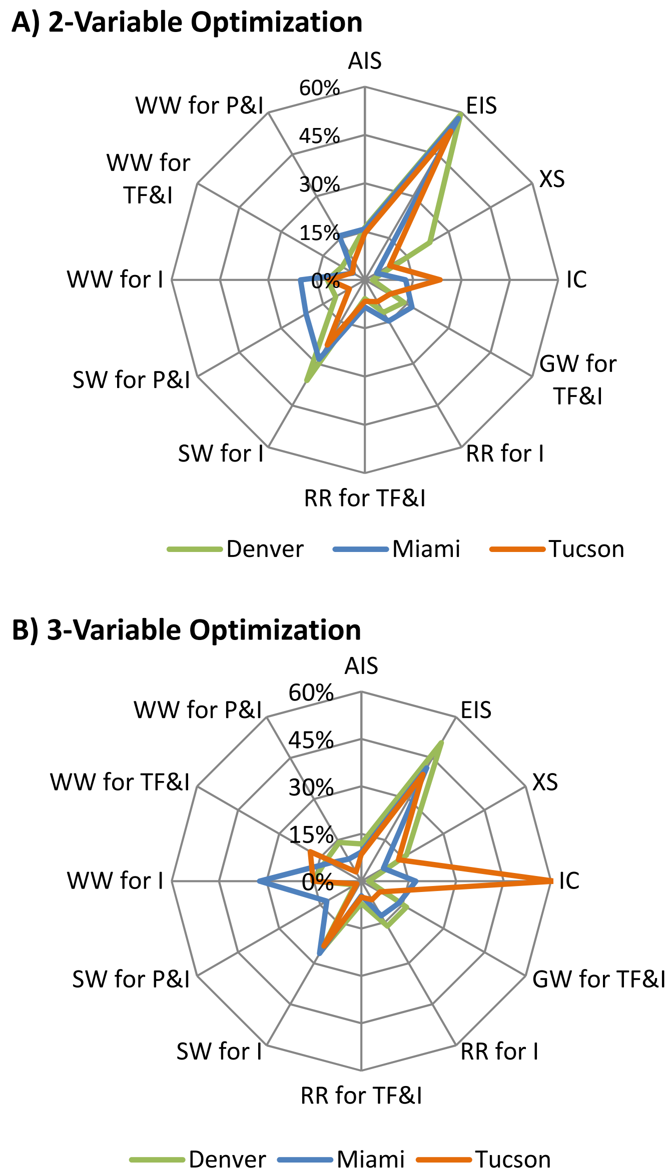
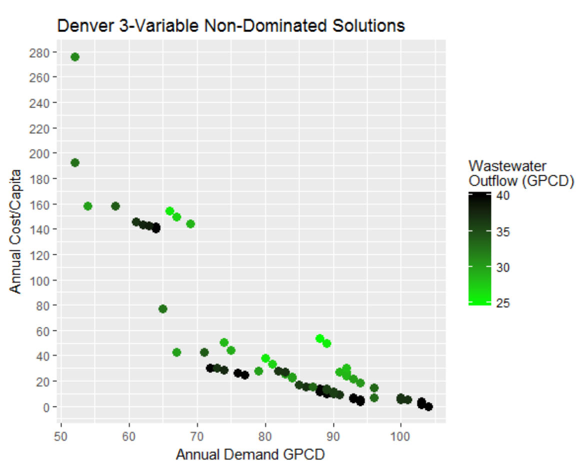


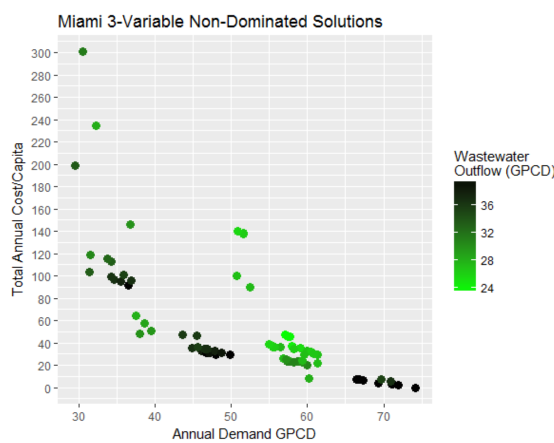
Figure 17: A) Relative frequency of strategies in non-dominated solutions resulting from the 2-variable optimization. B) Relative frequency of 1-3 ranking strategies in solutions resulting from the 3-variable optimization

Comparing the 3-variable frequency results to the 2-variable results; *Indoor conservation* is much more frequent in Tucson, sees a rise in frequency in Miami but does not see much change in Denver. *Indoor conservation* takes part in 60% of Tucson's 3-variable non-dominated solutions likely due to the cost factor being on a per household basis. The number of households in Tucson's modeled area is less than Miami and Denver. This keeps the total cost of *IC* low relative to water demand and wastewater reduction allowing *IC* to appear in more optimal solution scenarios in Tucson than in Denver and Miami.

We see that wastewater strategies increase in frequency when minimizing wastewater outflow with the 3-variable optimization. In Miami especially, due to its higher population density. Cost differences between wastewater end uses (Table 9) influence their frequency. Strategies differ between cities due to the proportion of indoor and outdoor demands and the volume of wastewater available for reuse. For example, Miami generates a larger amount of wastewater in proportion to its outdoor demands than Denver and Tucson.

The 3-variable non-dominated solutions for each city are presented in Figure 18; lightest green points are those with lowest wastewater output. Optimal solutions consist of combinations of; *wastewater for potable & irrigation*, *wastewater for toilet flushing*, *graywater for potable & irrigation* and *graywater for toilet flushing*. In addition, these strategies pair well with *efficient irrigation systems* and *indoor conservation* to decrease total demand at low relative cost.





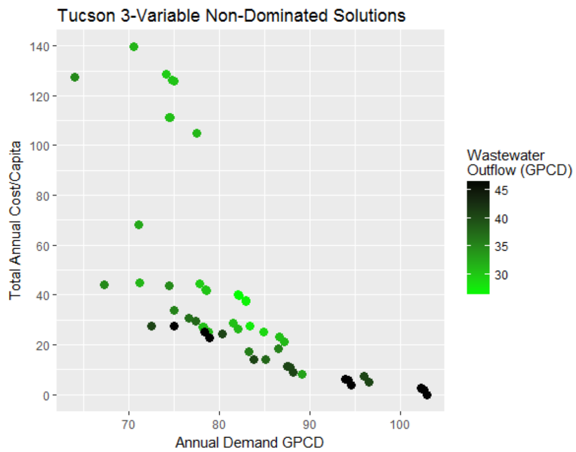


Figure 18 3-variable non-dominated solutions for Denver, Miami and Tucson. Plotted by annual cost ($)/capita and annual demand (GPCD). Wastewater outflow volume (GPCD) increases from light to dark green color shading.

Wastewater and graywater reuse come at a higher cost but provide a lower wastewater output when included in a combined solution. These are ideal strategies cities should consider if reducing wastewater is a goal to reduce the risk of CSOs and exceeding treatment plant capacity. Important to note that excessive wastewater volume in Denver and Tucson is not such a problem since the outflows are relied upon by downstream users. Wastewater outflow in the west is ultimately accounted for use downstream under prior appropriation laws.

3.4 Conclusions

This study presented a framework to assess the tradeoffs between several urban water demand reduction strategies and their associated costs modeled across three diverse U.S. cities. Cost-effective water conservation and reuse strategies meeting a wide range of water demand reduction targets were identified for each city. Results show discrete strategy performance in demand reduction between cities influenced by local climate, land cover and population density. The strategies that stood out in top performing solutions for each city were; *efficient irrigation systems* and *xeriscape* in Denver; *stormwater for potable & irrigation* and *graywater for toilet flushing & irrigation* in Miami; and *indoor conservation, graywater for irrigation* and *efficient irrigation systems* in Tucson.

The top performing strategies overall in terms of water savings and total cost were found to be *efficient irrigation systems* and *stormwater for irrigation.* While use of stormwater achieves large demand reduction relative to other strategies, it only occurred in non-dominated solutions that were characterized by higher cost. This strategy can be very effective for demand reduction but is also costly. On the contrary, efficient irrigation systems are frequently part of low-cost solutions across all three study cities. The most cost-effective solutions that also minimize wastewater outflow were identified to be *wastewater for potable & irrigation*, *wastewater for toilet flushing* and *graywater for potable & irrigation.*

Cities can effectively reduce potable water demand by targeting areas of high use. The proportions of landscape water demand and indoor demand to total influence strategy impacts. In Denver, outdoor use accounts for over half of total demand and it is in this use category the city should focus on finding approaches to increase efficiency and adopt alternative sources. Strategies that appeared in standout solutions and appeared most frequently in non-dominated solutions included outdoor demand reduction, while strategies reducing indoor demand rarely appeared in non-dominated solutions. While cities with higher per capita water use and a greater proportion of indoor use to total, as shown in Miami and Tucson, can focus more on indoor efficiency measures and incentive programs. Stormwater use was often part of non-dominated solutions across all cities but can achieve particularly large demand reduction in Miami where there is more precipitation and large impervious area resulting in higher generation of stormwater compared to other study cities. Of the three cities analyzed, Tucson is the city with the most limited supply of freshwater. There, graywater can be a consistent source of supply and most effective in supplementing low water use landscapes or xeriscape.

The solutions generated demonstrate the wide range of water demand reduction potential and associated costs with implementation of strategy combinations in each city. The non-dominated ranking procedure allows us to assess cost & demand-reduction tradeoffs between solutions of equal ranking. Using IUWM, strategy level of adoption and respective parameters can be tuned to match the system type and scale being modeled. This study presents one comparative set of strategies modeling certain types of systems and generating outputs at the city scale. IUWM and this methodology can help decision makers choose the most cost-effective demand reduction strategies to meet the goal of implementing total system solutions.

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APPENDIX

A.1 Miami Single-Family Residential Indoor Use Analysis

Miami has a warm tropical climate with little seasonal variation and outdoor irrigation occurs throughout the year. This presents a challenge as there isn't a straightforward way to separate indoor and outdoor use from the residential meters that make up a majority of the data. A set of 3,097 single family residential (SFR) homes have two meters that measure indoor and outdoor use separately. An analysis was conducted in attempt to assign indoor demand profile parameters to the indoor use data. The outdoor use data was analyzed to identify any seasonal trends in irrigation water use and attain the general allocation of outdoor to indoor use. Indoor use for each home was converted to gallons per household per day (GPHD) and plotted with the average household size in the block group the home resided in. The Residential End Use Study v2 (REUSv2) use data of all Level 1 surveyed homes is plotted with the Miami single family residence (SFR) indoor use data from the dual meter homes. The demand profile function of the REUSv2 Level 1 homes was used to model indoor use in Miami. Household size is estimated by respective block group population divided by the number of households (Figure 4).

The Miami dual SFR data has a large variation in use that is difficult to account for. Much of the data is considerably higher than normal indoor use and could be attributed to indoor labeled data that also includes outdoor use. In addition, the homes with double meters may not be a good representation of the greater set of households in the greater Miami study area. The use of average household size by block group does not represent true household size and makes it difficult to fit an adequate function to provide alpha and beta parameters for the daily household use function. Due to the uncertainty in this data, the national average indoor demand profile function of REUSv2 was applied in Miami for this analysis.

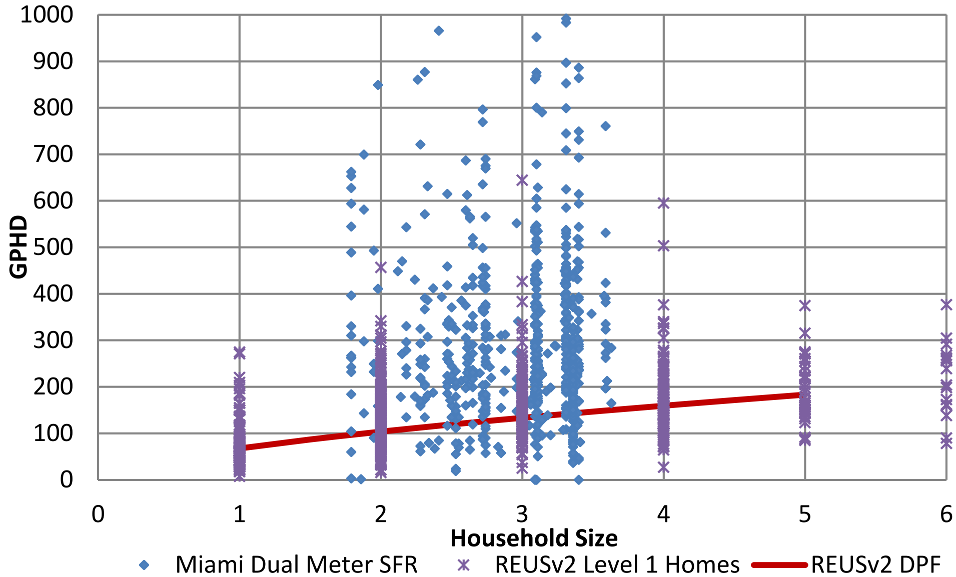

Figure 19: Comparison of Miami Dual Meter Single Family Residence (SFR) Indoor use and REUSv2 Level 1 surveyed homes average daily indoor use. REUSv2 Level 1 demand profile function (DPF) is used in this analysis

The set of dual meter homes was used to analyze any seasonal trends and attain the general allocation of outdoor to indoor use. Outdoor and indoor use for the entire set was summed by month. The percent of total by month is shown in the following chart. Outdoor use is fairly consistent throughout the year with no observable seasonal trend. On average use was found to be 58% outdoor and 42% indoor.

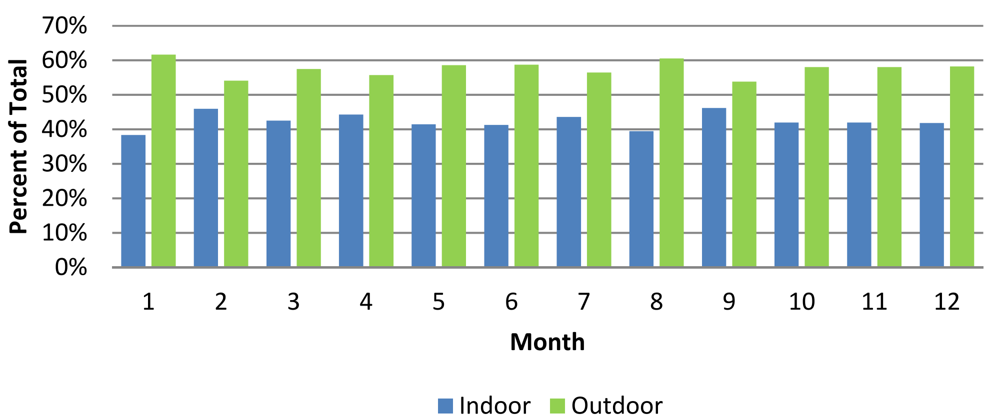

Figure 20: Monthly average indoor-outdoor percentage of single family homes with dual meters

A.2 Variable Discount factor comparison for Denver

Table 18: Variable discount factor comparison for Denver

	i = 5%	*i* = 6%	*i* = 7%
Conservation Strategies:			
Xeriscape	$ 109,012,000	$ 126,261,000	$ 144,203,000
Indoor Conservation	$ 19,977,000	$ 22,310,000	$ 24,748,000
Efficient Irrigation Systems	$ 3,500,000	$ 3,740,000	$ 3,989,000
Advanced Irrigation Systems	$ 5,457,000	$ 5,832,000	$ 6,219,000
Graywater Reuse for:			
Irrigation	$ 18,161,000	$ 20,282,000	$ 38,443,000
Toilet Flushing	$ 4,893,000	$ 5,465,000	$ 6,062,000
Toilet Flushing & Irrigation	$ 4,830,000	$ 5,394,000	$ 5,983,000
Roof Runoff for:			
Irrigation	$ 1,378,000	$ 1,431,000	$ 1,486,000
Toilet Flushing	$ 1,746,000	$ 1,831,000	$ 1,919,000
Toilet Flushing & Irrigation	$ 1,746,000	$ 1,831,000	$ 1,919,000