University of Nevada, Reno

Sustainability Assessments to Evaluate the Impact of Water Reuse in Urban Water Resource Management

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Abstract

This research develops planning-level sustainability assessment approaches for integrated urban water management (IUWM). The research focuses on the IUWM considerations that are most relevant to inland, water stressed cities; specifically using the Reno-Sparks metropolitan area to explore the wastewater management challenges that arise from growth in water demand due to urbanization. This region is hydrologically unique, with rapid growth occurring in hydrologically closed basins, and even the larger Truckee River watershed ends at a terminal lake. These characteristics enforce a close coupling of water and wastewater management as well as unique insights into how growth in water demand results in costs, or impacts, associated with wastewater production that are external to the water utility. In this basin, growth in wastewater production can have acute effects due to the close geographical relationship between the urban growth areas and the "downstream" communities and environments impacted by effluent discharge. Thus, the hydrological conditions of closed basins provide a valuable model for how these externalities around wastewater management can be incorporated into decision making for water resource management. This analysis is also meaningful for demonstrating the importance of taking an IUWM approach in other inland urbanizing regions, where the external effects of population growth on water resources impact communities further downstream.

First, a composite index was developed based on the concept of using the water-economy nexus as a strategy to characterize the sustainability of urban water use and reuse, particularly across non-residential water users. Triple bottom line analysis was used as a planning level approach to qualitatively compare the internal and external benefits of alternative IUWM scenarios. A microeconomic study then examined the economic efficiency of potable reuse based on the total costs of jointly managing water and wastewater resources for a region. Together, this research highlights the use of sustainability assessment approaches for examining goals such as evaluating regional water policy and comparing the economic, social and environmental impacts alternative resource management scenarios. This research uniquely demonstrates the application of sustainability assessment and the importance of considering water and wastewater resources jointly in urban water management.

Dedication

I dedicate this dissertation to Tom; thank you for all the discussions, debates, and inspirations that we shared. Insights from those conversations echo through this research. I will forever miss you.

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Acronyms

ASR – Aquifer storage and recovery

AWT – Advanced water treatment

AWTF – Advanced water treatment facility

BAC - biological activated carbon filtration

CBA – Cost-benefit analysis

CEC - Contaminants of emerging concern

CFCGMF - Coagulation-flocculation-clarification and granular media filtration

DBP – Disinfection byproduct

DWEL – Drinking water exposure level

GDP – Gross domestic product

HAA5 - Haloacetic Acids (5)

IPR – Indirect potable reuse

IUWM – Integrated urban water resource management

LCA – Lifecycle assessment

LPCD – Liters per capita per day

MB – Marginal benefit

MC – Marginal cost

MCL - Maximum contaminant level

MOS – Margin of safety

MTD – Minimum tolerated dose

NDMA - Nitrosodimethylamine

O3 – Ozone disinfection

O&M – Operation and maintenance

PC – Principal component

PCA – Principal component analysis

PPCP - Pharmaceuticals and personal care products

PR – Potable reuse

RBAL - risk-based action level

SQ – Status quo

TBL – Triple bottom line analysis

TCEP - Tris (2-Chloroethyl) phosphate

TTHM - Total Trihalomethanes

WAS – Waste activated sludge

WEI – Water-economy index

WRRF – Water reclamation and reuse facility

Chapter 1 Introduction

1.1 Problem Statement and Significance

Urbanization can reshape urban water systems in a multitude of ways that result in the loss of water use potential. Changes in land use to accommodate urban sprawl increases impermeable surfaces, which in turn can impact aquifer replenishment, the runoff generated by the area, erosion in streambeds, and flood risks; these alterations can also shift ecological regimes through degradation of water quality (Bertrand-Krajewski et al., 2000). Thus, there is a wealth of possible impacts to regional sustainability (e.g. growth that does not result in lost benefits for future generations), environmental resilience (e.g. to droughts, wildfire, flooding), localized water quality, and the potential costs associated with water and wastewater treatment that should be considered when planning for growth in water demand. Water reuse provides a strategy to impact some negative effects of urbanization on two fronts by benefitting both regional water supplies and effluent management. Often, reuse is sought to diversify water portfolios with a more drought-proof water supply, but it can also result in measurable reduction in pollutant load to the waterways that effluent is discharged to. This can not only benefit ecosystems but also local and downstream communities. Overall, reuse of water can reduce the pollution burden on ecosystems while also satisfying a variety of human needs.

When comparing water resource management scenarios to accommodate urbanization, decision making approaches should take a multifaceted approach to incorporate sustainability goals and address uncertainty around climate change, population growth, and future water resource conditions. Paradigms in water management have shifted, as demonstrated by conceptual frameworks such as the UN Sustainable Development Goals, the World Economic

Forum's Global Water Initiative, and the One Water movement. These ideas are increasingly altering the approaches taken to evaluate water infrastructure and policy, but there is a need to provide better frameworks for examining trade-offs and incorporating the qualitative and quantitative parameters that factor into these sustainability goals (Falkenmark and Rockström, 2010; Hoekstra et al., 2018; Lund, 2015; Srinivasan et al., 2017). The purpose of the research is to enhance the tools available for urban water resource managers to examine the sustainability impacts of water reuse. This is accomplished by demonstrating the application of three decision making tools that integrate social, economic, and environmental impacts into decision making for water reuse projects. Existing approaches to evaluate proposed water management projects often fail to consider urban water resources holistically; instead they often consider only water or wastewater management.

This research demonstrates the importance of holistic, integrated urban water approaches that consider the interdependencies of water and wastewater systems when evaluating urban water systems and proposed management strategies or infrastructure. The first paper develops the Water-Economy Index, which examines the differences in how urban areas allocate, conserve, and reuse existing water resources. The second paper is a planning-level evaluation of a proposed advanced water treatment system for indirect potable reuse through a triple bottom line analysis. The third paper presents a conceptual economic model to evaluate the conditions under which indirect potable reuse is a cost-minimizing water management project with an integrated urban water approach.

1.2 Background

The relationship between urbanization and water resources is complex and dependent on a multitude of factors that influence the magnitude on interlinkages between land use, soil impermeabilization, stormwater runoff, aquifer replenishment, streamflow, and the riparian zone benefits to flood control and water quality (Bertrand-Krajewski et al., 2000). The need to evaluate urban water security and urban water footprints more holistically has been recognized for several decades and has resulted in concepts such as greywater, green water, virtual water, and urban water metabolism (Behzadian and Kapelan, 2015; Falkenmark and Rockström, 2010; Hoekstra and Chapagain, 2006; Rushforth and Ruddell, 2015; Venkatesh et al., 2017). These ideas have provided novel ways to understand how urban growth can be better managed to improve regional sustainability and water security. There is a need to develop decision-making approaches that operationalize these ideas while also allowing for representation of the diverse social, economic, and environmental factors that influence how cities can sustainably grow and manage water resources.

1.2.1 Composite Indicators of Vulnerability

Composite indicators, or indexes have been widely adopted as decision making tools for policy development because of their ability to integrate a broad spectrum of variables into a numeric score for performance benchmarking. Composite indicators have been applied in numerous contexts related to water resource management. For example, Akkoyunlu and Akiner (2012) developed a composite indicator to evaluate water pollution and eutrophication throughout streams; whereas Sullivan et al. (2003) developed the water poverty index to aid international institutions and non-governmental organizations in evaluating community level water management needs. In the context of urban water management, indexes are appropriate for developing an understanding of the underlying systems that govern how water is used by a city with applications. This can be used to compare how efficiently the study area uses water resources compared to similar cities, or to evaluate the potential for enhancing water resource management based on its water use characteristics.

1.2.2 Triple Bottom Line Analysis

Triple Bottom Line (TBL) analysis is a flexible, more qualitative approach that allows us to leave the perspective of the institution. We can look at diverse aspects of regional values and needs and incorporate parameters that can't be quantified or qualified into a dollar value. It can provide a strong basis for the desirability or undesirability of a project or policy that can't be evaluated in financial terms. TBL is often associated with Life Cycle Assessment (LCA), although the goals of the two methods are different. LCA provides a somewhat holistic approach to evaluate the cost-effectiveness of a project compared to alternatives. This approach does not encompass the dollar value of benefits provided by the alternatives under consideration. For example, when each alternative may result in the same annual benefits and the same non-monetary affects, LCA can be an appropriate approach to evaluate the trade-offs between alternatives. Some studies utilize LCA to account for environmental impacts within the framework of a triple bottom line assessment (Chhipi-Shrestha et al., 2017; Godskesen et al., 2013; Rygaard et al., 2014).

In situations where the non-monetary impacts of alternative scenarios are different TBL provides a more consistent approach to examine trade-offs. TBL assessments primarily rely on qualitative techniques to determine the benefit or loss of social and environmental criteria under different management scenarios. While previous TBL studies have formulated approaches that examine the feasibility of urban water reuse in water-stressed regions (Garcia and Pargament, 2015; Makropoulos et al., 2008; Rygaard et al., 2014), these studies have not emphasized drivers that characterize the needs of closed-basin inland regions described above.

1.2.3 Cost-efficiency, Economic Optimization, and Cost-Benefit Approaches

Net present value analysis and financial benefit-cost analysis evaluates the cost-effectiveness of projects by accounting for both the project costs and the direct monetary benefits that would result from a project. Similar to LCA, the scope for evaluating projects is limited to the project; it does not account for the potential impact to the regional economy or changes across the entire water sector. LCA and financial benefit-cost analysis are appropriate for evaluating cost-effectiveness of projects under the conditions described and are relevant for private investments in which a project may be selected based on the private returns and benefits. Net present value analysis is often incorporated into TBL as a criterion for the water utility, or institution.

Economic benefit-cost analysis evaluates a project based on the potential impact across the regional economy. In the case of IPR the largest impact to the economy would likely be the population increase that the additional freshwater resource could support. Thus, in a water scarce region with population growth limited by available water resources, the regional demand for housing would correlate to demand for the new water resource. Economic benefit-cost analysis should also account for the significant externalities that may result from the project, such as impacts to flood risk resulting from changes to water flows or health risk resulting from changes to water quality. Thus, this approach accounts for both “market goods” (such as water rights) and non-market goods and non-uses (e.g. Molinos-Senante et al., 2010). Economic benefit-cost analysis is more appropriate than financial benefit-cost analysis for public utilities and government agencies, which must ensure that the public investments that fund projects and policies are allocated in a way that enhances returns to society. While economic cost-benefit modeling is useful for evaluating the trade-offs between alternative scenarios, it does not provide insight into how variables, such as the cost of energy or uncertainty in population growth, may influence which option is optimal or cost-minimizing.

Models to evaluate the economic efficiency of water resource use have largely been applied to agricultural water management challenges. For example, Expósito and Berbel (2016) developed a cost model to demonstrate the economic impacts of reducing irrigation to olive groves. Microeconomic models related to urban water management have not been widely explored and none that were found incorporated an integrated urban water approach that included wastewater, which is often treated prior to environmental discharge at a higher cost than the cost of treating potable water. Research by Angulo et al. (2014) explored the microeconomics of water use by hotel and restaurant sectors. Additionally, the microeconomics of residential water use has been widely studied (Dalhuisen et al., 2003; Hung and Chie, 2013; Moeltner and Stoddard, 2004; Olmstead and Stavins, 2009).

1.3 Research Objectives

The objective of this research was to demonstrate how integrated urban water management approaches and sustainability goals can be operationalized using existing decision making frameworks. The research focused on evaluating the trade-offs and efficiency of water reuse to enhance regional water sustainability. A comparative analysis of urban water conservation and reuse strategies was carried out by developing a composite index that also identifies correlations between water management strategies and socio-economic and environmental characteristics of cities in the United States. A triple bottom line analysis was then carried out to evaluate the social, institutional, and environmental trade-offs of a potable reuse system compared to conventional water-wastewater management practices. The final objective was a conceptual microeconomic model of cost-minimizing integrated urban water management, considering scenarios with and without potable reuse.

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Chapter 2 The Water-Economy Nexus: A Composite Index Approach to Evaluate Urban Water Vulnerability

2.0 Abstract

Resource scarcity is a driving force behind water conservation and reuse as urban areas seek strategies to adapt to population growth and environmental challenges. Although there are numerous indicators that examine urban water resource and demand characteristics, these approaches do not tie together how aspects like economic health, environmental conditions, and population growth correlate with local water conservation to demonstrate a city's ability to cope with water resource vulnerability. This research develops a conceptual framework that for Water-Economy Index (WEI) which characterizes social, economic, and environmental dynamics of water reuse and conservation. The application specifically utilizes a principal component analysis (PCA) to evaluate how hydro-economic indicators (including water demand intensity, demand for recycled water, economic productivity of water, unemployment, and allocations of water resources) are correlated and can impact sustainability goals. The PCA method aggregates indicators into three groups: socio-economic, water allocation, and socio-environmental indicators. The most influential indicators within each group are economic productivity of water, wastewater reuse, and consumptive water demand, respectively. The WEI ranks of 49 cities are compared to identify shared traits across individual indicators and to demonstrate the application of the WEI for benchmarking. The results provide insight into the complex relationship between the characteristics of an urban area's water demand and socio-economic performance.

2.1 Introduction

As urban areas concentrate growing populations, cities situated in arid and semi-arid regions are increasingly challenged with problems related to water and wastewater management. The growth in water demand and wastewater production can intensify the risk of water shortages and flooding, particularly in drought prone basins and those with limited drainage capacity. These risks contribute to concerns regarding the resilience and sustainability of regional economic activity, food, and water resources (Hoekstra et al. 2018). To address the challenges arising from water scarcity, water utilities have embraced a variety of management strategies, including demand management, water storage, water recycling, efficiency and leak reduction (Chelleri et al. 2015). Vulnerability assessments provide a strategy to assess if water infrastructure and policies achieve an acceptable buffer against short- and long-term risks, such as droughts and population growth.

Urban planners and researchers utilize numerous vulnerability and sustainability assessment techniques to evaluate water and wastewater management policies including composite indices, simulation modelling, multi-objective modelling, and forecasting (Brown et al. 2015). The development of these tools and strategies has become increasingly important due to the uncertainties arising from climate change. Future hydrologic conditions may increasingly vary from historical statistics, weakening the ability of water planners to optimize management based on forecasts (Milly et al. 2008). Recent reviews of water management indicators identify 50 indices or tools to assess water vulnerability and 24 indicators fulfil sustainability criteria through inclusion of social, economic, and environmental dimensions (Plummer et al. 2012; Pires et al. 2017). However, none of these indicators specifically examine the interactions between water resources and the economy in an urban context, which is a critical aspect of urban resilience to water scarcity.

Researchers and urban planners have demonstrated the role of a variety of adaptive strategies to address exposure to short- and long-term stressors like droughts and population growth (Olmstead and Stavins 2009; Gober et al. 2016; Ashoori et al. 2016; Breyer et al. 2018). Water institutions, laws, and policies can provide solutions or present barriers against beneficial adaptations to reduce vulnerability (Srinivasan et al. 2017). Water pricing structures and water use restrictions are tools commonly used by water utilities to influence water demand and encourage conservation. However, the interactions between water conservation, water reuse, and the socio-economic characteristics of an urban area are not well understood.

This study demonstrates a water-economy nexus framework to evaluate a region’s ability to cope with water resource stressors through water conservation policies and reuse. The research utilizes indicators of water resource utilization, conservation, climate, and demographic pressure on water resources to create a more holistic picture of the interconnections between water scarcity and water demand. Within an urban water management context, the water-economy nexus can be defined as the interaction between water resource utilization and economic sectors (e.g. commercial, industrial, service) to produce products, facilitate trade, provide services, support employee needs, and to generate economic outputs (e.g. gross domestic product). The resilience of urban water supplies can be transformed through drivers from environmental, governmental, and socio-economic systems (Daniell et al. 2015). This study uses indicators to characterize these drivers and develop an understanding of their influence on urban water vulnerability through a principal component analysis (PCA). PCA is widely used for exploratory data analysis because of its ability to simplify correlated multivariate data into a few key dimensions (OECD et al. 2008). This strategy has been used in numerous studies to characterize sustainability and resource vulnerability (Doukas et al. 2012; Cîrstea et al. 2018; Greco et al. 2019; Uddin et al. 2019).

The proposed framework evaluates how environmental, social, and economic characteristics of urban areas correspond to their adoption of water conservation and reuse. The method uses socioeconomic indicators of welfare, water demand intensity, and environmental parameters to encompass a holistic view of how conservation can offset water stress and the changing relationship between economic outcomes and water demand in response to water stress. Principal component analysis identifies correlations between the selected indicators. Using this research approach, a composite indicator is developed to characterize the sustainability of urban water use, the WEI.

2.2 Conceptual Framework

In integrated water resource management, indexes have been proposed to evaluate water quality, water resource vulnerability, and to compare water policies. For example, the Sustainability Index for Water Planning and Management (Sandoval-Solis S. et al. 2011) was developed to compare water management policies. This index incorporated dimensions of performance criteria to evaluate alternative policies including reliability, resilience, and vulnerability. The Water Poverty Index was constructed from 5 categories of indicators (resource, access, capacity, use, and environment) to connect water and poverty. This index has been applied at both community and national scales for an integrated approach to assess the link of welfare to water resource access and scarcity in developing regions (Sullivan et al. 2003). The Water Scarcity Index (Falkenmark 1989) incorporated four indicators of water availability and demand to evaluate anthropogenic stress on water resources. The WEI was built on these foundations by developing an indicator that can aid in comparison and advocacy for urban water conservation and reuse through benchmarking against other cities.

This research compiled a dataset of metropolitan water use and economic statistics available through federal United States databases for 49 metropolitan areas. Data availability has been a major challenge for developing and applying proposed composite indicators (OECD et al. 2008; Juwana et al. 2012); this was overcome by limiting urban water-economy indicators to those that were measured according to a standardized protocol or could be calculated by statistics of public databases. An initial cluster analysis was conducted to evaluate the similarities between metropolitan areas based on water use and economic statistics. The cluster analysis identified that cities that generated the highest economic productivity of water demand had lower than average consumptive water demand and water demand intensity (Liters demand per capita per day). The cities with the highest consumptive water demand were in arid climates and generally had low water demand intensity and low to moderate economic productivity of water demand. Cities with low to moderate consumptive water demand and low to moderate economic productivity generally had the highest water demand intensity.

The dataset included five metropolitan areas in arid or semi-arid climates that receive an average annual precipitation under 250 mm (Albuquerque, NM; El Paso, TX; Phoenix-Mesa-Scottsdale, AZ; Reno, NV; Yuma, AZ). All five of these cities were categorized as having a high percent of consumptive water demand but low water demand intensity and low to moderate economic productivity of water demand. This preliminary analysis indicated that these sustainability indicators identified differences in how water resources are managed across cities. It also revealed that there may be limitations to a city’s water demand characteristics based on environmental characteristics (e.g. precipitation).

The pairwise examination of the selected water-economy indicators identified strong correlations between several indicators and differences in how water resources were managed to address water scarcity and the sustainability in different cities. For example, the demand for non-potable reuse of reclaimed water correlated to higher percent consumptive use of water resources. Thus, cities that allocated a relatively large amount of water resources to more consumptive demands, like irrigation, were more likely to adopt non-potable reuse of reclaimed water. The WEI composite indicator examined these interrelationships between components to develop a more complete characterization of how the direct use of water resources within an urban area contributed to sustainability goals.

2.2.1 Indicator Selection

A literature review of water resource vulnerability and sustainability indicators was first performed to identify common indicators or variables used to characterize urban water demand characteristics. These indicators were then assessed to determine if they satisfied the quality criteria for good indicators: integrity, methodological soundness, accuracy and reliability, serviceability, and accessibility, etc. (OECD et al. 2008). Pires et al. (2017) identified 170 indicators related to water demand and management, 24 of which fulfilled sustainability criteria. This research evaluated those indicators to identify which complied with the selection criteria for good indicators, as identified above. Overall, the accessibility and serviceability criteria presented the greatest limitations to the indicators reviewed. These limitations were most significant for the inclusion of economic and environmental indicators, such as sector-based earnings and drought vulnerability. Specifically, only indicators that could be calculated from data measured regularly (a minimum of every 5 years) and reported on publicly accessible databases were retained. The candidate indicators (Table 2-1) captured information describing aspects of how urban communities vary in water conservation and reuse, water demand characteristics, and economic productivity. Selection criterion (e.g. Spearman rank-order correlations) were used to determine which candidate indicators were selected for inclusion in the WEI.

Indicators with positive resilience impacts are those that may result in lower water vulnerability. For example, the percent of water resources allocated to public municipal supplies has a positive resilience impact by enhancing the ability of the regional utilities to efficiently manage local supplies through integrated water management strategies like conjunctive use. Conversely, consumptive water demand has a negative resilience impact by reducing the availability of water resources for reuse and downstream communities.

Table 2-1 List of selected and candidate indicators

Candidate Indicators	Units	Dimension			Resilience Impact
		Env.	Soc.	Econ.	
Precipitation	mm/year	X			Positive
Population Density*	Persons/m^2	X	X		Positive
Economic Productivity of Water*	US$/L		X	X	Positive
Water Demand Intensity*	L/person/day (LPCD)		X		Negative
Wastewater Reuse*	m^3/year	X	X	X	Positive
Consumptive Water Demand*	% of total	X	X		Negative
Municipal Water Demand*	% of total	X	X		Positive
Water Demand for Thermal Energy	m^3/MJ		X	X	Negative
Water Productivity in Jobs	Employment/m^3		X	X	Positive
Average Water Cost	$/$m^3$		X	X	Positive
Unemployment Rate*	%		X	X	Negative

* Indicates variables selected for index development

2.3 Methods

Following the conceptual framework, the indicators are calculated from statistics of water demand, population, GDP, and employment. The subsequent sub-sections describe data sources, calculation of the indicators, indicator standardization, and the methodology to develop the WEI through a PCA analysis.

2.3.1 Data Sets

The indicators listed in Table 2-1 were available or were calculated through databases that report economic and water demand statistics for U.S. metropolitan areas. This study analyzed the 10 indicator variables for 49 cities based on economic and water demand statistics reported for U.S. metropolitan statistical areas in 2015. Population, metropolitan land area, and GDP statistics were retrieved from the US Bureau of Economic Analysis (BEA) database. Employment and unemployment statistics were retrieved from the US Bureau of Labor Statistics. All water demand characteristics (total withdrawal, withdrawal for public supplies, consumptive use, reuse for agriculture and recreation) were obtained from the US Geographic Survey Water Use Report (Dieter et al. 2017).

2.3.2 Indicator Calculations

Water allocations within urban areas were characterized by the percent of total regional water demand that is allocated to public supplies (municipal demand), the percent of water demand that is consumptive, energy production from water supplies, and reclaimed wastewater reuse (including agriculture, thermoelectric power, and recreational purposes). These indicators represented social-environmental aspects of water demand that can be related to land use within the metropolitan areas, climate, and water scarcity. Economic characteristics and thermoelectric power also may have had strong influences on these characteristics, such as reclaimed wastewater reuse and consumptive water demand. To increase the significance of reclaimed wastewater reuse reported for metropolitan areas, the total volume reused annually was determined based on the reported annual use for agriculture, recreation, and thermoelectric energy, reported in Dieter et al. (2017). Water demand to support thermoelectric power generation was normalized based on reported energy production (mega Joules), reported in Dieter et al. (2017). Finally, water demand intensity was calculated by standardizing the average daily municipal water demand to the population served by public supplies within the metropolitan area.

Apart from economic productivity of water resources, the socio-economic indicators of unemployment rate (%) and employment (number of jobs) were obtained from public databases (Bureau of Economic Analysis 2018; Bureau of Labor Statistics 2018). Calculation of the economic productivity of water followed the method described in Gleick (2003) by normalizing GDP (reported in millions of US Dollars) to water demand. Thus, annual GDP statistics for 2015 (Bureau of Economic Analysis 2018) were divided by the total annual water withdrawal for public supplies (Dieter et al. 2017) to calculate the economic productivity of water.

2.3.3 Standardization of Indicators

PCA is a multivariate method that requires transformation of indicators into a standardized range (0-1). This step ensures that differences in the scale and variance in different indicators do not exert undue influence in the analysis. This step is standard across literature that utilizes PCA to carry out vulnerability or sustainability assessments (Doukas et al. 2012; Cîrstea et al. 2018; Uddin et al. 2019). The Shapiro-Wilk test is first used to confirm that the assumption of normality cannot be rejected for each indicator. Then, indicators that have a positive impact on reducing vulnerability are standardized following equation 2-1, while indicators that enhance vulnerability are standardized with equation 2-2. Refer to the last column of Table 2-1 for these directional relationships.

$$Index,\ X_{d,t}^{+v} = \frac{X_{d,t} - X_{min}}{X_{max} - X_{min}} \qquad (2\text{-}1)$$

$$Index,\ X_{d,t}^{-v} = \frac{X_{max} - X_{d,t}}{X_{max} - X_{min}} \qquad (2\text{-}2)$$

The correlation coefficients for paired indicators are used to identify if any indicators are highly correlated, which can create an undesired bias in the final analysis. The correlation matrix indicates the strength of relationships among the indicators as well as the direction of these relationships. The Spearman rank-order correlations are calculated between the indicator variables to detect if any are highly correlated and redundant (Spearman > 0.9). This analysis is used to select the final indicators to be included in the PCA and for development of the composite index. When determining which highly correlated indicators to eliminate, the metrics that are most widely used throughout literature or that best characterize the underlying relationship of interest should be retained.

2.3.4 Principal Component Analysis (PCA)

The ability to distill an array of indicators into a low dimensional space has made PCA well adapted to sustainability and vulnerability assessments, such as assessment of climate vulnerability (Uddin et al. 2019), and development of composite indexes of energy sustainability (Doukas et al. 2012; Cîrstea et al. 2018). A similar approach is followed in this research to develop the WEI.

The significance of intercorrelations between indicators in the dataset is tested with Bartlett's test and the Kaiser-Meyer-Olkin (KMO) measure of sampling adequacy, removing any variable with a KMO score below 0.5 (Field et al. 2012). The correlation matrix developed in the prior step is developed into a matrix of eigenvectors and eigenvalues when the data are rotated to the principal component coordinate system. Thus, the eigenvalues correspond to the values in the correlation matrix. The first principal component is the linear combination of variables that explains the largest amount of variation and corresponds to the largest eigenvalue. Often, most of the variability observed across the variables (> 80%) can be explained in two to three principal components.

The PCA results were then examined to determine the combination of indicator variables that are most important under each retained principal component. PCA generated scores for each indicator variable within each principal component. The indicator variables within each principal component were evaluated to identify those with a significant contribution to variability within that component. Within each principal component, the PCA method assigned a loading and contribution to every indicator variable. For an analysis with n indicator variables, the indicator variables that had a contribution greater than $1/n$ were considered significant to that principal component. These results allowed for identification of the underlying processes described by each principal component, for example socio-economic aspects, water management and allocations, and land use aspects. Each principal component variable was then summarized as a linear combination of the important indicator variables, using the indicator variable scores for weighting.

The retained principal components combined the correlated indicators into a variable that characterized the regions ability to cope with water scarcity. Each retained principal component

(N=3) was calculated for the 49 metropolitan areas in the dataset to evaluate how hydro-economic factors influence water resource management. Component indices were derived by combining the significant indicator variables using their respective loadings. The final index was calculated as follows:

$$WEI = \frac{\lambda_1 PC_1 + \lambda_2 PC_2 + \lambda_3 PC_3}{\lambda_1 + \lambda_2 + \lambda_3} \quad (2\text{-}3)$$

where PC_i is the aggregate score of the indicator variables for the *i*th principal component, and the eigenvalues (λ_i) are the weights assigned to each component, as determined through PCA analysis.

2.3.5 Non-compensatory Aggregation

PCA is a linear aggregation method; this type of method has theoretical weaknesses in how the weights assigned to indicators are interpreted. Theoretically, weights on indicators are intended to measure importance. However, linear aggregation implicitly assumes a compensatory logic in which investment in one indicator can be substituted for a shortcoming in another. For example, loss in economic productivity of water could be compensated for by decreasing water demand intensity.

Aggregation approaches that are consistent with a non-compensatory logic can preserve the interpretation of weights as true importance coefficients. The Condorcet-Kemeny-Young-Levenglick ranking procedure was selected as a non-compensatory method to aggregate the indicators (OECD et al. 2008; Munda 2012). This approach utilizes an outranking matrix to carry out a pair-wise comparison between each city across all indicators. In this method, equal weights are given to each indicator. Thus, rather than emphasizing the importance of different indicators, this method emphasizes the performance of each city across each indicator to determine the final WEI ranks.

2.4 Results

Prior to conducting the PCA analysis, the correlations between indicator variables were tested, following the methods described in the previous section. This analysis identified that water demand for thermal energy was not sufficiently correlated to other variables (KMO < 0.5) and water productivity in jobs was too highly correlated (0.99) to the economic productivity of water. According to the selection criteria, these two indicators were not included in the WEI. All other indicators were found to have acceptable intercorrelations according to KMO criterion.

2.4.1 Calculation of WEI

The correlation matrix described the directions and magnitudes of correlations between indicators (see Appendix Table 6-1). The economic productivity of water demand had large negative correlations to the water demand intensity (75% correlated), unemployment rate (54% correlated), and consumptive water demand (40%). It was positively correlated to population density by 48%.

Water demand characteristics were encompassed by three indicators: wastewater reuse, consumptive water demand, and the percent of water demand allocated to public supplies. The variance across the data revealed a weak negative correlation between the percent of water allocated to public supplies and the fraction of demand that was consumptive. Consumptive water demand had stronger correlations to the socio-economic indicators: unemployment (38%) and negative correlations with economic productivity and population density (36%). Wastewater reuse was positively correlated to population density (29%) and negatively correlated to the percent of water resources allocated to public supplies.

The indicator variables were standardized to scale them and account for the different metrics of each indicator (Appendix Table 6-2). Then they were normalized using a log-transformation. The PCA analysis produced three principal components with large eigenvalues (2.79, 1.24, 1.13); these accounted for 75% of the total variance in the dataset. The KMO for the PCA was 0.64. Together, the KMO sampling adequacy test and the Bartlett test suggested that the PCA should result in distinct and reliable principal components (Field et al. 2012). Table 2-2 presents the eigenvalues for each of the three principal components (λ_i), which were used as weights to calculate the WEI.

Table 2-2 Eigenvalues for the three principal components

	PC1	PC2	PC3
Eigenvalue	2.79	1.24	1.13
Explained Variance (%)	39.9%	19.6%	15.2%
Cumulative Explained Variance (%)	39.9%	59.5%	74.7%

The three retained principal components corresponded to linear combinations of each indicator variable. The indicator variables were weighted from their loading factors (Table 2-3), which represented the relative importance of each indicator in describing variance across a given component. The percent contribution of each indicator variable across the principal components was evaluated to identify those with a significant contribution, as illustrated by highlighted text in Table 2-3. The first component described 40% of the total variance across the dataset. This component included the socio-economic indicators related to water use including the economic productivity of water demand, the unemployment rate, water demand intensity, and population density. The second component was comprised of indicators describing water and wastewater resource allocations based on the percent of water resources going to the public supply and wastewater reuse. The third component characterized socio-environmental aspects of water use including consumptive water demand, water demand intensity, and wastewater reuse.

Table 2-3 Loadings of indicator variables

Indicator Variable	PC1	PC2	PC3
Economic Productivity	0.76	-0.04	0.09
Unemployment	-0.61	0.01	0.02
Population Density	0.43	0.09	0.00
Water Demand Intensity	-0.51	0.16	-0.18
% Municipal Demand	0.11	0.43	-0.05
% Consumptive Demand	-0.33	-0.15	0.46
Wastewater Reuse	0.00	0.49	0.39

2.4.2 Urban Clusters in WEI

Cluster analysis on the PCA results identified shared indicator variable characteristics across the cities in the dataset. These clusters were illustrated across the first and second principal components (PC1 and PC2) in Figure 2-1a and across the first and third principal components (PC1 and PC3) in Figure 2-1b. In Figure 2-1a, the lower right quartile of the graph contained urban areas with the highest economic productivity from water demand, higher rates of wastewater reuse, and higher population densities. These cities generally had the lowest water demand intensity. Cities located in the upper right quartile also had high economic productivity from water demand and higher population densities with less consumptive water demand and a higher overall percent of water supplies allocated to the public supply. Cities located in the lower left quartile had lower than average economic productivity of water resources and a higher than average unemployment, consumptive water demand, and wastewater reuse. Cities located in the upper left quartile had higher than average water demand intensity, a larger percent of water allocated to municipal demand, but generally lower than average economic productivity from water resources and wastewater reuse.

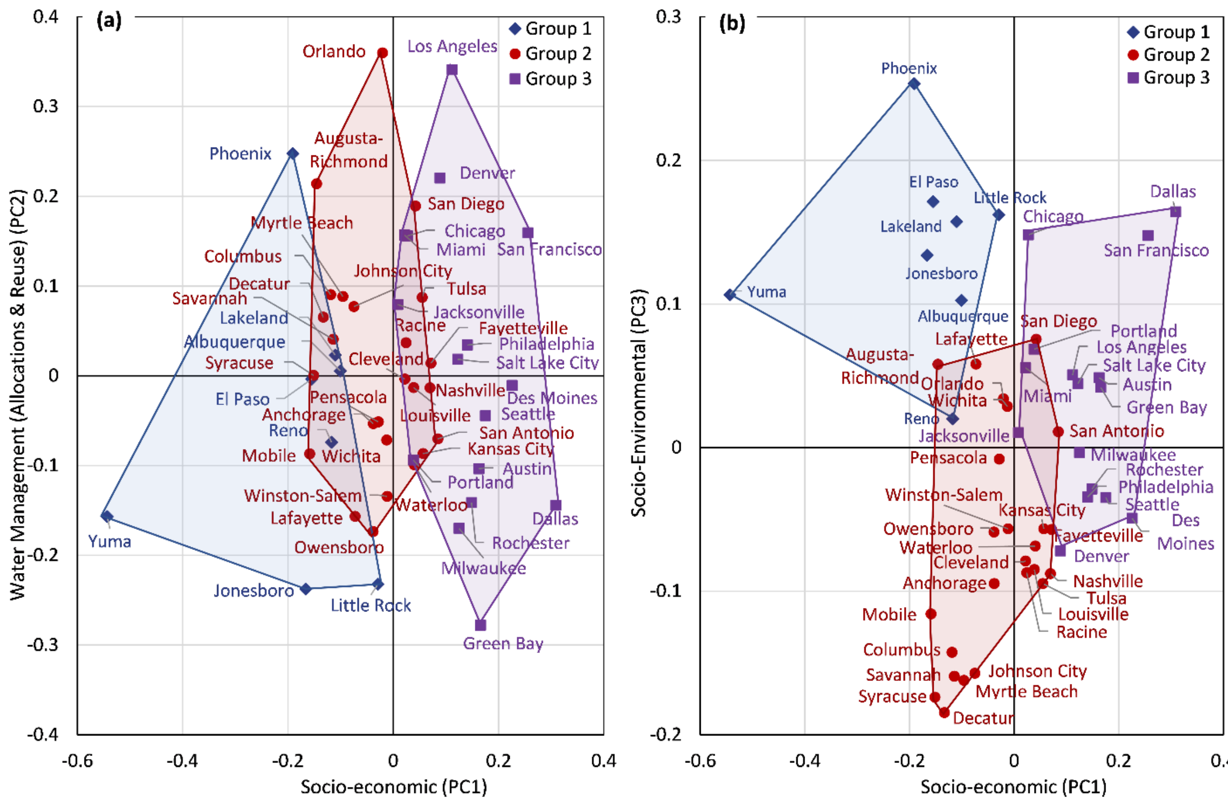

Figure 2-1 Orientation of city clusters along the WEI component axes (a) socio-economic indicators versus water allocation indicators, and (b) socio-economic versus socio-environmental indicators

The three urban clusters indicated cities with similar characteristics across the seven indicator variables. The most arid cities in the dataset (Yuma, AZ; El Paso, TX, Albuquerque, NM; Phoenix-Mesa-Scottsdale, AZ; and Reno, NV) were all located in the first cluster. Within this cluster it was observed that the cities with the highest economic productivity (> \$0.25/L water demand) (e.g. Albuquerque, NM and Reno, NV), had lower water demand intensity (near 500 LPCD) and higher rates of wastewater reuse and consumptive water demand (> 30%).

Cities in the second cluster were observed to generally have near average economic productivity ($0.4/L) from water demand and much lower consumptive water demand compared to the first cluster (< 20%). Cities in the third cluster had the highest scores for the WEI, driven by their high economic productivity for water resources (> $0.4/L). These cities often had higher population densities (>600 persons/km^2) and lower water demand intensity (< 450 LPCD) and consumptive water demand (< 5%).

2.4.3 Drivers of WEI Across Urban Clusters

The WEI indicator classified cities based on three major components: indicators of socio-economic influences on water demand, water allocation characteristics, and water conservation indicators. Examination of the city scores across the three principal components (Figure 2-2) provided greater detail into how the urban clusters were identified based on the indicators.

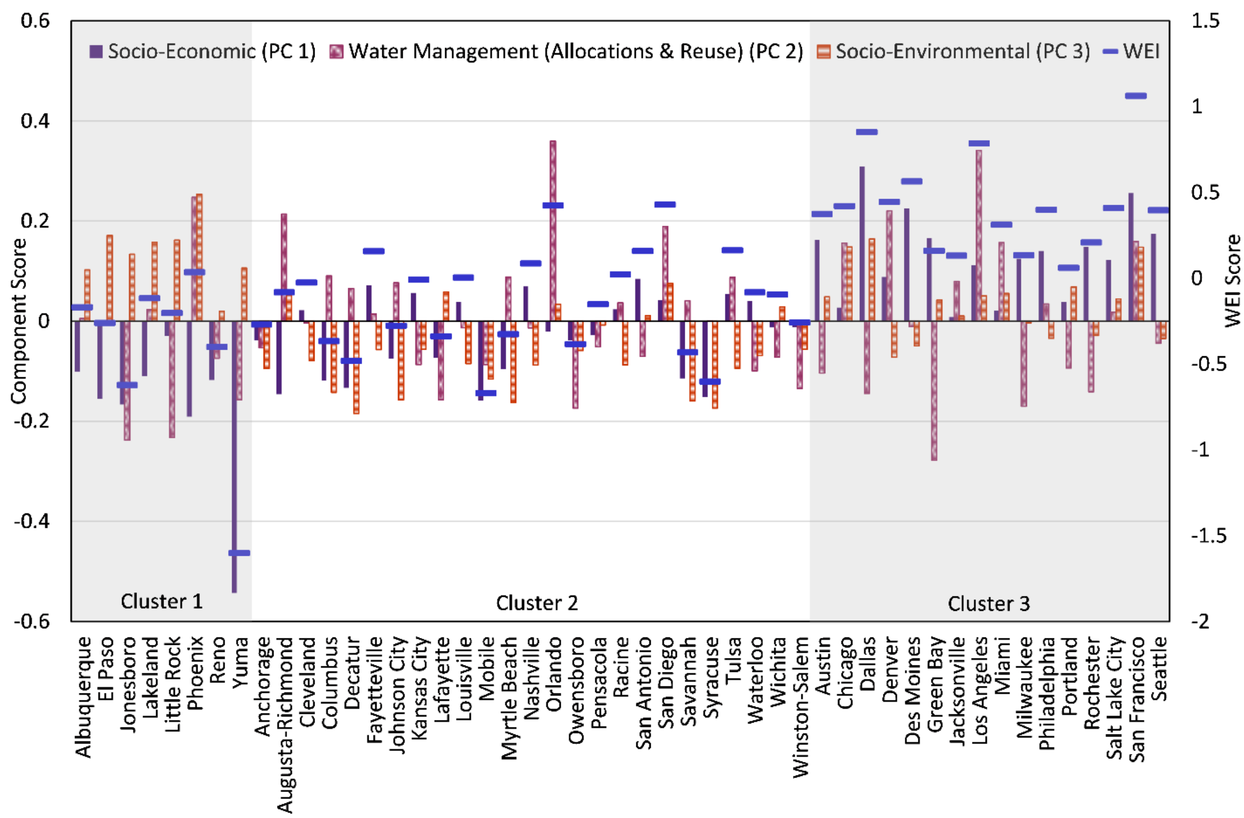

Figure 2-2 Principal component and aggregate WEI scores for the 49 U.S. cities

The first cluster was characterized by positive scores across the third component, reflecting higher than average wastewater reuse, consumptive water demand, and allocations to municipal supply. These cities had relatively low scores across principal components one and two, resulting in relatively low WEI scores due to lower than average economic productivity and higher than average water demand intensity. Most of these cities (60%) also had higher than average unemployment rates. The high scores across the third component, resulting from wastewater reuse and consumptive demand, may have reflected the influence of climate on altering water demand characteristics in arid or semi-arid climates.

On average, cities in the second cluster had more neutral scores across the first and second component but had low scores for the third component. Cities in this cluster had near average economic productivity from water resources, higher than average water demand intensity, lower than average consumptive water demand, and lower than average reuse of wastewater.

The third cluster of cities had the highest WEI scores. These cities consistently had lower than average water demand intensity and had high economic productivity from water resources. Along the second component, the cities in this group received positive or near neutral scores. Over 70% of the cities in this cluster had above average demand for wastewater reuse and low consumptive water demand. San Francisco, CA, USA received the highest WEI score, driven by a low unemployment rate, high economic productivity from water, low water demand intensity, and above average wastewater reuse.

2.5 Discussion

A major goal in the development of the WEI was to utilize data widely available, which better ensures that raw data and indicators satisfy criteria like timeliness of data collection, consistency in measurement methods, and data accessibility. This has been a major challenge for other indexes or tools (e.g. water footprint and water poverty index) developed to evaluate water sustainability (Plummer et al. 2012; Lovarelli et al. 2016). The WEI indicators provided local economic and social context for water use, resulting in a composite index that characterized aspects of sustainability relevant to water resource management at the local scale. The results demonstrated the importance of socio-economic indicators, like water demand intensity and the economic productivity of water, in driving urban water management sustainability.

2.5.1 Component Characteristics

The first principal component characterized socio-economic traits influencing water use: the negative correlation between water demand intensity and unemployment with population density and economic productivity. Thus, more urbanized metropolitan areas with lower unemployment rates were observed to conserve more water. Unemployment may be influenced by a wider variety of underlying processes. For example, the composition of industries prevalent in an urban area may impacted both unemployment rates and water demand characteristics. Overall, the first principal component gave the highest weight to the economic productivity of water, followed by the unemployment rate. These results indicated a significant relationship between the economic health of a region (e.g. GDP and unemployment rate) and water conservation.

The second component characterized how water and wastewater resources were allocated in municipal regions. The percent of water resources allocated to the public water supply had the largest weight and it was negatively correlated to wastewater reuse. Wastewater reuse may have offset some demand for potable water from the public supply for consumptive uses and the discharge of pollutants into ecosystems. The largest sources of consumptive water demand included agricultural use, landscape irrigation, and thermoelectric energy production use, which were also the largest end uses for wastewater reuse in the dataset. Wastewater reuse was highest in cities that allocated fewer water resources to the public supply.

The third component described consumptive water demand, wastewater reuse, and water demand intensity, which characterize socio-environmental dynamics of water demand. Cities with the highest scores along this principal component had the highest rates of wastewater reuse, higher than average population densities and lower than average water demand intensity. Cities located in arid climates were among the highest scoring cities along this component.

The largest cities in the dataset were all placed in the third cluster, based on their high scores across the first and third components. No clear relationship existed between precipitation and water demand or economic characteristics among cities in this cluster. The first cluster of cities was dominated by small to medium sized cities from arid and semi-arid climates. These observations were consistent with research which demonstrated that as cities grew in size there was a decrease in the regional heterogeneity in water use characteristics (Darrel Jenerette and Larsen 2006). While the inclusion of environmental and economic indicators was limited by the WEI selection criterion, the results were consistent with critiques of other indicators on the importance of these characteristics in describing urban water use (Plummer et al. 2012).

2.5.2 Aggregation Method

As described in the methods section, the linear aggregation of indicators through PCA implied a compensatory logic where weights could be viewed as trade-offs rather than measures of importance. The Condorcet aggregation approach was selected as a non-compensatory approach to aggregate indicators into the WEI (OECD et al. 2008). Table 2-4 compared the WEI ranks for cities obtained using the PCA and Condorcet aggregation approaches. Except for three cities, the rankings obtained from the Condorcet approach were within 25% of the rankings obtained from PCA. Phoenix had the largest deviation in rankings between the two methods. The largest difference was observed for Phoenix, AZ, USA, which was ranked near the middle by PCA method but among the lowest by the Condorcet approach. Most cities (35 of 49) received ranks within 10% of the ranks obtained by PCA. The differences in rankings obtained for the highest-ranking cities and lowest ranking cities were less than 10%. This was consistent with results from (Munda 2012), which observed that linear and non-compensatory methods resulted in the most notable differences for middle ranking positions. These results validated the WEI and enhanced the understanding of potential trade-offs between indicators, as illustrated by the similar results achieved using PCA and non-compensatory aggregation approaches.

Table 2-4 Comparison of WEI city ranks by PCA and Condorcet methods

Category 1 City Ranks				Category 2 City Ranks				Category 3 City Ranks			
City	**PCA**	**Condorcet**	**% Diff.**	**City**	**PCA**	**Condorcet**	**% Diff.**	**City**	**PCA**	**Condorcet**	**% Diff.**
Albuquerque	38	33	10.2%	Anchorage	28	37	18.4%	Austin	5	12	14.3%
El Paso	36	36	0.0%	Augusta-Richmond	42	28	28.6%	Chicago	17	8	18.4%
Jonesboro	47	47	0.0%	Cleveland	25	27	4.1%	Dallas	3	2	2.0%
Lakeland	34	31	6.1%	Columbus	41	41	0.0%	Denver	8	5	6.1%
Little Rock	30	34	8.2%	Decatur	34	45	22.4%	Des Moines	1	4	6.1%
Phoenix	44	23	42.9%	Fayetteville	15	18	6.1%	Green Bay	10	16	12.2%
Reno	40	43	6.1%	Johnson City	32	38	12.2%	Jacksonville	21	20	2.0%
Wichita	24	30	12.2%	Kansas City	16	26	20.4%	Los Angeles	4	3	2.0%
Yuma	49	49	0.0%	Lafayette	42	40	4.1%	Miami	19	13	12.2%
				Louisville	27	25	4.1%	Milwaukee	12	19	14.3%
				Mobile	48	48	0.0%	Owensboro	39	42	6.1%
				Myrtle Beach	33	39	12.2%	Philadelphia	7	10	6.1%
				Nashville	18	21	6.1%	Portland	26	22	8.2%
				Orlando	20	7	26.5%	Rochester	11	14	6.1%
				Pensacola	31	32	2.0%	Salt Lake	6	9	6.1%
				Racine	29	24	10.2%	San Francisco	2	1	2.0%
				San Antonio	13	17	8.2%	Seattle	8	11	6.1%
				San Diego	13	6	14.3%				
				Savannah	45	44	2.0%				
				Syracuse	46	46	0.0%				
				Tulsa	23	15	16.3%				
				Waterloo	22	29	14.3%				
				Winston-Salem	36	35	2.0%				

2.6 Conclusions

The WEI, developed with weights and aggregation through PCA, emphasized the importance of economic health in water conservation and allocation behaviors. For example, the socio-economic component described 40% of the variability in data observed across 49 metropolitan areas. Within this component, economic productivity of water resources and unemployment received the highest weights, followed by water demand intensity and population density. The second component was comprised of indicators for water and wastewater resources allocations. The third component encompassed the socio-environmental variables, with consumptive water demand and wastewater reuse receiving the highest weights.

The urban WEI ranks achieved by the PCA methodology were compared to a non-compensatory approach to evaluate if the theoretical differences in aggregation methods resulted in significant differences for WEI ranks. The difference in ranks obtained by the non-compensatory method were within 25% of the PCA WEI ranks for all but three cities. More than half the ranks were less than 10% different. Overall, the PCA approach provided valuable insight into the correlations between indicator variables and a strategic methodology for identifying variable weights and aggregation of components.

The WEI illustrated social, economic, and environmental factors that contributed to the dynamics of water demand in an urban area. This indicator provided novel insight into the connections between economic health and water conservation in urban areas. Due to limited data, no indicators were included to characterize downstream environmental needs, which was a limitation in evaluating the third principal component of socio-environmental water demand characteristics. The conceptual framework for the WEI provided a flexible strategy to examine how the urban water-economy nexus influenced water resource conservation and reuse and could aid cities in benchmarking and advocating for water management policies.

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Chapter 3 Sustainability Assessment for Indirect Potable Reuse: A Case Study from Reno, Nevada

3.0 Abstract

A triple bottom line (TBL) approach was used to examine the trade-offs between potential reclaimed water management strategies in a closed basin. The goals of the water management strategy included minimizing water source shortages, ensuring safe and resilient future water supplies, and protection of inland ecosystems through adequate surface flows. The TBL approach consisted of quantitative and qualitative impact assessments of social, environmental, and economic criteria. This research examined how potable reuse (PR) of reclaimed water addresses water needs in a closed basin such as water quality, reclaimed water disposal management, growing water demand, balancing groundwater extraction rates with inflows, preserving inland ecosystems, and ensuring a locally controlled safe drinking water source. The TBL assessment first evaluated water stress based on water demand and supply under status-quo conditions. The results were compared with the PR scenario which provides more environmental and social benefits than status-quo scenario.

3.1 Introduction

Water management has evolved over recent decades to become more holistic, integrating management of surface water, groundwater, storm water, and related resources such as land and energy use. Flows and contaminant loads to water resources are influenced by other aspects of the urban water cycle, which encompass a variety of activities impacting return flows, treatment, distribution, storage, and drainage (Termes-Rifé et al., 2013). Management decisions impacting the urban water cycle can influence social and environmental benefits from water use at a regional scale. The adoption of wastewater as an aspect of regional water management and planning has been facilitated by the need to sustainably address increasing water demands and supply uncertainties, particularly in water-scarce regions. To address the growing complexity of integrated water resource management for decision making, a triple bottom line (TBL) approach can be taken for trade-off analysis.

TBL assessments examine key inputs (e.g. energy requirements, chemical additions, finance) and outputs (e.g. effluent flows, contaminant loads, solids production, and greenhouse gas emissions) to assess how the surrounding community, ecosystems, and economy may be impacted by different water management or infrastructure decisions. These decision support tools use a variety of quantitative measurements and qualitative indicators to compare benefits and costs to the social, environmental, and economic systems (Rathnayaka et al., 2016).

There is no singular framework used to carry out a TBL assessment, and the approach followed may vary depending on the scale or stage of a project. Additionally, TBL can be integrated with other approaches such as life-cycle assessment (LCA) to account for some environmental impacts or can be integrated with economic models to develop a cost-benefit assessment (Raucher, 2006). All approaches involve utilization of quantitative and qualitative assessment techniques to determine the costs of each water management scenario and various environmental and social impacts or benefits. However, cost-benefit assessments may be limited in their inclusion of social and environmental effects due to limitations of modeling and data to convert non-economic impacts into monetary values (Liner and deMonsabert, 2011).

In a TBL assessment, environmental impacts may be assessed using an LCA model (Chhipi-Shrestha et al., 2017; Godskesen et al., 2013; Rygaard et al., 2014) or environmental criteria selected on the basis of the project and region (Rathnayaka et al., 2016). Social impacts for water management can examine factors that impact the public and employees impacted by the change examined, such as the risk of exposure to pathogens and public acceptance (Garcia and Pargament, 2015; Makropoulos et al., 2008; Rygaard et al., 2014). The impact of water reuse projects on outdoor watering restrictions may also be a significant social benefit (Mansur and Olmstead, 2012). However, impacts that may characterize this social benefit, such as water restrictions or supply resilience, are less commonly characterized in TBL analyses (Liner and deMonsabert, 2011; Rathnayaka et al., 2016). Economic criteria generally cover the financial expense of the project alternatives including capital, maintenance, and operational costs of water and wastewater treatment systems. Indirect costs and benefits, such as hydropower production and byproducts of treatment (e.g., phosphorus recovery), can be used to expand the economic criteria by including regional impacts that are external to water and wastewater treatment (Rathnayaka et al., 2016).

This paper develops a TBL approach that uniquely characterizes the drivers for potable reuse (PR) in a semi-arid inland community. In closed hydrologic basins, which are most commonly found in arid regions around the world, reclaimed water discharges can have lasting impacts on watersheds due to pollutant loads and return flow volumes. Integrated water and wastewater management strategies are needed to protect the ecosystems and species that rely on limited water resources and to ensure that reclaimed water discharge provides a beneficial impact ecosystem resilience (Cooper and Koch, 1984). Reuse of treated wastewater can provide solutions by both expanding water supplies and providing a beneficial use for the reclaimed water. While previous TBL studies have formulated approaches that examine the feasibility of urban water reuse in water stressed regions (Garcia and Pargament, 2015; Makropoulos et al., 2008; Rygaard et al., 2014), these studies have not emphasized drivers that characterize the needs of closed basin inland regions described above.

This study develops a TBL assessment strategy that characterizes the environmental and social criteria that act as drivers for water-stressed closed loop inland communities. This was achieved through a combination of quantitative and qualitative methods to assess economic, social, and environmental impacts of PR compared to a status-quo (SQ) scenario for future water and wastewater management in the metropolitan area of Reno-Sparks, Nevada. The goal of the study is to determine whether PR of treated water may provide a more sustainable water supply for the region. This is assessed through a trade-off assessment that reveals the benefits and costs of the two scenarios.

3.2 Methods

This project was carried out in three phases. In the first phase, a review of the regional water and wastewater resources was conducted, including projections for future demands. In the second phase, criteria representing potential environmental, social, and institutional impacts from water infrastructure management decisions were identified from previous TBL analysis studies (Garcia and Pargament, 2015; Makropoulos et al., 2008; Rathnayaka et al., 2016; Raucher, 2006); these criteria encompassed aspects such as pollutant emissions, biodiversity, water supply sustainability, land use change, employment, wages, costs, health and safety. The final criteria for the TBL analysis were then selected following a value tree approach to narrow down quantitative criteria and indicators related to the regional drivers for water and wastewater management strategies. The criteria assessments were normalized to a standard scale by setting goals, or benchmark values for each criterion and calculating the percent difference between the scenario scores and the benchmark. PR and SQ scenarios were then compared based on the trade-offs between social, environmental, and institutional criteria.

3.2.1 Water Challenges for Reno-Sparks Metropolitan Area in Nevada

Northern Nevada receives an average of only 18 cm of precipitation per year and faces challenges to sustainably manage water resources due to growing demands for residential and commercial uses, requirements for high quality discharge with limited disposal options of reclaimed water, and competition between residential, commercial, agricultural, and ecological demands for water. Rising global temperatures are expected to cause more frequent droughts and a shift towards a higher proportion of rain rather than snow precipitation, which is the primary source for potable water to the region (CDWR, 2015). Within the Reno-Sparks metropolitan area, the planning area of North Valleys was selected for this study; it is one of the most rapidly growing areas in the metropolitan region and receives a small natural supply of groundwater. However, over-extraction of groundwater resources has resulted in lowered groundwater tables.

Table 3-1 presents the projected water demand and supply flows for the planning year 2035 in the North Valleys. This data illustrates that most of the water supplies to this area are imported from either an adjacent basin within the municipality (imported surface water) or groundwater imported from outside the city; groundwater supplies local to the North Valleys comprise a relatively small fraction of its total water supplies. Residential water demand accounts for more than seventy percent of total water demands in the study area (Table 3-1). As this water balance illustrates, future water demands will rely largely on imported water resources. The conjunctive use plan for the North Valleys also utilizes surplus flows of imported surface water in an aquifer storage and recovery (ASR) program to stabilize the water table in the North Valleys area (TMWA, 2016a).

Table 3-1. Water balance for 2035 estimates in regional growth, adapted from (NNWPC, 2017)

Water supply/demand	North Valleys area flow (Mm^3/y)
Water supplies	
Local surface water supply	-
Import of Truckee River water	4.93
In-basin groundwater supplies	3.12
Imported groundwater	9.87
Total supply for area demands	18.01
Water demands in 2035	
Non-residential demand	1.77
Residential demand	14.25
ASR demand	0.62
Well conversion	2.99
Other demand	-
Total water demand	19.63
Water supply surplus (deficit)	(1.62)

Non-potable reuse plays a crucial role in reclaimed water management throughout the region; however, the scarceness of surface water flows creates a complicated dynamic for management of water and wastewater resources within this area. In the North Valleys, reclaimed water is used to fulfill non-potable water demands such as ecosystem support and irrigation. However, these demands are seasonal and are smaller than the flow that will be generated at the water resource recovery facilities (WRRF) by 2035. The Swan Lake Playa is a wetland habitat that relies on reclaimed water from the North Valleys area to maintain habitat crucial to migratory waterfowl. Maximum and minimum flows to the playa are specified through a discharge permit to preserve habitat and minimize the flood risk to nearby housing communities.

The use of reclaimed water for irrigation is limited by both customer demand and the lack of seasonal storage facilities.

Wastewater flows in the North Valleys planning area may increase to nearly 11 ML/d by 2035 primarily due to population growth (NNWPC, 2017). Evaporation ponds are currently used to manage reclaimed water flows that exceed demand but are an infeasible solution for future flow increases due to limited space and flood risks. Additionally, the discharge of effluent into larger watersheds in the region is infeasible due to the ecological sensitivity of these closed basins. Local demand for non-potable reclaimed water is not expected to increase significantly from present day. Strategies to manage the future increase in reclaimed water were explored in a regional water planning study (Eco:Logic, 2010). The most feasible solutions identified include exporting reclaimed water into a distant watershed (SQ scenario) and groundwater augmentation (PR scenario) for future use.

3.2.2 Determining Assessment Criteria

To assess the potential costs and benefits of the water and wastewater management scenarios, TBL criteria were identified across sustainability (Makropoulos et al., 2008; Rathnayaka et al., 2016) and economic (Raucher, 2006) decision making frameworks. To select criteria that were most relevant for the study area and were appropriate for a planning phase assessment, a value tree approach was followed. A value tree involves first identifying the regional goals and needs that are driving the project, then separating them into social, environmental, and institutional values to select decision making criteria (Figure 3-1). Institutional criteria included capital and operating costs as well as the value generated by increased local control of water resources. Environmental values reflected the desire to manage water resources sustainably to protect supplies for residential and environmental uses. The environmental criteria selected examined potential impacts to the Swan Lake ecosystem, the sustainability of groundwater demand, and the stress to water resources, which are drought prone and impact wildfire risk and wetland habitat. This study does not examine several design-specific environmental criteria (e.g. solid waste quantity and quality, greenhouse gas and other emissions, chemical use), which require design beyond a planning phase assessment that are specific to the water quality characteristics of influent to advanced water treatment (AWT) processes and the effluent quality goals at individual treatment processes. While these environmental criteria can play a large role in deciding between specific advanced water treatment options, they were excluded due to a lack of information about the chemicals and energy required to achieve the regulatory requirements through advanced water treatment processes in this case study. The social values focused on the potential impacts to the community within the planning area. Social criteria included a health risk assessment and a qualitative assessment of potential impacts to water conservation and water use restrictions.

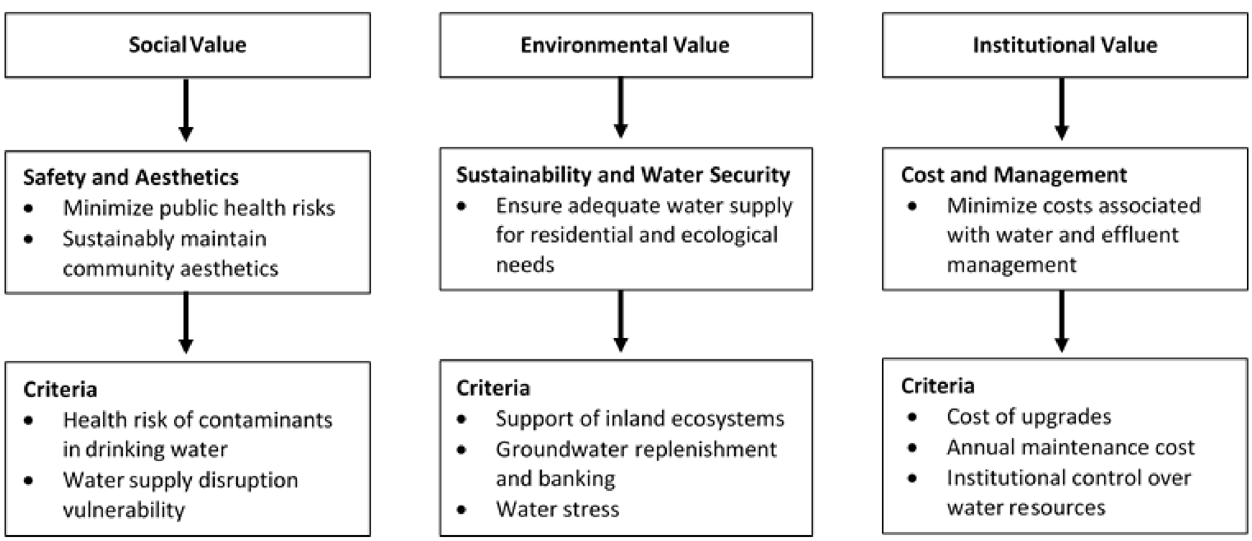

Figure 3-1 Value tree for sustainable water management in the Reno-Sparks metropolitan area

After generating quantitative or qualitative comparisons of the SQ and PR scenarios, the impacts were assessed as percent change from the criteria benchmark. These change measures were then converted into a standard score based on the magnitude and direction of the impact relative to the desired range and scale of impacts relative to the benchmark. The score scale ranges from -3 to +3 with negative numbers indicating a cost or loss, positive scores indicating a benefit. Benchmark selection and scaling for scores are described for each criteria below.

3.2.2.1 Environmental criteria

Environmental impacts were assessed based on projected future potable water demands (TMWA, 2016a) and the 2035 regional water and wastewater balance projections (NNWPC, 2017). The water supply available for aquifer replenishment under the PR scenario was assumed to equal the reclaimed water flows considered “non-beneficial” in the SQ scenario. Flow to Swan Lake in this scenario were assumed to be double the minimum discharge requirement, which was assumed to be an allocation that would provide additional environmental benefits but well under the maximum discharge allowed. Flow to swan lake in the SQ scenario was assumed to reach the maximum allowed annual discharge. Table 3-2 identifies the calculated volume of water available for advanced treatment for PR to be 2.76 Mm^3/y (8 ML/d). This flow was generated by eliminating flows of reclaimed water to evaporation ponds and export, while also reducing flows to the inland habitat to reduce flood risk. This number does not consider the hydrologic limitations of injection or spreading basins that would govern the actual supply of potable reuse water to groundwater aquifers. PR was assumed to have no effect on the flows of exported waste activated sludge (WAS) and non-potable uses, such as irrigation and landscaping. The environmental criteria of the groundwater balance, potable water supply availability, and flows to inland ecosystems indicated the largest environmental challenges for the region: stabilizing groundwater levels, meeting the increase in water demands, and preserving ecosystem services from the playa while minimizing flood risk.

Table 3-2. Reclaimed water flow available for groundwater replenishment.

Reclaimed water	SQ flow (Mm^3/y)[5]	PR flow (Mm^3/y)
Reclamation facility incoming flow	4.73	4.73
Swan Lake disposal flow[1]	3.24	1.17
Evaporation disposal	0.41	0.00
WAS export[2]	0.25	0.25
Purple pipe reuse	0.56	0.56
Supply available for potable reuse[3]	0.00	2.21
Surplus reclaimed water[4]	0.27	0.00

1. A minimum of 0.6 Mm^3/y and maximum of 3.24 Mm^3/y is allowed
2. WAS is transferred to a nearby WRRF for solids handling
3. Assumed to be 80% of the non-beneficial use in Status-quo scenario
4. Surplus reclaimed water must be exported out of basin
5. SQ flow utilizes the projected 2035 North Valleys wastewater balance from NNWPC (2017).

To evaluate the potential for PR to enhance the current conjunctive use program for water banking and to prevent aquifer depletion, a groundwater balance was examined based on water demand and inflow projections (TMWA, 2016a). The groundwater balance calculated the natural inflows and artificial aquifer replenishment through PR and ASR compared to the total groundwater demands in the region, which consist of municipal withdrawals and private domestic wells. The goal for the groundwater criteria was to achieve inflows greater than or equal to future groundwater demands for the North Valleys, which are identified to be 4.2 Mm^3/year (TMWA, 2016a). Flows greater than this benchmark received a score of 1 per every 25% change in groundwater supplies with magnitudes reflecting increases or decreases.

To evaluate the effect of the SQ and PR scenarios on wildlife habitat, the flow to Swan Lake was evaluated. Minimum and maximum flows into Swan Lake are mandated for the WRRF through discharge permits. These flows ensure that sufficient recycled water is supplied to provide scrub habitat and adequate water depths for migratory waterfowl. Development in the North Valleys has resulted in increased runoff drainage into Swan Lake, elevating the flood risk of nearby residential communities (NNWPC, 2017). The potential flows into Swan Lake were compared in the SQ and PR scenarios and were normalized based on the benchmark of discharging the 150% above the minimum required flow requirement (0.9 Mm^3/y). The benchmark value for this criterion was selected due to uncertainty related to future precipitation and evapotranspiration rates that control flood risk and ecosystem needs at Swan Lake. Results were scaled based on deviation from this goal, receiving a score of -1 for every doubling of flow, a score of +1 for achieving flow within 50% of the benchmark, a score of +2 for flows within 25% of benchmark, and a score of +3 if it achieves the benchmark.

Water stress index (WSI) was utilized to examine the sustainability of anthropogenic demands on potable water resources. Technical water stress identifies the ratio of water withdrawals for anthropogenic use to total sustainable inflow (Falkenmark and Lundqvist, 1998). Demographic pressure on water considered household water needs to identify the contribution of population requirements on available water resources (Vörösmarty et al., 2000). The index value is calculated assuming a household requirement of 100L per capita per day for an unstressed water resource, water resources were considered scarce when water availability drops below 1000 m^3 per capita annually and absolute scarcity below 500 m^3 per capita annually (Falkenmark and Lundqvist, 1998). In the PR scenario, regional water supplies were assumed to include the supply available for potable reuse, as specified in Table 3-2. The WSI achieved under SQ and PR scenarios were normalized based on a benchmark value of technical water stress less than 1, which would indicate that water supplies are greater than demand. Results were scaled based on a maximum score of 3 for a WSI of 40%, with a score of 0 if the benchmark is met and scores increasing or decreasing by a point for every 20% the WSI is decreased or increased, respectively.

3.2.2.2 Social criteria

Two social criteria were selected based on the regional values, health risk of potable water supplies and the impact of each scenario to water supply resilience. The health risk was assessed based on a literature review of pathogens and trace organic compound concentrations in various potable reuse treatment trains. Contaminant concentrations in drinking water were found to vary widely based on regional inputs as well as treatment trains and are thus a function of advanced treatment process design. For example, some emerging contaminants were not readily removed by many advanced treatment processes but their presence in wastewater vary widely and can influence treatment train design (Quiñones and Snyder, 2009). The health risk was assessed through microbial and chemical risk assessments.

The microbial risk assessment process evaluated the likelihood of adverse human health effects from each exposure to a drinking water source in which pathogens may be present. Dose-response models address the probability that an individual will be infected by a single exposure. The assessment assumes that individuals drink 1.2 L of un-boiled water per day from either the SQ groundwater or PR water over the course of a year to determine annual risk of infection (NRC, 2012). Uncertainties in this analysis arose from the variability in pathogen infectious units and pathogen density across seasons and geography, and neglect of predation and die-off (Fong and Lipp, 2005; Haas et al, 1999). However, pathogen removal efficiency during treatment was likely to be the largest source of uncertainty (NRC, 2012). Pathogen density in advanced treated water was calculated by identifying pathogen densities reported in literature for primary effluent (Huffman et al., 2006; Tchobanoglous et al., 2003) and applying the regulated pathogen log-removals (12-log removal of virus and 10-log removal of *Cryptosporidium* and *Giardia*). These pathogens have been widely studied in literature due to their importance in waterborne illness and all have reported dose-response relationships (Haas et al., 1999; Hunter et al., 2011) that identify the probability of infection as a proportionate response of the mean pathogen dose. A review of monitoring data identified $1.3x10^6$ gc/L *Norovirus* (Nordgren et al., 2009), 100 oocysts/L *Cryptosporidium* (Hachich et al., 2013; Huffman et al., 2006; McCuin and Clancy, 2006), and 400 cysts/L *Giardia* (Hachich et al, 2013; Huffman et al., 2006) as conservative estimates for pathogen density in primary effluent. Annual risk of illness was then determined based on the daily risk identified from the dose-response relationship according to the exponential dose-response model given in Hunter et al. (2011). The margin of safety for microbial risk was assessed based on a risk tolerance of 10^{-6} disability adjusted life years per person per year, which is equivalent to an annual diarrheal risk of illness of 10^{-4} per person per year (NRC, 2012). The microbial risk assessment used a benchmark margin of safety of 10, with results scaled to one point per every 100% deviation from this goal.

Chemical contaminant risk in drinking water was assessed based on the potential concentrations of contaminants of emerging concern (CECs) and disinfection byproducts (DBP) in each scenario. The health risks associated with DBPs in both scenarios were evaluated for Total Trihalomethanes (TTHM) and Haloacetic Acids (HAA5). The SQ assessment used the highest reported concentrations in local groundwater wellhead monitoring data, which have elevated DBP formation due to the ASR program (TMWA, 2016b). In the PR scenario, DBPs present in advanced treated water were determined based on maximum levels reported in literature for various treatment trains. The margin of safety for DBPs was determined based on the ratio of contaminant concentration in drinking water to the risk-based action levels (RBALs), such as a drinking water maximum contaminant level (MCL).

This assessment included CECs that were recommended for monitoring by multiple potable reuse frameworks (NRC, 2012; Soller et al., 2015; Tchobanoglous et al., 2015), including pharmaceuticals and personal care products (PPCPs), household chemicals, and flame retardants (Table 3-4) to the exclusion of contaminants for which insufficient data was available. There was not a consensus for safety guidelines of unregulated contaminants. A literature review identified RBALs for CECs as well as the potential concentrations in advanced treated water; the most conservative values were selected. In compounds that are suspected carcinogens, a slope factor gives the upper-bound risk estimate of chemical dose (Bruce et al., 2010).

The presence of several contaminants after advanced water treatment is poorly studied in literature, such as, Sucralose. Additionally, formation of N-Nitrosodimethylamine (NDMA) was found to be highly dependent on treatment train design (Quiñones and Snyder, 2009), and hence, data available from a regional pilot-scale study was utilized (Sundaram and Emerick, 2010). Pharmaceuticals were evaluated based on reported minimum tolerated doses (MTD), which were assumed to correspond to a cancer risk of 1 in a million (NRC, 2012). Doses were converted into a drinking water exposure level (DWEL), assuming an average person weight (70 kg) and drinking water intake (1.2 L) over a 70 year life span (Soller et al, 2015). The RBALs selected account for uncertainty factors that consider effects on sensitive subpopulations as well as variability in methods and uncertainty in hazardous levels. The chemical risk assessment was concluded by identifying a margin of safety expected for each contaminant. These were then compared to a benchmark margin of safety of 5. Results were scaled based on deviation from this goal, receiving a point change in score for every 100% change in the margin of safety from the benchmark. Assessment of CEC health risk was assumed to be negligible in the SQ scenario based on limited data for the conventional drinking water resources. A study of organic compounds in the Reno-Sparks area drinking water supplies examined several hundred chemical contaminants and detected few CECs, such as camphor and bisphenol A, at concentrations well below 0.1 µg/L (Thomas, 2009). However, no references were found that recommend monitoring of these CECs in potable reuse because drinking water is unlikely to be the greatest source of exposure (NRC, 2012).

The second social criteria selected examines the potential impact of each scenario (SQ and PR) on residential water service disruptions. Residential uses include household and landscaping, with household use representing a fixed water demand needed to fulfil the sanitary and survival needs of residents and landscaping use representing a dynamic demand that is responsive to changes in water price, use restrictions, and weather. Benefits of water for landscaping include maintaining community aesthetics and property values. Outdoor watering restrictions are used throughout the Reno-Sparks metropolitan area as a demand management strategy that encourages water conservation goals by reducing demand for potable water resources for residential and commercial landscaping. The resilience of the water supply was assessed through use of an indicator that signifies the fraction of total water consumption that comes from non-sustainable water resources (Wada et al., 2014). The blue water sustainability index (BlWSI) incorporates regional use of non-renewable groundwater and environmental flow requirements to assess water supply sustainability (Wada et al., 2014). In the North Valleys area there were no natural surface water flows used for potable water supply, so the BlWSI was the ratio of all non-renewable groundwater abstractions to the total water consumption for domestic, industrial, and agricultural sectors. The non-renewable groundwater abstraction for the North Valleys was equal to the difference between total groundwater abstraction and total groundwater recharge, which includes natural, return flow, and artificial recharge supplies. This calculation utilizes projections for future groundwater demands and natural recharge rates described in TMWA (2016a) in the SQ scenario, while the PR scenario includes additional artificial recharge based on the PR flow calculated in Table 3-2. The total water consumption for domestic, industrial, and agricultural sectors for the North Valleys was calculated based on the difference between the sum of sector water demands in the region and the inflows to the regional WRRF, with water demand and wastewater flows projected (NNWPC, 2017). A benchmark of 0.05 BIWSI was selected based on the present day index values identified in the region (Wada et al., 2014). For every 50% decrease or increase in the BIWSI the score of a scenario was increased or decreased by 1, respectively.

3.2.2.3 Institutional criteria

The capital and operating costs for the next 8 ML/d expansion of wastewater treatment capacity in the North Valleys were estimated in a previous report (Eco:Logic, 2010), including the potential costs for reclaimed water management solutions like export (equivalent to the SQ scenario) and advanced treatment with O_3-BAC prior to aquifer injection (equivalent to the PR scenario). Capital costs for the wastewater utility include the costs of upgrades needed in both SQ and PR scenarios to expand and meet discharge requirements with increasing wastewater flows. This study updates the capital and operating and maintenance (O&M) costs of PR and the SQ scenarios to present day costs based on changes to the projected future population, and the ENR Construction Cost Index for 2017.

The necessary expansion and upgrades to the WRRF to meet future flows was used as the benchmark for normalizing the additional cost burden that either scenario might impose. The benchmark capital costs included grit removal, expansion of secondary reactors and clarifiers, and tertiary filtration. Under the SQ scenario, additional capital costs included UV disinfection, a pump station, and distribution to the proposed discharge site Long Valley creek. Under the PR scenario, advanced treatment processes were assumed to utilize membrane-free technology due to the costs of brine disposal for inland communities. Capital costs in the PR scenario included filtration, UV disinfection, O_3-BAC, distribution to the proposed injection well, conveyance from the recovery site, and development of regulatory programs and public outreach. Conveyance and distribution system costs for the PR scenario were calculated based on expected distance to the injection well and extraction well in the PR scenario. The extraction well must be sufficiently far from the injection site to provide a minimum of six months retention time in the aquifer to satisfy the regulatory requirements of virus removal for indirect PR in Nevada, as well as beneficial chemical transformations (NRC, 2012). Scores for the capital cost criteria were assessed by determining the percent change of either scenarios cost compared to the benchmark cost, with a point deduction for every 50% cost increase.

O&M costs for potable water service, and WRRF processes through secondary treatment (including chemical use, power and in-plant pumping costs) were assumed to be necessary under any scenario. These costs were taken as the benchmark to compare the additional cost burden of the SQ and PR scenarios. Each scenario included additional pumping costs based on the headloss to the discharge site or injection and extraction wells. The SQ scenario included costs for pumping to the discharge site at Long Valley creek, O&M of the UV system, power costs for pumping and cooling, and O&M costs for UV disinfection. The PR scenario included costs for pumping to the injection well, pumping costs to retrieve PR water from the extraction well, O&M of the UV system, power costs for ozone generation and backwash pumping, and carbon media replacement. The PR scenario included an additional annual cost for regulatory oversight. Scores for the O&M cost criteria were assessed in the same manner as the capital costs, with a point deduction for every 50% cost increase above the benchmark.

The final institutional criteria was local control over water resources. Local control reduces the susceptibility of water supplies to the rules or policies of neighboring localities or other entities (Tchobanoglous et al., 2015). This criterion used the present day water balance as a benchmark based on the percent of total potable and recycled water resources supplied and retained locally. Local control over water and reclaimed water resources provided benefit by enhancing the opportunity for administrative and management decision making. In each scenario the percent of water and recycled water resources retained locally was examined to evaluate the impacts from a loss of reclaimed water under the SQ scenario and an increase in local potable water through PR. Benchmarks of 50% local control of potable water supplies and 100% local control over reclaimed water were selected as feasible goals for this criteria. Scores differentiated every 10% gain or loss in local control of either resource.

3.3 Results & Discussion

3.3.1 Environmental Impact Assessment

Future projections indicate that a water supply deficit may occur under the SQ scenario, which could potentially result in further depletion of stressed groundwater resources. While reclaimed water provides environmental benefits through flows to the wetlands and irrigation, future flows may exceed these beneficial uses. The potential environmental benefits through PR were examined to see if they could mitigate potential losses under the SQ scenario by offsetting groundwater withdrawal and providing sufficient supplies to meet future anthropogenic demands and ecosystem needs. These environmental criteria can also provide secondary social benefits that were not quantified in the social impact analysis such as flood risk mitigation, improved management of a recreational area, and reduced pumping costs to domestic and municipal wells.

3.3.1.1 Change in water quality and flow to groundwater

To assess the impact of PR groundwater replenishment on groundwater flows, the artificial recharge to groundwater was estimated from surplus reclaimed water that was not providing a beneficial use, as identified in Table 3-2. The increase in groundwater flows under the PR scenario is assumed to be 80% to the supply allocated for reuse, accounting for losses and inefficiency in aquifer replenishment (Table 3-3). The PR scenario is estimated to increase groundwater inflows by approximately 60% compared to the SQ scenario. However, actual replenishment volumes would depend on the limitations of injection or soil aquifer treatment strategies to recharge the aquifer.

Table 3-3. Future groundwater flow in status-quo and PR scenarios

Supply	SQ flow (Mm^3/y)	PR flow (Mm^3/y)
Groundwater, natural recharge	3.21	3.21
Groundwater, ASR	0.62	0.62
Groundwater, PR[1]	0	2.21

1. Groundwater replenishment through PR is assumed to be 80% of the available supply.

Groundwater inflow under the SQ scenario fell 25% below the benchmark flow of 4.2 Mm^3/y, while the PR scenario was estimated to increase groundwater supplies by more than 40%. This increase in groundwater supplies resulted in a score of 2 for the PR scenario and was expected generate benefits in conjunctive use of water resources, groundwater table stabilization, and water banking. Further hydrological analysis must be pursued to determine the suitability of aquifers in the region for water banking. However, the ASR program has successfully demonstrated use of unconfined aquifers in the North Valleys for aquifer replenishment. In the SQ scenario, the combined natural and artificial recharge flows were not sufficient to balance projected groundwater demands, potentially resulting in further depletion of the aquifer.

3.3.1.2 Water supply to inland ecosystems

Flow of water under either scenario was required to stay within the limits specified in the WRRF discharge permit. These flow requirements were assumed to be sufficient to ensure a healthy habitat. Under the SQ scenario, discharge of reclaimed water into the Swan Lake playa and wetland was projected to increase to the maximum allowed flow (3.4 Mm^3/y), as shown in Table 3-2. In the PR scenario, flows to Swan Lake were reduced to 1.2 Mm^3/y, approximately double the minimum flows required to support the habitat. Overall, flows to Swan Lake in the PR scenario were approximately one third the flows discharged in the SQ scenario. Increased flows and erosion under the high flow SQ scenario was identified as a risk factor for additional negative impacts, such as greater turbidity in the wetland habitat and increasing the 100-year flood hazards for the surrounding properties.

The benchmark of flow to Swan Lake (0.9 Mm^3/y) was exceeded in both scenarios. The percent difference between flows under the SQ scenario and the benchmark was 260%. Under the PR scenario flows were 30% above the benchmark. The scale for this criterion placed the benchmark as the highest achievable score; the resulting scores for the scenarios after scaling were -2 and +1 for the SQ and PR scenarios, respectively.

3.3.1.3 Water stress

Based on projections for water demands and population growth in 2035, the North Valleys area were found to pass the threshold of scarce water resources and into absolute scarcity, due to technical water stress under a SQ scenario (Figure 3-2). The technical water stress indicated that demand will outstrip supply due to population growth in the study area. In the PR scenario both demographic and technical water stress were reduced due to the larger water supply available to the area. In both scenarios the demographic water stress indicated that there was more than the necessary minimum of 500 m^3 per capita annually, indicating that while demographic pressure on this resource is high, other uses such as irrigation were driving water resources into absolute scarcity. Water resources imported from outside the Reno-Sparks area comprised a large portion of total supplies to the North Valleys in both scenarios, making up 55% of water supplies in the SQ scenario and 48% of supplies in the PR scenario. The SQ scenario fell 10% above the benchmark of a technical WSI of 100%, indicating that water resources could not be managed sustainably without additional resources being found. The WSI in the PR scenario was 5%, giving the scenarios respective scores of -2 and +1. The water stress under the PR scenario indicates that the small flow generated through PR would provide sufficient to meet the increase in future demands without requiring structural change to water allocations.

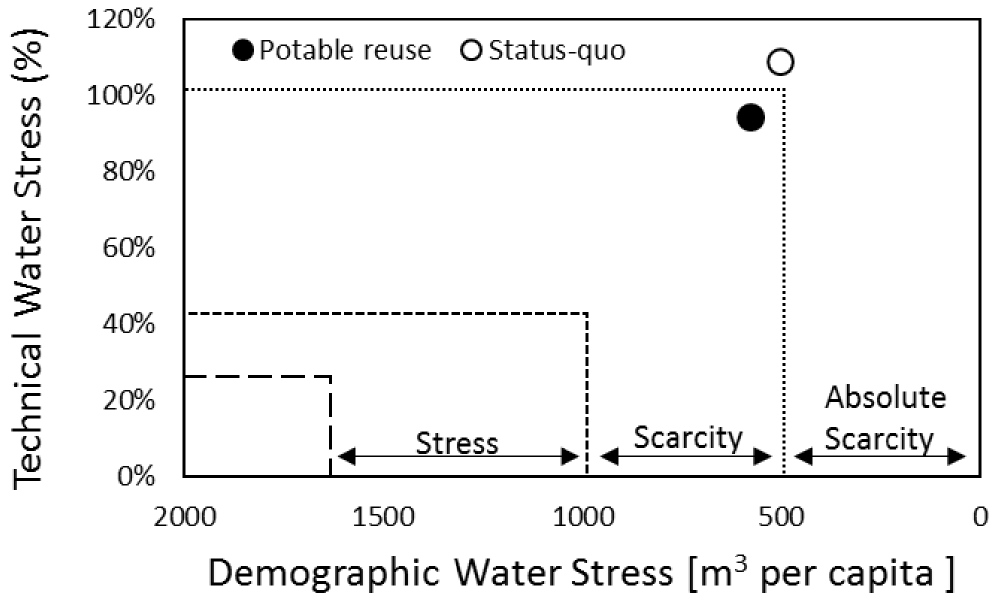

Figure 3-2 Technical and demographic water stress under water management scenarios

3.3.2 Social Impact Assessment

While social impacts from PR projects can be wide ranging, potential health effects and impact of water supply resilience to water utility customers were identified as the drivers in the semi-arid and water stressed study area. Other important social considerations, such as public acceptance are community specific and require additional study (Hartley, 2006).

3.3.2.1 Health risk

Microbial risk was assessed based on a risk tolerance of 10^{-6} disability adjusted life years per person per year, equivalent to an annual diarrheal risk of illness of 10^{-4} per person per year (NRC, 2012). Thus, the estimation of annual risk of illness was compared under SQ and PR scenarios with the standard acceptable annual risk of illness (10^{-4}) to determine the margin of safety (Hunter et al., 2011; NRC, 2012). Pathogen densities prior to wastewater treatment (Tchobanoglous et al. 2003) were used to extrapolate approximate pathogen densities after advanced water treatment based the Nevada Administrative Code (NAC) 445A.425 regulations, which require a 12-log enteric virus reduction, 10-log *Giardia* cyst reduction, and 10-log *Cryptosporidium* oocyst reduction. The resulting pathogen densities after advanced treatment were assumed to represent conservative estimates for pathogen density at the wellhead in a groundwater augmentation application of PR. Dose-response relationships have been reported for *Norovirus*, *Cryptosporidium* and *Giardia* based on empirical observations of infectiousness (Haas et al., 1999; Hunter et al., 2011). These relationships were used to estimate the risk of illness per capita on an annual basis. *Giardia* was estimated to have the highest annual risk, at 4.3×10^{-6} illnesses per capita, while *Norovirus* and *Cryptosporidium* had risks of 2.7×10^{-6} and 1.5×10^{-9}, respectively. The health risk of all three pathogens was well below acceptable risk levels of one in a million based on the estimated concentrations present in secondary effluent (Figure 3-3). Health risk from pathogens in the SQ scenario, identified from wellhead monitoring, observed no violations of pathogen MCLs. This indicated a margin of safety greater than or equal to one. Calculations to derive health risk from pathogens and CECs can be found in the Appendix (Tables 6-3 and 6-4) Margin of safety for the PR scenario ranged from 290 for norovirus to 4×10^{4} for *Cryptosporidium* (Figure 3-3). These values fell above the microbial benchmark margin of safety of 100.

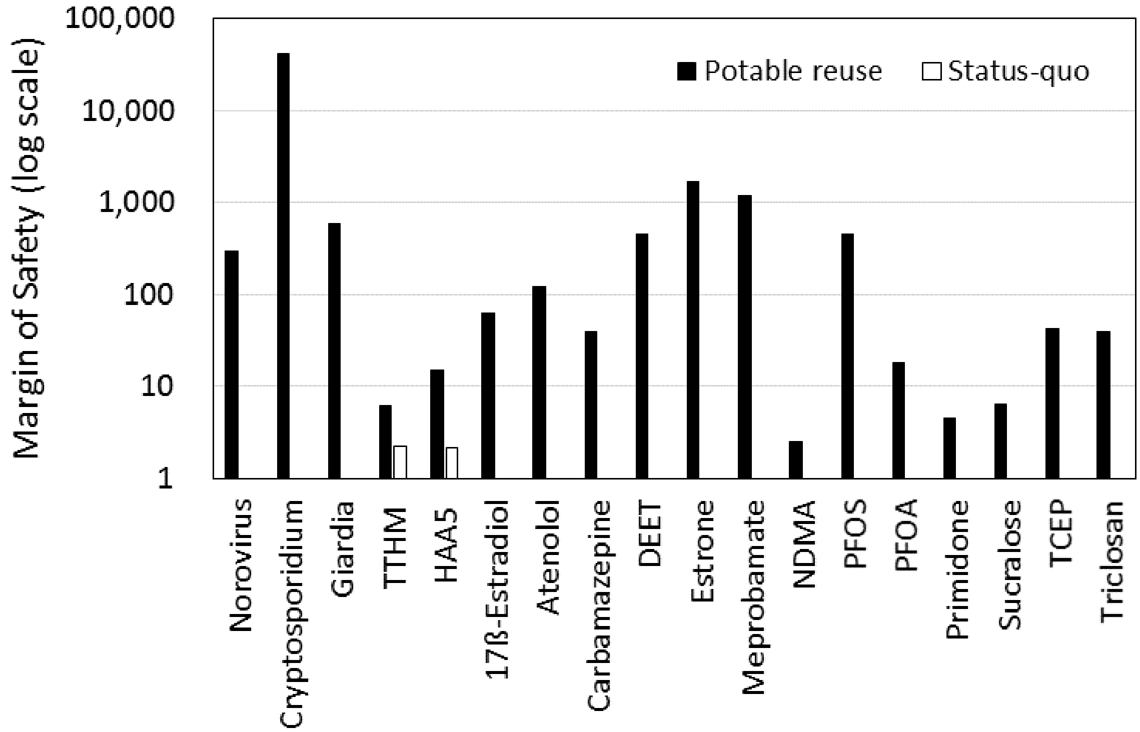

Figure 3-3 Margin of safety for drinking water contaminants

The margin of safety for DBPs was determined based on the highest concentration reported in literature for both TTHMs and HAA5 in the PR scenario, as shown in Table 3-4. Concentrations for DBPs in the PR scenario were not widely reported in literature and did not reflect additional dilution that would occur in the aquifer. However, these concentrations were used to represent a conservative estimate for possible health risk from DBPs in the PR scenario. Wellhead monitoring data from the North Valleys region (TMWA, 2016a) was used to identify the probable concentration of DBPs in the SQ scenario, as shown in Table 3-4. TTHM was found to have the lowest margin of safety, at 6 in the PR scenario and 2 in the SQ scenario. HAA5 also had a margin of safety near 2 in the SQ scenario. DBP risk in the SQ scenario fell approximately 50% below the benchmark minimum margin of safety.

Table 3-4. RBAL and concentrations of DBPs and CECs in PR water based on literature review. SQ concentrations are based on available wellhead monitoring data.

Chemical contaminant	RBAL (ng/L)	PR concentration (ng/L)	SQ concentration (ng/L)
HAA5	6.0E+04[1]	3.7E+04[3]	2.8E+04[2]
TTHM	8.0E+04[1]	13E+04[3]	3.6E+04[2]
17ß-Estradiol	5.8E+02[4]	9.3E+00[9]	-
Atenolol	4.0E+03[5]	3.3E+01[10]	-
Carbamazepine	1.0E+04[5,9]	2.5E+02[10]	-
Estrone	3.5E+02[9]	2.2E-01[9]	-
Meprobamate	2.0E+05[5]	1.7E+02[12]	-
N,N-diethyl-meta-toluamide (DEET)	2.5E+03[9]	5.5E+01[11]	-
NDMA	7.0E-01[6,7]	3.1E-01[13]	-
Perfluorooctane sulfonate (PFOS)	2.0E+02[5,6,9]	4.5E-01[9]	-
Perfluorooctanoic acid (PFOA)	4.0E+02[5,6]	2.2 E+01[12]	-
Primidone	8.4E+02[7,8]	1.9E+02[12]	-
Sucralose	1.7E+05[8]	2.7E+04[5,11]	-
Tris (2-Chloroethyl) phosphate (TCEP)	2.5E+03[9]	6.1E+01[11]	-
Triclosan	3.5E+02[5]	9.1E+00[12]	-

1. EPA Maximum contaminant levels in drinking water
2. TMWA, 2016a
3. Marfil-Vega, 2017
4. AWWA Research Foundation, 2008
5. Crook, 2010
6. NRC, 2012
7. Soller et al., 2015
8. McDonald and Nellor, 2015
9. Anderson et al., 2012
10. Reungoat et al., 2012
11. Lee, 2010
12. Trussell et al., 2015
13. Sundaram and Emerick, 2010

The most conservative RBALs were selected for assessment of CEC health risk. A comparison of the RBALs and the highest concentrations of CECs reported in water after advanced treatment is shown in Table 3-4. All contaminants were expected to be present at concentrations below the RBAL for drinking water (Figure 3-3). NDMA, primidone and sucralose were found to have the margins of safety below 10 in the PR scenario. It should also be noted that primidone RBALs were wide ranging and the lowest RBAL was determined from the maximum recommended therapeutic dose. The margin of safety for NDMA was near 3, which was approximately 50% lower than the benchmark for chemical margin of safety.

High concentrations of DBPs expected in the SQ scenario, and uncertainty in the pathogen densities resulted in an overall margin of safety that was 70% less than the overall health risk benchmark. When adjusted to scale, the SQ risk was determined to have a rank of 0. The health risk in the PR scenario was driven by CECs, particularly NDMA. The overall health risk for the PR scenario fell within the acceptable range for the margin of safety, resulting in a rank of 0.

Uncertainty about health risks from potable reuse water arise not only from the variability of contaminant loads in wastewater, but also due to the large portion of the compounds that comprise total organic carbon and DBPs in reclaimed water that have not yet been identified. Increasingly, bio-analytical screening assays have been recommended to understand how CECs and other organic matter that persists through advanced treatment may present health risk such as biological toxicity or antibiotic resistance (Anderson et al., 2012; Michael-Kordatou et al., 2015). A region-specific study of reclaimed water and advanced treated water characteristics was recommended to further understand health risks such as DBP formation during advanced treatment, site specific evaluations of DBP formation and NDMA, which cannot be adequately rated based on literature review. Despite these uncertainties, the conservative approach taken to estimate the health risk and the additional safety factors that were not accounted for in the PR scenario such as the environmental buffer, aquifer dilution, and extended storage, are likely to ensure adequate time to test and respond to contaminants exceeding safe levels at which no health effects are expected to occur.

3.3.2.2 Water conservation and sustainability

The sustainability of water resource utilization was assessed through the BIWSI indicator. Groundwater demands in this area were found to be over-allocated and were assumed to remain at 4 Mm^3/y under either the SQ or PR scenarios. Overall BIWSI calculated for the scenarios were differentiated by the aquifer recharge contributed by PR. The indicator value in the SQ scenario was 55% larger than the benchmark value, indicating an expected increase in groundwater supply stress. The PR scenario reduced the BIWSI to a value of zero, indicating that the groundwater resources should not be stressed due to supplies exceeding demands. The PR scenario achieved a scaled rank of 2, while the SQ scenario achieved a rank of -1.

Overall, lower water stress and higher water tables may decrease the need for residential water use restrictions when drought periods occur. Similarly, greater resistance to drought and less seasonal variability in supplies may reduce the occurrence or duration of outdoor watering restrictions for residential users. These results neglected geographic and hydrologic features that should be characterized through hydrologic modelling of groundwater replenishment through PR to determine the potential impact on water supply disruptions for domestic well users.

3.3.3 Institutional Impact Assessment

The economic feasibility is a critical aspect in determining whether a project will go forward but equal comparison requires consideration of how either water management strategy may have indirect impacts on other costs. This assessment accounted for costs that include capital and O&M costs related to reclaimed water management, drinking water management, and advanced treated water management as well as the non-economic impact to local control over resources.

3.3.3.1 Capital costs

Numerous advanced treatment trains can be used to generate safe and reliable water through PR, with varying costs. Capital costs in the SQ scenario were approximately 50% larger than the benchmark cost required for standard WRRF upgrades needed within the planning period (Table 3-5). The cost of conveyance to the discharge site was the largest factor to inflate SQ cost above the benchmark. In the PR scenario the costs of conveyance to the injection well, conveyance from the extraction well and O_3-BAC treatment systems were the largest additional costs, resulting in a cost increase of 100% above the benchmark value. The upgrades needed for advanced treatment processes from the WRRF were the driving costs in both scenarios.

Table 3-5. Percent increase in capital and annual costs relative to benchmark costs

Cost item	SQ value (2017)	PR value (2017)
Capital cost		
Development of programs for political, regulatory, public processes, and public outreach	0%	7%
Costs for upgrading WRRF treatment train	11%	32%
Cost of reclaimed water distribution systems ($)	38%	62%
Total capital costs ($)	49%	101%
Operating cost		
Operating costs for potable water service ($/y)	0%	0%
Operating costs for RSWRF service (includes pumping costs to either Long Valley creek or to and from Bedell Flat in addition to wastewater treatment O&M costs)	17%	14%
Cost of ongoing regulatory oversight	0%	15%
Total annual costs ($/y)	17%	28%

The scaled ranks of capital costs were found to be -2 in the PR scenario and -1 in the SQ scenario. These results did not consider the distribution of this cost between water and wastewater management authorities. It also did not include the potential water rights that could be developed through PR. The costs of water rights are significant and variable in the Reno-Sparks metropolitan area because water resources are fully allocated. Several costs relevant to the SQ scenario were also not included in this analysis, including the lost opportunity to reuse the reclaimed water and the potential expense of water storage during the winter.

3.3.3.2 Operating and maintenance costs

Unlike membrane based advanced treatment systems, O_3-BAC based advanced water treatment configuration is operated with significantly fewer energy requirements and maintenance cost. However, regulatory oversight was expected be the largest annual cost increase above the benchmark value in the PR scenario (Table 3-5). Costs associated with pumping to the discharge sites (and from the extraction well for PR) were found to be the largest factor inflating annual costs of reclaimed water management. They comprised a large portion of O&M costs in either scenario and may change based on likely locations for discharge in the SQ scenario and the injection well location in the PR scenario.

In the SQ scenario the costs were calculated to be approximately 20% larger than the benchmark. In the PR scenario utilizing O_3-BAC treatment, most costs were calculated to originate from energy requirements for ozone generation. These costs were approximately 30% larger than the benchmark cost. After scaling, the ranks of both the SQ and PR scenario were determined to be -1. Importantly, these analyses do not examine potential annual cost savings that may occur in the PR scenario by offsetting the water supply shortage calculated in the SQ scenario. Overall, annual costs for operating PR are expected to be marginally higher due to the expense of additional ongoing regulatory oversight and monitoring requirements.

3.3.3.3 Institutional control over water resources

Potable reuse serves as a mechanism by which local water resources are expanded, potentially reducing the need for non-local water sources. In the North Valleys planning area, the need for imported water to meet customer demand is expected to decrease in the PR scenario proportionally to the 2.2 Mm^3/y increase in flow generated by aquifer replenishment. The SQ scenario resulted in local control of potable water by 17% below the benchmark value and reclaimed water control 6% below the benchmark value. Local control of drinking water resources was calculated to be 11% above the benchmark value in the PR scenario, and to satisfy the benchmark for reclaimed water. The resulting ranks were calculated to be -1 for the SQ scenario and +1 for the PR scenario. The enhanced local control over potable and reclaimed water resources in the PR scenario were anticipated to enhance local autonomy by ensuring better control over water resource planning, monitoring, and protection. Local control over reclaimed water was also anticipated to prevent uncertainty in rules and regulations that would otherwise be incurred if reclaimed water was exported out of the region.

3.3.4 Trade-off Assessment

To compare costs and benefits of each water management scenario for all the criteria examined, impacts were assessed based on the benchmarks and scaling factors described previously to generate ranks ranging from -3 to +3. The resulting trade-off comparison between the criteria under each scenario is illustrated in Figure 3-4. No additional weighting factors were used. The health risk criteria were not anticipated to change depending on the scenario selected, although further study of the health risks associated with either management scenario was identified as necessary to accurately assess this criterion. Local data on SQ potable water quality was limited, and both scenarios should be evaluated with bio-analytical methods to account for cumulative risks from chemical contaminants. Capital costs were approximately twice the benchmark value in the PR scenario and 50% larger in the SQ scenario. However, these costs did not include the potential costs to obtain additional water rights in the SQ scenario so that adequate supplies were available to meet projected demand. O&M costs were larger than the benchmark value under both scenarios due to the additional pumping costs that will arise for reclaimed water or PR water conveyance. Regulatory oversight costs were expected to result in larger O&M costs under the PR scenario. The institutional control of local water and reclaimed water resources was strongly weighted towards the PR scenario. Environmental effects also strongly favored the PR scenario due to the resilience that will be achieved through groundwater augmentation, decreased reliance on inland ecosystems for reclaimed water disposal, and increased potable water supplies to meet future demands. The additional flows to groundwater were also found to decrease the risk of water service failure, particularly for domestic well owners, as the PR scenario estimates that groundwater augmentation could ensure that supplies surpass groundwater demand. Overall, the criteria that related to water supplies and sustainability generated the highest benefits with PR, while other criteria exhibited only small differences between the scenarios.

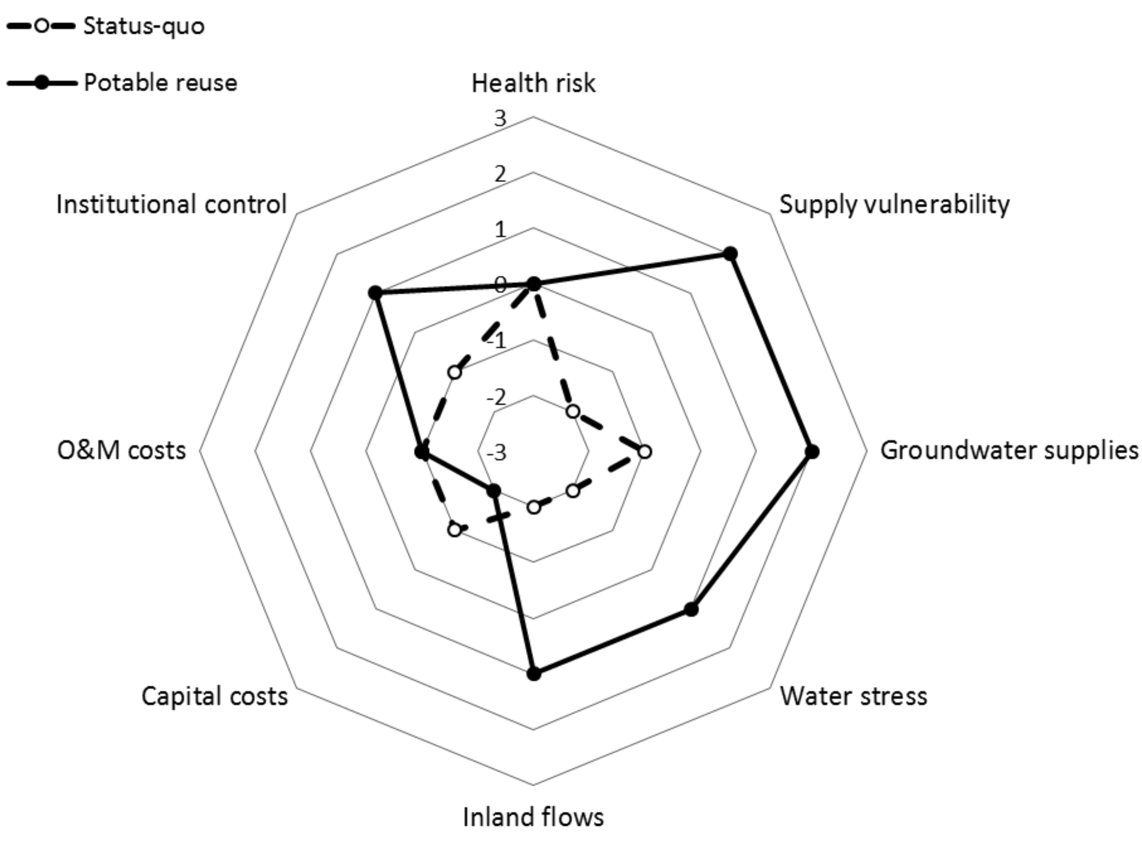

Figure 3-4 Decision making trade-offs between criteria for SQ and PR scenarios

3.4 Conclusions

This TBL analysis of water management in the study area indicates that PR can bring many benefits to the region compared to SQ water management. Environmental criteria were found to generate the largest benefits in the PR scenario. The increase in water resources to the North Valleys area of the Reno-Sparks region would nearly double local water resource availability, reducing reliance on imported water resources. This was also expected to generate other environmental benefits due to the energy requirements of water importation. Social impacts were expected to be small due to uncertainty in how water use restrictions might be impacted by PR and the expectation that advanced treated water in the region would achieve desired removals of CECs and provide equivalent safety to the conventional water resources already used. Pathogens were expected to be removed to acceptable levels through PR and health risk can be reduced through PR if de-facto, or unplanned potable reuse is occurring due to upstream WRRFs (NRC, 2012). Although this research assumed that de-facto reuse was not significant, septic tanks can be a source of de-facto reuse in basins like the study area. Additionally, some de-facto reuse occurs with the surface water resources imported into the study area, which include flows from a small upstream WRRF. Institutional criteria indicate small losses in both scenarios. In the PR scenario, capital costs were expected to be offset through connection fees paid by developers. Similarly, O&M costs may also be partially offset by water use fees charged to customers of the water and wastewater utilities. Institutional control over water resources was found to gain benefits in the PR scenario. Under SQ management control over water supplies would reduce as the region relies increasingly on imported water to meet customer needs.

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Chapter 4 Economic Evaluation of Indirect Potable Reuse for Inland Communities

4.0 Abstract

This research presents an analysis of the microeconomics of indirect potable reuse for urban management of water and wastewater resources. Few prior studies have developed economic models to describe urban water system complexity. Urban areas often comprise a sizeable portion of the potable water resource demand in watersheds. Additionally, the treatment of wastewater produced by urban demand is costly and is often not considered in conjunction with the costs of expanding urban water supplies. The importance of jointly modeling the costs of water and wastewater in an urban water management context is even more important when scenarios of potable reuse are considered. The interconnected water flows, total costs, and capacity limitations imposed on both water and wastewater management systems should be jointly considered due to the impacts that potable reuse can have in addressing the externalities associated with urban wastewater. Our research presents a preliminary economic analysis joining the costs and constraints of water and wastewater management for an inland urban region. The study develops a model that describes the economic feasibility of indirect potable reuse under varying costs and water resource management scenarios. The model demonstrates the importance of the comparative costs associated with water and wastewater management in determining the optimal capacity at which indirect potable reuse should be pursued for long-term cost minimization.

4.1 Introduction

This study undertakes an exploratory analysis of the microeconomics of potable reuse to manage urban water resources. Cities are adopting various strategies of water reuse of reclaimed or purified wastewater due to catalysts including water stress, drought vulnerability, and the ecological impacts of wastewater management. Particularly in inland regions, wastewater management may be the driving factor in water management decision-making due to the costs associated with nutrient removal, the value of wastewater components (e.g. nutrients and water), and regulatory requirements on the discharge of wastewater effluent. While water scarcity has long been the most common driver of innovative water management strategies like potable reuse, wastewater management is increasingly factored into urban water policy as urbanization and population growth increase the negative effects of wastewater effluent discharge as well as stormwater runoff due to land use changes.

Over the last three decades applications of indirect potable reuse (IPR) have rapidly increased in order to address water scarcity and uncertainty of water supplies. Although research is still ongoing into the capabilities of specific treatment train configurations to achieve desired water quality through removal of pathogens, contaminants of emerging concern, and disinfection byproduct formation potential, the individual treatment processes used to achieve water of potable quality are well established (known as advanced water treatment technologies). Thus, quality IPR water has been shown to be equivalent to or better than conventional water sources, and factors influencing health risk of water consumption sourced from reclaimed water may even be superior in some cases. IPR can address numerous parameters of urban water resource vulnerabilities by providing a drought-resistant water supply, increasing local potable water resources, and providing a high-value use for wastewater resources. Wastewater effluent management may also be a key driver in adoption of IPR in some cities, particularly for inland regions that discharge into sensitive ecosystems or to water bodies that impact the risk of flooding to surrounding areas. Population growth may not only intensify these impacts.

The costs and benefits of IPR are region specific to some extent, due to regulatory requirements, water scarcity, geospatial characteristics, and the water demands of local economic sectors. Few papers have explored the economic trade-offs of water reuse projects (e.g., Arborea et al., 2017; Molinos-Senante et al., 2011), but no studies were found that examined the scenarios of potable or non-potable reuse to meet urban water demands and effluent management needs. Recycled water is water of lower quality to groundwater and certain surface water and is produced from conventional methods of wastewater treatment. In coastal regions, recycled water may carry lower costs because minimal treatment is be required to meet regulatory requirements for discharge into the ocean, and the environmental benefits of water reuse can outweigh the costs of the additional infrastructure (Molinos-Senante et al., 2011). These regions also widely reuse recycled water for irrigation due to the scarcity of potable water sources, the demands of their thriving agricultural sectors, and the risk of saltwater intrusion if groundwater resources are over-extracted. Inland regions can have higher costs associated with recycled water due to environmental impacts. For example, groundwater mining to offset shortages in surface water can result in significant economic losses (Holland and Moore, 2003; Medellin-Azuara et al., 2015; Roseta-Palma, 2002; Roumasset and Wada, 2010). In water scarce and inland cities, IPR requires significant investment in infrastructure compared to non-potable water recycling, but it may provide greater benefits to the region by enhancing management of wastewater effluent for environmental benefits and supporting growth in potable water demand. Often, highly purified wastewater equal to or better than natural source water is recycled through an environmental buffer such as an aquifer or surface reservoir/lake prior to reuse as source water for potable use. This practice is termed as indirect potable reuse (IPR) (Haak et al., 2018). IPR may impact the shadow price of water resources and may also cause an outward shift in the demand curve if it is used to support population growth. Economic analysis should also consider the externalities that result from the IPR project, such as the impacts of reallocating wastewater effluent flows which may contribute to flood risk or provide essential water to support ecosystems.

This research presents an economic evaluation of IPR of advanced treated or purified wastewater in a hydrologically closed basin. The research investigates three scenarios for water and effluent management, including: (i) wastewater reuse limited to non-potable uses and population growth is limited by the availability of existing potable water supplies (Do Nothing Scenario), (ii) wastewater can be purified with advanced water treatment methods to generate a potable water supply that is stored in an aquifer, but new water rights are not granted thereby limiting population growth to existing potable water supplies, and (iii) a new potable water supply is generated and water rights are granted, thereby allowing further population growth in the region. A key driver for IPR is residential growth in water scarce and inland regions, with constraints to how many new units can be built based on available water supplies but also based on effluent disposal/reuse capacity. This research examines under what growth scenarios does each possible solution might provide the greatest benefits.

This research seeks to provide local policy makers with an economic analysis of IPR in an urban or suburban development. The results are expected to provide a broader relevance to other arid and semi-arid cities, especially in regions with appropriative water rights. The economics of urban water management and reuse have historically focused on the impacts of water pricing and restrictions on urban water demand (Mansur and Olmstead, 2012; Moeltner and Stoddard, 2004; Olmstead and Stavins, 2009). However, demand management policies are unlikely to address needs in urban areas where wastewater effluent presents a constraint on water management because wastewater largely originates from indoor water demands that are generally inelastic and less responsive to water pricing policies.

4.2 Case Study Description

The water demands of growing populations in closed basins can make management of both water resources and wastewater coupled drivers towards water reuse. Management of wastewater effluent in closed basins can present several challenges due to limited outlets for discharge and the sensitivity of closed basins to the quantity and quality of inflows. Demands for non-potable uses are limited by the costs of additional service connections and the demand from commercial and recreational users. Expanding potable water resources through investment in advanced water treatment processes may provide an optimal solution for management of both effluent and water supplies.

These challenges are all catalysts for investigating IPR as a water supply and effluent management strategy in the North Valleys region of Reno. This study area is a closed sub-basin in which stormwater runoff and wastewater effluent discharge collect in several playa wetlands. This region is among the most rapidly growing in the city, resulting in increased runoff due to land use change and higher wastewater effluent flows. The projected water balance for 2035 indicates that water supply deficits will occur in the planning area unless new water resources are imported. Growth in water demand and depleted groundwater tables are expected increase the area's reliance on the 9.87 Mm^3 of imported groundwater rights available to the study area under status-quo conditions.

Short-term growth will be limited by the capacity of the region to reuse or discharge reclaimed water. If no action is taken, future growth will be constrained based on the local capacity to discharge effluent (“Do Nothing Scenario”). Two options are considered most viable: exporting effluent out of the region (“Effluent Management Scenario”) or purifying it to meet drinking water quality so that it can be reused to meet residential water demands through IPR (“One Water Management Scenario”). These scenarios are illustrated in Figure 4-1. If the IPR scenario is pursued, it can be treated as generating a new water resource; producing water rights that can be sold for planned residential expansion.

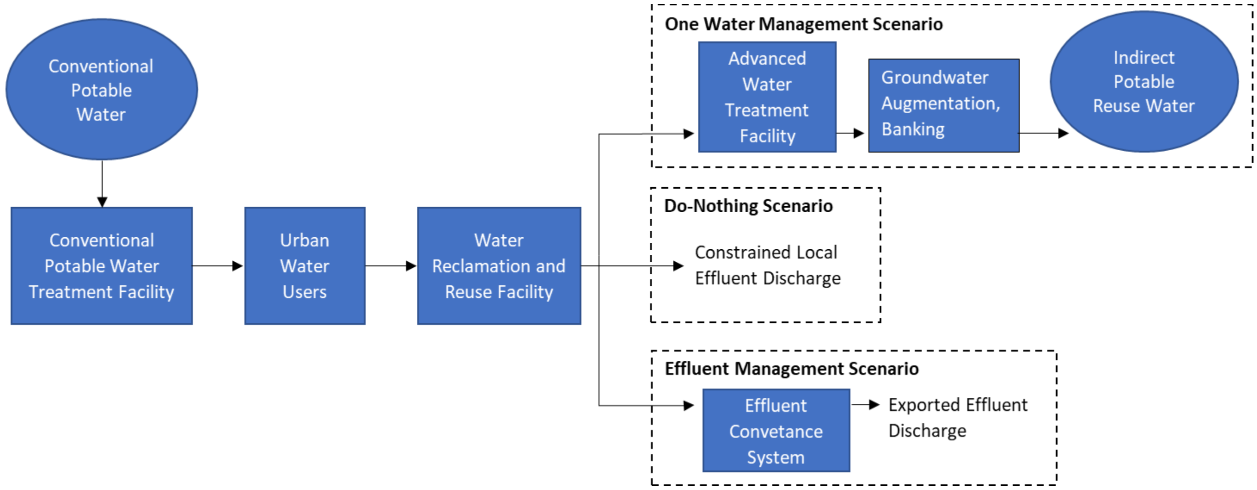

Figure 4-1 Water and wastewater management scenarios

4.2.1 Resource Management Scenarios

The scenarios considered would each limit the region’s population growth constraints in different ways. If no project is pursued, regional growth is limited by effluent discharge capacity to prevent flood risk from increases in the discharge of effluent to the environment. In this “no project” scenario, the total water resources available to the region would never be required due to this effluent management constraint. Alternatively, the region has two feasible effluent management scenarios (see Figure 4-1): IPR and export of surplus effluent (flows that exceed regional discharge capacity). It is important to note that previous studies have evaluated other options to address the effluent management constraint and have found that solutions such as expanding non-potable reuse infrastructure to increase regional use of effluent for irrigation are not feasible (Eco:Logic, 2010). Although previous regional studies identified that export of effluent has a lower average cost than IPR, it may not provide as many benefits to this water scarce region (Haak et al., 2018). Finally, through IPR it is also possible to recover water rights, which are a property right that can be used to allow for an additional increase in population growth, beyond the limit imposed by the conventional water resources available to the region. Thus, each scenario results in a different limitation to future population growth, based on the capacity to discharge wastewater effluent to the environment and the availability of potable water resources to support development in the region.

The value of potable water resources reflects its scarcity and the benefits resulting from its use. In a region with an appropriative water rights system, the generation of a new water resource would result in a property right to a specified volume of water resources that are assumed to be equivalent to an annual sustainable yield. Effluent management without IPR generates a low-quality resource that provides limited environmental benefits (benefits are generated up to the local environmental discharge capacity) and a low value market good through non-potable wastewater recycling for landscape irrigation. IPR may serve as an alternative to expanding non-potable wastewater recycling by converting that resource into a potable water resource. The costs of treatment for IPR are higher than wastewater recycling, however these water resources could be delivered to customers via the existing water conveyance systems. Conversely, wastewater recycling requires an additional water conveyance system to housing communities or commercial areas. Benefit-cost analysis provides a strategy to compare the net benefits generated by alternative water management scenarios.

IPR would impact both the regional water management cost curves and the demand curves. The changes in these functions will reflect a change in the overall welfare that water resources generate for the local community. To understand how the scenarios are expected to impact the net benefits generated by each of the management scenarios, we developed generalized depictions of the average cost and water demand (marginal benefit) curves. The average costs associated with water and wastewater management are assumed to follow a linear trend (Tracy et al., 2014). Figure 4-2 illustrates how these changes in welfare can be evaluated based on the changes to regional water supplies and demand. The marginal benefit (MB) curves illustrate the outward shift in the water demand function that results from an increase in population and housing. The area under the MB_0 and AC curves is the total benefit generated by water and wastewater resources allocated to meet demands in the study area by the utility. This assumes that regional population is initially limited to the capacity of the water management system. The area under the MB_1 curve is then the benefits that would be generated if the population increases due to an increased water supply generated by IPR. The area under the average cost curves (AC) reflect the costs of water and wastewater management when the system reaches its capacity with current water management systems (AC up to w_o) and in the future when the IPR system reaches its build-out capacity (AC up to w_{max} or $w_{max,IPR}$).

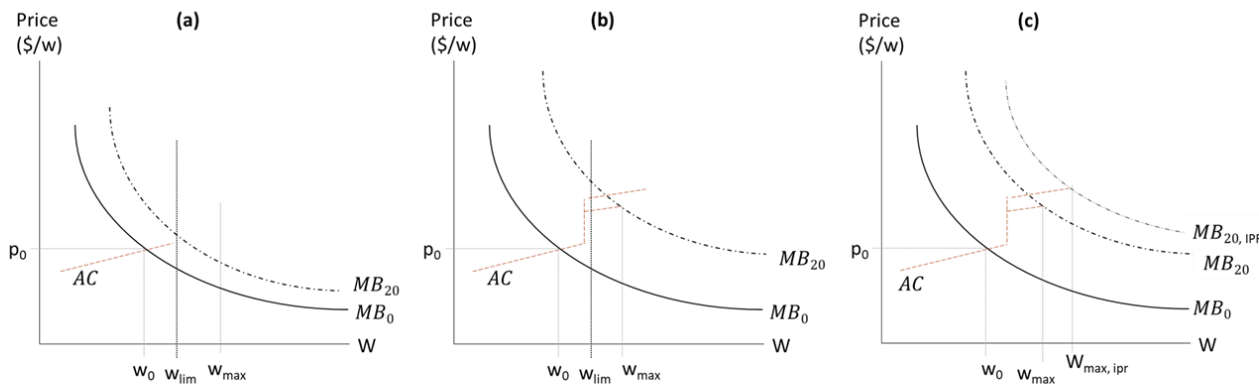

Figure 4-2 Illustration of theoretical cost and benefit curves under the three scenarios: Do Nothing in which future growth is constrained by effluent management capacity (available water resources are also constrained) (a), Effluent Management which involves exporting surplus effluent to address the effluent discharge capacity (b), and One Water Management which utilizes IPR to increase the capacities of both effluent management and potable water resource(c). Over the two periods, the demand curves are expected to shift outward to population growth. New water management strategies must accommodate this growth, resulting in higher average costs (AC).

Scenario 1: Do Nothing

Figure 4-2a illustrates the marginal costs and benefits expected in a do nothing scenario. Although there are w_{max} units of water available in the region for residential supply, population growth is constrained based on the regional effluent disposal capacity, which then limits water demand in the region to w_{lim}. Thus, future growth is constrained based on available effluent “disposal” capacity. The future benefits curve will shift out from population growth to a point limited by the effluent management constraint.

Scenario 2: Effluent Management

Scenario 2 is illustrated in Figure 4-2b. This describes the possibility that the region would pursue effluent management infrastructure, comparing costs between advanced water treatment (AWT) to recycle reclaimed water and exporting effluent. This allows population growth to rise further over 20 years, shifting the demand curve outward more dramatically. In this scenario, population growth is constrained by w_{max}, which is the currently available supply of water resources. The additional water resources created by IPR are not used to increase this limit.

Under this scenario, the region would select an effluent management scenario based on the average costs for each effluent solution (exporting has a lower average cost than IPR). The marginal benefits function will be the same with either effluent management solution, shifting outward based on the water supply constraint, w_{max} (water rights limit on future housing).

Neglecting any possible external costs of exporting, the region would pursue the solution with the lowest average cost effluent management option (exporting). However, external costs or benefits may also be substantial. For example, if IPR provides benefit by offsetting environmental risk, which could be treated as a cost averted to shift its AC function down. Although IPR will not increase the available water resources (water rights) in this scenario, it will increase the percent of those water resources that are potable, increasing local water security. Additionally, if it reduces the discharge of water to the inland ecosystem on the playa wetland, it may have a beneficial impact by reducing flood risk.

Scenario 3: One Water Management

The third scenario is illustrated in Figure 4-2c. Here, the region is assumed to pursue AWT to recycle reclaimed water and recover water rights, avert external costs (e.g. flooding or pollution), versus exporting (assumed not to cause any external costs). This system is constrained by the available water resources. Although exporting effluent may provide a lower cost solution, in the long run IPR provides greater benefits by increasing the water resource constraint to $w_{max,IPR}$.

In this scenario, there are now significant benefits produced by IPR compared to exporting because it allows greater future population growth, shifting the marginal benefit curve (water resource demand) out further than when it is constrained by w_{max}. Depending on how significantly population is expected to increase (an outward shift in the benefit curve), IPR is expected to provide greater net benefits to the region long-term.

4.3 Economic Efficiency

The economic analysis of water use examines the benefits generated through water demand compared to the costs of inputs and any external costs or benefits that are generated under alternative water management scenarios. For water management planning related to residential areas, water resources are inputs that support an output population. This relationship is referred to as the production function. At a given increase in inputs (water resources) the outputs will increase; however, the relationship is not necessarily proportional across the range of possible inputs. The production function can be used to identify the point at which benefits are maximized by optimizing input levels relative to profit. This can be achieved by minimizing the costs.

4.3.1 Cost Minimization

Public utilities are expected to manage water supplies through cost minimization strategies. For example, water supplies that are available most locally, such as groundwater, may be of lower cost to deliver than supplies that have to be conveyed further or imported into the project area. The differing quality of available water supplies will also influence the cost of water delivery by influencing the necessary water treatment processes and the respective cost of chemicals, labor, and other materials to ensure that the water meets or exceeds drinking water regulations. Public utilities will choose the water supplies that minimize net costs, thereby maximizing the overall benefits of the water resources over time.

4.3.2 Public Water Management and Reuse

In developed urban areas, water conveyance systems are established so that individual water demands are not limited by access and all demand points are metered for assignment of water use fees. However, inefficiencies in the conveyance system do arise from leakage in aging systems, flat-rate water users, and un-metered water demands such as fire hydrants. These characteristics result in revenue losses for public water agencies and reduce the overall productivity of the regional water supply. Overall, inefficiencies reduce the feasibility of cost recovery and the economic efficiency of public water systems.

External impacts from inefficient public water systems may include higher pumping costs for water conveyance, lower recovery of water resources at water reuse and reclamation facilities, greater strain on regional water supplies such as aquifers, and the loss of water resources to downstream communities outside the urban area. Localized environmental impacts may also be observed as a result of inefficient water systems and may include dewatering of aquifers, changes to flood risk, degradation of water quality, and decline of water flows to ecosystems and natural habitats. Although the magnitude of these impacts may be difficult to measure, they are important considerations when assessing changes to water infrastructure and management strategies.

Both potable and non-potable cases of water reuse are unique cases of water management. Reuse of reclaimed water for non-potable purposes occurs widely, particularly in de-facto or unplanned scenarios. A common example of de-facto reuse occurs when upstream communities discharge reclaimed water into streams which contribute to the water supply of downstream communities. The hydrogeologic analysis of such reuse can be difficult to characterize because of the influences of these surface water supplies on aquifers and the seasonal variations in relative flow of potable water and reclaimed water. Planned reuse of reclaimed water for non-potable uses may be more straightforward to characterize because often the area generating the reclaimed water is also reusing it. This generally requires an alternative conveyance system that delivers the non-potable water supply to demand loci such as parks and other green spaces. The demand for non-potable water supplies can thus be used to reduce the demand for potable water in specific applications.

IPR of reclaimed water effectively generates a new water supply that is equivalent to potable water supplies. Once treated to a standard that meets or exceeds drinking water regulations, this resource can be returned to existing water conveyance systems for the public supply. Conventional water supplies, such as surface water, generally require chemical and mechanical processes to generate a potable water supply. The production of potable water through advanced treatment uses similar processes but with additional barriers and redundancies due to the lower quality of the reclaimed water compared to conventional water resources.

4.4 A Model of Water and Wastewater Management

This research considers a two-period water and wastewater management model for the finite planning horizon of a residential area. The planning horizon, hereafter referred to as build-out, represents the year that proposed housing projects will be completed, bringing population in the area to the approximate capacity. The static model considers the long-run optimal water-wastewater management strategy. It is important to jointly consider water and wastewater management because IPR directly uses wastewater to create a drinking water resource. Figure 4-

3 provides a generalized depiction of the interconnections between water and wastewater resources from a municipal water management perspective. The Do Nothing Scenario described in the previous section corresponds to the areas of Figure 4-3 shown in blue, while the green arrow illustrates how wastewater management constraints can be relieved through the Effluent Management Scenario. Finally, the purple arrows illustrate how both wastewater management constraints and potable water resource constraints can be relieved through indirect potable reuse in the One Water Management Scenario. Although conjunctive use programs such as aquifer storage and recovery influence the value of the regional water supply (Booker et al., 2012; Krishnamurthy, 2017; Roseta-Palma, 2002; Roumasset & Wada, 2010), this component of a system is neglected in the water management model developed below. To simplify the complexity of the urban water system, all existing potable water resources (which may include surface water, groundwater, and imported resources) are combined together in the following model. However, the economics of imported water resources is another complex topic that should also consider the impacts to the region the water is transferred from (Booker et al., 2012; Tello & Ostos, 2012; Vaux & Howitt, 1984; Zarghami & Akbariyeh, 2012). This study assumes that flows of reclaimed water to advanced water treatment for IPR will be driven by restrictions and physical limitations to surface water return flows and wetland discharge. From Figure 4-3 it is apparent that water demands will result in a proportional wastewater flow, and the physical characteristics between regional water demand, wastewater generation (from indoor water use and economic sector water demands), and limitations to discharge of reclaimed water will result in a linear relationship between water demand and the diversion of reclaimed water for IPR. Several previous studies have considered optimality in urban water resource management (Angulo et al., 2014; Malla & Gopalakrishnan, 1999; Mansur & Olmstead, 2012; Rosenberg et

al., 2008) but have neglected the impacts of increased water demand on wastewater production. Wastewater management is generally more expensive than water management, so it is also likely to be significant in the cost minimization problem that regional utilities face.

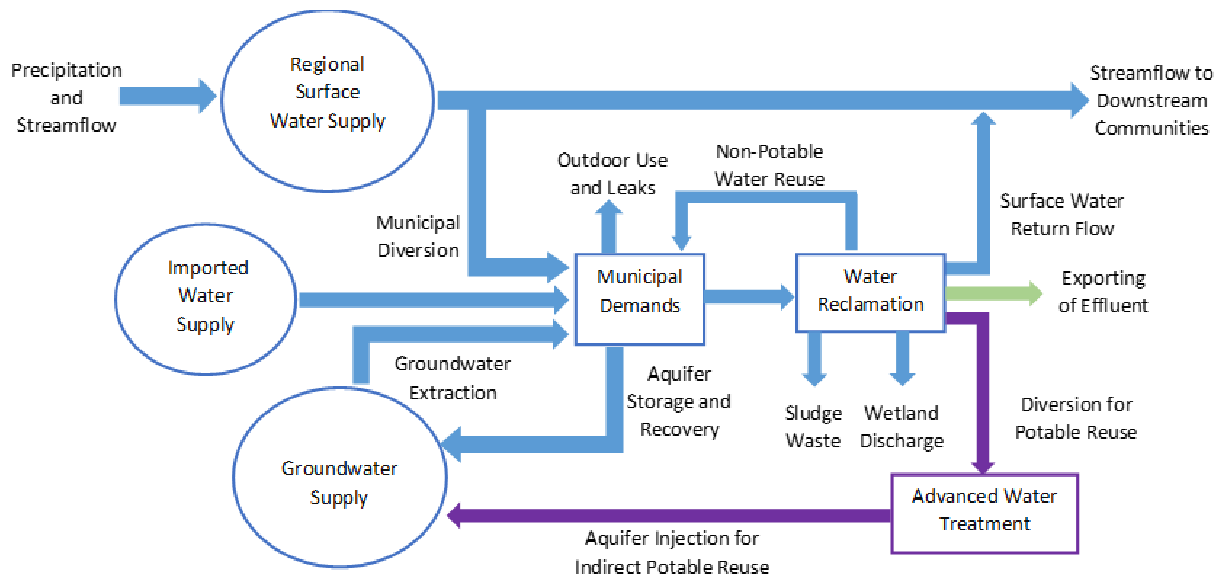

Figure 4-3 Water supplies, diversions, reuse, and return flows that generally comprise the hydrological balance for municipal water and wastewater management. Purple arrows correspond to additional flows that occur in the One Water Management scenario with IPR and the green arrow corresponds to the Effluent Management scenario.

Although separate utilities may manage the water and wastewater resources, this study assumes that their costs are considered jointly. The total costs are the sum of the costs associated with producing the water resources (w_1 and w_2), treating the resulting wastewater (r), and the costs associated with managing reclaimed water flows that are not discharged to the local environment. In this study, water not discharged to the local environment must be either exported out of the basin (e) or processed through advanced water treatment to produce water suitable for IPR (i). Although this closed-system for water and wastewater management is most relevant to closed basins, it provides a more holistic approach to managing urban water resources and may also be useful for inland communities and study areas that desire to manage all water flows within a sewershed. Water-wastewater utilities are assumed to be public entities that charge water fees to customers only for cost recovery; thus, the optimization problem only considers costs associated with resource management.

Three constraints are also critical to characterizing the water supply system. First, the water supply generated must be equal to the anticipated demand at project buildout. This assumes that population growth in the study area will be constrained due to limitations in buildable area. The relationship between future population and water demand can be characterized based on local historical data, but this approach is likely to over-estimate the water demand in newer housing that has more water conserving appliances and irrigation. Additionally, the lot size for new housing is likely to have significant influence over residential per capita water demand due to irrigation requirements. This effect would be larger in more arid regions that allocate a greater percent of residential water use to irrigation.

The water resources, w_1 and w_2, are both subject to physical constraints on water production. In this case, w_1 represents the conventional sources of potable water, or natural water supplies available to the region, which is related to precipitation, aquifer recharge rates, and snowmelt. Year to year rates for these three parameters can widely fluctuate, but the use of upstream reservoirs and aquifer replenishment are often used to ensure that there is adequate water storage to compensate for seasonal variability. In practice, the potable water supply capacity should correspond to the water rights historically available to the region. This constraint is accounted for by introducing variable s. Finally, the water resource generated from aquifer replenishment is subject to a capacity constraint, which is the capacity of the advanced water treatment facility. This assumes that the capacity of the aquifer being replenished is significantly greater than the advanced water treatment facility and does not impose any additional constraint on generating this water resource. Thus, the utility's problem is described by Equation 4-1 below, subject to the three constraints that include the requirement that long-term demand for potable water resources cannot exceed the total available potable water resources, a freshwater supply constraint (s), a constraint on the capacity of the IPR system (c).

$$\min_{w_1, w_2} [p_1 w_1 + p_2 w_2 + p_3 r + p_4 e + p_5 i] \ \textit{subject to} \tag{4-1}$$

$$q_T - f(w_1, w_2) = 0,$$

$$w_1 - s^2 = 0$$

$$w_2 - c^2 = 0$$

4.4.1 A Production Function of Conventional and Potable Reuse Water Resources

In the scenarios explored, the two water resources (potable water from conventional resources like groundwater and potable water from IPR) can be viewed as the inputs to satisfy residential water demand, or in this case water rights for residential uses. Any combination of the two water resources can be used to provide the desired output of water rights; these combinations can be pictured as an isoquant map. For a water-wastewater utility the isoquant map for substitution of conventional water (w_1) and water from IPR (w_2) likely has a convex shape in regions such as inland communities that face limitations on discharge of reclaimed water (Figure 4-4a). At high levels of consumption of w_1, the demand for a unit of w_2 is likely very high because that unit shift not only satisfies potable water demand but also addresses the need for managing large volumes of reclaimed water. However, as w_1 is marginally decreased, the marginal demand for w_2 would increase at a slower rate. The exact shape of the isoquant curves will depend on local characteristics and the alternative water resource management options available to a region. According to this relationship, the management of potable water resources by the water utility is likely to follow a Cobb-Douglas production function (Equation 4-2). This type of production function is likely to represent production of conventional and IPR water resources in regions that adopt IPR for purposes such as increasing resilience to drought but may not be appropriate for modeling production in areas facing serious constraints on water or wastewater management, as described in the following paragraph.

$$q = f(w_1, w_2) = A w_1^a w_2^b \tag{4-2}$$

A special case of the production function may apply to different regions. For example, if IPR is exclusively being considered as a way to manage effluent a fixed-proportion production function may be appropriate (Figure 4-4b). In this case IPR must be used to accommodate indoor water demands that result in effluent beyond the regional capacity, so the water production would operate at a fixed ratio of w_1/w_2 based on the fraction of w_1 that is returned to the water system as effluent and the regional effluent capacity. For example, if a region wishes to have zero effluent discharge and 60% of water demand is for indoor uses that are then returned to the water system as wastewater, an IPR system would be required to have a capacity of 60% of regional water demand. Some water loss would occur in the IPR treatment and storage processes and demand for conventional potable water may decrease once the IPR resource is incorporated into the water portfolio. In this case the effluent production, which would then be converted into w_2 (IPR resource) would be the binding constraint in the process of producing water resources for such a region (i.e. $bw_2 < aw_1$). Production in a fixed-proportion case would occur at the vertex where $aw_1 = bw_2$, which would be the cost-minimizing place to operate. To produce water resources at any given level of production (e.g. q_1), increasing w_2 beyond the vertex would not be possible because it is limited by the amount of effluent generated by water demand, and additional w_1 would not be beneficial because it would result in effluent beyond the regional capacity. Any increases in water demand (e.g. from q_1 to q_2 in Figure 4-3b) for this region would then result in an increase of w_1 and w_2 at a constant ratio. This study will assume a fixed proportion production function, which most closely reflects the water and wastewater management in the case study area.

$$q = f(w_1, w_2) = \min(aw_1, bw_2) \tag{4-3}$$

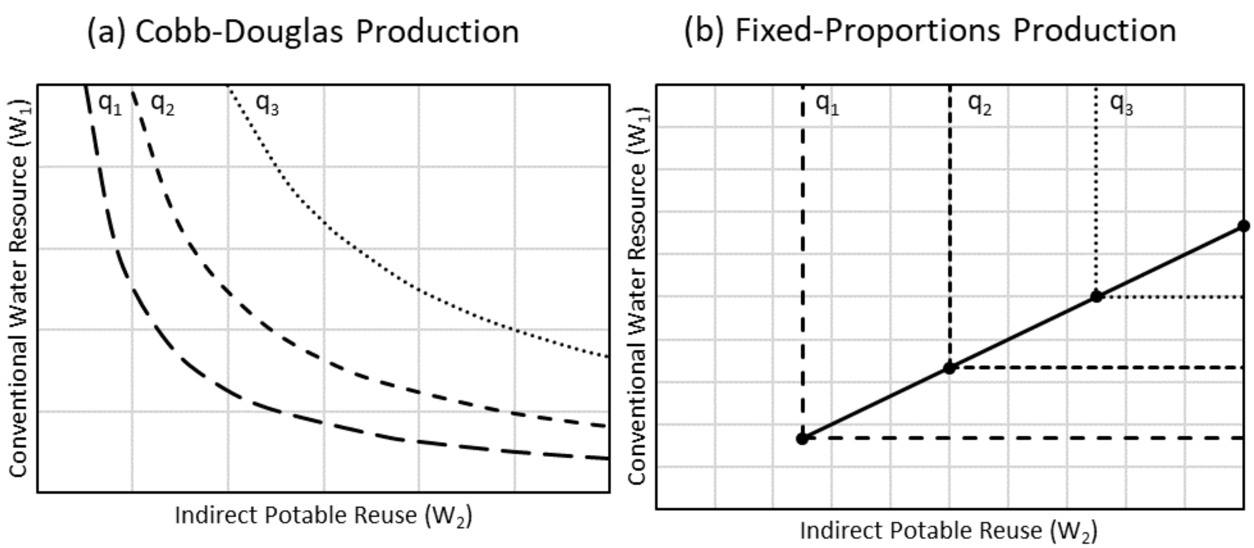

Figure 4-4 Production functions that can represent different cases of substitution for conventional and IPR water resources. In (a) substitutability of conventional water (w_1) and IPR (w_2) to satisfy a given level of demand (q_1, q_2, or q_3) have a convex shape. In (b) a production at each level of demand will occur at a constant ratio and it is not possible to substitute a unit of one water resource for another due to the properties of production.

4.4.2 Cost of Indirect Potable Reuse

The IPR treatment system includes coagulation-flocculation-clarification and granular media filtration (CFCGMF), ozonation (O_3), biological activated carbon filtration (BAC) with an empty bed contact time of 20 minutes, granular activated carbon (GAC) filtration, and ultraviolet disinfection (UV). Within the planning horizon, reclaimed water flows in the case study area are not expected to exceed 13,000 m^3/day (3,838 acre-feet per year). As mentioned previously, the majority of this water (approximately 80%) is used to supplement environmental flows. Thus, it is anticipated that there are fewer than 2,700 m^3/d (800 acre-feet per year) available for IPR; the majority of which is currently beneficially reused for irrigation. Overall, an AWT facility in the study area would be designed with a capacity smaller than 3,000 m^3/d. Cost surveys have included AWT processes of a similar scale for BAC, UV, GAC, and CFCGMF processes (Guo et al., 2014; Plumlee et al., 2014), but for ozone treatment, a comparable scale-cost comparison was only found in drinking water applications (Sharma, 2015).

The capital costs for ozone treatment are driven by organic loading, which depends on influent water quality and flow rate. In addition to system capacity, other variables related to treatment process capital costs included construction materials, equipment, borrowing costs, project planning and project design services. Total costs for each treatment were regressed across the variables and model selection tools (r-squared, AIC, BIC) were used to determine the best model to describe operating and maintenance cost variability in the treatment systems reported in literature. This analysis determined that capital costs could accurately be modeled based on system capacity with a simple linear relationship. Figure 4-5 depicts the linear models for each treatment process with 95% confidence and prediction intervals. The F-test for overall significance of each total cost model demonstrated that the data from literature provided significant evidence for the ability of process capacity to predict treatment costs within the range of capacities examined.

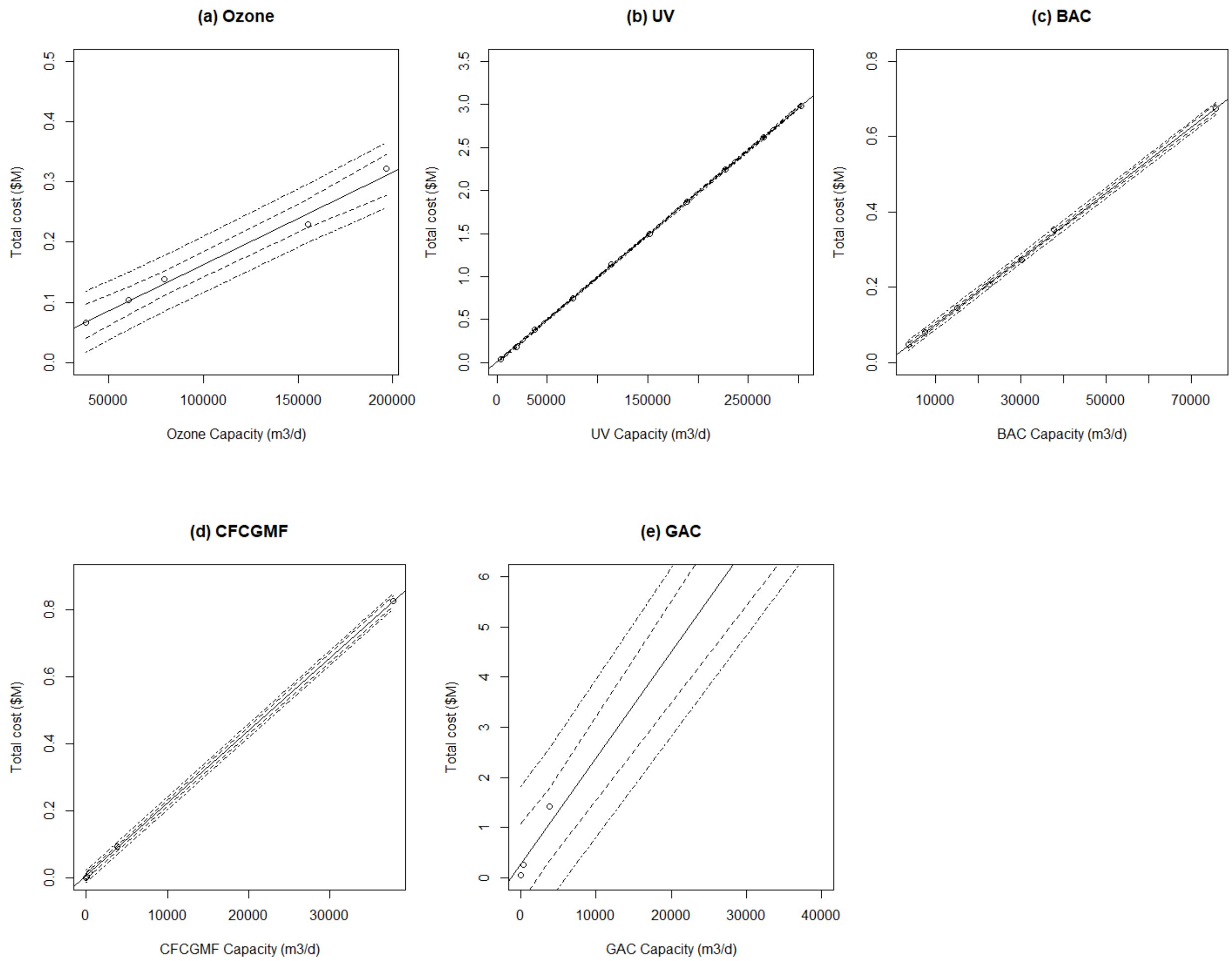

Figure 4-5 Linear models of total cost versus capacity for the five treatment processes considered for an inland region's IPR system (data from Guo et al., 2014; Plumlee et al., 2014; J. R. Sharma, 2010).

Although the minimization of long-run costs is the focus of this study, average and marginal costs of the water supplies are more relevant when considering the relationship between water supply and demands. Overall, the average costs of treatment exhibit a non-linear relationship with system capacity (Figure 4-6). For all treatment processes the best models of average costs exhibited a non-linear relationship with capacity. Specifically, at large system capacities a linear fit can accurately predict the average cost of treatment, but when considering a range of small treatment capacities (e.g. <5,000 m^3/d) a non-linear average cost model is appropriate for most treatment processes.

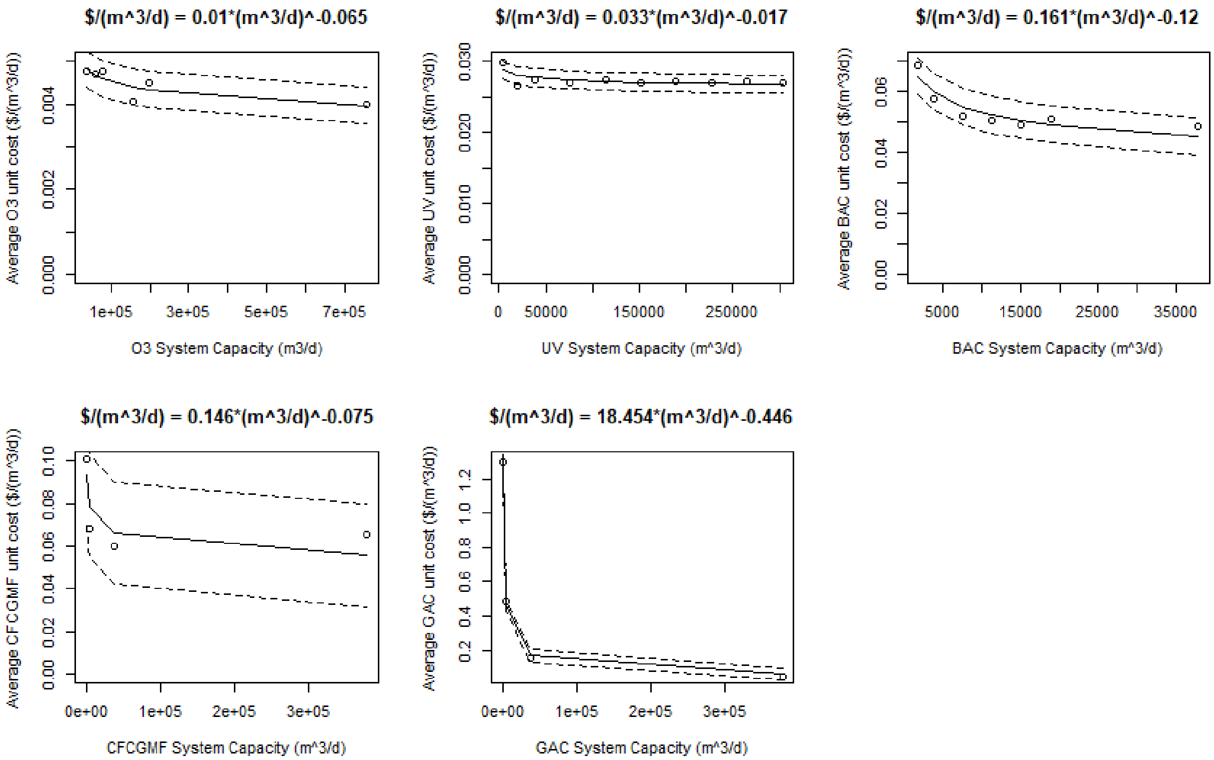

Figure 4-6. Average cost models for AWT processes likely to be used in inland region.

As described previously, due to the limited water available for IPR, the cost equations for AWT processes were simplified to linear models according to a regression of capital costs versus system capacity from advanced water treatment facility (AWTF) and drinking water system survey data. Additionally, the system would incur pumping costs from conveyance to an aquifer, injection of treated water. Thus, the average cost of treating *i* units of reclaimed water to produce potable quality water (w_2) is described by the following linear model, from the CFCGMF, Ozone, BAC, GAC and UV cost component models described below. Parameters A, C, E, G, and I define the slopes that characterize the increase in cost per unit of capacity for each treatment process and parameters B, D, F, H, and J describe where the models intercept the cost axis.

$$AWT\ Cost_i = CFCGMF_i + O3_i + BAC_i + GAC_i + UV_i + Pump_i = p_5 * i \quad (4\text{-}4)$$

Where,

$$CFCGMF_{w_2} = A * i + B$$

$$O3_{w_2} = C * i + D$$

$$BAC_{w_2} = E * i + F$$

$$GAC_{w_2} = G * i + H$$

$$UV_{w_2} = I * i + J$$

4.4.3 Physical Characteristics Water and Wastewater Flows

This analysis simplifies its representation of all the existing conventional water resources available to the region as w_1, which can include existing potable water resources such as local groundwater and surface water, and imported groundwater and surface water. The analysis neglects the variability in natural water resources that may present additional constraints on water supplies due to environmental phenomena like droughts, and instead assumes that water management strategies are able to provide the water resources up to the regional entitlement such as water rights. The water resource from aquifer replenishment in w_2 represents the recoverable stock of water. The analysis simplifies the dynamics of groundwater extraction and neglects to account for how the costs of recovering this water may vary throughout the year due to variations in aquifer height and how the size of w_2 may be constrained by aquifer capacity.

Indoor water use comprises the majority of water demand, but its relative allocation can vary greatly depending on climate, income, and other local considerations. Nearly all of the water used indoors is returned to the public utilities as sewage. On average, cities across the United States return 60% of the water demand as sewage (r):

$$r = 0.6\ (w_1 + w_2) \quad (4\text{-}5)$$

Many cities rely on environmental discharge to manage the reclaimed water, or effluent, that is created through conventional wastewater treatment of sewage. Although this may often be the lowest cost strategy to manage reclaimed water, there may be a limitation imposed by permitting or desired to address environmental concerns. This analysis assumes that effluent management is a driver for exploring IPR as an alternative resource management strategy as the region has reached the capacity for environmental discharge (d). A limited capacity for environmental discharge of reclaimed water may be more commonly encountered in inland regions, where flood risk and other environmental conditions may be adversely impacted by excessive discharge of reclaimed water. The two alternative options to manage effluent that are explored include exporting the effluent out of the region (e), or treating it through advanced processes for IPR (i):

$$r = d + e + i \tag{4-6}$$

Aquifer replenishment through IPR may is a complicated hydrogeologic process that is impacted by numerous factors such as the flow rate of water artificially pumped into the aquifer, aquifer shape, and aquifer composition. These factors and others may influence loss from the aquifer as well as the percent of water that is recoverable. A common assumption is that 80% of the water injected into the aquifer can be recovered (Equation 4-7).

$$w_2 = 0.8 * i \tag{4-7}$$

4.5 Minimizing the Cost of Water-Wastewater Management

The total costs of managing water and wastewater resources can be evaluated from the production function and cost functions. The physical characteristics of water and wastewater flows (Equations 4-5 to 4-7) can be used to express total costs as a function of the two water resources. The fixed proportion production function reaches the optimum at the vertex, where production of both w_1 and w_2 are minimized (Equation 4-3), which can be expressed as potable water production q = $aw_1 = bw_2$, which can be substituted into the total cost function (Equation 3-8) to express the relationship between costs and water production. The total cost function demonstrates that the costs are simply a function of the component costs of water and wastewater management (p_1 through p_5), the water demand (q). This can be expressed as $Total\ Cost = qC(p_1, p_2, p_3, p_4, p_5)$, which states that the total costs are the volume of potable water produced to meet demand (q) times the costs of producing 1 unit of potable water, including the costs of managing the resulting wastewater.

$$Total\ Cost = p_1 w_1 + p_2 w_2 + p_3 r + p_4 e + p_5 i \qquad (4\text{-}8)$$

$$= q\left[\frac{p_1}{a} + \frac{p_2}{b} + 0.6 p_3\left(\frac{1}{a} + \frac{1}{b}\right) + p_4\left(\frac{0.6}{a} + \frac{1}{0.8b}\right) + \frac{p_5}{0.8b}\right] - p_4 D$$

Given that production must occur at q, we can identify the optimal solution to the cost minimizing water management strategy through the Lagrangian expression (Equation 4-9). The production function for water supplies appears here as $f(w_1, w_2)$, which is assumed to be characterized by fixed proportions of w_1 and w_2 as described by Equation 4-3.

$$\mathcal{L} = [p_1 w_1 + p_2 w_2 + p_3 r + p_4 e + p_5 i] + \lambda_1[q - f(w_1, w_2)] + \lambda_2(w_1 - s^2) + \lambda_3(w_2 - D^2) \qquad (4\text{-}9)$$

Substitution was used so that variables r, e, were expressed in terms of w_1, while variable $_i$ was expressed in terms of w_2. The critical point at which cost was minimized was then determined by differentiating to identify the first-order conditions.

$$\frac{\partial \mathcal{L}}{\partial w_1} = p_1 + 0.6 p_3 + 0.6 p_4 - \lambda_1 f_1 + \lambda_2 = 0$$

$$\frac{\partial \mathcal{L}}{\partial w_2} = p_2 + 0.6p_3 - \frac{p_4}{0.8} + \frac{p_5}{0.8}\lambda_1 f_2 + \lambda_3 = 0$$

$$\frac{\partial \mathcal{L}}{\partial b} = -2b\lambda_2 = 0$$

$$\frac{\partial \mathcal{L}}{\partial \lambda_1} = q - f(w_1, w_2) = 0$$

$$\frac{\partial \mathcal{L}}{\partial \lambda_2} = w_1 - s^2 = 0$$

$$\frac{\partial \mathcal{L}}{\partial \lambda_3} = w_2 - D^2 = 0$$

The return to scale (RTS) function, which describes the long term costs/benefits of the water resource management model, was developed for w_1 and w_2 so that the water production should be equated to water and wastewater as well as the shadow values of the freshwater supply and the aquifer replenishment water supply.

$$RTS = \frac{f_1}{f_2} = \frac{p_1 + 0.6p_3 + 0.6p_4 - \lambda_2}{-0.8\left(\frac{p_2}{p_5}\right) - 0.48\left(\frac{p_3}{p_5}\right) + \frac{p_4}{p_5} + \frac{\lambda_3}{1.25p_5}} \quad (4\text{-}10)$$

4.5.1 Effect of Increasing IPR Treatment Cost

The RTS is inversely affected by the cost if the AWT processes. Because the cost of AWT is likely to be as much as an order of magnitude higher than other costs, in most cases higher than anticipated increases in AWT costs will have small effects on the resulting RTS. The general relationship between RTS and AWT cost is illustrated in Figure 4-6a. The precise shape of the curve is expected to be driven by the relative cost of water treatment (p_1 and p_2) relative to the wastewater cost components; a larger denominator from relatively high wastewater treatment costs would drive the curve to a more linear shape.

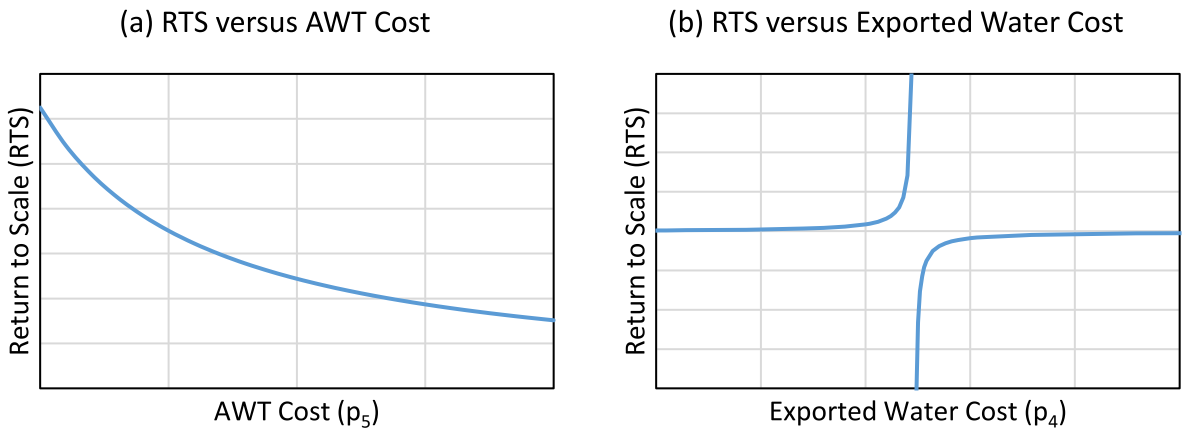

Figure 4-6 Analysis of how the return to scale is impacted by AWT process costs (a) and pumping costs to export surplus effluent (b)

4.5.2 Effect of Increasing Wastewater Treatment Cost

Rearranging the RTS function to study the effect of exported water costs yields a function with a vertical asymptote, as illustrated by Figure 4-6b. This figure illustrates that exporting water will result in increasing benefits at costs below the asymptote, but beyond that point there will be negative returns, indicating that IPR would be pursued if long term costs associated with exporting the water exceed the asymptote. The asymptote is a ratio with a numerator determined from the costs of IPR treatment and the other wastewater treatment processes (p_2, p_3, p_5) and the shadow value of storing the aquifer replenishment water from IPR (λ_3) and the denominator determined by the fraction of potable water that is discharged into sewers (assumed to be 60%) and the fraction of IPR water that is recovered from the aquifer (assumed to be 80%). Thus, exporting water becomes more economically feasible as the wastewater management processes, particularly IPR (p_5 and p_2), increase relative to the cost of exporting the water (p_4).

4.6 Conclusions

The production of potable water to meet municipal water demands is a complicated process and generally requires operation based on physical limitations in water supplies that can vary from year to year. Although several important studies have explored the microeconomics of urban water production, the production of wastewater has generally been a neglected externality. As water scarcity and climate change increasingly drive urban areas to adopt potable and non-potable water reuse systems based on constraints on wastewater discharge or conventional potable water supplies, there is a need to develop a greater understanding for the economic impacts of these strategies for urban water utilities. This study has developed a preliminary model to describe how IPR can be a long-run cost minimizing water-wastewater management strategy for urban areas facing a constraint in wastewater discharge under numerous scenarios. The costs of wastewater management are estimated to comprise the majority of the total costs of water-wastewater management. Importantly, the costs of advanced water treatment will impact the characteristics of the return-to-scale for water-wastewater management infrastructure but would depend on the relative cost of water treatment processes compared to wastewater treatment processes (including advanced water treatment). The general models presented in this paper should be further examined with empirical evidence, but the results provide insight into the situations under which IPR can be an economically optimal long-term strategy.

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Chapter 5 Conclusions

5.1 Summary

The studies together demonstrate how sustainability and IUWM principles can be operationalized to assess the ways in which urban areas use, reuse, and conserve water resources as well as to evaluate proposed water or wastewater management strategies. This research demonstrates the importance of holistic, integrated urban water approaches that consider the interdependencies of water and wastewater systems when evaluating urban water systems and proposed management strategies or infrastructure. This is accomplished using three decision making tools that integrate social, economic, and environmental impacts into decision making for water reuse projects.

Cluster analysis and the development of a composite index was demonstrated to provide unique insights to characterizing the differences in how urban areas utilize water resources. Principal component analysis identified how water-economy characteristics of cities correlate to one another and identified three components that described the 7 variables that were most significant to describing the differences between cities. Smaller semi-arid cities in the US were found to have similar water-economy characteristics. The socio-economic component included four variables, with the highest weights on the economic productivity of water resources and unemployment, followed by water demand intensity and population density. The second component was comprised of indicators for water and wastewater resources allocations. The third component encompassed the socio-environmental variables, with consumptive water demand and wastewater reuse receiving the highest weights. Evaluation of 49 U.S. metropolitan areas through the Water-Economy Index (WEI), illustrated social, economic, and environmental factors that contributed to the dynamics of water demand in an urban area. This indicator provided novel insight into the connections between economic health and water conservation in urban areas. City clusters based on WEI characteristics revealed the similarities in how the largest cities in the U.S. were able to maintain a lower per capita water demand which correlated to higher population density and water reuse. Smaller semi-arid cities also formed a cluster based on their relatively low population densities, higher rates of reuse and higher per capita water demand.

The triple bottom line analysis (TBL) of water management was developed using sustainability and IUWM principles to identify decision making criteria including water, wastewater, social, and environmental, and resource management utility impacts. The criteria were evaluated for two water-wastewater management scenarios: potable reuse versus a status-quo scenario that required exporting wastewater effluent that exceeds local wastewater management capacity. A rapidly urbanizing hydrologically closed sub-basin within the Reno-Sparks metropolitan area was selected as a case-study to evaluate the trade-offs of managing water and wastewater resources through potable reuse compared to status-quo management practices. Potable reuse was estimated to surpass status-quo management practices in both environmental and social criteria, but also resulted in larger direct costs to the joint water and wastewater utilities.

The microeconomic study of indirect potable reuse as a cost-minimizing IUWM strategy demonstrated that in the long-term, indirect potable reuse (IPR) can be a lower cost solution compared to conventional water management. The results demonstrated economic feasibility of indirect potable reuse under varying costs, and the importance of the comparative water management and wastewater management costs in determining the optimal capacity at which indirect potable reuse should be pursued for long-term cost minimization. Effluent disposal capacity was treated as a fixed parameter; it was found to influence the timing of when IPR would become a cost minimizing effluent management strategy. Although effluent disposal capacity is a uniquely closed-basin parameter, it may also have parallels with other inland communities and could be used to characterize how other regulatory requirements, like the quality of wastewater effluent, constrain the amount of effluent an area can discharge. Wastewater management costs were estimated to comprise the majority of the total costs of water-wastewater management and had a critical role in determining the timing of when or if IPR would be a cost minimizing strategy. The research assumed that wastewater management costs comprised the majority of the total costs of the IUWM system based on the costs of treating water and wastewater resources in the Reno-Sparks metropolitan area. However, IPR would never become feasible for regions with low wastewater treatment costs that are unaffected by wastewater discharge constraints.

5.2 Next Steps

The social, economic, and environmental impacts of indirect potable reuse can be very localized. The body of research in this dissertation presents a preliminary approach to understand how these impacts can be incorporated to evaluate sustainability, to compare practices across cities, to assess the trade-offs of alternative resource management scenarios, and to determine when IPR is a cost-minimizing IUWM strategy. While this work made progress in jointly accounting for the interrelationships between water and wastewater systems in an urban area, it neglected to include stormwater and the impacts of land use change from urbanization. Additionally, the general models presented in the fourth chapter should be tested with case-study cost data to demonstrate how the theoretical model can be applied to an actual urban water management challenge with uncertainty around water and wastewater treatment prices as well as different scenarios of population growth.

Chapter 6 Appendix

The first step in a PCA analysis towards identifying the rotation of the principal components is to identify the correlations between variables (Table 6-1). This step is also used to identify if any variables are too highly correlated or are uncorrelated to the others, which can skew the resulting principal component dimensions.

Table 6-1 Correlation matrix for the selected indicators

Parameter	Water Demand Intensity	Economic Productivity	Population Density	Consumptive Demand (%)	Unemployment Rate	Municipal Demand (%)	Wastewater Reuse
Water Demand Intensity	1.00	-0.75	-0.22	0.09	0.50	-0.05	0.10
Economic Productivity	-0.75	1.00	0.48	-0.40	-0.54	0.15	0.17
Population Density	-0.22	0.48	1.00	-0.36	-0.36	0.18	0.29
Consumptive Demand (%)	0.09	-0.40	-0.36	1.00	0.38	-0.37	0.31
Unemployment Rate	0.50	-0.54	-0.36	0.38	1.00	-0.32	0.01
Municipal Demand (%)	-0.05	0.15	0.18	-0.37	-0.32	1.00	0.28
Wastewater Reuse	0.10	0.17	0.29	0.31	0.01	0.28	1.00

Table 6-2 presents all the data compiled to define the candidate indicators for each metropolitan area. Note that the metropolitan areas correspond to the metropolitan statistical areas as defined by the U.S. census bureau and are comprised of one or more counties.

Table 6-2 Summary of water use, water reuse and socio-economic data for US metropolitan areas (Data from 2015)

Metro Area Name	State	County	Population	Current dollar GDP	Unemploy-ment rate	County Land area (sq mi)	Pop Dens (2015)	GPCD	% Agri-culture Reuse	% Rec-reation Reuse	% Potable Reuse
Albuquerque, NM (Metropolitan Statistical Area)*	NM		903489	41539	6.1	9282.5	693.5014	137.4135	0.4%	84.0%	100.0%
	NM	Bernalillo				1160.83	580.7052	143.2885			
	NM	Sandoval				3710.65	37.27837	107.6226			
	NM	Torrance				3344.85	4.628608	166.6825			
	NM	Valencia				1066.17	70.88926	112.3008			
Anchorage, AK (Metropolitan Statistical Area)	AK		399082	28069	5.6	26312.58	178.9304	167.9075	0.0%	100.0%	100.0%
	AK				1704.68	174.8234	170.0259				
	AK				24607.9	4.106974	139.2195				
Augusta-Richmond County, GA-SC (Metropolitan Statistical Area)	GA-SC		589688	23273	6.3	2512.42	1299.952	194.8154	0.0%	100.0%	100.0%
	GA				324.33	621.5799	200.0163				
	SC	Aiken, SC				1071.03	154.5708	249.9141			
	GA	Burke				826.97	27.37947	102.2741			
	GA				290.09	496.4218	133.5302				
	SC	Edgefield				500.41	53.37423	192.8141			
	GA	Lincoln				210.38	36.70026	1.964637			
	GA	McDuffie				257.46	83.36052	146.1873			
Austin-Round Rock, TX (Metropolitan Statistical Area)	TX		2000784	127804	3.4	4219.89	474.1318	93.682	0.6%	0.0%	82.7%
	TX	Travis				990.2	1189.954	108.6185			
	TX	Hays				677.98	286.8772	45.87797			
	TX	Caldwell				545.26	74.05091	94.40943			
	TX				1118.3	453.7423	70.91207				
	TX	Bastrop				888.15	90.29781	132.9654			
Chicago-Naperville-Elgin, IL-IN-WI (Metropolitan Statistical Area)	IL-IN-WI		9557503	639033	5.9	7196.81	1328.019	128.3192	3.8%	94.3%	100.0%
	IL	Cook				945.33	5549.206	159.3892			
	IL	DeKalb				631.31	165.3815	78.86614			
	IL	DuPage				327.5	2849.206	7.896876			
	IL	Grundy				418.04	120.4311	85.78401			
	IL	Kane				520.06	1019.104	129.6318			

	IL	Kendall				320.34	383.6299	76.538			
	IL	Lake				443.67	1588.895	96.68687			
	IL	McHenry				603.17	510.5692	77.36768			
	IL	Will				836.91	820.7812	64.10116			
	IN	Jasper				559.62	59.8281	83.91737			
	IN	Lake				498.96	978.6676	172.0565			
	IN	Newton				401.76	34.80685	91.48832			
	IN	Porter				418.15	400.4066	135.7199			
	WI	Kenosha				271.99	617.2359	125.0442			
Cleveland-Elyria, OH (Metropolitan Statistical Area)	OH		2062842	128887	5	1997.3	5034.935	154.4617	0.0%	0.0%	0.0%
	OH	Cuyahoga				457.19	2751.497	177.121			
	OH	Geauga				400.16	234.6436	65.4598			
	OH	Lake				227.49	1009.148	104.8974			
	OH	Lorain				491.1	621.6473	143.6233			
	OH	Medina				421.36	417.9989	29.97023			
Columbus, GA-AL (Metropolitan Statistical Area)	GA		310545	13762	6.9	1936.14	1150.328	183.1306	0.0%	0.0%	0.0%
	GA				248.74	44.5244	93.93939				
	GA	Harris				463.87	71.49632	244.1879			
	GA	Marion				366	23.43169	253.8711			
	GA				216.39	919.0905	183.3193				
	AL	Russell				641.14	91.78495	152.4335			
Dallas-Fort Worth-Arlington, TX (Metropolitan Statistical Area)	TX		7101031	491879	4.1	9277.78	8946.212	72.62483	0.0%	0.0%	0.3%
	TX	Collin				841.22	1087.204	3.417259			
	TX	Dallas				871.28	2931.587	133.169			
	TX	Denton				878.43	887.4606	41.4792			
	TX	Ellis				935.49	174.5449	93.15517			
	TX	Hood				420.64	131.5377	104.9745			
	TX	Hunt				840.32	106.7141	54.54383			
	TX	Johnson				724.69	219.9423	65.16076			
	TX	Kaufman				780.7	146.229	7.666637			
	TX	Tarrant				863.61	2296.919	13.41016			
	TX	Parker				903.48	138.9826	79.10842			
	TX	Rockwall				127.04	709.3907	0			
	TX	Somervell				186.46	46.19221	239.5704			
	TX	Wise				904.42	69.50753	48.74636			
	IL		107419	6260	6.3	580.69	184.9851	199.1881	0.0%	0.0%	0.0%

Decatur, IL (Metropolitan Statistical Area)	IL	Macon				580.69	184.9851	199.1881			
Denver-Aurora-Lakewood, CO (Metropolitan Statistical Area)*	CO		2807211	192498	3.7	8346.07	8828.137	167.0059	1.3%	16.9%	34.7%
	CO	Adams				1167.65	419.4939	164.6938			
	CO	Arapahoe				798.1	787.9138	226.0359			
	CO	Douglas				840.25	382.6052	128.1172			
	CO	Denver				153	4455.02	239.2077			
	CO	Elbert				1850.85	13.2658	137.5405			
	CO	Gilpin				149.9	38.63242	172.1519			
	CO	Jefferson				764.21	738.4947	42.11625			
	CO	Park				2193.85	7.619026	149.8127			
	CO			33.03	1961.732	87.14363					
Des Moines-West Des Moines, IA (Metropolitan Statistical Area)	IA		622566	50853	3.6	2883.7	1082.194	101.2244	0.0%	0.0%	0.0%
	IA	Dallas				588.45	136.7848	19.35331			
	IA	Guthrie				590.62	18.02174	124.9632			
	IA	Madison				561.01	28.00841	28.44276			
	IA	Polk				573.79	814.3258	125.0237			
	IA	Warren				569.83	85.05344	21.54244			
El Paso, TX (Metropolitan Statistical Area)	TX		836326	27867	5.2	5583.67	149.7807	128.1975	4.7%	0.0%	50.0%
	TX	El Paso				1012.69		128.409			
	TX	Hudspeth				4570.98		74.44169			
Fayetteville, NC (Metropolitan Statistical Area)	NC		384554	16981	3.7	1043.05	643.7136	115.6247	0.0%	0.0%	0.0%
	NC				652.31	508.5803	117.4155				
	NC	Hoke				390.74	135.1333	102.896			
Green Bay, WI (Metropolitan Statistical Area)	WI		315687	18336	4.2	1870.22	583.7462	85.90507	0.0%	0.0%	0.0%
	WI	Brown				529.71	486.823	83.28198			
	WI	Kew				342.52	59.37172	100.5199			
	WI	Oconto				997.99	37.55148	132.4503			
Jacksonville, FL (Metropolitan Statistical Area)	FL		1445986	68810	5.4	3201.08	451.7182	129.3041	8.3%	23.7%	0.1%
	FL	Baker				585.23		139.9689			
	FL	Clay				604.36		103.7093			
	FL	Duval				762.19		141.8391			
	FL	Nassau				648.64		148.2768			
	FL	St. Johns				600.66		87.11054			
	TN		200217	6691	6	853.83	646.877	156.8928	0.0%	0.0%	0.0%
	TN	Carter				341.2	165.1993	168.6827			

Johnson City, TN (Metropolitan Statistical Area)	TN	Unicoi				186.17	95.49874	90.82991			
	TN				326.46	386.179	161.7045				
Jonesboro, AR (Metropolitan Statistical Area)	AR		128307	4965	4.6	1465.6	179.1359	132.7037	0.0%	0.0%	0.0%
	AR	Craighead				707.21	147.4767	140.3858			
	AR	Pointsett				758.39	31.65917	97.97753			
Kansas City (Metropolitan Statistical Area)	KS-MO		2085221	125765	4.8	7255.59	4830.095	130.1658	0.0%	0.6%	0.0%
	KS	Johnson				473.37	1222.976	19.88483			
	KS				462.83	170.7106	362.6048				
	KS	Linn				594.06	16.12632	114.3037			
	KS	Miami				575.66	56.89296	175.2488			
	KS				151.6	1077.731	380.4271				
	MO	Bates				836.69	19.56161	76.29256			
	MO	Caldwell				426.39	21.13323	53.96825			
	MO	Cass				696.84	145.4997	15.38156			
	MO	Clay				397.3	592.1294	527.9646			
	MO	Clinton				418.96	49.10254	14.95513			
	MO	Jackson				604.46	1136.853	39.15415			
	MO	Lafayette				628.43	51.91668	71.12069			
	MO	Platte				420.19	229.4319	25.25569			
	MO	Ray				568.81	40.03094	105.2162			
Lafayette, LA (Metropolitan Statistical Area)	LA		490041	23776	6.4	3408.8	1240.333	131.479	0.0%	0.0%	0.0%
	LA	Acadia				655.12	95.70155	119.5007			
	LA	Iberia				574.11	128.6043	141.7173			
	LA	Lafayette				268.72	891.8763	122.8732			
	LA	S. Martin				737.65	73.0265	106.799			
	LA	Vermillion				1173.2	51.12428	217.592			
Lakeland-Winter Haven, FL (Metropolitan Statistical Area)	FL		649644	19611	6.3	1797.84	361.3469	114.2287	5.2%	20.9%	44.1%
	FL	Polk				1797.84	361.3469	114.2287			
Little Rock-North Little Rock-Conway, AR (Metropolitan Statistical Area)	AR		730107	37284	4.5	4085.18	1004.738	108.2991	0.0%	0.0%	0.0%
	AR	Faulkner				647.88	186.9374	1.248232			
	AR	Grant				631.81	28.5149	60.29851			
	AR	Lonoke				770.73	92.46818	83.89168			
	AR	Perry				551.4	18.67791	7.509387			
	AR	Pulaski				759.76	517.4094	102.4686			

	AR	Saline				723.6	160.7297	287.5158			
Los Angeles-Long Beach-Anaheim, CA (Metropolitan Statistical Area)	CA	Santa Rosa	13283824	967100	6.1	4848.45	6492.558	128.2067	41.6%	39.8%	0.3%
	CA	LA			4057.88	2494.713	123.8405				
	CA	LA									
	CA	LA									
	CA	LA									
	CA	LA									
	CA	LA									
	CA	LA									
	CA	Orange				790.57	3997.845	142.2638			
	CA	Orange									
	CA	Orange									
	CA	Orange									
	CA	Orange									
Louisville/Jefferson County, KY-IN (Metropolitan Statistical Area)	KY-IN		1277992	71699	4.6	3578.35	4032.954	142.8812	0.0%	0.0%	0.0%
	KY	Bullitt				297.02	264.9889	0			
	KY	Henry				286.28	54.37334	0			
	KY	Jefferson				380.42	2009.203	175.5028			
	KY	Oldham				187.22	344.4931	65.1374			
	KY	Shelby				379.64	120.5142	69.0419			
	KY	Spencer				186.68	96.33598	0			
	KY	Trimble				151.65	57.65908	348.4541			
	IN	Clark				372.86	308.231	199.2125			
	IN	Floyd				147.93	517.2649	16.36528			
	IN	Harrison				484.52	81.56732	81.24577			
	IN	Scott				190.4	124.2542	151.1301			
	IN				513.73	54.06926	128.9504				
Lubbock, TX (Metropolitan Statistical Area)	TX		310125	12935	3.4	2687.67	346.2693	292.0696	4.8%	0.0%	100.0%
	TX	Crosby				900.2	6.571873	292.0696			
	TX	Lubbock				895.6	333.2548	5.10506			
	TX	Lynn				891.87	6.442643	31.90132			
Miami-Fort Lauderdale-West Palm Beach, FL (Metropolitan Statistical Area)	FL		6026044	315624	5.4	5077.27	3714.712	132.5948	3.8%	39.5%	1.6%
	FL	Broward				1209.79	1563.647	117.1626			
	FL				1897.72	1425.912	128.9132				
	FL				1969.76	725.1533	162.4061				

Milwaukee-Waukesha-West Allis, WI (Metropolitan Statistical Area)	WI		1576376	99343	4.9	1454.75	5379.879	109.3696	0.0%	0.0%	0.0%
	WI				241.4	3972.075	122.0838				
	WI	Ozaukee				233.08	376.699	87.62003			
	WI				430.7	310.8289	93.60947				
	WI				549.57	720.2759	74.18914				
Mobile, AL (Metropolitan Statistical Area)	AL		414588	18520	6.9	1229.44	337.2169	175.2972	0.0%	0.0%	0.0%
	AL	Mobile				1229.44	337.2169	175.2972			
Myrtle Beach-Conway-North Myrtle Beach, SC-NC (Metropolitan Statistical Area)	NC-SC		431502	16064	7.2	1980.87	417.0851	164.0548	0.0%	0.0%	0.0%
	NC				846.97	144.3935	24.00837				
	SC	Horry				1133.9	272.6916	243.3377			
Orlando-Kissimmee-Sanford, FL (Metropolitan Statistical Area)	FL		2391028	123054	5.2	3478.48	3473.887	174.4089	23.5%	39.4%	76.3%
	FL	Lake				938.38	347.0865	195.9537			
	FL	Orange				903.43	1430.114	182.2642			
	FL				1327.45	244.2194	180.1975				
	FL	Seminole				309.22	1452.467	135.8649			
Owensboro, KY (Metropolitan Statistical Area)	KY		117408	5494	4.6	898.47	300.0843	131.7621	0.0%	0.0%	0.0%
	KY	Daviess				458.35	216.7863	140.2051			
	KY	Hancock				187.65	46.06981	81.30081			
	KY	McLean				252.47	37.22819	78.00136			
Pensacola-Ferry Pass-Brent, FL (Metropolitan Statistical Area)	FL		475894	16913	5.3	1668.06	635.9696	119.991	31.2%	54.1%	0.0%
	FL	Escambia				656.46	471.5124	129.0531			
	FL				1011.6	164.4573	102.0234				
Philadelphia-Camden-Wilmington, PA-NJ-DE-MD (Metropolitan Statistical Area)	PA-NJ-MD-DE		6066589	418605	5.4	4602.11	23761.26	127.0638	0.0%	0.0%	0.0%
	PA	Bucks				604.31	1036.238	199.7784			
	PA	Chester				750.51	686.5012	138.4141			
	PA	Delaware				183.84	3066.28	45.16306			
	PA				483.04	1693.17	96.48326				
	PA				134.1	11711.46	159.2021				
	NJ	Burlington				798.58	562.0526	140.2953			
	NJ	Camden				221.26	2308.637	88.40461			
	NJ				322.01	905.7203	73.30145				
	NJ	Salem				331.9	192.0157	100.3147			
	MD	Cecil				346.27	295.8732	90.1781			

	DE				426.29	1303.308	121.2709			
Phoenix-Mesa-Scottsdale, AZ (Metropolitan Statistical Area)	AZ		4558145	221570	5.2	14565.75	526.8303	185.8117	3.8%	22.8%	72.6%
	AZ	Maricopa				9200.14	451.5231	188.229			
	AZ	Pinal				5365.61	75.30719	160.5019			
Portland-Vancouver-Hillsboro, OR-WA (Metropolitan Statistical Area)	OR-WA		2382181	156869	5.2	6683.75	3787.458	123.1705	0.0%	0.0%	0.0%
	OR				1870.32	213.7736	411.2587				
	OR	Columbia				657.36	75.52635	131.3436			
	OR				431.3	1832.379	19.37857				
	OR				724.23	790.376	95.51569				
	OR	Yamhil				715.86	141.6716	106.4137			
	WA	Clark				629	726.8601	123.4231			
	WA	Skamania				1655.68	6.871497	155.1749			
Racine, WI (Metropolitan Statistical Area)	WI		194763	7917	5.6	332.5	585.7534	118.6778	0.0%	0.0%	0.0%
	WI	Racine				332.5	585.7534	118.6778			
Reno, NV (Metropolitan Statistical Area)	NV		448230	23359	6.2	6302.37	70.50649	134.3159	1.2%	15.4%	0.0%
	NV	Washoe				6302.37	70.50649	134.3159			
Rochester, MN (Metropolitan Statistical Area)	MN		213829	11366	3.2	2476.91	343.0001	100.8168	0.0%	0.0%	0.0%
	MN	Dodge				439.28	46.44873	74.85761			
	MN	Fillmore				861.3	24.05201	108.1081			
	MN	Olmstead				653.35	231.6278	98.46107			
	MN	Wabasha				522.98	40.87154	142.4172			
Salt Lake City, UT (Metropolitan Statistical Area)	UT		1167013	80644	3.4	7683.63	1496.835	115.9662	1.9%	19.2%	0.0%
	UT	Salt Lake				742.28	1487.811	107.9551			
	UT	Tooele				6941.35	9.024325	256.6796			
San Antonio-New Braunfels, TX (Metropolitan Statistical Area)	TX		2379054	115020	3.8	6508.91	2133.167	112.6947	0.3%	0.0%	0.2%
	TX	Atascosa				1219.54	39.64528	116.5506			
	TX	Bandera				790.96	26.7346	62.20993			
	TX	Bexar				1239.82	1528.295	116.3652			
	TX	Comal				559.48	229.951	134.7144			
	TX				711.3	211.7039	47.69033				
	TX	Kendall				662.45	60.33361	71.37022			
	TX	Medina				1325.36	36.50329	140.3163			
San Diego-Carlsbad, CA	CA	San Diego	3290044	211807	5.2	4206.63	782.1092	124.9707	9.4%	39.8%	2.1%
	CA				4206.63	782.1092	124.9707				

(Metropolitan Statistical Area)											
San Francisco-Oakland-Hayward, CA (Metropolitan Statistical Area)	CA		4657985	445124	4.3	2470.55	24484.55	103.1923	4.8%	31.9%	26.4%
	CA	Alameda				739.02	2215.286	100.2209			
			East Bay Dischargers Authority								
			Hayward City								
			Union Sanitary District								
			Livermore, City of								
			Pleasanton, City of								
			San Leandro, City of								
	CA				715.94	1572.795	134.8625				
			Brentwood city								
	CA	Marin				520.31	503.004	109.1333			
			Marin Municipal Water District								
			North Marin Water District								
			Novato Sanitary District								
	CA				46.87	18483.46	78.47484				
	CA				448.41	1709.995	92.03858				
			Palo Alto, City of								
			Pacifica, City of								
			Redwood, City of								
Savannah, GA (Metropolitan Statistical Area)	GA		378664	17053	5.7	1340.11	871.8269	195.611	0.0%	0.0%	0.0%
	GA	Bryan				435.97	79.81742	101.4609			
	GA	Chatham				426.44	672.5893	101.5596			
	GA	Effingham				477.7	119.4201	955.6861			
Seattle-Tacoma-Bellevue, WA (Metropolitan Statistical Area)	WA		3728606	317153	4.8	5872.35	1873.661	108.4901	0.0%	0.0%	0.0%
	WA	King				2115.57	1001.005	99.39249			
	WA	Pierce				1669.51	504.0197	141.9075			
	WA				2087.27	368.6365	96.80122				
	NY		659850	32271	5.4	2384.88	837.0509	264.5297	0.0%	0.0%	0.0%
	NY	Madison				654.84	109.5199	22.78755			

Syracuse, NY (Metropolitan Statistical Area)	NY	Onondaga				778.39	601.5404	263.847			
	NY	Oswego				951.65	125.9906	423.5651			
Tulsa, OK (Metropolitan Statistical Area)	OK		980208	60543	4.4	6269.24	1572.477	141.519	0.0%	0.0%	0.0%
	OK	Creek				950.14	74.42798	90.91891			
	OK	Okmulgee				697.35	56.03356	149.2002			
	OK	Osage				2246.36	21.06964	423.3264			
	OK	Pawnee				567.95	28.94973	80.28335			
	OK	Rogers				675.63	133.2815	888.1643			
	OK	Tulsa				570.25	1122.125	0			
	OK	Wagoner				561.56	136.5891	431.505			
Waterloo-Cedar Falls, IA (Metropolitan Statistical Area)	IA		170602	9361	4.4	1503.11	317.4259	122.2196	0.0%	0.0%	0.0%
	IA				565.77	235.8467	133.7803				
	IA	Bremer				435.48	56.8591	85.02252			
	IA	Grundy				501.86	24.72004	30.83857			
Wichita, KS (Metropolitan Statistical Area)	KS		642782	32081	4.6	5012.42	651.4041	122.8825	0.1%	2.1%	0.0%
	KS	Butler				1429.86	46.33321	215.5578			
	KS	Harvey				539.75	64.4428	298.3905			
	KS	Kingman				863.36	8.835248	183.2687			
	KS	Sedgwick				997.51	511.9899	104.5842			
	KS	Sumner				1181.94	19.80304	89.22881			
Winston-Salem, NC (Metropolitan Statistical Area)	NC		657101	28677	5.5	2008.62	1570.715	133.4137	0.0%	0.0%	0.0%
	NC	Davidson				552.67	296.1804	95.46264			
	NC	Davie				264.11	157.8017	112.5255			
	NC	Forsyth				408.15	901.4382	171.5799			
	NC	Stokes				448.86	102.8227	9.368559			
	NC	Yadkin				334.83	112.472	114.894			
Yuma, AZ (Metropolitan Statistical Area)*	AZ		203558	6357	21.7	5513.99	36.91664	186.7394	0.1%	23.0%	0.0%
	AZ	Yuma				5513.99	36.91664	186.7394			

Calculation of health risk was carried out by estimating the concentrations of pathogens that could be present after advanced water treatment based on reported removal rates (log reductions). Health risk is modeled from dose-response relationships, to determine an annual risk. U.S. EPA specifies that an acceptable annual risk for infection from each pathogen is 1E-4 infections per year.

Table 6-3 Pathogen Health Risk Calculation from Dose-Response Relationships

Pathogen	**Enterovirus (gc/L)**	**Cryptosporidium (oocysts/L)**	**Giardia (cysts/L)**
Raw sewage maximum density	1E+05	1E+04	1E+04
Required log reduction	12	10	10
Pathogen density after advanced treatment	1.0E-07	1.0E-06	1E-06
Dose-response relationship[1,2,3]	$p = 1 - \left(1 + \frac{d}{\beta}\right)^{-\alpha}$	$p = 1 - \left(1 + \frac{d}{\beta}\right)^{-\alpha}$	$p = 1 - \exp\,(-kd)$
Risk of illness (illness/(capita*d))	7.3E-11	4.2E-12 6.5E-10	1.2E-08
Annual risk (Hunter) $p_a = 1 - (1 - P_d)^{365}$	2.7E-08	1.5E-09	4.3E-06
Acceptable annual risk	1E-04	1E-04	1E-04
Margin of safety	4E+03	7E+04	23

1. Enterovirus estimated based on Norovirus kinetics: α=0.04 β=0.055 (Haas et al., 1999)
2. Cryptosporidum estimated from the most conservative kinetics reported in literature: α=0.0042 (Haas et al., 1999) and α=0.115 β=0.176 (Hunter et al., 2011)
3. Giardia estimated based on Teunis and Havelaar (2002): β=0.0119

Table 6-4 presents the results of the health risk assessment for each CEC considered.

Table 6-4 Health Risk Calculation for CECs

CEC	Concentration after AWT (ng/L)	RBAL (ng/L)	Margin of Safety (ng/L)
17ß-Estradiol	9.33	580	62
Atenolol	33.1	4,000	121
Carbamazepine	254.5	10,000	39
DEET	5.5	2,500	455
Estrone	0.21	350	1667
NDMA	0.28	0.7	2.50
Meprobamate	170.0	200,000	1176
Perfluorooctane sulfonate (PFOS)	0.45	200	444
Perfluorooctanoic acid (PFOA)	22.0	400	18
Primidone	186.0	840	5
Sucralose	27000	170,000	6
Tris (2-chloroethyl) phosphate (TCEP)	60.5	2,500	41
Triclosan	9.0	350	39

For the analysis in Chapter 3, the costs were treated qualitatively due to uncertainty around actual costs, with the SQ cost defining the expected cost and the PR cost and an anticipated percent increase or decrease. Table 6-5 presents the planning level costs estimated that were used to develop the qualitative cost analysis.

Table 6-5 Capital and Operating & Maintenance Cost Evaluation

	Benchmark wastewater treatment cost	**SQ**	**PR**
Capital costs	Benchmark value (WRRF costs)	$53,200,000	$53,200,000
	Additional WRRF treatment costs	$6,000,000	$17,000,000
	Costs of reclaimed water distribution system	$20,000,000	$33,000,000
	Program development (political, regulatory, public processes, and public outreach)	0	$3,500,000
	Total capital costs	$79,200,000	106,700,000
O&M	Benchmark	$1,700,000	$1,700,000
	Operating costs for potable water service ($/y)	$ 0	$0
	Operating costs for RSWRF service (includes pumping costs to discharge point	$290,000	$230,000
	Cost of ongoing regulatory oversight	$0	$250,000
	Total annual costs ($/y)	$1,990,000	$2,180,000