THESIS

INTEGRATED WATER RESOURCES MANAGEMENT UNDER UNCERTAINTY: EXPLORING INTERCONNECTED TECHNOLOGICAL, INFRASTRUCTURAL AND INSTITUTIONAL SOLUTIONS

Submitted by

Benjamin Wostoupal

Department of Civil and Environmental Engineering

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Master's Committee:

Advisor: Mazdak Arabi

Christopher Goemans
André Dozier

ABSTRACT

INTEGRATED WATER RESOURCES MANAGEMENT UNDER UNCERTAINTY: EXPLORING INTERCONNECTED TECHNOLOGICAL, INFRASTRUCTURAL AND INSTITUTIONAL SOLUTIONS

Rapidly growing populations in many of the world's semiarid regions intensify competition for increasingly scarce freshwater resources. Growing urban demands, land-use change, and a changing climate will further exacerbate regional vulnerability to water scarcity. The intensification of these trends creates several challenges for the future planning and management of water resources. In this work we employ the use of an integrated socioeconomic, hydrologic, and ecological modeling framework to quantify the effects of water rights allocation on a representative semiarid river basin. Through this framework we analyze the tradeoffs of several water management practices, institutional settings, and regional policies on municipal and agricultural sectors. Generally, the agent-based adoption of water management strategies can alleviate the harm of water scarcity while providing positive feedbacks to reducing municipal costs and increasing agricultural profit from production. Household adoption of xeriscaping is considered the most important technology to lower urban demands and offset the negative externalities of rural-to-urban water transfers. Additionally, an uninhibited water market leads to the most effective allocation of water rights, providing benefits to both rural and municipal communities. The future allocation of water rights under climatic, institutional, agricultural, and technological uncertainty shows significant sensitivity to fluctuations in water conveyance infrastructure costs. Such changes in infrastructure costs (i.e. 50% to 150%) can nearly double

the expected costs of reliably supplying water to urban households. However, urban water supply planners can incentivize the adoption of water management practices to stabilize these costs. Further, required water purchases for land developers set by urban planners can be used as a key policy tool for keeping costs low. This work contributes to existing literature in integrated water resources management to help understand the effects of water scarcity and provide practical solutions for urban water planners in rapidly urbanizing semiarid regions.

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CHAPTER 1: INTRODUCTION

1.1 Overview

Water scarcity is one of the greatest challenges facing our society in the present era (Postel 2000; Seckler et al. 1999). Over the course of the last century rapid population growth, variations in climate, and land-use change have created severe problems for managing our increasingly-scarce freshwater resources (Mekonnen and Hoekstra 2016; Vorosmarty 2000; White and Nackoney 2003; World Economic Forum 2017). The presence of water scarcity can create serious harm to many facets of a community by negatively impacting the economy, environment, and those in poverty (UN-Water 2007). Further, the competing and growing demands between disparate sectors can exacerbate these impacts (Flörke et al. 2018). Yet, in the face of rising scarcity, opportunities exist to better plan and manage water resources. An important stage in addressing water scarcity is to first categorize and quantify where its effects will be most pronounced. Next, several water management practices, institutional settings, and policy changes can be incorporated to alleviate and better manage water scarcity. In this study we analyze several water management practices to address water scarcity in semiarid regions through an integrated social, ecological, and technological framework. Practical water management solutions, evaluated under deep uncertainty, are key outcomes of this work. Through this study we provide policymakers and water planners with information to help effectively manage water resources in semiarid regions across the globe.

1.2 Background

Over one billion people in the world lack access to clean and safe drinking water (Vorosmarty 2000). Several water scarcity analyses, summarized well by Rijsberman, show that nearly two-thirds of the global population will be affected by water scarcity in the coming decades (Rijsberman 2006). Unsurprisingly, these populations and their livelihoods are severely impacted by their access to clean water. The effects of water scarcity permeate into local and global economies. Numerous studies have quantified the effects of water scarcity and water trading on economies at the regional scale. Notable works include restricted-supply computable general-equilibrium models, geospatial water stress frameworks, and "virtual water" trading (Berrittella et al. 2007; Hoekstra and Hung 2002; Vörösmarty et al. 2010). As noted by Vörösmarty et. al. (2010), regional water scarcity can have severe impacts on a nation's ecological biodiversity, gross domestic product, and vulnerability to extreme weather events. Locally, inequitable water allocation, particularly by way of water scarcity, can cause detrimental harm to several economic sectors. In semiarid regions this reallocation of water often occurs in the form of rural-to-urban water transfers. As a result, agricultural communities which depend on the economic activity of irrigated agriculture are negatively impacted (Dozier et al. 2017; Howe et al. 1990; Howe and Goemans 2003; Start 2001; Thorvaldson and Pritchett 2006). Such results can have somber effects on standards of living: income, education, health, childhood development, and crime among several others (Brooks-Gunn and Duncan 1998; Hsieh and Pugh 1993; Wagstaff 2002; Yoshikawa et al. 2012). However, effectively managing water resources remains a clear alternative to the persistence of water scarcity.

Effectively managing freshwater resources is imperative for the health of our environment, economy, and societal wellbeing. Historically, the reliable delivery and treatment of water supplies were considered best practice for the management of water resources. As such, tremendous amounts of infrastructure were created to deliver these goals (Ansar et al. 2014). This traditional "hard path" approach often emphasized large infrastructure projects through dams, pipelines, and treatment plants (Gleick 2003). While these water supply plans were largely successful, such infrastructure inadvertently caused detrimental impacts to many vulnerable environmental and ecological systems (Ansar et al. 2014; Pittock and Lankford 2010; Poff et al. 2016). Impacts to these systems include losses to biodiversity, biological integrity, ecosystem health, declines to native fisheries, and several large rivers drying up before meeting their historically flowing deltas (Gleick 2003). Thus, to meet the growing demands of future populations, a purely infrastructure-first approach may be misguided. Evaluating the impacts of water supply infrastructure, alongside more general water resources management, are needed to quantify the tradeoffs of supply portfolios. Next, an integrated approach can be considered to holistically manage our water resources and eliminate water scarcity.

1.3 Integrated Water Resources Management

While traditional infrastructure solutions are important a “soft path” approach of water resources management may be more effective for the complex challenges water planners face. This “soft path” approach calls for the use of water conservation, more efficient water markets, the protection of environmental flows, and an integrated approach to water resources management (Gleick 2003; Gleick et al. 2003). This integrated approach couples several models to account for the interdisciplinary nature of water resources. Proper water resources planning and management requires long-term climate forecasting, hydrologic modeling, ecological assessments, large-scale infrastructure, and stakeholder involvement among several other factors (Karr 1991; Lubell and Edelenbos 2013; Mitchell 2005; Thomas and Durham 2003). This systems approach to water resources engineering is commonly referred to as Integrated Water Resources Management (IWRM) (GWP TAC 2000). IWRM holds much promise to effectively manage water resources. However, the mutually-dependent systems required for accurate IWRM can lead to ineffective, if not contradictory, water resources management portfolios due to the complexity of these systems (Biswas 2008; Giordano and Shah 2014). Although IWRM may not be in itself the best pathway forward, a combination of ideas from IWRM alongside robust optimization and modeling frameworks seem to offer practical solutions for resolving water scarcity issues.

Several creative and practical management practices, optimization methods, and modeling frameworks have been proposed to provide solutions for water scarcity. First, appropriate water management practices are key to effectively managing water resources. Several water management practices include water conservation, efficient water markets, green infrastructure, and alternative transfer methods (DiNatale Water Consultants 2013; Larson et al. 2009; McMahon and Smith 2013; Richter et al. 2013; Tzoulas et al. 2007; Vaux and Howitt 1984; Wilkins-wells and Anderson 2002). However, water management strategies alone are a necessary but not sufficient condition to effective IWRM, due to the multitude of tradeoffs which exist for each opportunity. A combination of an integrated modeling approach alongside water management practices can help to quantify and recognize these tradeoffs. Multitudes of integrated modeling approaches have been created to help solve water scarcity. Notable examples include multi-objective optimization programs (Kasprzyk et al. 2009; Nicklow et al. 2010), decision support systems (Andreu et al. 1996; Fredericks et al. 1998; Zagona et al. 2001), and hydroeconomic modeling frameworks (Harou et al. 2009; Maneta et al. 2009). To evaluate solutions to water scarcity in semiarid regions we chose a methodology which incorporates water management practices, hydroeconomic modeling, an integrated modeling framework, and a robust optimization engine.

Past modeling frameworks thoroughly advanced the present-day approach to creating solutions for water scarcity. However, additional research is required to fully enumerate the impacts of water management practices, population growth, water supply uncertainty, and land-use change at a spatiotemporally fine, agent-based scale. This gap in the current research requires integrating several municipal, agricultural, climatic, agronomic, and land-use change models. Each of these driving models categorizing water supplies, agricultural production functions, land-use, and water demands must be parameterized using locally-calibrated inputs. Next, water management practices such as household conservation, water markets, and policy objectives evaluated through the modeling framework should be included. Finally, all of the preceding model drives (e.g. climate, land-use) can be evaluated under future uncertainty. As such, there exists an opportunity to add to the field of water resources management through this paper.

1.4 Outline of Methodology

Building on several advances in water resources management, we employ the use of an integrated modeling approach with selected water management practices. "Computational Semiarid Water Sustainability" (CSaws) is an agent-based, partial-equilibrium hydroeconomic model with locally-calibrated urban water demand and agricultural production functions. The model evaluates water allocation in a spatially heterogeneous manner, representing municipalities and agricultural producers as decision-making agents. Several integrated models characterizing crop production, water supply and demand, water rights, and land-use change are included. Next, we incorporate several water management practices to evaluate the tradeoffs of these practices on water scarcity. New supply expansion projects, agricultural irrigation technology, urban household conservation, water markets, and regional policy objectives are evaluated with the integrated modeling framework. Lastly, uncertainty in model drivers are evaluated with CSaws and a global sensitivity analysis. The important impact of this work is to assess the sensitivity of water resources planning and management under future uncertainty in water supplies, water management practices, agricultural production functions, institutional change, and municipal growth. By incorporating an integrated modeling framework (CSaws) with water management practices under future uncertainty a comprehensive analysis of the tradeoffs for water resources management in semiarid regions can be assessed.

1.5 Research Objectives

This thesis is a compilation of four separate chapters. The first chapter provides an overview of IWRM and previous literature. The second chapter evaluates the effects of water management practices, water markets, and regional policy on the future allocation of water rights for a representative semiarid river basin. Included in this chapter is a fully-characterized methodology of the integrated modeling framework, CSaws. Key research objectives of this chapter are to: (i) explore the most effective water management solutions for sustaining irrigated agriculture, (ii) evaluate the tradeoffs of market-driven and goal-oriented water market institutions, and (iii) identify policies which affect municipal water purchases and impact the benefits of water management strategies. The purpose of the second chapter is to identify water management solutions to sustain agriculture and rural economies in rapidly growing regions under water scarcity.

The third chapter addresses challenges in water resources planning and management by explicitly modeling uncertainty with CSaws. Specific research objectives are to: (i) discuss the effects of uncertainty on satisfying future urban demands, (ii) discover critical parameters that affect municipal and rural success indicators, (iii) identify key thresholds of critical parameters, and (iv) evaluate future changes in climate and policy for municipal water supply planning. The last chapter succinctly summarizes the key findings of chapters two and three while providing a general synopsis of this work.

CHAPTER 2: CAN IN-BASIN WATER MANAGEMENT STRATEGIES SAVE AGRICULTURE?

2.1 Introduction

Semiarid river basins in rapidly growing regions face arduous water supply challenges due to the effects of population growth, land-use change, and complex institutional agreements (Mekonnen and Hoekstra 2016; Vorosmarty 2000; White and Nackoney 2003; World Economic Forum 2017). The competing uses of water between municipal, agricultural, industrial, and environmental sectors intensify the need for the efficient management of water resources under water scarcity. A changing climate and rising population creates additional challenges for meeting these competing demands (Flörke et al. 2013, 2018; Richter et al. 2013). The role of irrigated agriculture in semiarid regions is often called upon to meet these rising demands without sacrifices to food security, rural economies, or agrarian culture. Increases in irrigation efficiency and agricultural-to-municipal water transfers are the primary pathways for solving water scarcity. Consequently, decreases in rural economic activity occur from water transfers out of agriculture.

The purpose of this study is to identify water management solutions to sustain agriculture and rural economies in rapidly growing regions under water scarcity. We specifically ask: Can new water supply and demand management strategies save agriculture in semiarid regions over the 21st century? Saving agriculture here means to keep agricultural producers in business to reduce the negative consequences of municipal water acquisition. To accomplish this goal, specific objectives of this study are to: i) explore the most effective water management solutions for sustaining irrigated agriculture, ii) evaluate the tradeoffs of market-driven and goal-oriented water market institutions, and iii) identify policies which affect municipal water purchases and impact the benefits of water management strategies. Results indicate the adoption of water management strategies can sustain irrigated agriculture, decrease costs to municipalities, and quantitatively define tradeoffs between water market intuitions and governing policies.

Globally, irrigated agriculture accounts for nearly 70% of freshwater withdrawals, yet contributes 40% of food production on only one-fifth of all cultivated lands (FAO 2016). However, as a result of urbanization, cities are acquiring water for future generations by drying up historically irrigated cropland through water rights transfers, also referred to as buy-and-dry (Thorvaldson and Pritchett 2006). In the American West nearly 1.75 million acres of irrigated cropland have been taken out of production since 2002, likely attributed to urbanization and growing water demands[1]. Land development policies often exacerbate the situation by requiring large volumes of water acquisition per unit of new urban land development. Additionally, the sale of water rights from agriculture to municipalities is a potentially profitable enterprise for agricultural producers in regions under prior appropriation management. Past literature (Dozier et al. 2017) has revealed that the sale of farmers' water rights in the Front Range of Colorado can compensate for profits from nearly 40 years of crop production.

The drying of irrigated cropland is of great importance for policymakers looking to minimize buy-and-dry impacts on rural economies (Colorado Water Conservation Board 2016). The sale of water rights from agricultural communities to urban areas often results in detrimental externalities due to lost agricultural land and production (Howe et al. 1990; Howe and Goemans 2003; Start 2001). Thorvaldson and Pritchett (Thorvaldson and Pritchett 2006) have shown decreases in irrigated acreage can have negative direct (primary) and indirect (secondary) effects

[1] Calculations include states of Arizona, California, Colorado, Idaho, Montana, Nevada, New Mexico, Oregon, Utah, Washington, and Wyoming. Data obtained from the USDA Census of Agriculture, available at: https://www.agcensus.usda.gov/.

on local economies. These secondary consequences have a multiplier effect on economically-linked sectors, perpetuating the loss of economic activity and rural livelihood. Consequently, if an irrigator does not reinvest water sales profits locally (e.g. areas with high farm debt), these economies suffer from the lost benefits of economic linkages. Such results can have somber effects on standards of living: income, education, health, childhood development, and crime among several others (Brooks-Gunn and Duncan 1998; Hsieh and Pugh 1993; Wagstaff 2002; Yoshikawa et al. 2012).

Complex water institutions define the water management regime in numerous semiarid regions. Due to these institutions, uncertainty in future supplies, and expectations of future prices municipalities seek "firm-yield" water, or water with a high likelihood of delivery. Under the pressures of water institutions, price expectations, and transaction costs it is of little surprise that water markets operate inefficiently. Alternatives exist to traditionally inefficient water allocation markets, notably the Colorado-Big Thompson (C-BT) project in Northern Colorado, U.S. Additionally, alternative transfer methods (ATMs) are substitutes to an imperfect water market system and alleviate some of the deadweight loss created by transaction costs and strict water court rules (DiNatale Water Consultants 2013; McMahon and Smith 2013; Western Governors' Association 2012; Wilkins-wells and Anderson 2002). Leasing agreements, water banking, cap and trade programs, rotational fallowing, and deficit irrigation are among many proposed through prior research efforts (Colorado Water Conservation Board 2016; Thorvaldson and Pritchett 2006; WestWater Research 2016). ATMs help to alleviate the welfare loss of inefficient markets but growing municipal demand will eventually result in lower acreage in production.

We integrate supply-side and demand-side management practices within a robust modeling framework driven by municipal growth, land-use change, and climate to assess impacts on agricultural production across time and space (Dozier et al. n.d., 2017). Previous literature has explored the effects of municipal pricing, drought-induced restriction policies, and potential conservation yields (DeOreo et al. 2016; Gleick et al. 2003; Kjelgren et al. 2000; Larson et al. 2009). This work expands previous studies through the inclusion of independent economic decision making across agricultural and municipal sectors in addition to enhanced temporal and spatial resolution. Supply infrastructure improvements, agricultural irrigation technology upgrades, indoor urban water conservation (e.g. efficient toilet renovations), and outdoor urban water conservation (e.g. xeriscaping landscape conversions) represent the opportunities available to municipal and agricultural agents when planning for water supplies between 1980 and 2070. The framework enables the exploration of technologies to alleviate critical tipping points through institutional controls, new supplies, and water-conserving technology.

2.2 Methodology

2.2.1 Integrated Modeling Framework Overview

To evaluate the effects of technology adoption on agricultural water transfers we developed a robust modeling framework illustrated in Figure 1. "Computational Semiarid Water Sustainability" (CSaws) is an agent-based, partial-equilibrium hydroeconomic model with locally-calibrated urban water demand and agricultural production functions. The model evaluates water allocation in a spatially heterogeneous manner, representing municipalities and agricultural producers as decision-making agents. Agricultural producers individually represented by cropping systems are driven by future climate conditions, local soil properties, and historical water rights. Producers maximize future profit through the sale of crops and water rights, motivating their decision to continue in production or retire their acreage. Urban water demand, pressured by rapid population growth and land development, must be satisfied by cost-minimizing municipal and industrial (M&I) agents. These municipalities respond to urban water demand by transferring water rights from agriculture or pursuing investments in technology. A water market between M&I and agricultural agents is solved by the hydroeconomic model and various economic institutions, policies, and governing systems are incorporated. The model is evaluated from 1980-2070 to include training (1980-2010) and future assessment (2020-2070) periods. Components and description of the model formulation, data inputs, and optimization methods are equivalent to those of Dozier et. al. (Dozier et al. 2017) and can be found in the supplementary information (SI).

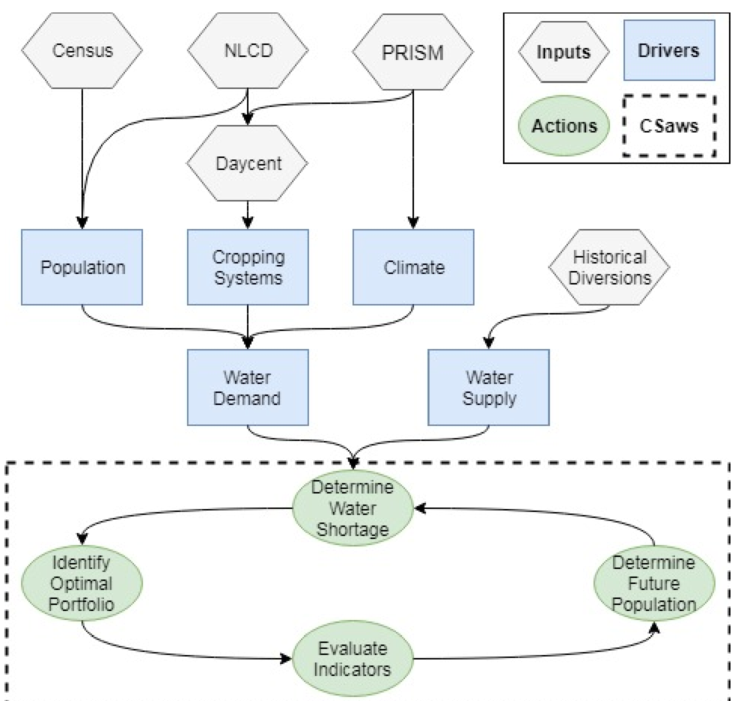


Figure 1 – Model diagram of CSaws framework.

2.2.2 The South Platte River Basin (SPRB), Colorado

The study river basin is a semiarid region experiencing water scarcity. The South Platte River Basin (SPRB) is home to the Front Range Urban Corridor, a growing mega-region facing massive population growth and urbanization (Hagler 2009). The Colorado State Demographer predicts a population of over 6 million by 2050 in the SPRB, nearly doubling in population since 2000 (Colorado Department of Local Affairs 2015). Further, a gap between M&I water supply and demand estimated to reach 400,000-500,000 acre-feet (AF) by 2050 exists (Colorado Water Conservation Board 2016). As a result of population growth and future supply shortages agriculture in the SPRB is declining. Trends in irrigated acreage from 1965 to 2010 illustrate a decrease in irrigated acreage in the SPRB, likely attributed to supply limitations, the competitive uses of water, and urbanization (Leonard Rice Engineers Inc. 2008). The SPRB is an ideal case study to evaluate various water management strategies, market institutions, and water-related policy, though the framework is applicable to all semiarid river basins throughout the world.

The South Platte River Basin is split into five separate sub-regions which represent various counties in Northeastern Colorado and levels of agricultural, municipal, and industrial presence (Figure 2). The North sub-region consists of Boulder, Broomfield, and Larimer counties and is represented by a near equal amount of municipal and agricultural presence. The North Central sub-region encompasses Weld County and has a small municipal presence with a large amount of irrigated acreage. The Central sub-region is represented by Adams, Arapahoe, Clear Creek, Denver, Gilpin, and Jefferson counties and has the largest municipal presence out of all sub-regions, with a small amount of irrigated agriculture. The South Metro sub-region consists of Douglas, Elbert, and Park counties with an equal amount of municipal and agricultural land. Lastly, the East sub-region includes Morgan, Logan, Sedgwick, and

Washington counties and consists of nearly all agricultural land, with a few small municipalities. The C-BT project serves the North, North Central, and East sub-regions due to ease of gravitational conveyance and existing canals, streams, and rivers. Sub-region boundaries were created by aggregating various counties and combing their respective political county lines.

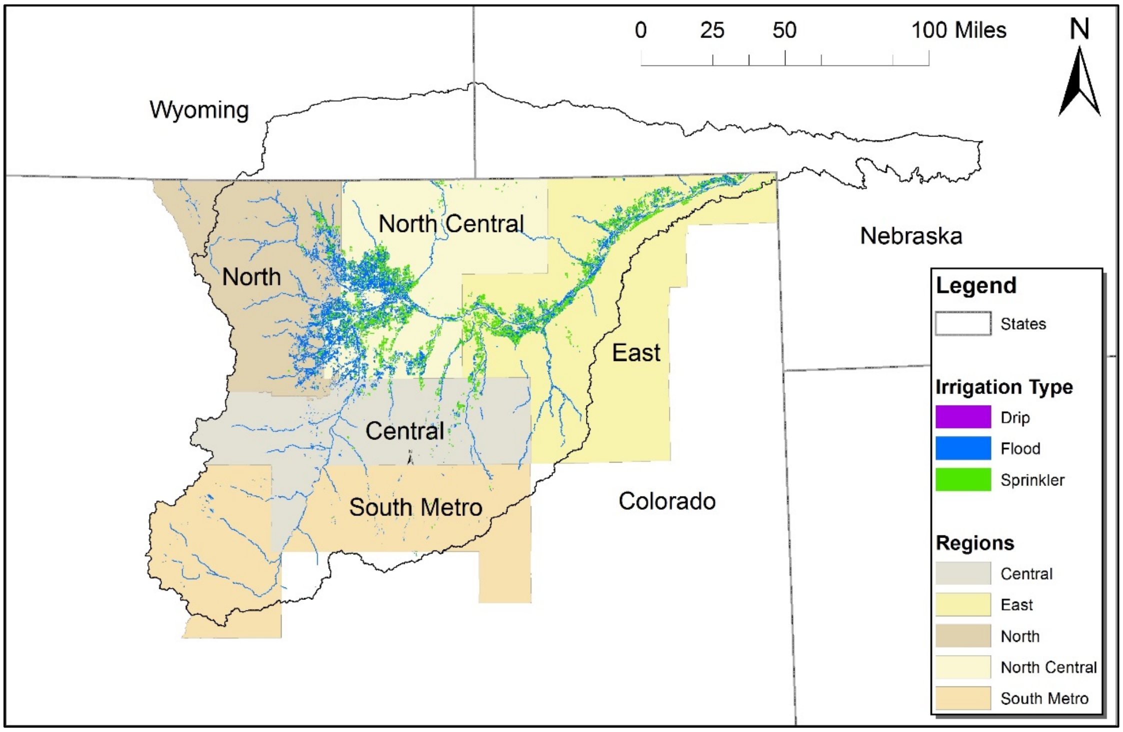


Figure 2 – The South Platte River Basin of Northeastern Colorado, U.S. All irrigated fields in 2010 are illustrated according to their respective irrigation types.

2.2.3 Management Strategies

New supply and conservation technologies provide pathways for sustaining agriculture by reducing the burden of urban water demand on irrigated crop production. Supply infrastructure improvements increase the quantity of reliable water within the river basin and were selected by five proposed reservoir developments in the SPRB (City of Fort Collins Utilities 2017; Denver Water 2017; Northern Colorado Water Conservancy District 2008, 2015a; US Army Corps of Engineers 2013). Conservation practices such as farmland irrigation technology improvements, efficient toilet renovations, and xeriscaping effectively reduce water usage and act as alternatives to traditional supply expansion. Upgrades to irrigation technology increase farmland application efficiency through investments in center pivot, center pivot with attachment, and linear move systems ranging from $139,000 to $214,000 per 160-acre field (Scherer 2015). Toilet efficiency renovations decrease household indoor demand by replacing inefficient models with highly-efficient 1.28 gallon per flush toilets. Residential outdoor water use is reduced through the conversion of traditional lawns to water efficient landscapes through xeriscaping. The replacement of high water-use grasses to drought-adapted or native plants is often recommended and subsidized in regions with scarce water supplies. A rebate of $2 per square foot was assumed to estimate household willingness-to-accept payments for landscape conversion and is similar to several xeriscaping rebates across the Western U.S. (California Division of Water Resources 2015; Las Vegas Valley Water District 2016).

Agricultural producers improve their application efficiency by investing in irrigation technology upgrades. Improved application efficiencies allow producers to continue crop production, thereby maintaining economic linkages in rural communities while freeing up conserved water for municipal purchase. Agricultural water use and application efficiencies were estimated using downscaled climate projections and locally-calibrated ecosystem properties in combination with an agro-ecosystem model, DayCent (Parton et al. 1998; Zhang 2016). Investing in conservation technologies allows municipal agents to capture historical demand to satisfy future water acquisition requirements. Household water uses were estimated using the Integrated Urban Water Model (IUWM) for calibrated water use data in the city of Fort Collins, CO at a block group resolution (Sharvelle et al. 2017). The adoption of technologies are dependent on individual opportunity costs and only implemented when it is economically beneficial for each agent. The hydroeconomic model estimates these levels and does not make adoption compulsory or set basin-wide objectives. Additional parameters such as technology costs, water savings yields, and maximum savings parameters are equivalent to Dozier et. al. (Dozier et al. n.d.) and can be found in the SI.

Technologies were combined into separate bundles to assess the importance of individual tactics and quantify tradeoffs. The water management strategies are as follows:

A. All technologies without acreage targets
B. All technologies with acreage targets.
C. All technologies besides xeriscaping.
D. All technologies besides toilet upgrades.
E. All technologies besides irrigation technology.
F. All technologies besides supply efficiency improvements.
G. No action.

Future water plan goals of several water scarce regions in the Western United States attempt to mitigate the decline of irrigated agriculture. Specifically, the Colorado Water Plan outlines several water policies and market institutions to decrease the removal of irrigated cropland in order to sustain rural communities (Colorado Water Conservation Board 2016). A market institution scenario (Scenario B) was created to test the effects of such acreage targets. We examine the welfare gains of restricting acreage removal and assess its viability in semiarid regions. All water management strategies besides Scenario B evaluate the adoption of technology without acreage goals. Conversely, Scenario B creates a best-case scenario for rural communities by requiring the total acreage in production to be at least 94% of irrigated acreage in 1980. The 94% bound represents the largest amount of acreage that can be kept in production through 2050 despite municipal growth. After 2050 this constraint is reduced to 70% of cropland in 1980 to ensure model feasibility. Scenarios C-F do not enforce minimum acreage requirements and represent the impact of removing selected technologies to allow the exploration of specific tradeoffs. All scenarios include buy-and-dry requirements in which a producer must retire lands associated with water rights sales (as is required by local water transfer laws). Scenario G represents the worst-case scenario by disallowing any selection of technologies, requiring all increased water demand from M&I users to be met by agricultural water purchases.

2.2.4 Water Management Targets and Tradeoffs

Success metrics were collected to evaluate the effectiveness of technology adoption for each agent. Irrigated acreage, aggregated both spatially and temporally, was considered a key metric for producers. The net-present value (NPV) of agricultural profit from production (over a 40-year planning period with a 3% discount rate) simulates the strength of the rural economy assuming economic linkages remain intact over the modeling period. Going forward, the NPV of agricultural profit from production will be referred to simply as agricultural profit. M&I success metrics are based solely on the costs of satisfying urban water demand and the average water price in the river basin. M&I costs are calculated as the aggregate cost of investing in technologies and water rights purchases. These costs are transformed into costs per new population in the SPRB, as it is assumed new utility users (those who create the increase in demand) will pay for either technology adoption or water acquisition. The average water price is the spatial average across the SPRB during the final planning period. A water price which is lower is assumed to be beneficial for M&I agents as it will mean future water acquisition will be less expensive. While higher water prices may result in increased water sales revenue for producers, it is assumed higher prices would be harmful to the rural economy if these profits are not reinvested locally. Welfare is defined as the total effect a policy has on agricultural and municipal indicators. Higher amounts of irrigated acreage and agricultural profit and lower values of M&I cost and water prices would result in better welfare for the SPRB. All cost, profit, and price metrics are relative to 2010 dollars.

2.2.5 Institutional Tradeoffs

Policies regarding water resources often aim to lessen the impacts of water scarcity in semiarid regions. Goals to decrease water scarcity through education, conservation, water use restrictions, alternative water institutions, environmental regulations, and additional supply construction often begin through water-related policies. With CSaws and the Multiobjective Evolutionary Algorithm[2] framework (MOEA) we can quantitatively test the impacts of water-related institutions provided by state water plans on water management scenarios. The resulting impacts on these scenarios provide better information for water utilities and policymakers, hence influencing their decisions and goals for future water management in semiarid regions. All institutional settings include the opportunity for investments in new reservoir developments, agricultural irrigation technology, xeriscaping, and toilet upgrades. In these scenarios policymakers create targets for future water supply options and agents choose their optimal level of participation. Policies are compared against two selected indicators: municipal water acquisition cost and rural expenditures, both in 2050. Rural expenditures are calculated as the sum of crop production profit (excluding the costs of land and water) and water sales revenue. We assume water sales revenues are reinvested into the local rural economy for the MOEA scenarios. Model objective functions, parameterizations, and constraints are equivalent to those of Dozier et. al (Dozier et al. n.d.).

[2] Available at: http://moeaframework.org/

Two institutional settings were compared to the baseline (i.e. no institutional setting): i) reducing municipal raw water requirements imposed on land developers, and ii) removing buy-and-dry practices. Numerous semiarid water plans advocate for the reduction of urban water demands between 17-30% (Albuquerque Bernalillo County Water Utility Authority 2016; California Department of Water Resources 2010; Colorado Water Conservation Board 2016; SLCDPU 2009). To simulate the effects of urban water demand reductions the raw-water requirement for land development in the SPRB was decreased by 20%, comparable to Colorado Water Plan goals. Raw-water requirements are compulsory to land developers in the SPRB and are equated as the amount of water a land developer must secure per acre of developed land. Reducing these requirements by 20% is effectively the same as reducing urban water usage by 20%. Removing buy-and-dry practices incentivizes farmers to participate in deficit irrigation and dryland farming. This flexible approach to current market institutions allow farmers to sustain cropland production alongside the opportunity to sell water rights to municipalities.

2.2.6 Water Rights Model

A hydroeconomic model was created to represent a water rights market for the South Platte River Basin (SPRB) of Northeastern Colorado. The hydroeconomic model acts as a water allocation model and allocates water to buyers and sellers accordingly. The hydroeconomic model is applicable to semi-arid regions experiencing water scarcity and can operate under several economic and political institutions. In this case, the hydroeconomic model employs the Prior Appropriation Doctrine according to Colorado water law. The SPRB is split into five separate sub-regions at a multi-county scale. A single municipality and four agricultural producers are present in each of the respective sub-regions. Each producers' goal is to maximize the net-present value (NPV) of profit through the production and sale of crops or water rights. Producers represent the four most prevalent crops in the SPRB: alfalfa, corn, sugar beets, and wheat. Additionally, producers have the option to purchase upgrades to irrigation technology and additional supply reserves. Municipalities minimize the total cost of purchasing water rights, investing in conservation measures, or constructing new supply reservoirs to sustain increases in population growth. As such, the model is driven by the differing objectives between the producers and municipalities. Additional drivers of the model including climatic forcings, population growth models, and land-use change forecasts will be discussed in subsequent sections.

1.9.2 Mathematical characterization of the water rights model

The hydroeconomic model is solved through a set of non-linear equations formulated as a mixed-complementarity problem and solved using the General Algebraic Modeling System (GAMS Development Corporation 2013). The partial equilibrium model solves a water rights market in a set of r sub-regions for p producers and m municipalities. The solution methodology is similar to the approach used by Britz et. al. (Britz et al. 2013) to solve Multiple Optimization Problems with Equilibrium Constraints.

The goal of each municipality is to minimize the cost of water and technology purchases driven by increases in population and subsequent water demand. Population growth is translated into raw water purchase requirements (RWR), formulated and calculated by municipalities in the SPRB. A raw water purchase requirement is the amount of water (in acre-feet) land developers must purchase per acre of developed land. The municipal objective function is defined as:

$$\min_{B_{m,r,d}, X^{\text{stor}}_{m,r,d}, X^{\text{tlt}}_{m,r}, X^{\text{xeri}}_{m,r}} C_{m,r}$$

$$= \sum_{d \in D} \left[P^{\text{W}}_d \cdot B_{m,r,d} + c^{\text{tran}}_{r,d} \cdot B_{m,r,d} + [d = r] \cdot \frac{b^{\text{tran}}}{\sum_{p,r} u^{\text{endow}}_{p,r,d}} \cdot B^2_{m,r,d} \right. \quad (1)$$

$$\left. + \left(c^{\text{stor}}_d + c^{\text{tran}}_{r,d}\right) \cdot X^{\text{stor}}_{m,r,d} \right] + \left(c^{\text{tlt}} \cdot X^{\text{tlt}}_{m,r}\right) + \left(c^{\text{xeri}}_{m,r} \cdot X^{\text{xeri}}_{m,r}\right)$$

$$B_{m,r,d}, X^{\text{stor}}_{m,r,d}, X^{\text{tlt}}_{m,r}, X^{\text{xeri}}_{m,r} \geq 0$$

where $C_{m,r}$ is the cost to municipality m in sub-region r. Municipalities can purchase water rights $B_{m,r,d}$ at price P^W_d, additional reservoir storage $X^{\text{stor}}_{m,r,d}$ at price c^{stor}_d, outdoor conservation in the form of xeriscaping $X^{\text{xeri}}_{m,r}$ at price $c^{\text{xeri}}_{m,r}$, and indoor conservation in the form of toilet renovations $X^{\text{tlt}}_{m,r}$ at price c^{tlt}. The demand-saving methods of outdoor and indoor conservation help to decrease the reliance on water rights transfers from agricultural sector.

An imperfect water market is modeled through the inclusion of transaction costs $c^{tran}_{r,d}$ which represent both physical and legal conveyance costs associated with water purchases. Transaction costs $c^{tran}_{r,d}$ are calibrated by b^{tran} to match past literature, historical observations, and expert interviews (Bauman et al. 2015). Transaction costs generally increase with the distance between sub-region r and ditch company d as more physical infrastructure is needed to convey the newly purchased water. To account for the simplified spatial representation of agents within the water market model, a quadratic term ($\frac{b^{\text{tran}}}{\sum_{p,r} u^{\text{endow}}_{p,r,d}} \cdot B^2_{m,r,d} \cdot [d = r]$) is included. This quadratic term only applies when an agent purchases water within the same sub-region ($[d = r]$) and is scaled by the total water right endowment of agricultural producers $u^{\text{endow}}_{p,r,d}$. This term simulates how an agent explores nearby supplies first until large purchases in other sub-regions become cheaper due to economies of scale. Further, the inclusion of the quadratic term ensures an accurate representation of historical water prices from 1980-2010. The calibration and parameterization methods for transaction costs $c_{r,d}^{tran}$ and b^{tran} are further described in Appendix D of Dozier et. al. (Dozier et al. 2017).

Land developers are required to purchase a specific quantity of firm water q_r^{rwr} (in acre-feet) per acre of developed land $a_{m,r,t}^{devel}$ for municipality m in sub-region r and future planning period t. Firm water is subject to a reliability factor k_d^{rel} which is estimated as the diversion volume with a 75-year return period divided by the average annual diversion volume $\bar{q}$. Municipalities must optimize the above objective function subject to the following land development constraint:

$$\sum_{d\in D}\left[u_{m,r,d}^{endow} + k_d^{rel} \cdot B_{m,r,d} + X_{m,r,d}^{stor}\right] - \sum_{d\in D}\left(u_{m,r,d}^{endow}\right) \cdot \left(1 - k^{tlt} \cdot X_{m,r}^{tlt}\right) \cdot \left(1 - k_{m,r}^{irr} \cdot X_{m,r}^{xeri}\right) - q_r^{rwr} \cdot a_{m,r,t}^{devel} \geq 0$$

The historical, or endowed, ownership of water $u_{m,r,d}^{endow}$ can be reduced by a fraction of total water use from toilets k^{tlt} and outdoor irrigation $k_{m,r}^{irr}$. This effectively captures historically used water by municipality m in sub-region r. The above constraint forces municipalities to purchase water from agricultural producers or construct reservoir storage (1st term) and/or conserve water to capture historically used demand (2nd term) to balance the raw water requirements for newly developed land due to population growth (3rd term). The inequality ensures there is always enough water for new populations.

The Northern Colorado Water Conservancy District (NCWCD) restricts the speed of Colorado-Big Thompson (C-BT) water purchases from municipalities, and is represented though the following constraint:

$$g_{m,r}^{rwr} = 2 \cdot \left(q_r^{rwr} \cdot a_{m,r,t}^{devel} - \sum_{d \in D_n} k_d^{rel} \cdot B_{m,r,d} \right) - B_{m,r,cbt} \geq 0$$

Where water rights purchases from native ditch companies D_n are distinguished from C-BT purchases $B_{m,r,cbt}$ (i.e. water rights purchases from non-native ditch companies). The set of six ditch companies to buy water rights D consists of five native water rights companies $d \in D_n$ and one nonnative water rights company $d = cbt$. In accordance with NCWCD policy, the above constraint slows municipal C-BT water purchases at an upper bound of 2 times their average annual demand ($q_r^{rwr} \cdot a_{m,r,t}^{devel}$) minus the average firm annual yield of other water purchases (Northern Colorado Water Conservancy District 2015b).

A constraint on the total amount of municipally-owned water is added to the hydroeconomic model to allow for policy interventions. For each model run there is no municipal cap, though this parameter k_d^{mcap} can be used to enforce and test water policy. The only municipal cap which exists in conservation and baseline runs is a C-BT cap at 0.8, as is enforced by NCWCD. The municipal cap constraint is as follows:

$$g_d^{CAP} = k_d^{mcap} \cdot \sum_{p \in P, r \in R} \left(u_{p,r,d}^{endow} + B_{p,r,d} - S_{p,r,d} \right) - \left(1 - k_d^{mcap} \right) \cdot \sum_{m \in M, r \in R} \left(u_{m,r,d}^{endow} + B_{m,r,d} \right) \geq 0$$

Where P is the set of all producers, R is the set of all sub-regions, and M is the set of all municipalities.

The goal of each prouder p in sub-region r is to maximize the NPV of profit over the planning period, and is defined through the following objective function:

$$\max_{\mathrm{V_{p,r}}, \mathrm{A_{p,r}}, S_{p,r,d}, B_{p,r,d}, X_{p,r,d}^{\mathrm{stor}}, \eta_{p,r}^{\mathrm{a}}} \pi_{p,r} =$$

$$k^{\mathrm{NPV}}\left[p^{\mathrm{crop}} \cdot f_{p,r}\left(V_{p,r}, A_{p,r}\right) - C_{p,r}^{\mathrm{W}}\left(V_{p,r}, \eta_{p,r}^{\mathrm{a}}\right) - C_{p,r}^{\mathrm{L}}\left(A_{p,r}\right) - \left(c_{p,r}^{\mathrm{eff}} - b_{p,r}^{\mathrm{eff}}\right) \cdot \eta_{p,r}^{\mathrm{a}}\right]$$

$$+ \sum_{d \in D}\left[P_d^{\mathrm{W}} \cdot S_{p,r,d} - P_d^{\mathrm{W}} \cdot B_{p,r,d} + c_{r,d}^{\mathrm{tran}} \cdot B_{p,r,d} + \frac{b_{r,d}^{\mathrm{tran}}}{\sum_{p,r} u_{p,r,d}^{\mathrm{endow}}} \cdot B_{p,r,d}^2 \cdot [d = r]\right]$$

$$-\left(c_d^{\mathrm{stor}} + c_{r,d}^{\mathrm{tran}}\right) \cdot X_{p,r,d}^{\mathrm{stor}}$$

$$\mathrm{V_{p,r}}, \mathrm{A_{p,r}}, S_{p,r,d}, B_{p,r,d}, X_{p,r,d}^{\mathrm{stor}}, \eta_{p,r}^{\mathrm{a}} \geq 0$$

Where $V_{p,r}$ is the amount of irrigation volume (AF), $A_{p,r}$ is the amount of acreage in production, $S_{p,r,d}$ is the amount of water sold, $B_{p,r,d}$ is the amount of water bought, $X_{p,r,d}^{stor}$ the amount of water from reservoir purchases, and $\eta_{p,r}^{a}$ the fractional amount of improved application efficiency from purchases of more efficiency irrigation technology for each producer p in sub-region r. The planning period factor k^{NPV} represents a 40-year planning period with a constant 3% discount rate. An exogenous crop price p^{crop} is included using data from the National Agricultural Statistics Service. Production costs of water $C_{p,r}^{W}$ and land $C_{p,r}^{L}$ are described below. The cost of application efficiency improvements $c_{p,r}^{eff}$ is calibrated towards the true cost ($c_{p,r}^{eff} - b_{p,r}^{eff}$) of improvements through a calibrated parameter $b_{p,r}^{eff}$ which sets the marginal profit to zero for historically observed levels of irrigation efficiency. During each model run each producer in each sub-region chooses their amount of acreage in production, volume of irrigation, amount of water to buy and sell, volume of reservoir storage to purchase, and irrigation technology investments to improve application efficiency.

The crop production function $f_{p,r}$ (in tons of production per year), cost of irrigation $C_{p,r}^{W}$, and cost of farmland $C_{p,r}^{L}$ are defined as:

$$f_{p,r}\left(V_{p,r}, A_{p,r}\right) = k_p \cdot p_{0_{p,r}} \cdot \left[p_{1_{p,r}} \cdot A_{p,r}^{p_{2_{p,r}}} + \left(1 - p_{1_{p,r}}\right) \cdot V_{p,r}^{p_{2_{p,r}}}\right]^{\frac{p_{3_{p,r}}}{p_{2_{p,r}}}}$$

$$C_{p,r}^{W}(V_{p,r},\eta_{p,r}^{a}) = (c_{p,r}^{W} - b_{p,r}^{W}) \cdot V_{p,r} \cdot (\hat{\eta}_r^a - \eta_{p,r}^{a})$$

$$C_{p,r}^{L}(A_{p,r}) = (c_{p,r}^{L} - b_{p,r}^{L}) \cdot A_{p,r}$$

$$\sum_{d\in D}(u_{p,r,d}^{\text{endow}} + B_{p,r,d} - S_{p,r,d} + X_{p,r,d}^{\text{stor}}) - \frac{V_{p,r}}{\eta_r^{\text{c}}} \cdot (\hat{\eta}_r^{\text{a}} - \eta_{p,r}^{\text{a}}) \geq 0$$

Where $p_{i_{p,r}} \forall\, i \in \{0,1,2,3\}$ are the parameters of a constant elasticity function for each producer p in sub-region r. The production function for each producer is calibrated to the output of an agro-ecosystem model that simulates irrigated and dry-land crop production (Parton et al. 1998; Zhang 2016). Cost of water $c_{p,r}^{water}$ ($\$/AF$) and land $c_{p,r}^{land}$ ($\$/AF$) were estimated from Colorado State University Extension reports. Parameters $b_{p,r}^{water}$ ($\$/AF$) and $b_{p,r}^{land}$ ($\$/AF$) represent calibrated "intrinsic benefits" to match the historical water diversions and land in production for agriculture, and are similar to $b_{p,r}^{eff}$. The factor k_p converts crop production from dry mass to wet mass to match output price data in $\$\backslash ton$. The term $V_{p,r} \cdot (\hat{\eta}_r^a - \eta_{p,r}^a)$ represents the total amount of water used on planted acres $A_{p,r}$, where $\eta_{p,r}^a$is the improved application efficiency through irrigation technology investments and $\hat{\eta}_r^a$ is the historical application efficiency. Application efficiencies and efficiency improvements are characterized in the model as values greater than 1. The amount of water diverted from the stream before conveyance $\frac{V_{p,r}}{\eta_r^c} \cdot (\hat{\eta}_r^a - \eta_{p,r}^a)$ must always be less than the amount of water owned after the market clears $\sum_{d\in D}(u_{p,r,d}^{endow} + B_{p,r,d} - S_{p,r,d} + X_{p,r,d}^{stor})$ (i.e. the buying and selling of time period t is finished), where η_r^c represents the channel and evaporation losses of conveyance. The inequality in the last term ensures water balance constraints are met. This final constraint ensures producers do not use more water than they currently own. Modeled crop production functions as a function of irrigation volume (AF) were created for each producer in each sub-region using their maximum endowed amount of acreage (Figure 3). A linearly-spaced vector of irrigation volume was created by using each producers' minimum and maximum values of irrigation depth (ft.). These values were transformed into irrigation volumes by multiplying irrigation depths with their respective maximum acreage values. Illustrated lines represent the values of producer crop production functions, calculated by $f_{p,r}\left(V_{p,r}, A_{p,r}\right)$.

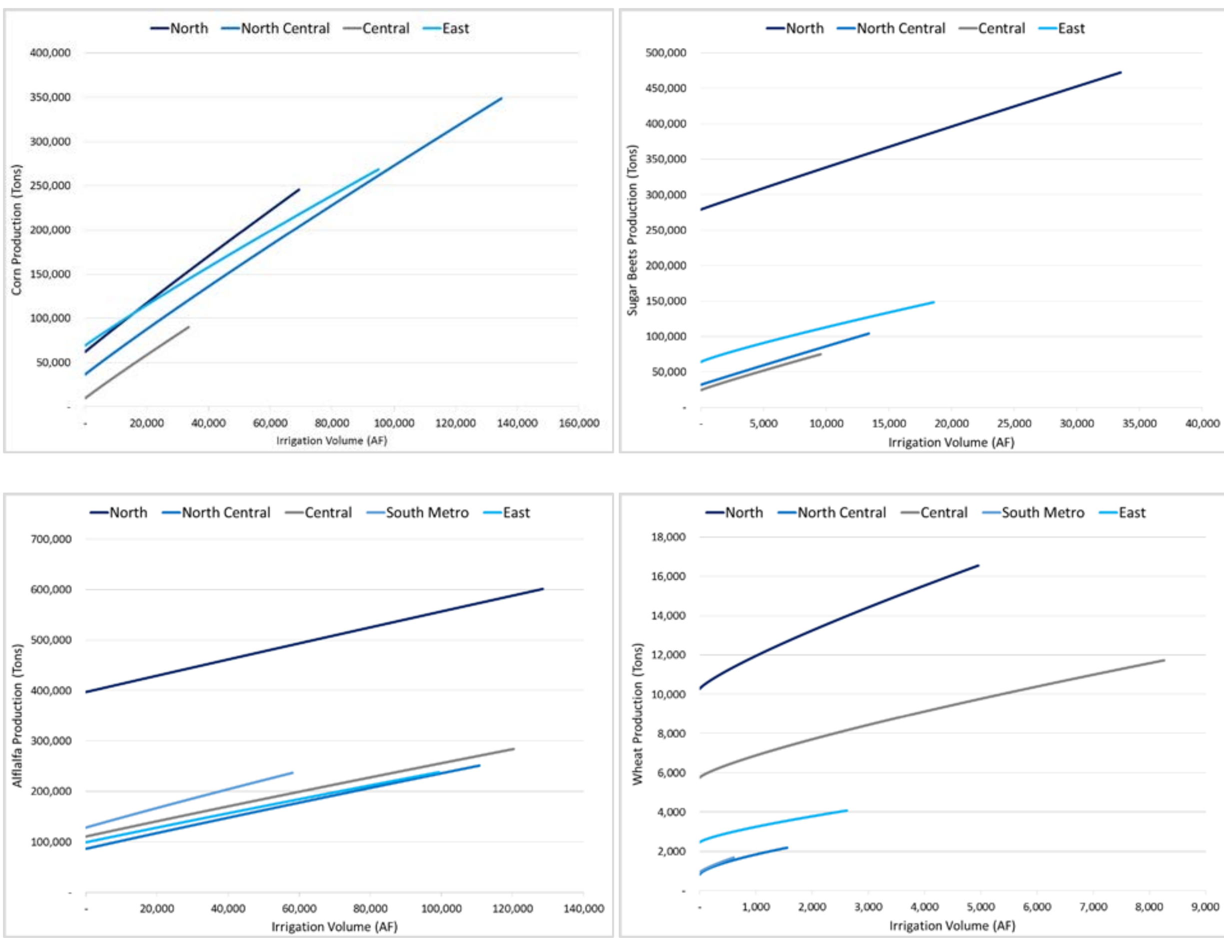


Figure 3 – Modeled crop production functions for A) Corn (Top Left), B) Sugar Beets (Top Right), C) Wheat (Bottom Left), and D) Alfalfa (Bottom Right).

To simulate current water market institutions in Colorado a buy-and-dry constraint is included in the hydroeconomic model. This constraint forces agricultural producers to retire their land ($a_{p,r} - A_{p,r}$) in combination with their consumptive use of water ($v_{p,r} - V_{p,r}$):

$$g_{p,r}^{B\&D} = k^{B\&D} \cdot \left(v_{p,r} - V_{p,r}\right) - \left(a_{p,r} - A_{p,r}\right) \cdot I_{p,r}^{NIR} = 0 \; (or \geq 0 \; for \; different \; policies)$$

$$V_{p,r}, A_{p,r}, B_{p,r,d}, S_{p,r,d} \geq 0$$

Where $I_{p,r}^{NIR}$ represents the annual average net irrigation requirement of crop p that a producer in sub-region r must irrigate to meet the water demand of the crop after accounting for effective precipitation. Upper bounds for acreage $a_{p,r}$ and irrigation volume $v_{p,r}$ are set at 1980 levels. The buy-and-dry factor $k^{B\&D}$ simulates current Colorado water market institutions by forcing producers to retire their land alongside the sale of water rights. The buy-and-dry factor can be used to simulate other water institution scenarios and irrigation techniques such as deficit irrigation, rotational fallowing, lease fallowing, alternative transfer methods, and other crop-choice and institutional scenarios.

A market clearing or market equilibrium constraint must be met in order to balance the total amount of sold and bought water among the independent agents. This forces the market to "clear" and the appropriate amount of water to be sold and bought among agents, which yields the optimal solution at each time scale. The constraint is as follows:

$$\sum_{p \in P, r \in R} S_{p,r,d} - \sum_{m \in M, r \in R} B_{m,r,d} = 0$$

Where $S_{p,r,d}$ and $B_{m,r,d}$ represent the amount of water sold by producer p in sub-region r to municipality m in sub-region r.

All technologies are subject to the following constraints which ensure new supplies or technologies are not adopted when not available. For example, the second constraint ensures a producer cannot improve their application efficiency beyond the limitations of the modeled technology. The constraints are as follows:

$$0 \le X_d^{\mathrm{stor}} \le X_d^{\mathrm{stor,max}}$$
$$0 \le \eta^{\mathrm{a}} \le \eta^{\mathrm{a,max}}$$
$$0 \le X^{\mathrm{tlt}} \le X^{\mathrm{tlt,max}}$$
$$0 \le X^{\mathrm{xeri}} \le X^{\mathrm{xeri,max}}$$

Where $X_d^{\mathrm{stor,max}}$, $\eta^{\mathrm{a,max}}$, $X^{\mathrm{tlt,max}}$, and $X^{\mathrm{xeri,max}}$ are the maximum bounds on reservoir storage purchases, irrigation technology improvements, toilet upgrades, and xeriscaping.

2.2.7 Water Appropriation Institutions

Complex water rights institutions define the water management regime in the South Platte River Basin. Water rights are governed by the prior appropriation doctrine in Colorado, which shares similarities to many water institutions in the Western U.S. (Squillace 2013). The prior appropriation doctrine in Colorado defines water rights by a “first in time, first in right” system which water rights holders obtain their right by using water for a beneficial use (Benson 2012). Water users who obtained their right first hold precedence over those who obtained one subsequently and water is allocated according to these ranks. Risk-averse municipalities seek “firm yield” water, or senior and valuable water with high likelihood of delivery. M&I users who purchase agricultural water rights incur transaction costs for legal processes and conveyance, thus hindering the market by driving up the price of water for participating agents. In order to avoid injury to other water rights holders, as is required by water law in Colorado, historically irrigated acreage is often permanently dried alongside water purchases, a term named “buy-and-dry”. Numerous water institution scenarios can be included in the hydroeconomic model to test various tradeoffs, many of which were tested in the previous work by Dozier et. al. (Dozier et al. n.d., 2017).

Agricultural producers are represented by aggregating irrigated land in the SPRB. Every irrigated parcel in the SPRB is mapped by the Colorado Division of Water Resources (CDWR) alongside the respective acreage, crop type, irrigation system, and other key agricultural data. The four crops characterized in the hydroeconomic model represent around 97% of all irrigated cropland. Using county-level data for irrigated croplands, producers were aggregated into the multi-county sub-regions represented in the model. That is, there are four producers of the predominate crops in each of the five sub-regions of the hydroeconomic model. A subsequent section detailing the parameterization of producer crop production functions follows for more detail.

Municipalities in the SPRB were modeled using NLCD land-use data alongside Census data from 2010 for households and population metrics. Using the sub-regions created in the hydroeconomic model, a single municipality was represented in each sub-region by aggregating the total amount of developed land, population, and households for each county within the respective sub-region. Developed land was categorized as the total amount of open, low density, medium density, and high density land-use categories from NLCD data. Municipal water supplies, water demands, and other economic drivers can be found in subsequent sections.

2.2.8 Water Supply Model

Colorado water rights and diversions data are not easily linked between owners, uses, and sectors and as such an automated methodology for assigning water rights and diversions was created. All water rights and diversions data originate from the Colorado Hydrobase (Colorado

Division of Water Resources n.d.). Our methodology is similar to the South Platte Historic Crop Consumptive Use Analysis (Leonard Rice Engineers Inc. 2008). For each right and diversion record a “USE” code was assigned to the following uses: agricultural, environmental, industrial, municipal, and unknown. Diversion usage and yearly water rights ownership acquired through Hydrobase were queried using specific algorithms from Dozier et. al. 2017 (SI).

The Colorado-Big Thompson project is a local, share-driven market in which water transfers are provided in the form of tradable water units based on available water supplies. As a result of this trans-basin project, the C-BT system has become one of the most active water markets in the Western United States (Grafton et al. 2012). Managed by the NCWCD, the C-BT project delivers about 310,000 AF annual in additional water supplies to the SPRB and serves the North, North Central, and East sub-regions. Because the C-BT system is managed by the NCWCD no legal transaction costs are assigned to water purchases. The NCWCD limits municipal ownership of C-BT shares at 80% of total shares and is used throughout all scenarios in this study.

Water supply inputs into the hydroeconomic model are considered as static endowments for individual agents in each sub-region. Constant endowments are used to drive water supplies because water rights native within the SPRB have reached a plateau in which no new water rights have been created (Dozier et al. 2017). Further, trans-basin (C-BT) water transfers have become the predominant supply of new water rights.

The sum of direct flow rights with annual storage rights to calculate endowments is not feasible due to the model solving at an annual time scale (Figure 4). As such, annual historical diversions are used to calculated endowments (Figure 5). Endowments for agricultural producers are calculated as the historical average annual diversion from 1981–2014. Producer endowments are scaled by individual crops within each sub-region where each crop receives a portion of the total endowment, weighted by total irrigated land for each individual crop in 1980. M&I endowments are set to equal the same proportion of total endowment as water rights. That is, the portion of M&I ownership of total water rights is used to calculate their endowment as a fraction of the producer endowment (M&I ownership fraction times producer endowment in the same sub-region) divided by the un-owned portion of water rights (one minus M&I ownership fraction of total water rights).

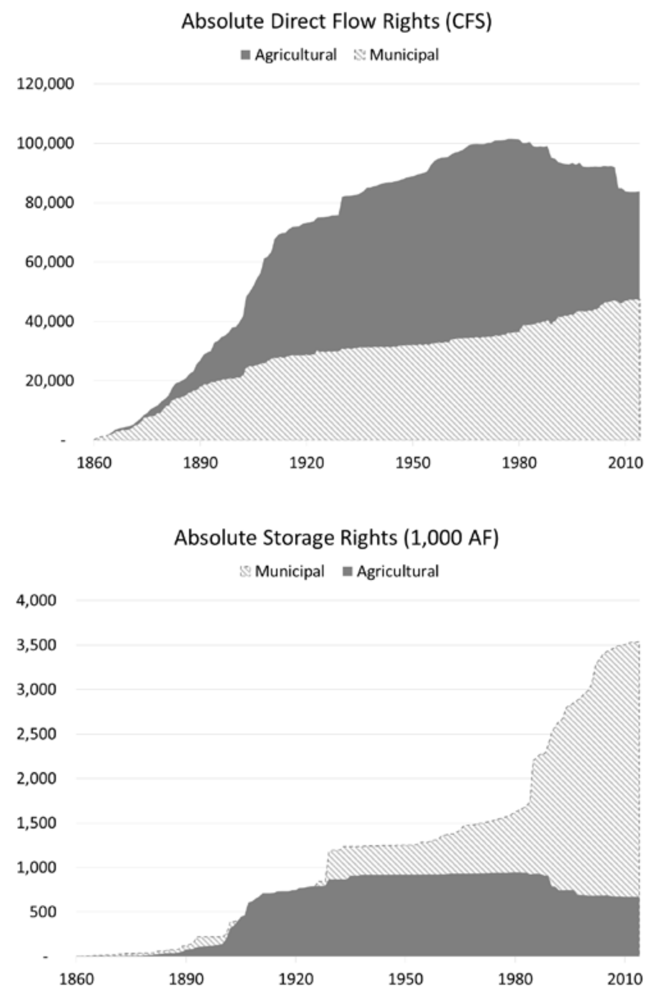


Figure 4 – Absolute direct flow (cfs) and storage (AF) rights for municipal and agricultural sectors in the SPRB.

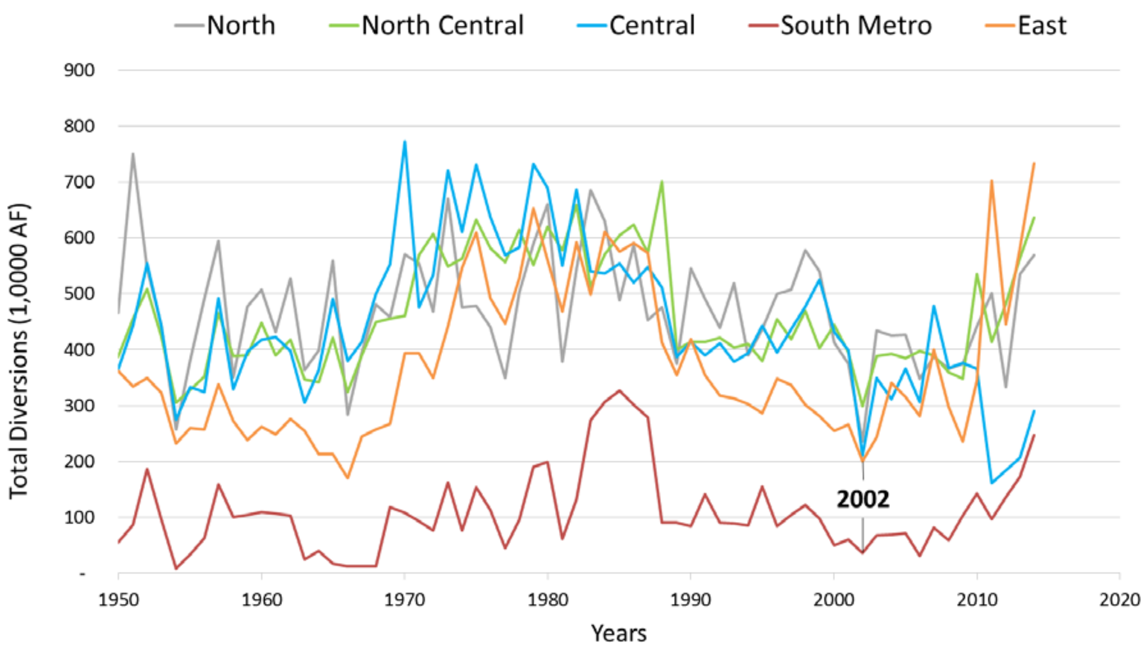


Figure 5 – Total diversion amounts in the South Platte River Basin. Diversions include industrial, agricultural, municipal, and environmental sectors.

C-BT endowments are calculated based on fractional ownership of the total amount of C-BT shares available of 310,000 AF. The C-BT project maintains extensive records of ownership by sector and ownership fractions are easily calculated. Agricultural producers represent around 34% of total C-BT shares (105,400 AF). Agricultural endowments of C-BT shares are weighted by sub-region crop fractions against the total amount of crop fractions from sub-regions which the C-BT serves. Municipal ownership of C-BT shares represents the remaining 204,600 AF and is weighted the same as agricultural C-BT shares.

Water purchases in the SPRB are subject to hydrological and climatic uncertainty. As such, ditch companies whom provide water may not be able to deliver the full amount allocated to each agent. To account for uncertainty in water deliveries a reliability factor for each ditch company in the hydroeconomic model was assigned. Reliability fractions for each sub-region were calculated as the diversion volume at a 75-year return period (q_{75}) divided by the average annual diversion volume ($\bar{q}$). Hydrobase diversion data between 1981-2014 were used to estimate both q_{75} and $\bar{q}$. Interestingly, the 75-year return period of historical diversions often corresponds with the severe 2002 drought in Colorado. The C-BT reliability factor was calculated from the average annual yield of C-BT deliveries, estimated at 70%. These estimated reliability measures are the fractional amount of water a municipality can expect to be delivered on any given year. Reliability factors for each sub-region are estimated in Table 1. Empirical and normal cumulative distribution functions of total historical diversion volumes for each sub-region are shown in Figure 6. Blue circles represent the median total historical diversion which very closely resembles $\bar{q}$. Green squares show the total diversion amount of a 75-year return period, q_{75}. As discussed previously, the reliability factor created for each sub-region is $\bar{q}/q_{75}$. Figure 6 closely illustrates these values of reliability by the ratio of the blue circle and green square for each sub-region. Generally, total historical diversion volumes follow a normal probability distribution function. Kolmogorov-Smirnoff normal probability distribution tests for each sub-region pass a confidence level of $\alpha = 10\%$, with the North, North Central and Central sub-regions passing at a confidence level of $\alpha = 5\%$.

Table 1 – Water supply reliability (delivery) factors for each sub-region in the SPRB.

Sub-region	**Reliability (k^{rel})**
North	0.50
North Central	0.63
Central	0.37
South Metro	0.48
East	0.71
C-BT	0.70

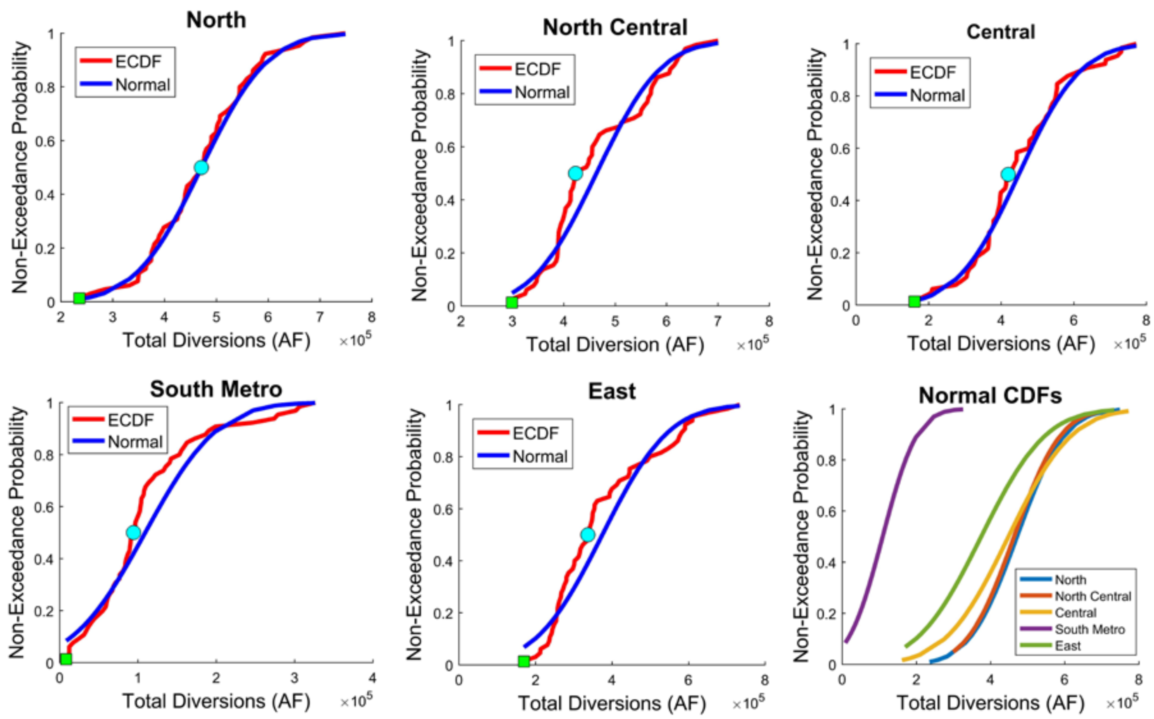


Figure 6 – Empirical (red line) and normal (blue line) cumulative distribution functions for all modeled sub-regions in the SPRB in regards to total historical diversion amounts (AF). Blue circles represent the median diversion and green squares represent the diversion volume with a 75-year return period.

Transaction costs have a significant impact on the price and opportunity costs of water and can create barriers to entry for water purchases. As such, transaction costs are primarily a function of legal costs, infrastructure (conveyance) costs, and location. Although transaction costs are imperative to modeling water rights, data remains limited. To cope with the lack of data transaction costs were calibrated to reflect expected water rights price trends from historical water purchases (WestWater Research 2016). Any water rights purchases incur a transaction costs $c_{r,d}^{tran}$ to reflect both legal and physical infrastructure costs from sub-region r in pool d. All transaction cost calibrations include scaling for inflation.

Water court fees were assumed to be \$4,000 for purchases within each sub-region r and an additional $0.25 \cdot 4000x_{r,d}$ for each pool d at a distance $x_{r,d}$ from sub-region r. The additional costs of buying water outside a sub-region are derived from a local interview with a water management expert[3]. Infrastructure costs $c_{r,d}^{infra}$ were calculated using the following formula:

$$c_{r,d}^{infra} = c^{infra} \cdot \left(1 + b^{infra} \cdot x_{r,d} \cdot \left[x_{r,d} > 0\right]\right)$$

where $c^{infra} = \$10{,}000$ is the calibrated cost of infrastructure for all within-sub-region purchases. The calibrated additional cost for infrastructure from a lower elevation water purchase $b^{infra} = \$12{,}500$ and was corroborated with local interviews and news sources[1,4]. Distance parameter $x_{r,d}$ was used as the calibration parameter to calibrate total transaction costs $c_{r,d}^{tran}$ according to the methodology found in Appendix D of Dozier et. al. 2017. The quadratic cost term b^{tran} was calibrated to slow the amount of water purchases within a sub-region to the modeled water price to historical water price levels. The cost term simulates an agent exploring additional water purchase options within the same sub-region to account for the heterogeneity of purchases within that sub-region. The quadratic cost term b^{tran} is calibrated through the following formula when ditch d is equal to sub-region r:

$$b^{tran} = \frac{k^{tran}}{\prod_{t=1980}^{2013} k_t^{CPI}} \left(b^{legal} + b^{infra}\right)$$

where $b^{infra} = \$12{,}500$ and $b^{legal} = \$1{,}000$. The calibration factor k^{tran} scales the costs of buying the last remaining water rights within a sub-region to be more than the cost of buying water rights from a nearby sub-region and is equal to 1.5. The consumer price index factor k_t^{CPI} accounts for inflation.

[3] Personal communication with Kelly DiNatale of DiNatale Water Consultants in Boulder, CO.
[4] Duggan, K. "Thornton plans pipeline to tap into Poudre water," *Coloradoan*, Oct. 19, 2015. Available on http://www.coloradoan.com/story/news/2015/10/19/thornton-plans-pipeline-poudre-river-water/74247032/.

2.2.9 Water Demand Model

Individual crop production functions were parameterized for producers within the SPRB with a locally-calibrated version of DayCent, an agro-ecosystem model. This version of DayCent was calibrated for semi-arid regions and crops experiencing deficit irrigation at the county level (Dozier et al. 2017; Parton et al. 1998). Input parameters were obtained from the calibrated DayCent model, though an individual calibration of radiation use efficiency for total biomass was adjusted to match historical yields for producers in the Northern sub-region. These calibrated parameters were then applied for all producers in all sub-regions. Crop production functions were created for each producer in each sub-region using constant elasticity of substitution (CES) functions. Yearly water budget and production outputs from DayCent, alongside acreage, were combined to create CES production functions from annual DayCent outputs. Input parameters of the CES production functions were fit to crop production curves for the four dominant crops in the SPRB: alfalfa, corn, sugar beets, and wheat. Appendix D of Dozier et. al. 2017 illustrates these CES production functions for each producer in each region.

Important inputs into the DayCent model include soil properties, management strategies for crop production (i.e. irrigation and tillage), and daily maximum or minimum temperature and precipitation data. Soil properties were included from SSURGO and soil hydraulic properties estimated from the pedotransfer function (Saxton et al. 1986; SSURGO 2011). Past weather data from 1981-2014 was used to inform the analysis and was extracted from 4 km by 4 km gridded PRISM data (PRISM Climate Group 2004). Each combination of soil, 32 km climate region (defined by the North American Regional Reanalysis (NARR) grid cells), and crop type was simulated assuming they respond hydrologically similar to one another as hydrologic response units (HRUs). These HRU's reduce the number of spatial units from 25,200 to 4,500 (Mesinger et al. 2006). All data was stored in MongoDB for reproducibility[5].

Typical crop management practices such as automatic fertilization to eliminate nutrient stress and tillage were assumed in order to create crop production functions. Within each HRU, values of the net irrigation requirement (NIR) and maximum allowable depletion (MAD) were simulated for each crop to quantify yield changes in response to deficit irrigation. NIR and MAD values were applied to non-critical and critical irrigation periods during the reproductive period of the crop to estimate optimal irrigation strategies. Output yield of enumerated levels of NIR and MAD were excluded from crop production functions assuming farmers act optimally during the critical growing period.

Crop production function outputs of dry biomass were converted to wet biomass, both in units of tons. Assumptions for the conversion of dry biomass to wet biomass and volumetric bushels (for corn and wheat) to tons are found in Appendix D of Dozier et. al. Crop prices for each decadal model run were scaled to 1980 dollars to remain consistent with other cost metrics. Crop prices were exogenously updated each decade from a simple linear regression to match historical price changes using data from NASS. After accounting for inflation, only alfalfa was updated to include crop price increases due to other crop prices (i.e. corn, sugar beets, and wheat) decreasing over time. Historical crop yields have improved over time for all modeled crops besides alfalfa. To account for increases in crop yield over time, individual regression analyses of corn, wheat and sugar beets were modeled and updated during each decade. Application efficiencies were assumed at 60% for flood irrigated fields and 80% for sprinkler irrigated fields (Howell 2003; Leonard Rice Engineers Inc. 2008). Aggregated application efficiencies $\hat{\eta}_r^a$ were

[5] Available at: https://www.mongodb.com/.

estimated from DayCent output for each sub-region by taking the quotient between the regional NIR and the regional gross irrigation requirement. Yearly average depth of net irrigation requirement $\hat{I}^{NIR}_{p,r}$ was estimated from DayCent output. Average application efficiencies $\eta^{a}_{p,r}$ and surface water net irrigation requirements $I^{NIR}_{p,r}$ for each crop and sub-region were calibrated as:

$$\eta^{a}_{p,r} = \max\left(\hat{\eta}^{a}_{r}, \sum_{d}\left(u^{endow}_{p,r,d}\right) \cdot \frac{\eta^{c}_{p,r}}{A_{p,r} \cdot I^{NIR}_{p,r}}\right)$$

$$I^{a}_{p,r} = \max\left(\widehat{I}^{a}_{r}, \sum_{d}\left(u^{endow}_{p,r,d}\right) \cdot \frac{\eta^{c}_{p,r}}{A_{p,r} \cdot \eta^{NIR}_{p,r}}\right)$$

which ensures the plant does not receive more than the gross irrigation requirement. We assume that if the net irrigation requirement cannot be met with surface water, groundwater is pumped to meet plant growth requirements. Conveyance efficiency $\eta^{c}_{p,r}$ was assumed to be 80% based on past literature values.

Municipal and industrial (M&I) agents within the hydroeconomic model optimize their decisions in order to minimize the cost of meeting future urban water demands. As population increases, hence increasing urban land development, specific requirements must be met by land developers to secure water for the newly developed land. These requirements are known as raw water requirements and are designed as the amount of water in AF a developer must secure per acre of land developed. Consequently, these raw water requirements and increases in urban land are the driving forces behind municipal water rights acquisition. Urban land development, or new urban area $a^{devel}_{m,r,t}$, is projected from 2020-2100 by autoregressive models. These regression models were created to estimate future urban land development as a function of future population and previous land development:

$$a^{devel}_{t,std} = \alpha \cdot pop_{t,std} + \beta \cdot a^{devel}_{t-1,std} + \epsilon$$

Where $a_{t,std}^{devel}$ is the standardized historical amount of urban area in acres and $pop_{t,std}$ is the population. Both variables are estimated for the county level in the SPRB. Past land use data for 2001 and 2011 were queried from the National Land Cover Dataset and combined with county-level population estimates from the U.S. Census of years 2000 and 2010 (Homer et al. 2015). The regression model was estimated for each municipality m in sub-region r at time t for the years 1980-2100. Population and new urban area estimates are shown in Table 2 and Table 3 below. Further, differences between rural and urban populations over time are highlighted in (Figure 7) using data from the Colorado State Demographer. Projected totals are extrapolated for municipal and rural populations by using the ratio between respective population types and the total population in for the SPRB. These static values for municipal and rural populations were calculated as 77% and 23% of the total population from 1980-2016, respectively.

Table 2 – Regional estimates of population during the modeling period

Year/Region	North	North Central	Central	South Metro	East
1980	338,809	123,438	1,413,432	37,336	50,883
1990	413,174	131,985	1,577,391	78,589	46,879
2000	542,782	180,936	1,947,595	210,161	55,348
2010	650,086	252,825	2,162,836	324,757	58,061
2020	789,000	361,800	2,581,900	461,400	70,500
2030	904,800	487,800	2,879,400	564,600	83,400
2040	1,002,547	586,933	3,156,466	630,125	94,292
2050	1,110,800	655,000	3,456,500	688,100	104,200
2060	1,266,646	824,653	3,877,012	797,971	118,956
2070	1,445,504	1,038,247	4,351,513	964,950	135,926
2080	1,650,911	1,307,165	4,887,295	1,167,453	155,449
2090	1,886,963	1,645,736	5,492,682	1,413,200	177,916
2100	2,158,404	2,072,001	6,177,177	1,711,626	203,779

Table 3 – Regional estimates of new urban area (acres) during the modeling period

Year/Region	North	North Central	Central	South Metro	East
1980	103,859	92,306	262,521	96,992	129,568
1990	113,909	99,066	294,734	102,966	137,560
2000	125,440	106,316	330,784	109,784	145,959
2010	134,799	108,369	384,457	111,868	159,770
2020	151,976	117,580	439,042	122,860	165,539
2030	171,599	129,181	500,211	135,908	171,691
2040	193,434	142,787	567,809	150,436	178,203
2050	217,721	157,932	642,407	166,359	185,070
2060	245,334	176,373	726,238	184,592	192,381
2070	276,757	198,947	820,479	206,164	200,189
2080	312,546	226,706	926,461	231,773	208,550
2090	353,343	260,973	1,045,696	262,261	217,530
2100	399,891	303,415	1,179,899	298,659	227,203

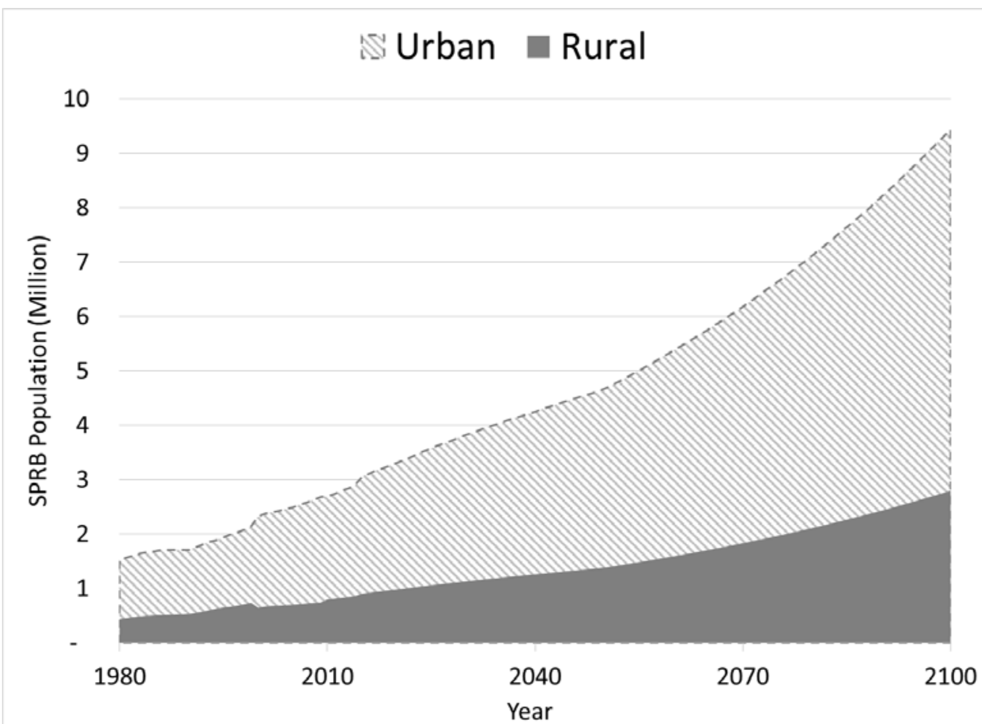


Figure 7 – Urban and rural population trends for the South Platte River Basin. All data is estimated by the Colorado State Demographer.

Raw water requirements (RWR) were estimated from Longmont and Fort Collins, two cities located in the SPRB whose RWR data is readily available. Firm raw water requirement q_r^{rwr} is calculated by multiplying the total water requirement by the corresponding reliability factor k_d^{rel} for native water rights. Resulting firm raw water requirements form Longmont, Fort Collins, and C-BT were estimated as 1 AF/acre, 1.56 AF/acre, and 1.75 AF/acre. The average of these values $q_r^{rwr} = 1.44$ AF/acre was used as a constant across all regions in the SPRB. Firm raw water requirements in combination with new urban area estimates for each decade effectively capture urban water demand and drive M&I water acquisition or technology adoption. Household demands were estimated from the Integrated Urban Water Model (IUWM) and are discussed in the following section.

2.2.10 Alternative Technologies

New water supply characterization includes the construction of two new reservoirs and the expansion of three existing reservoirs in the SPRB and are shown in Table 4. The reservoirs included in the hydroeconomic model are in various planning and construction phases. New water supplies provide agricultural and municipal agents with the option of purchasing additional water in the form of water storage rights. The newly purchased water storage rights are a function of the firm annual yield $X_d^{stor,max}$ from a reservoir (i.e. water that can be expected to be delivered each year). Investments in reservoir storage are included in the cost term c_d^{stor}. Though reservoirs add new supplies to the SPRB, scarcity is present due to annual yields being much less than projected increases in water demands. As such, water rights trading and investments in conservation will still occur. It is noted reservoir construction here is only for the purchase of new supplies and does not include benefits for flood control, drought storage, and other societal benefits of reservoirs (e.g. recreation). Data for reservoir capacity (AF), annual firm yield (AF), and total construction costs were estimated from NCWCD, CDM Smith, and Fort Collins

Utilities (City of Fort Collins Utilities 2017; Denver Water 2017; Northern Colorado Water Conservancy District 2008, 2015a; US Army Corps of Engineers 2013).

Table 4 – New supply reservoir project cost and supply information.

Sub-region	**Planned Reservoir Project**	**Annual Yield**	**Total Cost**	c_d^{stor} **($/AF)**
North	Halligan Reservoir Expansion	7,000	$ 30,000,000	$ 4,300
Central	Gross Reservoir Expansion (Moffat)	18,000	$ 380,000,000	$ 21,300
	Chatfield Reservoir Reallocation	8,500	$ 186,000,000	
C-BT	Windy Gap Firming Project	26,000	$ 223,000,000	$ 11,900
	Northern Integrated Supply Project	40,000	$ 600,000,000	

Improvements in application efficiency for agricultural producers can be gained through investments in irrigation technology. Aggregate costs of irrigation technology purchase and installation were acquired through Scherer (Scherer 2015) which include center pivot, center pivot with attachment, linear move, big gun, and wheel roll irrigation technologies. Big gun and wheel roll technologies were excluded due to their high costs and low application efficiencies. Costs of switching to efficient irrigation technology were estimated from modeled changes in demand with new irrigation technology compared to historical flood or furrow irrigation. Costs were parameterized for each producer in each region based on their historical application efficiency and transformed into cost per fractional increase in application efficiency using DayCent and gross irrigation requirements. To allow independent agent adoption of irrigation technology upgrades, application efficiencies were not exogenously updated in accordance with historical improvements to application efficiency.

Household indoor water conservation was included in the hydroeconomic model in the form of efficient toilet upgrades. That is, upgrading an inefficient toilet (assumed to be all toilets in 1980) to a highly efficient one. Using the most recent Residential End Use (REU) Study database, which includes data for 185 homes for two cities in the SPRB, we were able to estimate the cost of upgrading to efficient toilets and the maximum amount of water savings per household (DeOreo et al. 2016). The REU study includes survey information for each household on the number of toilets, flush volume of toilets, household size, and type of toilet. The average cost per household to upgrade to efficient toilets was found through the average cost per efficient toilet and the number of toilets in each household (HomeAdvisor 2016). The average potential water savings per homes were estimated through the difference in the average volume of toilet water use for inefficient homes to the average volume of toilet water use for efficient homes (those with an efficient toilet). This resulted in an upper bound in toilet water savings at about 35% of toilet water use. That is, households can expect to save about 35% of their current toilet water use by upgrading to efficient toilets. Indoor household toilet water use was estimated from the Integrated Urban Water Model (IUWM) at the block-group level and calibrated for the city of Fort Collins, CO (Sharvelle et al. 2017). This fraction of indoor household toilet water use was estimated at 4% of total household demand. All costs metrics are scaled by the consumer price index to transform data into 1980 dollars. Further, the average volume of toilet water use was scaled by household size.

Greywater and storm water reuse programs were initially included in the hydroeconomic model, though eventually not incorporated in the final analysis due to incomplete cost data. Much promise holds in the adoption of greywater and storm water capture, though current cost information is incomplete or prohibitive. Future adaptations to the hydroeconomic model will include storm water capture and grey water reuse.

Outdoor water conservation for each household in the SPRB was included in the form of xeriscaping. Xeriscaping converts high water-use grass lawns into efficient landscapes through the conversion of grass lawns to desert-adapted plants and other water savings features. Rebates for xeriscaping conversion were estimated by the Las Vegas Valley Water District (2016) and California Division of Water Resources (2015) to be about $0.59 per square foot (1980 dollars). Similarly, the installation costs and 40 years of maintenance for xeriscaping lawn conversions estimates to about $0.67 per square foot (1980 dollars) from a xeriscaping study in Colorado by Medina and Gumper (Medina and Gumper 2004).The average irrigated lawn size is estimated at 5,400 sq. ft. by the previous study, and in combination with the REU study, the total lawn size in the SPRB was found to be 4.3 billion sq. ft. by multiplying the average lawn size by the number of households in the SPRB. The maximum water savings that can be obtained through xeriscaping was also estimated from Medina and Gumper at about 59% of 1980 water use. The fraction of water savings gained from xeriscaping in each sub-region was estimated by a multi-county assessment of outdoor water use using IUWM. The amount of outdoor water use as a percent of total household water use was estimated between 47-94%, depending on urban versus rural locations.

In this study we tested four institutional settings. Two of these settings were constructed as market institutions to test the differences different economic policies. A market-driven institutional setting was created and to test the effects of a semi-perfect water market where transaction costs are still included for water rights purchases. Testing this market-driven scenario against a goal-oriented, or target, institution allows for tradeoff exploration between the two institutions. The target institution limits the amount of acreage that can be removed from production through water rights purchases. The amount of acreage is calculated as the sum of all acres across each sub-region and producer for the entire SPRB. Acreage goals were set at the total 1980 level of acreage. From 1980-2050 the acreage goal was set at 94% of total acreage in 1980. Thus, the total amount of acreage between the decades of 1980-2050 must be at least 94% of the total in 1980. To reach this goal each producer in each sub-region must produce at least 90% of their original acreage throughout each decade between 1980-2050. From 2050-2060 this goal was lowered to 75% of 1980 acreage levels to ensure model feasibility. This goal was met by requiring each producer in each sub-region to meet at least 64% of their original acreage. By aggregating goals by each producer and sub-region the model can ensure at least 94% of acreage remains in production from 1980-2050, and at least 75% of acreage remains in production from 2050-2060.

The next two scenarios were constructed as regional policies. Each of these institutional settings is targeted at either the municipal or agricultural sectors. First, raw water requirements imposed on land developers were decreased by 20%. These reductions acted as a policy substitute for reducing household water demands by 20% and is complimentary to Colorado Water Plan goals (Colorado Water Conservation Board 2016). Raw water requirement reductions were applied equally to across all municipalities and set at 1.151 AF/acre. Second, the removal of agricultural buy-and-dry was implemented to test and encourage the use of deficit irrigation and dryland farming. Colorado water law currently prohibits the use of previously irrigated acreage for crop production purposes alongside the sale of corresponding water rights. To incentivize the usage of dryland farming and deficit irrigation, a policy was created to remove current buy-and-dry restrictions. A buy-and-dry flag is used within the model to coordinate whether buy-and-dry restrictions are in place. If buy-and-dry restrictions are occurring, this flag is set to one. To test this policy the flag is set to 0, which allows famers to sell part of their water right and deficit irrigate or dryland farm. Agricultural production functions were created and calibrated to effectively manage these alternative irrigation practices. Permitting these practices allows famers the flexibility to sell part of their water rights while keeping a portion of their acreage in production. All four of these scenarios were tested using the hydroeconomic model, and the last two combined with the Multiobjective Evolutionary Algorithm framework (Hadka 2016).

2.2.11 Sustainability Indicators

Success metrics for rural communities were evaluated to quantify the sustainability of irrigated agriculture from 1980-2070. To assess the sustainability of rural economies, specific metrics were calculated. Because rural economies depend on the direct and indirect effects of local investment from agricultural producers, key metrics are associated with the health of irrigated agriculture. As such, the amount of irrigated acreage in production was identified as an important sustainability indicator. The more irrigated acreage in production, the higher the amount of profit from the sale of crops a producer can earn and reinvest locally into the economy. Additionally, the net-present value of agricultural profit from production (over a 40-year period at a 3% discounting rate) is calculated as a surrogate for the strength of the rural economy. These two metrics in combination effectively measure the strength and sustainability of the rural economy over the modeling periods. Both success indicators are tested against all water management scenarios evaluated and allow for the exploration of tradeoffs between technology adoption and policy inclusion.

Municipal sustainability indicators are different from agricultural metrics in that they pertain to the cost of supplying households with reliable water. From a water utility perspective, success metrics revolve around supplying secure water at a low cost with a high likelihood of delivery. Included in M&I costs are the investments made from the adoption of water management technologies. A lower cost is more efficient for municipalities and M&I cost per utility user is calculated as a key success metric for water utilities. Water management strategies which yield an overall lower cost per person are quantified as being more efficient than higher cost scenarios. To maintain lower costs at longer time scales, the average cost of a water right (in $/AF) is calculated. This metric is calculated as the average across each ditch company an M&I user purchases from. Lower water rights costs are associated as better for M&I users because future water transfer costs will be cheaper. Alternatively, lower water rights prices may harm agricultural producers who wish to retire their acreage.

MOEA indicators were included to create nondominated solutions across each of the institutional settings included in our analysis. The first indicator corresponds to the municipal cost of supplying water to satiate future urban water demands. This cost metric is calculated as the total cost of water rights purchases, new reservoir storage investments, xeriscaping rebates, and toilet upgrade rebates. The aggregate of these costs corresponds with the amount a utility must pay to ensure urban water demand is being met. This cost metric is calculated for each municipality and aggregated across space during each model run in the year 2050.

Agricultural indicators correspond with the strength of the rural economy and are calculated as rural expenditures. Rural expenditures include the total revenues from crop production and water rights sales from agricultural to municipal users. Crop production revenues do not include the cost of land and water (e.g. irrigation, fertilizer, etc.) as we assume these costs are purchased locally and would stimulate rural economic activity. Further, we assume water rights sale revenues are reinvested locally into the economy. As such, both success metrics create tradeoffs between each policy scenario and allow for the full exploration and creation of optimal portfolios in regards to policy choices, institutional settings, and water technology adoption.

2.2.12 Multiobjective Evolutionary Algorithm

A framework based on Computational Semi-Arid Water Sustainability (CSaws) optimization framework and the Multiobjective Evolutionary Algorithm framework assesses tradeoffs of various solutions across multiple sectors (Dozier set al. n.d.; Hadka 2016). The CSaws framework was used to evaluate impacts of different institutional settings on both sectors, and to identify institutional settings most effectively remove barriers to better practices. In CSaws, state-of-the-art computing technology is combined with advanced hydroeconomic modeling to produce a multi-layered approach to modeling that mimics the policy-making hierarchy. Institutional settings provide a backdrop over which regional or statewide planners develop many different portfolios of policy targets. The framework exposes policy-maker objectives that are slightly different than those of socio-economic actors within the system who act independently from policy-makers while being restricted by various institutional settings and policy constraints. Two different institutional settings were compared to the baseline: i) adapting common “buy and dry” processes to spur more deficit irrigation and dryland farming, and ii) reducing municipal raw water purchase requirements imposed on land developers. Under every institutional setting, urban and agricultural actors consider four different water supply options: new water storage reservoir development, xeriscaping, toilet flushing, and efficient agricultural irrigation technologies. Policy-makers make targets of water supply options to meet while actors choose how much to individually participate in adoption of any water supply option. Policies compare objectives across two sectors: city water acquisition cost and estimated expenditures in the rural economy. Shifts in the Pareto optimal frontier from high-cost and low rural expenditures to low-cost and high rural expenditures represent benefits to both urban and rural sectors of economy. So, even under uncertain adoption of specific water supply options, certain policies can be shown to be more effective for all sectors influenced by the policy.

2.3 Results and Discussion

Technology adoption can save land in agriculture and keep valuable water in rural communities by sustaining irrigated acreage and profit from production (Table 5). All management strategies (Scenarios A-E) vastly increase the amount of irrigated acreage and agricultural profit as compared to no adoption (Scenario G). The amount of irrigated acreage increases by an average of 480,000 across Scenarios A-E, resulting in an increased agricultural profit of $1.74 billion. The adoption of water management strategies decreases the externalities of urban growth on rural economies. These externalities result in large decreases of irrigated acreage and profit as shown in Scenario G. Thus, the rapidly growing SPRB can sustain irrigated agriculture while meeting the needs of growing populations through technology adoption.

Table 5 – Irrigated land (2050) and the NPV of profit from production (2010-2050) for each scenario.

Scenario	Irrigated Land (acres)	Agr. Profit ($ billion)
B	812,200	$ 7.140
A	747,500	$ 6.850
D	747,200	$ 6.850
F	738,600	$ 6.760
E	536,500	$ 6.530
C	404,400	$ 6.180
G	186,200	$ 4.980

The tradeoffs between modeled scenarios are explored in Figure 8A for selected sustainability indicators. Acreage and agricultural profit from production impact producers (higher is better), while water prices and M&I costs regard M&I agents (lower is better). Lines representing each scenario are more beneficial towards the outside of the chart and are normalized by the maximum value for each indicator. The largest tradeoffs occur when xeriscaping is not available for adoption (Scenario C). Large amounts acreage are removed which subsequently decreases agricultural profit. The only scenario which performs worse is Scenario G, which excludes all technology adoption. M&I users can help producers stay in production through converting grass lawns to water-efficient landscapes. The foregone water savings of not adopting xeriscaping cause M&I costs and average water prices to be among the highest out of all conservation scenarios at $15,000/person and $7,400/AF, respectively. Outdoor water demand, particularly in semiarid regions, represents a disproportionate amount of household water use (Sharvelle et al. 2017). By converting away from high water-use plants like grasses and towards desert-adapted landscapes, M&I users can achieve substantial water and costs savings. Xeriscaping is considered the most important technology to adopt and yields the biggest number of tradeoffs for each agent if not implemented. The second largest tradeoff occurs when irrigation technology is not adopted (Scenario E). Decreased irrigation efficiencies lead to less water available for municipal purchase, driving up costs to M&I users and water prices, as well as decreasing irrigated acreage and profit from production. Interestingly, agricultural producers investing in irrigation technology help M&I users. This symbiotic relationship between rural and urban conservation results in welfare gains or tradeoffs for both agents.

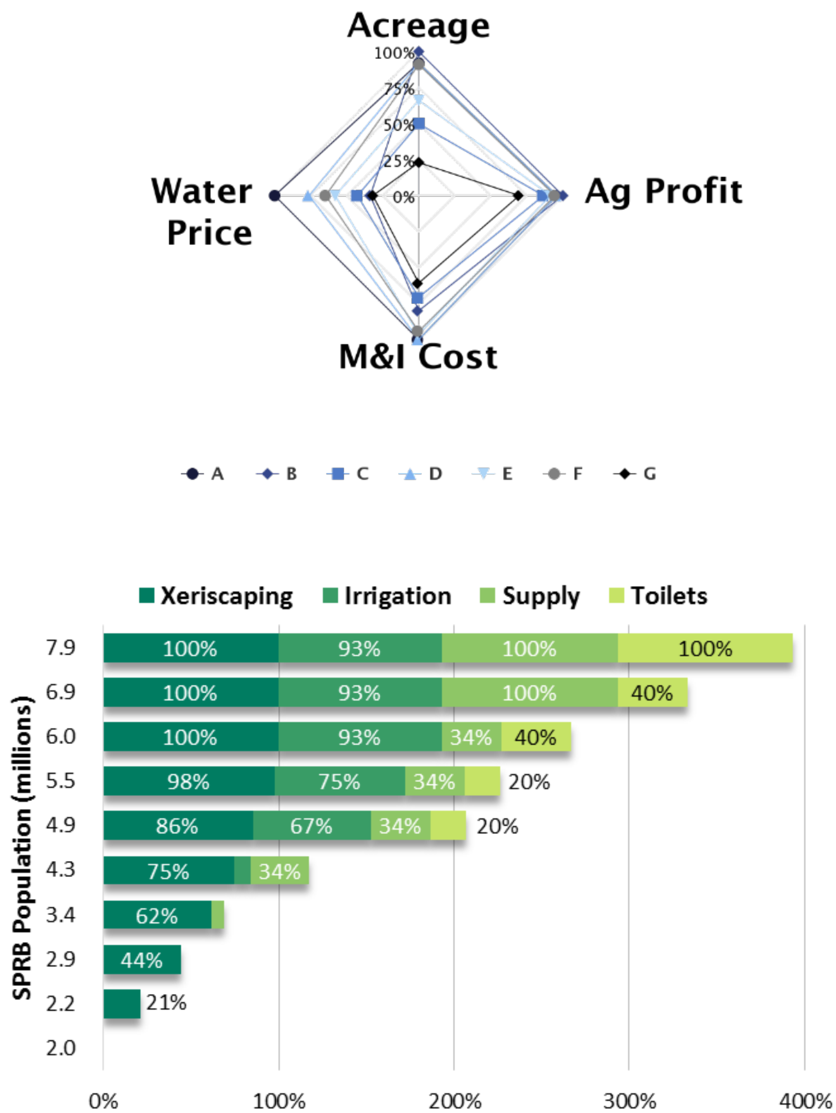


Figure 8 – A) Tradeoffs between conservation scenarios for municipal and agricultural agents (metrics are relative to 2050) and B) rates of adoption for Scenario A.

As the population in the SPRB expands to almost 8 million residents by 2070 the adoption of technology grows (Figure 8B). Population values are analogous to each decade the model was run (i.e. a population of 2 million representing 1980 and 7.9 million representing 2070). Xeriscaping, the most cost-effective and efficient measure at satisfying urban demands, is adopted first. The population in the SPRB has to exceed 3 million before other technology adoption and water transfers are needed to satisfy M&I demand. Further, no other technology is fully adopted before xeriscaping has reached its maximum. It is important to note both new supply developments and conservation technologies are adopted, highlighting the importance of additional supply reservoirs alongside urban and agricultural conservation to sustain population growth. Producer investment in irrigation technology reaches an upper bound of 93% of the total available due to specific producers selling nearly all their water and subsequently drying their acreage by 2020 (i.e. population of 4.3 million). To increase the welfare of the SPRB the adoption of xeriscaping, new supplies, and irrigation technologies are imperative. While toilet upgrades increase welfare the benefits remain relatively minimal. Making these three technologies available through incentives, decreased barriers to entry, public awareness, and education will allow semiarid regions to manage water scarcity effectively.

Given the opportunity to invest in water management strategies (Scenarios A-F) or rural-to-urban water transfers (Scenario G), M&I users are investing in conservation and supply expansion. The economic incentives of M&I agents shift away from water transfers towards conservation due to the cost-effective gains of conservation. Each agent in the hydroeconomic model is purely motivated through economic rationality, and each choice yields the most inexpensive solution. M&I costs and water prices decrease by an average of $5,400/person and $4,200/AF in 2050 when technology is available for adoption. This results in M&I users relying on a much smaller amount of water purchases from the agricultural sector. M&I agents purchasing less water from agricultural communities helps to sustain the future of irrigated crop production in the SPRB. This can benefit rural economies by decreasing the effects of externalities driven by rural-to-urban water transfers, as can be evidenced by gains in irrigated acreage and agricultural profit in Figure 8A. Water conservation can decrease water treatment and energy costs, reduce pollution, and offset carbon dioxide emissions, providing many beneficial externalities for urban areas (Spang et al. 2018). As such, the hydroeconomic model may overstate the total costs to M&I users by not including these benefits. Municipalities in the SPRB should focus on investing in cost-effective conservation and new supply reservoirs instead of purchasing water from the agricultural sector. Through urban and agricultural conservation, alongside reservoir construction, the buy-and-dry paradigm of agriculture in the SPRB can subside and irrigated agriculture can continue in production. M&I adoption of new supplies and conservation strategies is cheaper than traditional water acquisition and leaves both rural and urban communities better off.

A remarkable tradeoff occurs between Scenario A (no acreage targets) and Scenario B (acreage targets). Although there is a decline in irrigated land when removing acreage constraints, the decline is relatively minimal. Allowing the market to select the amount of acreage results in a mere 8% decrease in irrigated acreage compared to Scenario B. The some 65,000 less acres in production affects agricultural profit by about $300 million. Though more acreage is available for production in Scenario B, the price of water and M&I cost of meeting urban water demand increase exponentially. The price of water and M&I costs increase by $6,200/AF and $2,600/person respectively, ranking among the highest across each scenario. These outcomes suggest that market-driven solutions perform extremely well when compared to the optimal solution for producers. Though agricultural producers face a small tradeoff in irrigated acreage, municipalities are able to sustain future urban water demands at much lower costs. What is important here is the path taken by M&I users to sustain population growth. Though acreage targets may be better for rural economies these constraints effect the path and timing of M&I conservation and consequently drive up costs and water prices. Future socioeconomic policies targeting rural economies should aim efforts in creating a more efficient water market rather than prohibiting loss of acreage to ensure gains in welfare.

The nearly inevitable decline in irrigated acreage is indicated by the rapid fall of cropland at a population near six million in the SPRB (Figure 9). This large reduction in acreage represents a key threshold in irrigated agriculture. For the SPRB, technology adoption can no longer sustain irrigated acreage at a population over six million. Notably, the rapid loss of irrigated acreage at this population is directly linked with the maximum adoption of xeriscaping (Figure 8 – A) Tradeoffs between conservation scenarios for municipal and agricultural agents (metrics are relative to 2050) and B) rates of adoption for Scenario A.. Once xeriscaping opportunities are fully adopted, M&I users turn towards agriculture for future water acquisition. While additional supply reservoir adoption does increase between a populations of 6-6.9 million, acreage falls as demands continue to grow. The carrying capacity of six million residents in the South Platte River Basin of Colorado should act as a signal to municipalities, decision-makers, and rural communities. If the population reaches six million people, which will be met by current trends by 2050, irrigated agriculture in the SPRB may sharply decline. Supply improvements, irrigation technology, toilet upgrades, and xeriscaping can only sustain agriculture for so long and inevitably the rapid growth in water demand overwhelms the available technology in the model. Advances in technological innovation, alternative transfer methods, creative policymaking, and additional conservation measures must be taken to sustain agriculture in semiarid regions.

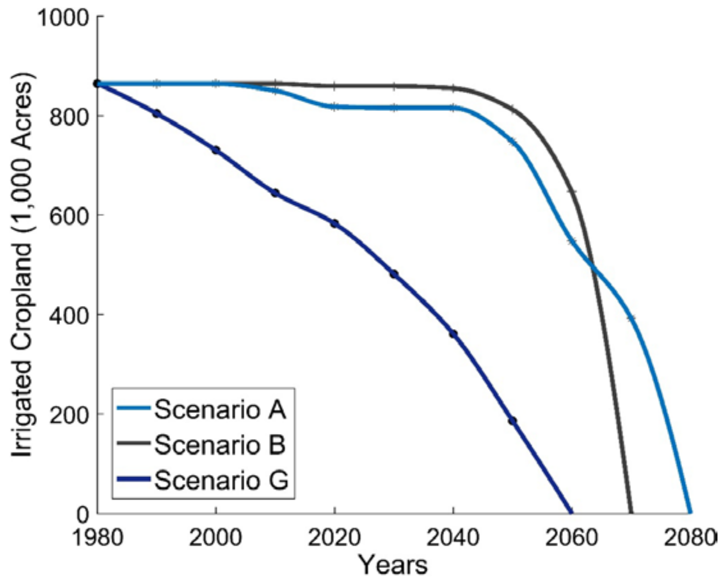


Figure 9 – Trends in irrigated cropland for Scenarios A, B, and G over modeled timescales.

Pareto frontiers were created for each policy considered in Figure 10. Pareto frontiers display uncertainty in the adoption of water management strategies and create a set of nondominated solutions. These solutions help planners build policy portfolios to help reach specific targets without tradeoffs to municipal or agricultural agents. Unsurprisingly, water management technologies increase rural expenditures and decrease municipal costs across nearly all of Pareto frontiers. Reducing raw water requirements significantly reduces costs to municipalities of satisfying urban demands. Reducing these requirements by 20% yields larger savings (near 30%) for low values of rural expenditures. Although reducing raw water requirements decreases cost to municipalities, the range of possible rural expenditures decreases from $8-10.5 billion to only $8-9.5 billion. The reduction of raw water requirements favors municipalities and can hinder rural expenditures on the upper end of the Pareto frontier.

The removal of buy-and-dry practices helps to sustain irrigated crop production. Creating the option to deficit irrigate or dryland farm allows producers to increase rural expenditures.

Though the difference in expenditures is small on the left-hand side of the Pareto frontier, opportunities to increase expenditures and slightly reduce municipal costs exist on the right-hand side. Further, the range of possible rural expenditures increases significantly to $8-11.5 billion, the largest out of all policies considered. Municipal costs remain largely unaffected by the removal of these requirements and stay within original baseline ranges. Removing buy-and-dry practices would largely benefit rural expenditures and should be considered to maintain rural economies in the long-run. Including these attainable policy goals creates a portfolio of solutions for the consideration of water management planners, and can help reach a solution where municipal water demands and agricultural production can be equally sustained. To reach these goals, the adoption of various water management technologies are imperative.

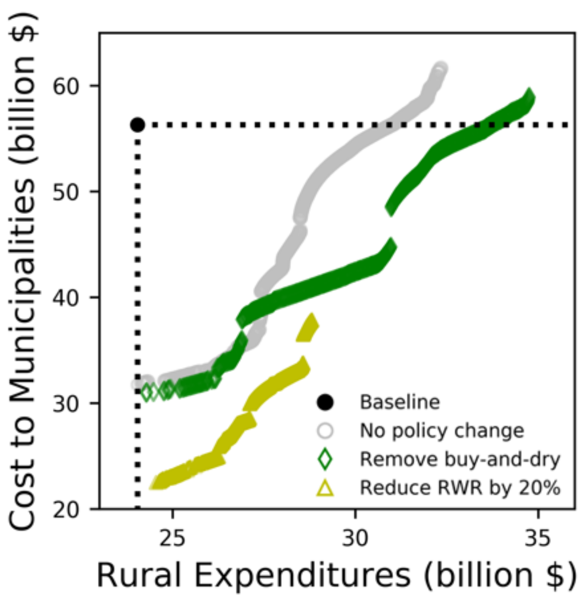


Figure 10 – Nondominated solutions for differing policies.

2.4 Conclusions

Water management strategies can save land in agriculture and increase the longevity of rural livelihood in semiarid regions. Conservation technologies, acting as substitutes for agricultural-to-urban water transfers, keep valuable water in agriculture. The adoption of technology (Scenario A) leads to an increase of irrigated acreage by 560,000 acres, resulting in profits from production reaching $6.8 billion – a $1.9 billion increase compared to zero adoption. The sustainment of production and irrigated acreage will help rural communities in semiarid regions avoid economic loss. Further, these communities will no longer bear the burden of the rapid population growth in nearby cities.

Not only do conservation technologies help agriculture, they cost less to M&I users than status-quo practices. While water utility incentives may not always align with societal goals the cost-effectiveness of water conservation is enough to justify the need for adoption. Producer and M&I conservation decreases costs for M&I users by nearly 40%, indicating water conservation is economical and efficient. Technologies drop the average water price by an average of $4,200/AF, leaving space for the future procurement of water. For M&I users, xeriscaping is the most effective technology at reducing water transfers. Large decreases in outdoor water demand results in proportional increases in irrigated acreage, creating a symbiotic relationship with urban outdoor water use and rural economic vitality.

The success of market-driven solutions is a notable outcome of this work. By clearing the market of acreage targets near-optimal performance is observed. A decline of 8% in irrigated acreage is diminutive compared to the likely costs of time and energy to enforce such regulations. The goal of future economic policy should be to free the market of constraints and instead focus on social policy targeted at sustaining rural economies past 2050. The practicality of this result is of importance for water management professionals in semiarid regions.

Policies affecting water purchases can have large impacts on the benefits of technology adoption. Reducing raw water requirements significantly reduces municipal costs but can limit the opportunity for high rural expenditures. Removing buy-and-dry constraints allow producers to maintain rural expenditures without causing large increases in municipal users. In combination these two policies could help to reduce externalities driven by rural-to-urban water transfers and keep costs low for municipal users.

Once a threshold of six million people in the SPRB is reached, agriculture inevitably falls. Going forward, rural economies can prepare for the loss of irrigated lands with plentiful time for robust policies, human endeavors, and additional water conservation to be enacted. The magnitude and interdisciplinary impacts of these losses are key focuses of future work. Studies of spatial heterogeneity and scale, input-output modeling, and atmospheric and groundwater feedbacks are of main importance to fully measure the impact of agricultural decline in semiarid regions. Further work should be conducted to include adoption uncertainties and nuanced agent behaviors. Additionally, the exploration of household water reuse and green infrastructure should be considered to sustain the future of agriculture in semiarid regions.

CHAPTER 3: EXPLORING THRESHOLDS AND CRITICALITIES OF WATER MANAGEMENT STRATEGIES UNDER CLIMATIC, TECHNOLOGICAL, AND INSTITUTIONAL UNCERTAINTY

3.1 Introduction

Water scarcity is one of the greatest challenges facing our society in the 21st century. A changing climate alongside rapid population growth and urbanization further complicate the systems we use to manage water resources. As such, creative solutions are essential to effectively plan and manage our increasingly-scarce freshwater supplies. Proper water resources planning and management requires long-term climate forecasting, hydrologic modeling, ecological assessments, large-scale infrastructure, and stakeholder involvement among several other factors (Karr 1991; Lubell and Edelenbos 2013; Mitchell 2005; Thomas and Durham 2003). This systems approach to water resources engineering is commonly referred to as Integrated Water Resources Management (IWRM) (GWP TAC 2000). IWRM holds much promise to effectively manage water resources. However, the mutually-dependent systems required for accurate IWRM can lead to ineffective, if not contradictory, water resources management portfolios (Biswas 2008; Giordano and Shah 2014). One area to improve the effectiveness of IWRM can be realized through the implementation of uncertainty in water management systems.

Uncertainty is inherently linked with water resources management through the interconnected nature of water resource systems. These water resource systems are often represented through a series of hydrologic, climatic, ecological, and socioeconomic models. Unfortunately, accurately estimating these models can lead to substantial uncertainty. Variability in model drivers, alongside population growth, land-use change, and growing water demands can cause severe water shortage vulnerability for municipal, agricultural, and environmental sectors (Brown et al. 2013; Foti et al. 2014; Stakhiv 2011). We define vulnerability as the state where water demand exceeds water supply. Further, rigid water appropriation institutions and competing demands between municipal and agricultural communities can intensify these risks (Ansar et al. 2014; Flörke et al. 2018; Rosegrant 2000). One of the most important factors surrounding water resources management is the required infrastructure needed to obtain, treat, and deliver water. Large infrastructure projects around the world are vulnerable to uncertainty due to a variety of factors (e.g. timing, natural hazards, planning fallacies, etc.) which can exponentially increase expected costs and completion times (Buehler et al. 1994; Flyvbjerg 2014; Touran and Lopez 2006). Such cost uncertainty can create potentially hazardous situations for growing municipalities. In combination, uncertainty in water resources management and its consequences can expose vulnerable populations to water shortage risks.

Urban planners around the world are tasked with the challenge to secure water for growing populations. Affecting these water supply portfolios are variations in climate, hydrology, politics, land-use, and population growth. An association between uncertainty and water resources management is imperative to accurately prepare for future water demands. Volumes of literature have given planners the opportunity to cope with risk driven by uncertainty. A handful of examples in regards to water resources include stochastic programming (Li et al. 2008), decision support systems (PALLOTTINO et al. 2005; Weng et al. 2010), multi-objective optimization programs (Nicklow et al. 2010; Reed and Kasprzyk 2009), and traditional statistical and risk analyses (Ang and Tang 2006). However, the impacts of uncertainty are not often evaluated at fine spatial and temporal resolutions with locally-calibrated parameters. Evaluating uncertainty of an integrated social, ecological, and technological analysis is needed to fully quantify the effects of variability in future water resources planning and management. Using an integrated modeling approach allows for urban planners to obtain the information to be prepared for, rather than reactive to, future uncertainty in water resources planning and management.

Several water management practices can alleviate the consequences of uncertainty when planning for future water resources. Specifically, a blend of in-basin strategies including conservation, institutional change, and infrastructure can be adopted to decrease water shortage vulnerability. Additional opportunities include alternative transfer methods, water markets, and crop management practices (DiNatale Water Consultants 2013; McMahon and Smith 2013; Vaux and Howitt 1984; Western Governors' Association 2012). In this study we evaluate the effects of water supply and demand management strategies on water supply planning under future uncertainty. The sensitivity for the adoption of water management practices are quantified using a robust-agent based modeling framework driven by municipal growth, land-use change, and climate. Municipal and agricultural agents are independently parameterized within the framework to model future changes in the allocation of water rights for a representative semiarid river basin. The important impact of this work is to assess the sensitivity of water resources planning and management under future uncertainty in water supplies, water management practices, agricultural production functions, institutional change, and municipal growth.

The purpose of this study is to address challenges in water supply and demand planning and management by explicitly modeling uncertainty with an integrated, locally-calibrated modeling system. Specifically, we ask: what are the criticalities and thresholds of future water supply planning in semiarid regions under rapid population growth? To effectively answer this research question, the objectives of this study are to: (i) discuss the effects of uncertainty on satisfying future urban demand, (ii) discover critical parameters that affect municipal and rural success indicators, (iii), identify key thresholds, or tipping points, of critical parameters, and (iv) evaluate future changes in climate and policy for municipal water supply planning. Results indicate uncertainty in the costs of new water supply infrastructure can significantly impact the cost of supplying water to future populations, while the adoption water management practices can increase the stability of water supply costs. Additionally, a key policy measure is identified which can be used to provide sustainability in water supply planning and lower its subsequent costs.

3.2 Methodology

3.2.1 Global Sensitivity Analysis

To evaluate the sensitivity of conservation adoption, water supply, and water demand to uncertainty a robust modeling framework known as the Computational Semiarid Water Sustainability (CSaws) platform was employed (Dozier et al. n.d., 2017). The CSaws framework is an agent-based, partial-equilibrium hydroeconomic model with capabilities to be applied at a variety of spatial scales and climate regimes. CSaws uses the framework of Britz et. al. (Britz et al. 2013) to solve the optimization problem as a Multiple Optimization Problem with Equilibrium Constraints (MOPEC), and is formulated as a nonlinear, mixed-complementarity problem. CSaws balances different goals for two objective functions and optimizes these functions towards a local optima based on a variety of agricultural, urban, climate, land-use, institutional, and hydrologic parameters. The first objective function maximizes the net-present value (NPV) of agricultural profit from production for agricultural producers. Producers in CSaws are motivated solely through profit maximization and can do so by producing and selling crops or selling water rights to growing municipalities. Alternatively, municipalities minimize the cost of acquiring water to sustain projected increases in water demands through future population growth and subsequent land-use change. Municipal and industrial (M&I) agents have the option to purchase water rights, invest in new supply storage projects, or conserve household water use. Agricultural producers may purchase additional water for on-farm use from new supply projects and invest in irrigation technology to improve their irrigation application efficiency. We performed a global sensitivity analysis of CSaws to evaluate the effects of input uncertainty on expected outcomes (Figure 11).

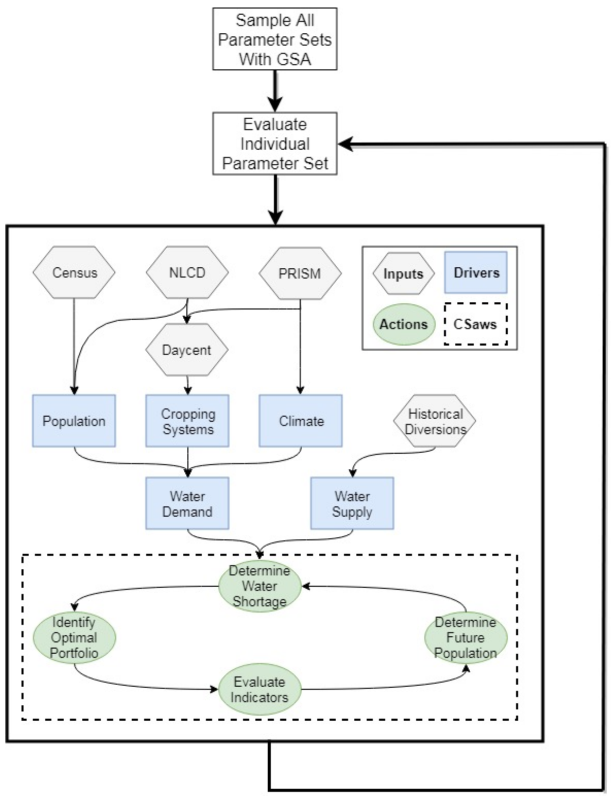


Figure 11 – Model diagram of global sensitivity analysis and CSaws. During each iteration CSaws evaluates a new parameter set from the global parameter set.

The Sobol method of global sensitivity analysis (GSA) was used to evaluate the effects of parameter uncertainty on key success indicators for agricultural and municipal sectors. A variance-based sensitivity analysis is necessary to calculate the sensitivity for the entire parameter input space and due to the nonlinearity of CSaws. A total of 16 parameters were included in the GSA and evaluated using SimLab (Tarantola and Becker 2017). SimLab readily provides the number of executions needed for a GSA given a specific sampling and sensitivity analysis method. The Sobol pseudo-random sampling method was used to create a total of 32,768 parameter sets and were evaluated using CSaws alongside the General Algebraic Modeling System (GAMS Development Corporation 2013). The CSaws framework was evaluated at each parameter set and optimized for the given changes in input parameters. Although GSA methods can be computationally-prohibitive for nonlinear optimization models, the Sobol method of GSA was determined to be the most robust measure of sensitivity given the parameter space. All model executions were evaluated using MATLAB [*The MathWorks, Inc. ®, 2017*] and various scripts created by SimLab to link MATLAB outputs with SimLab for the calculation of first and total order sensitivity indices.

3.2.2 Case Study Area

CSaws and the GSA are evaluated for a representative semiarid region experiencing significant water scarcity, population growth, and land-use change. The South Platte River Basin (SPRB) in Northern Colorado is a unique region with both highly-dense urban areas and a strong agricultural economy (Figure 12). The allocation of water rights are subject to the Prior Appropriation Doctrine which determines local water laws and transfers (i.e. water rights). This rigid water allocation system requires the permanent dry-up of irrigated acreage alongside the sale of water rights, a process referred to as buy-and-dry (Squillace 2013). CSaws incorporates these institutional settings alongside robustly parameterized agricultural and municipal sectors to model the future allocation of water rights. The SPRB is separated in to five sub-regions with contrasting levels of municipal and agricultural density at a multi-county scale. Within each sub-region four producers of the four most prominent crops in the SPRB (alfalfa, corn, sugar beets, and wheat) exist alongside a single municipality. Producers are aggregated as the total amount of irrigated cropland in each sub-region. Municipalities are represented equivalently by aggregating all cities within the same sub-region. A water market is created within CSaws to evaluate the trading of water rights within and across sub-regions in accordance with urbanization and subsequent water demand drivers. Water rights transfers are subject to various legal and infrastructural transaction costs, the latter of which can increase exponentially across different sub-regions. Parameterization, calibration, and estimation of all input parameters are discussed comprehensively in separate papers by the authors (Dozier et al. n.d., 2017; Wostoupal et al. n.d.).

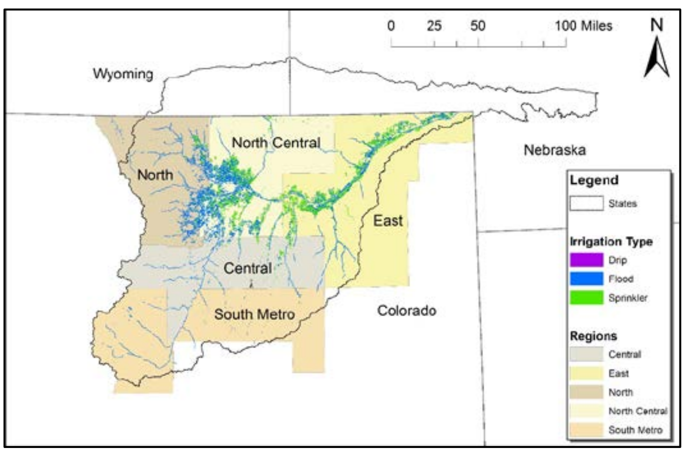


Figure 12 – The South Platte River Basin of Northern Colorado. All irrigated lands (2010 survey) are displayed with respect corresponding irrigation types. Sub-regions are highlighted by color and labeled by location in the SPRB.

3.2.3 Parameter Uncertainty

CSaws uses a variety of agricultural, institutional, municipal, and hydrological parameters to effectively model the future of water rights allocation in semiarid regions. All input parameters are subject to uncertainty and variability. The importance of a GSA on an integrated modeling system is to assess which parameters are critical to key success metrics. Next, thresholds of modeled outcomes can be investigated based on parameter criticalities. Accurately estimating probability distribution functions for a wide range of parameters can be extremely difficult and lead to spurious results. As such, unknown prior probability distribution functions were assumed to follow a wide uniform distribution. A uniform distribution is common in global sensitivity analyses for unknown prior probability distribution functions (Ajami et al. 2007). A summary of included all parameters, assumed distributions, average values, units and bounds are included in Table 6.

Table 6 – GSA parameter distributions for technological, institutional, climatic, and agricultural parameters (all dollar values in 2010 dollars).

Parameter	**Distribution**	**Lower Bound**	**Avg. Value**	**Unit**	**Upper Bound**
Storage Cost	Uniform	-50%	3,600	$/Acre-Feet	+50%
Storage Maximum	Uniform	-15%	33,200	Acre-feet	+15%
Irrigation Technology Cost	Uniform	-25%	151	$/Acre	+10%
Irrigation Tech. Maximum	Uniform	-15%	58%	-	+15%
Toilet Upgrades Cost	Uniform	-40%	375	$/Household	+30%
Toilet Upgrades Savings	Uniform	-10%	40%	-	+10%
Toilet Upgrades Maximum	Uniform	-15%	35%	-	+15%
Xeriscaping Cost	Uniform	-25%	2.0	$/sq. foot	+25%
Xeriscaping Savings	Uniform	-20%	8%	-	+20%
Xeriscaping Maximum	Uniform	-15%	75%	-	+15%
Infrastructure Costs	Uniform	-50%	21,000	$	+50%
Raw Water Requirements	Uniform	-5%	1.44	A-F/Acre	+5%
Firm Yield Ratio	Uniform	-5%	57%	-	+5%
Crop Production Function	Normal	-30%	7.67	-	+30%
Crop Output Prices	Uniform	-10%	46	$/ton	+10%

The available adoption of water management technologies is an important element of the CSaws model. New supply infrastructure projects, agricultural irrigation technology improvements, indoor household conservation (toilet renovations) and outdoor household conservation (xeriscaping) are opportunities available to agents within the model. During each model evaluation, agents may choose to adopt these technologies and supply options at a specific cost. As such, each technology has an upper bound on the amount of adoption and a factor describing its impact on water supply or demand. Unsurprisingly, all parameters regarding water management strategies (e.g. cost, yield, etc.) are uncertain. To inform parameter uncertainty both literature reviews and expert opinion were employed.

New supply infrastructure improvements allow municipalities to invest in highly-reliable water via reservoir construction and expansion projects. Supply infrastructure improvements parameterized in the model are derived from existing reservoir construction and expansion plans in the SPRB (Table 7). Supply projects have estimated costs which are subject to a variety of uncertain factors such as construction, timing, materials, legal fees, etc. Due to the heterogeneous nature of cost distributions, supply infrastructure costs were assumed to follow a uniform distribution. Upper and lower bounds of the uniform distribution were calculated as the average percent change between the most and least expensive supply projects. The cost of an acre-foot (AF) of firm-yield water supplied by the reservoir was used for these bounds, calculated as the ratio of the total reservoir project cost divided by the expected annual firm yield. The average percent change across all supply projects resulted in a 45% difference between expected costs per AF of firm-yield storage, which was rounded up to 50% as an upper and lower bound for simplicity. Expected values of annual firm yield are subject to a high amount of uncertainty given climate conditions, reservoir operations, water demands, and other unknown factors. To avoid over-prediction a uniform distribution with upper and lower bounds of 15% was assumed to explain the variance of expected annual firm yield.

Table 7 – New supply reservoir project cost and supply information.

Sub-region	Planned Reservoir Project	Annual Yield	Total Cost	c_d^{stor} ($/AF)
North	Halligan Reservoir Expansion	7,000	$ 30,000,000	$ 4,300
Central	Gross Reservoir Expansion (Moffat)	18,000	$ 380,000,000	$ 21,300
	Chatfield Reservoir Reallocation	8,500	$ 186,000,000	
C-BT	Windy Gap Firming Project	26,000	$ 223,000,000	$ 11,900
	Northern Integrated Supply Project	40,000	$ 600,000,000	

Improvements to agricultural irrigation technology can be adopted by producers to increase irrigation application efficiencies. Irrigation technology improvements occur in the form of transferring from flood irrigation practices to center pivot, linear move, and center pivot with corner systems. These irrigation systems range between $139,000 to $214,000 per 160-acre field, averaging $151/acre (Scherer 2015). Following the equivalent methodology of storage cost uncertainty, irrigation technology costs were assumed to be bounded as the average percent change between the most and least expensive systems. An additional study was implemented to include additional cost data to assess variability (Amosson et al. 2002). The upper and lower bounds on irrigation technology costs ranged from +11% to -25%, respectively. The final bounds on irrigation technology costs were assumed to follow a uniform distribution with an upper bound of +10% and a lower bound of -25%. The maximum achievable application efficiency improvements are assumed to be a uniform distribution between -15% and +15%, equivalent to storage maximum bounds.

Municipalities can capture previously used household indoor water use through the investment in toilet efficiency improvements. Indoor conservation improvements replace 3.65 gallon per flush toilet models (assumed to be all toilets in 1980) with highly-efficient 1.28 gallon per flush toilets. Toilet installation costs were gathered from HomeAdvisor (HomeAdvisor 2016) which maintains a user database on the average cost to install high efficiency toilets per household. The percent change between the average toilet renovation cost and the minimum and maximum costs were found to calculate the lower and upper bounds of uncertainty. Following a uniform distribution, toilet renovation costs were assumed to be between -40% and +30% of the average for all households. Savings gained from installing efficient toilets were informed through the Integrated Urban Water Model (IUWM), which gathers these savings fractions from a residential end-use study (DeOreo et al. 2016; Sharvelle et al. 2017). Using the residential end-use study, 5th and 9th percentiles on the savings gained from installing efficient toilets were found to be between +/- 6.4%. Assuming toilet upgrade savings followings a uniform distribution, these bounds were increased to +/- 10%. Following similar methodology for the storage and irrigation technology maximums, the toilet maximum savings parameter is uniformly distributed with bounds between +/-15%.

Alongside the adoption of indoor conservation, households can invest in outdoor irrigation through improvements in landscape irrigation practices. Previously-used outdoor water use can be captured for future use by replacing high water use plants with desert-adapted landscapes, a practice known as xeriscaping. Household willingness-to-accept (WTA) payments to convert grass lawns towards xeriscaping were assumed to be equal to current municipal xeriscaping rebate incentives. To classify bounds of uncertainty around WTA, various Western U.S. state's xeriscaping rebate plans were identified and compiled. Rebate data from cities in Arizona, California, Colorado, New Mexico, and Nevada ranged from $0.75-2.5/square foot (2017 values, Table 8). The average percent change from originally modeled xeriscaping rebate values ($2/square foot) were calculated based on additional rebate values. Resulting in a percent change of 28%, xeriscaping rebate costs were assumed to follow a uniform distribution with an upper and lower bound changing by 25%. Water demand savings gained from xeriscaping lawns are calculated through IUWM as the outdoor fraction of total household water uses, which averages at 74% across all sub-regions in the SPRB. Using the average percent change between each sub-region, the outdoor irrigation fraction was assumed to follow a uniform distribution with an upper bound of +20% and a lower bound of -20%. The maximum achievable savings from adopting xeriscaping follows a similar methodology to all other water management practices, and varies between +/- 15% through a uniform distribution. In general, the use of nationally-available rebate and cost data for conservation allows these methods to be applicable to a variety of semiarid regions, particularly across the Western U.S.

Table 8 – Xeriscaping rebates for cities in the Western United States. All dollars are relative to 2010 values.

Municipality	**Xeriscaping Rebate ($/ft^2)**	**Percent Change from $2/ft^2**
Fort Collins, CO	$0.68	67%
Castle Rock, CO	$0.90	56%
Phoenix, AZ	$1.81	11%
Las Vegas, NV	$1.81	11%
Santa Barbara, CA	$1.81	11%
Gallup, NM	$2.26	12%
Average	$1.55	28%

3.2.4 Institutional Settings

Institutional parameters within CSaws are influenced by current policy in the SPRB. Variations in these parameters can have ripple effects on municipal costs, water prices, rural expenditures, and water demands. Two key parameters were added to the GSA to determine the effects of institutional change. Transaction costs directly influence water prices in CSaws and are quantified as the sum between legal and infrastructure costs which correspond with water rights sales. While we assume legal transaction costs remain relatively constant across sub-regions, infrastructure costs can increase and vary exponentially due to water conveyance projects. Infrastructure costs were assumed to follow a uniform distribution with upper and lower bounds varying by 50%. Legal transaction costs were kept constant to original values during model evaluations.

The second institutional parameter, raw water requirements, is a main driver of municipal water acquisition. Set by municipalities and calculated as the amount of water a developer must secure (in AF) per acre of developed land, raw water requirements are parameterized in the model to quantify municipal water demand. Uncertainties regarding raw water requirements are extremely important to effectively model municipal demands and subsequent water rights transfers. Raw water requirements were assumed to follow a uniform distribution differing by 5%. It is essential to note that these raw water requirements were highly sensitive to model feasibility, and as such were not changed more than 5% alongside firm yield ratios to keep total model failures below 5%. It is noted raw water requirements may vary by more than 5% in the future and increased institutional variability, with bounds changing by +/- 50%, are explored in the discussion.

3.2.5 Water Supply

Water supplies in semiarid regions vary significantly on an annual basis. To account for the uncertainty in water supplies municipalities often purchase water according to a firm yield ratio. The firm yield ratio corresponds with the expected amount of water a water user may receive during a given year. This metric is quantified as the ratio between the diversion volume during the 2002 drought and the average annual diversion volume between 1950 and 2014. The 2002 drought, occurring with a 75-year return period, was used as the baseline assuming municipalities purchase water with highly inelastic demand. Firm yield ratios were calculated for each sub-region in the SPRB and range between 34-70%. To inform uncertainty in future water supplies a uniform distribution was applied to the water supply firm yield ratio ranging between -5% and +5%. Following uncertainty in raw water requirements, firm yield ratios are extremely sensitive to model feasibility and as such could not change by more than 5%. Variations larger than 5% are explored in the discussion alongside broad changes in raw-water requirements.

3.2.6 Agricultural Producers

Agricultural producers in the model are represented by fitted crop production functions which are inherently subject to uncertainty. Crop production functions were fit through a constant elasticity of substitution function with output from a locally-calibrated agro-ecosystem model, DayCent (Parton et al. 1998). Parameter uncertainty of crop production functions were found to follow a normal distribution with a mean and coefficient of variation of 1 and 0.1, respectively (Zhang 2016). Crop prices affecting producer profit are subject to various uncertainties. Price data from the National Agricultural Statistics Service from 1950-2015 was compiled for alfalfa, corn, sugar beets and wheat. The average change in modeled crop price per year (from 1950-2015) was found to be -$0.41. Translating these changes into decadal price variations resulted in an average percent change of 10% across each modeled crop. Crop price uncertainty was assumed to follow a uniform distribution with upper and lower bounds of +/-10%.

3.2.7 Scenario Evaluation

Two key scenarios were evaluated to isolate the impacts of the adoption of water management strategies. Scenario A allows for the adoption of supply improvements, agricultural irrigation technology upgrades, efficient toilet renovations, and xeriscaping lawn conversions.

Individual adoption of water management strategies are not compulsory in the model and based on individual opportunity costs. As such, Scenario A allows for the exploration of the sensitivity of conservation adoption under cost, climate, and intuitional uncertainty. Scenario B does not allow for the adoption of water management strategies and acts as the baseline, or no-action, scenario in which all municipal demand must be satiated through rural-to-urban water rights transfers. Scenarios A and B are evaluated from a 1980-2070 timescale until the model is no longer feasible for each parameter set. During each parameter set evaluation, the model optimizes a water rights market until municipal demands are met during each decadal run. To assess larger uncertainty in water supplies and institutional change a final scenario was created. This scenario evaluates the effects of uncertainty in firm yield ratios and raw water requirements, both of which change by -50% and +50%. All other modeled parameters are held constant at their original expected values.

3.2.8 Success Indicators

Sustainability indicators are evaluated for each scenario for both municipal and agricultural sectors. Scores of literature have evaluated the effects of rural-to-urban water transfers on rural economies (Dozier et al. 2017; Howe et al. 1990; Howe and Goemans 2003; Start 2001; Thorvaldson and Pritchett 2006). Yet, the magnitude and impact of the reinvestment of water sales revenue into local economies is unknown. In order to strike a balance between a 0% and 100% reinvestment of water sales revenue in local economies, both irrigated acreage and rural expenditures were evaluated as key indicators of rural economic health. Irrigated acreage is calculated as the total amount of acreage in the SPRB during each model evaluation. Higher amounts of irrigated acreage correspond with a more vibrant rural economy, equivalent to goals for future policy in the SPRB (Colorado Water Conservation Board 2016). Rural expenditures are quantified as the total amount of agricultural revenue from crop production (excluding crop production costs) and water rights sale revenues. We assume all water rights sales are reinvested into the local economy. Municipal success metrics are directly related to municipal costs of meeting future household water demand. These costs are calculated as the total amount of water rights purchases and investments in water management strategies. Municipal cost and rural expenditure metrics are discounted to reflect dollar values in 2010 due to model evaluations beginning in 1980, and are discounted by the product of annual inflation values between 1980 and 2010.

First and total order Sobol indices are calculated through SimLab and quantified to explore which parameters are critical for specific success indicators (Saltelli et al. 2010). First order indices represent the effect of an individual parameter on the expected variance in the output, normalized by the total variance of all parameters. Parameters with a high first order index are identified as critical to the expected variance in the output. That is, if a parameter with a high first order index is perturbed slightly it will have a large effect on the expected variance of selected success metrics. Total order indexes combine first order effects with interactions between parameters and are normalized by total parameter variance. Parameters with a high total order index may not be critical to the expected variance in the output, but their interactions with other parameters are important. We calculate “higher order” indexes to isolate the effect of interactions among parameters. These higher order effects are the difference between the total and first order indices, thus accounting solely for parameter interactions. Due to the large amount of indices from all parameters, scenarios, and timescales evaluated, only key total and first order indices are displayed.

3.3 Results and Discussion

Expected values for municipal costs and rural expenditures were aggregated across the entire SPRB for each scenario and decade (Figure 13). Shaded areas for each scenario represent 5th and 95th prediction intervals of expected values, estimated from empirical cumulative distribution functions. Years between modeled decades are interpolated using a shape-preserving piecewise cubic function. Both prediction intervals and expected values are interpolated between each decade. Generally, expected values and prediction intervals for both scenarios increase as time progresses. The widening of prediction bounds are driven by the uncertainty between water supplies, raw water requirements, transaction costs, and water management technologies.

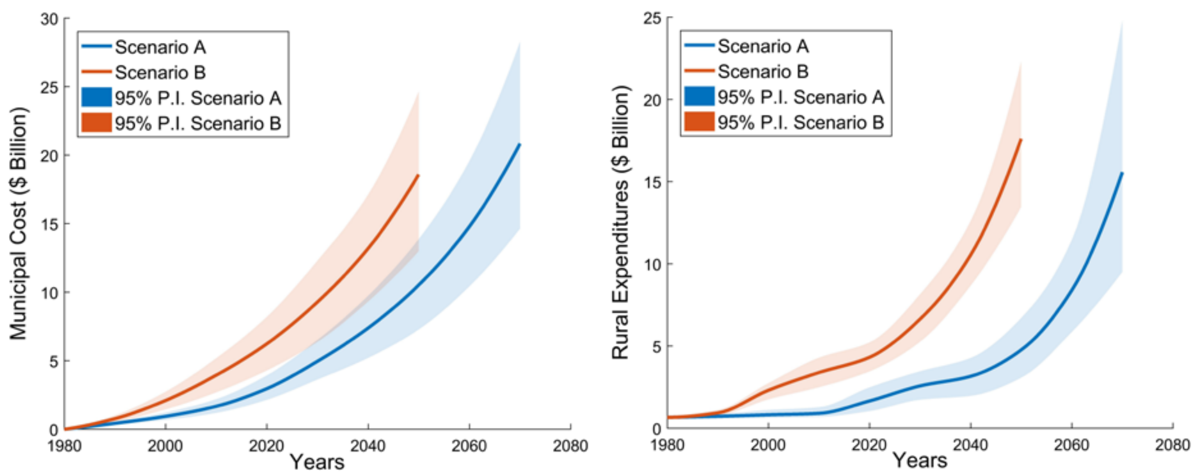


Figure 13 – Expected values functions for A) municipal cost and B) rural expenditures for Scenarios A-B over all GSA evaluations.

Investments in new supplies, agricultural irrigation technology, and household conservation can substantially lower municipal costs. As municipalities adopt these technologies current demands increase at slower rates when compared to no adoption (Scenario B). These demand "lags" result in lower costs over longer planning timescales. Further, water acquisition prices are complementarily lagged with demands due to decreases in water transfers. As such, expected municipal costs and their corresponding prediction intervals do not increase significantly until 2050. Conversely, municipal costs for Scenario B are influenced only by the cost of water acquisition. Because water demands are not decreased through household conservation, water transfers and their prices increase rapidly over time. Uncertainty in infrastructure costs, raw water requirements, and firm yield ratios widen prediction intervals over time. Thus, the adoption of water management technologies can help to offset municipal demands and lower their susceptibility to future uncertainty. Increasing the adoption of water management technologies and incentivizing their costs can help create a more sustainable solution for meeting future water demands.

A tradeoff from lowered municipal costs due to technology adoption occurs between modeled scenarios. Across all years Scenario B outperforms Scenario A in regards to rural expenditures. The primary reason for higher rural expenditures in Scenario B are due to increased water rights transfers. The water demand created by M&I agents when water management technologies are not available for adoption drives up the price of water sooner than in Scenario A. Consequently, rural expenditures dependent on water rights sales fall. If it is assumed a key indicator of rural economic health is only the agricultural revenues from crop production, the opposite story unfolds. Increased water rights transfers cause irrigated acreage to decline at a massive rate in Scenario B, causing agricultural crop revenues to drop rapidly. This can be evidenced by previous studies by the authors (Dozier et al. 2017; Wostoupal et al. n.d.). Further investigation into the reinvestment of water sales revenues into local economies is necessary to fully enumerate the impacts of rural-to-urban water transfers on rural expenditures.

Empirical cumulative distribution functions were evaluated for the year 2050 across Scenarios A-B (Figure 14). It is assumed all outcomes are equally likely for the calculation of empirical cumulative distribution functions. Interestingly, the stability of municipal costs for Scenario B are much lower than Scenario A, which can be evidenced by the large range of expected municipal costs. Scenario A, no longer dependent upon water rights transfers, creates a hedging effect in which the expected municipal costs vary by about $6.5 billion[1]. Instability in municipal costs for Scenario B, likely due to uncertainty in infrastructure costs, cause costs to vary by more than $11 billion[6]. Although Scenario A includes the same uncertainty in infrastructure costs the adoption of water management technologies allows costs to be less sensitive to parameter uncertainty.

Risk-averse municipalities planning for future water supplies should aim to protect their water management plans against cost, population growth, and water supply uncertainties. As these results show, a method for hedging these bets is to incentivize the adoption of water management practices. Not only do costs remain lower across all modeled years, the stability of expected costs are more consistent with less uncertainty. Further, creating a wedge in growing household demands through technology investments allow for the acquisition of water at future timescales. These lags in water transfers will allow rural and municipal communities to plan for imminent population growth. Combined with lower costs and less risk the adoption of water management technologies helps to create an ideal solution for semiarid regions experiencing severe water scarcity.

[6] Based on the difference between the 95th and 5th percentiles of expected municipal cost in 2050.

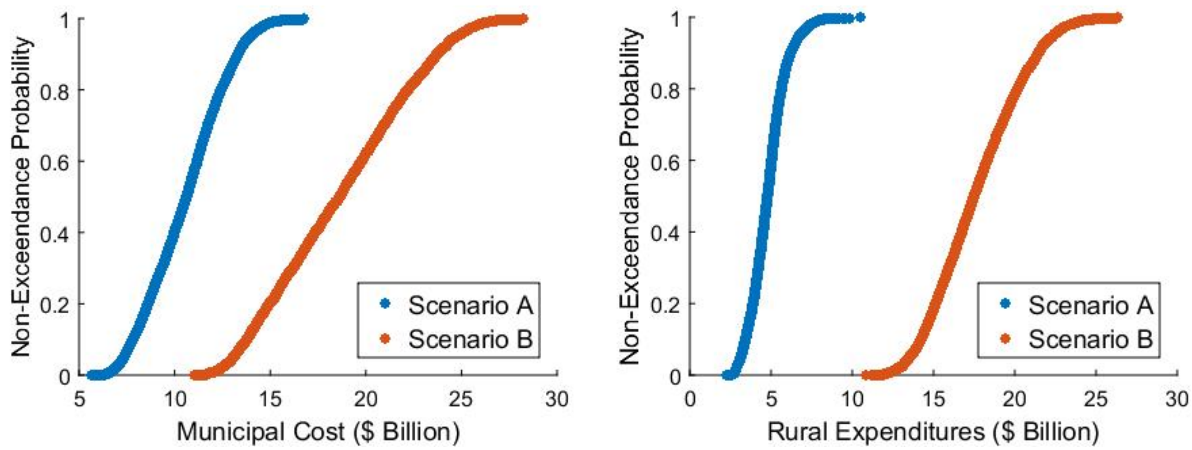
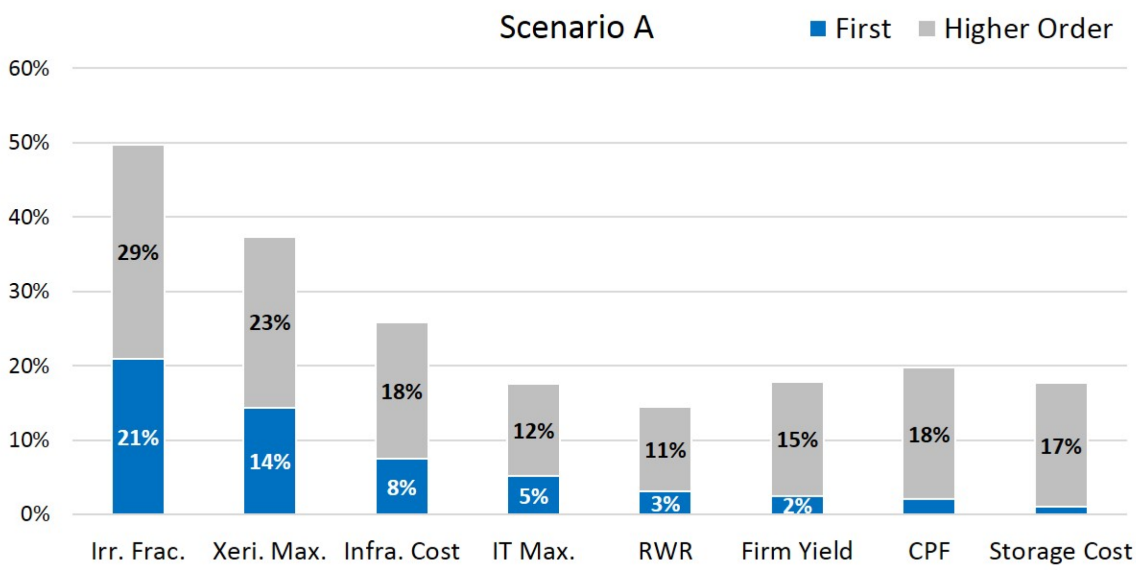


Figure 14 – Empirical cumulative distribution functions for A) municipal cost and B) rural expenditures for Scenarios A-B in the year 2050.

The selection of critical parameters for total irrigated acreage in 2050 can be explored by calculating first and higher order sensitivity indices (Figure 15). Water management technology parameters, particularly in regards to xeriscaping, are critical to keeping irrigated acreage in production for Scenario A. In particular, the irrigation fraction and xeriscaping maximum exhibit the highest first and higher order sensitivity indices. Because household outdoor water use represents a large fraction of total municipal water use, the irrigation fraction becomes one of the most sensitive parameters. The xeriscaping maximum parameter, which characterizes the upper bound on the amount of water than can be saved from xeriscaping, is the second-most critical parameter. Higher order sensitivity indices for both xeriscaping parameters indicate large interactions between other parameters in the model, further enhancing their importance. Institutional and water supply parameters of raw water requirements and firm yield ratios are less important but create considerable interactions between parameters. Lastly, agricultural parameters of the irrigation technology maximum and crop production functions have significant higher-order indices but are irrelevant for first order effects.

These results highlight the effectiveness of xeriscaping to maintain irrigated acreage and rural economies. One key parameter which causes minimal sensitivity to irrigated acreage is the xeriscaping cost parameter. Until xeriscaping costs outweigh the price of water transfers xeriscaping will remain adopted. Therefore, incentives and rebates for xeriscaping lawns can be better estimated based on current water price transfers. This may allow for increased adoption and buy-in from households. Runfola et. al. has shown lawn area is a significant indicator of annual household water use (Runfola et al. 2013). Though not investigated directly in this study, municipalities can use both lawn area and the irrigation fraction as methods to lower household demands. Further, these results and those of Runfola et. al. highlight the importance of land-use planning when managing water resources. Higher density housing structures, such as those found in dense urban areas, may lower lawn size and influence future water supply planning.



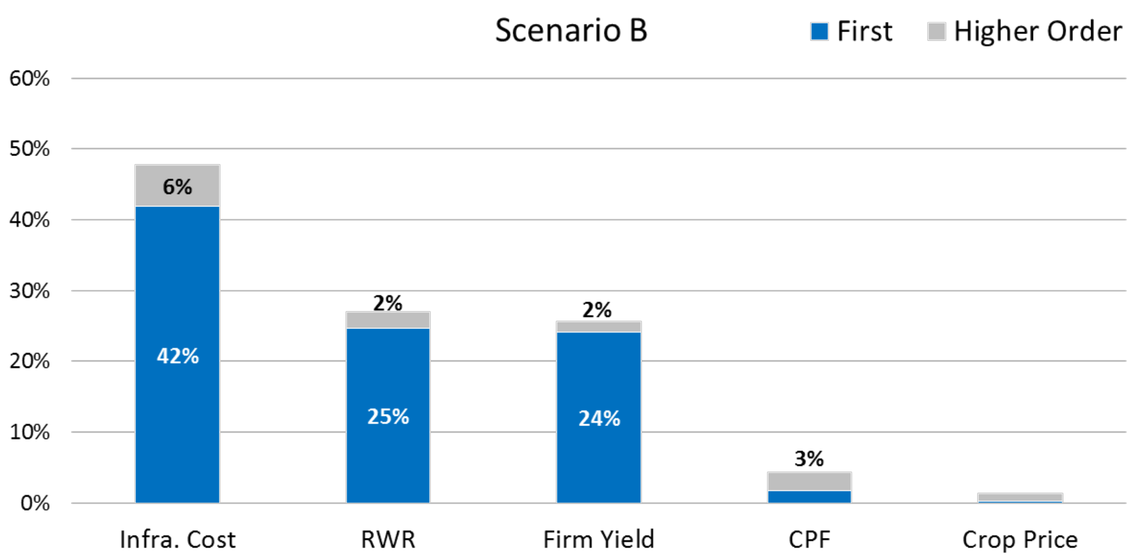


Figure 15 – First and higher order sensitivity indices for irrigated acreage in 2050 for A) Scenario A and B) Scenario B.

Infrastructure cost, raw water requirement, and firm yield ratio parameters are the most critical parameters to sustaining irrigated acreage in 2050 for Scenario B. All three parameters yield high first order sensitivities with minimal interactions. These sensitivities predominately affect the municipal sector in regards to water transfer costs, amount of transfers needed, and the firm yield ratio of water supplies. These effects ripple through the agricultural sector and directly impact irrigated acreage – a key indicator of rural economic health. Firm yield ratios are largely dependent on inter-annual fluctuations and climate and thus cannot be directly managed by municipalities. However, infrastructure costs and raw water requirements are two areas where local municipalities and state governments can influence. To increase the amount of irrigated acreage in production it is important that these two controls be considered. Household water conservation can allow municipalities to decrease current raw water requirements and their impact on rural areas. Additionally, efficient water conveyance structures can be constructed and improved. Selecting policy and designing future water plans which influence these two parameters will directly impact rural economies. Through these effects the tradeoffs of future policy and water supply planning can be considered for both municipal and agricultural communities.

Thresholds of expected municipal costs in 2050 for each scenario were quantified using regression classification trees (Figure 16). Each branch represents a decision based on the parameter given in each node. These decisions, or branches, quantify where important tipping points occur. Further, these decision trees provide a method for planners to evaluate outcomes based on the parameters found in Figure 5 and identify where important thresholds are expected. Node densities for each terminal node are estimated based on the total number of regressed values for each decision tree. Generally, density values tend to be uniform across expected terminal values, indicating the decision trees are a good indicator of where thresholds occur.

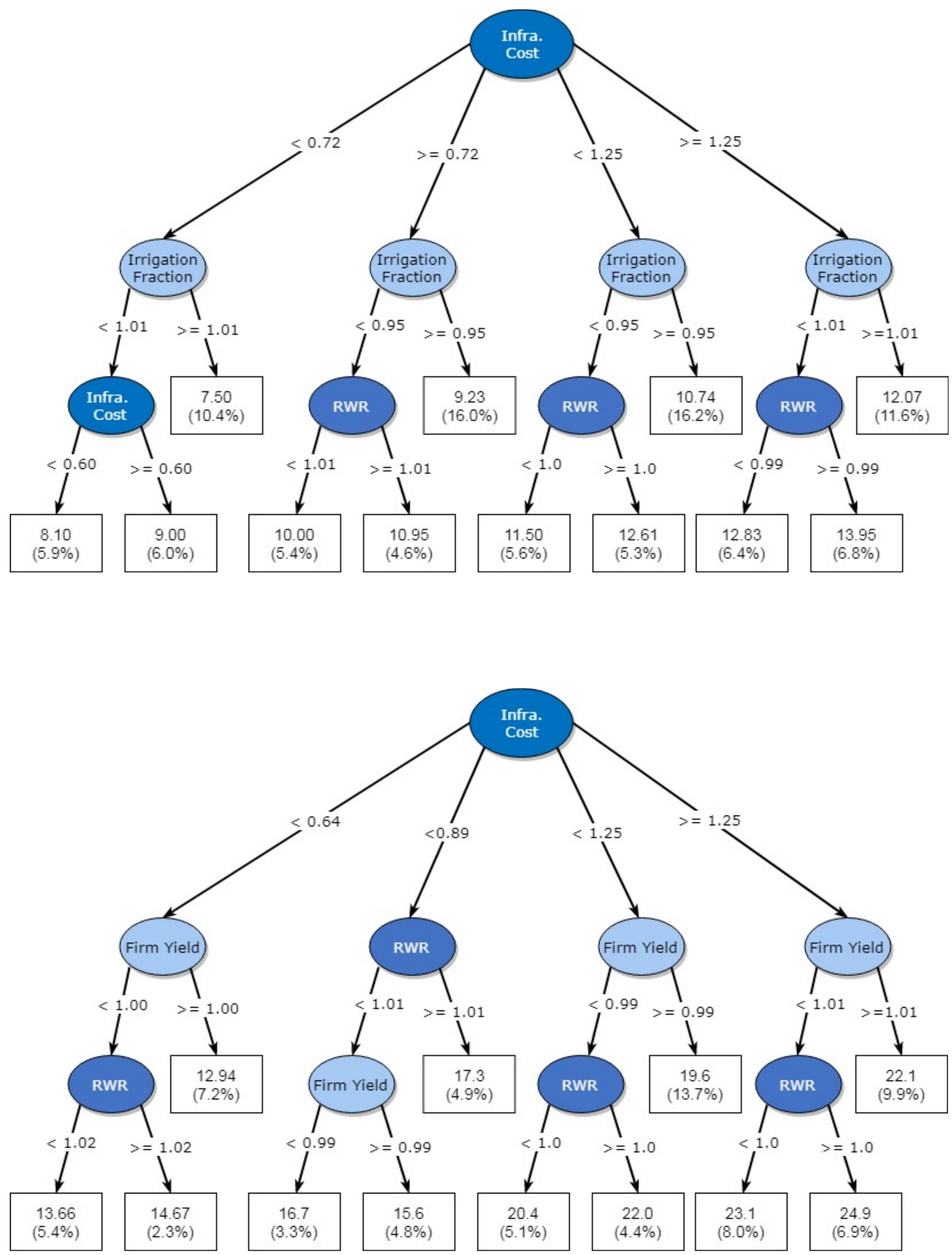


Figure 16 – Decision trees for total municipal cost ($ billion) for Scenario A (left) and Scenario B (right) in 2050. Top values in each terminal represent the expected value of municipal cost, while the bottom values represent the node density.

Thresholds of expected municipal costs for Scenario A are a result of the variability in infrastructure costs, irrigation fractions, and raw water requirements. These parameter uncertainties create an extensive range of municipal costs between $7.5-14 billion. Initial branches begin with infrastructure costs ranging between +25% and -28% with resulting costs varying by about $6.5 billion. Because infrastructure costs are the top branch of the decision tree they are the most important parameter in creating cost thresholds. The irrigation fraction is the second most important parameter for classifying thresholds. Interestingly, decision tree branches only vary between -5% and +1% from expected values and result in large municipal costs fluctuations. With 25% higher infrastructure costs and a 1% difference in the irrigation fraction parameter municipal costs can differ by over $2 billion. These small variations represent large thresholds in municipal costs and further highlight the importance and effectiveness of household water conservation through xeriscaping. Alongside estimating infrastructure costs accurately, additional research should be conducted to gain more efficiency from xeriscaping. Lastly, small variations in raw water requirements result in expected municipal costs varying by nearly $4 billion. Because raw water requirements are the main driver of household demand it is of little surprise small variations result in large changes to municipal costs. However, revisiting the calculation of raw water requirements should be a key priority for municipalities to accurately define future household demands.

Thresholds of expected municipal costs for Scenario B in 2050 are defined by uncertainty in infrastructure costs, firm yield ratios, and raw water requirements. Wide variances are observed from municipal costs, ranging between $13-25 billion based on changes between -36% and +25% in infrastructure costs. Parameters which affect the expected availability of water supplies and household demand represent the next largest thresholds. It is important to note these parameters are extremely sensitive due to high first order indices (Figure 15), small windows of model feasibility, and thresholds hinging on parameter variability of only 1-2%. As such, firm yield ratios differing by only 1% can lead to costs varying by almost $9 billion. Additionally, small changes in raw water requirements can result in increases to municipal costs by almost $2 billion. In combination with the effects of raw water requirements on municipal cost for Scenario A, it seems increasingly imperative to accurately define their values and update calculations frequently. Doing so would result in more accurate, and potentially lower, municipal costs. In general, municipalities can better plan their expected costs, budgets, and policies with accurate estimates in firm yield ratios, infrastructure costs, and raw water requirements.

Decision scaling plots in regards to municipal costs in 2050 for Scenario A and B were created as shown in Figure 17. These plots allow for the exploration of wider uncertainty in raw water requirements and firm yield ratios. Decision scaling plots were created using a high resolution grid of raw water requirement and firm yield ratio values. Contours were fit to a linearly interpolated surface given values of raw water requirements, firm yield ratios, and municipal costs in 2050. Colored areas represent ranges of municipal costs while regions in white represent model infeasibility. Small fluctuations in linear contour lines are due to interpolation techniques and optimization instability. The x-axis represents the average firm yield ratio for the entire SPRB and the y-axis represents the average raw water requirement across all modeled sub-regions. The red square characterizes current values of firm yield ratios and raw water requirements. Municipalities can use these decision scaling plots as a method to explore expected costs given changes in the future availability of water.

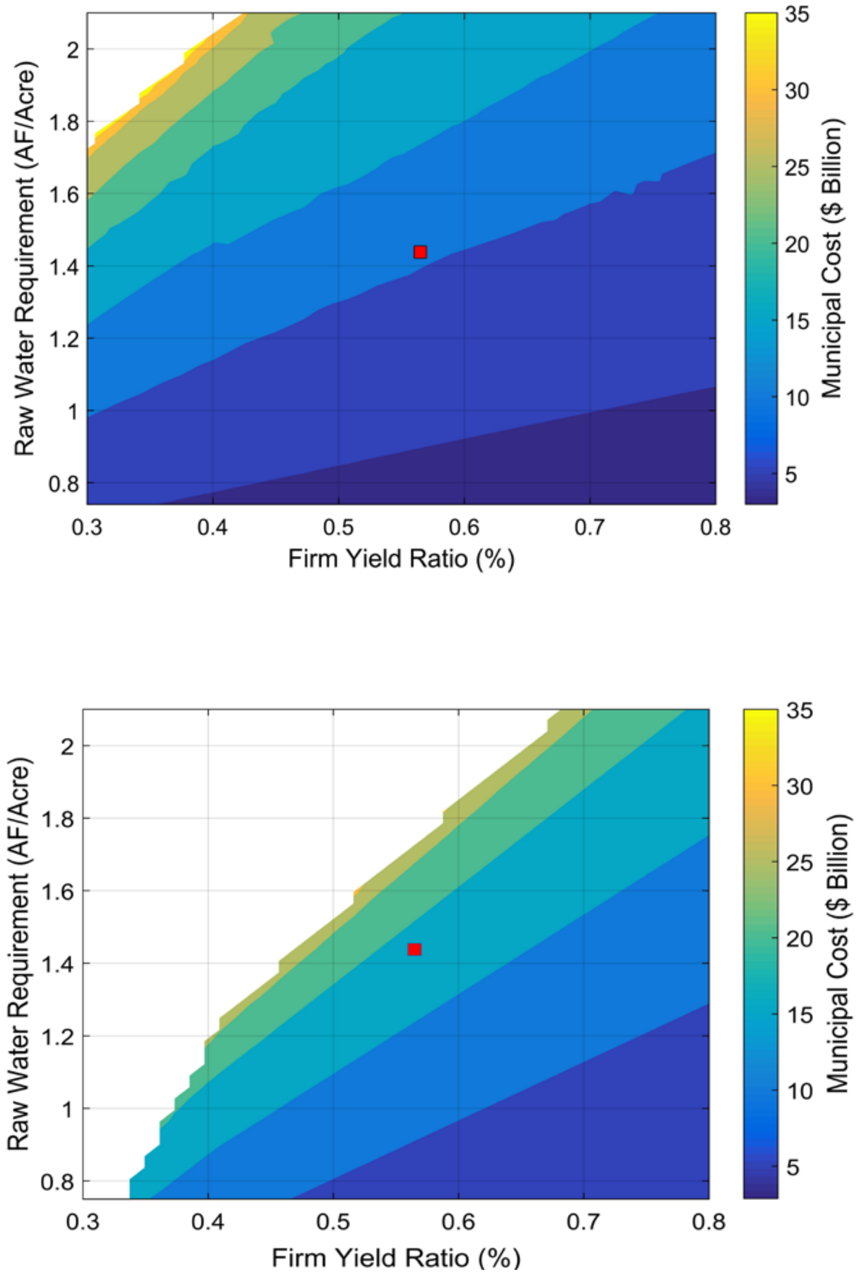


Figure 17 – Decision scaling plots for municipal cost in regards to Scenario A (left) and Scenario B (right). The red square represents current (2010) conditions.

While municipalities and policymakers may have little control over firm yield ratios due to hydrologic variability, raw water requirements can be used as a key method for influencing municipal cost. Currently, raw water requirements in the SPRB average near 1.44 AF/acre. As per capita demands decrease, particularly by way of water conservation, reductions in raw water requirements are feasible. Using this flexibility in raw water requirements allows municipalities to better manage expected costs given future changes in climate and water availability. Modeled municipal costs increase complementary to decreases in firm yield ratios and increases in raw water requirements. When municipal costs are low (i.e. $2-10 billion) changes in raw water requirements drive fluctuations in cost. However, once expected municipal costs increase past $15, further changes in municipal cost are largely driven by changes in firm yield ratios. These results can be evidenced by contour slopes corresponding to the distribution of municipal costs. In order to keep costs low when planning for 2050 municipalities can use their influence over raw water requirements to maintain lower costs. Current municipal costs are estimated near $10 billion, indicating the opportunity to maintain low municipal costs in the future given available decreases in raw water requirements.

Decision scaling plots for Scenario B display much higher municipal costs and regions of infeasibility. With less water management technologies to quench municipal demands it is of little surprise municipal costs rise. However, the lack of opportunity to keep costs low is important. Contour lines for Scenario B are much steeper than Scenario A, indicating more vulnerability to uncertainty and less ability to cope with these uncertainties. Further, areas of infeasibility begin after a mere 10-15% absolute decrease in the firm yield ratio when holding raw water requirements constant. As such, the adoption of water management strategies can lower costs and increase the feasibility to keep future costs low.

Once municipal costs rise due to decreased firm yield ratios or increased in raw water requirements, municipalities lose some of their control over future costs. Maintaining lower costs in the short run will allow municipalities to incorporate a safety factor into their planning portfolios. This water planning safety factor can protect municipalities against future climatic, institutional, and socioeconomic uncertainties. However, the safety factor is entirely dependent on the feasibility to change raw water requirements. Attaining feasibility to lower raw water requirements become available when households adopt water conservation tactics, particularly xeriscaping. Providing incentives and education to increase buy-in for household water conservation will allow decreases in raw water requirements, and subsequent increases in the water planning safety factor. Further, recalculating currently used raw water requirements to account for reductions in per capita demand will increase the magnitude of this safety factor and allow for lower municipal costs.

3.4 Conclusions

Investments in new supplies, agricultural irrigation technology, and household conservation can substantially lower municipal costs. Driven primarily by future urban demands, municipal costs decrease due to the adoption of water management strategies. These strategies create demand “lags” which offset future increases in water prices, hence lowering water acquisition costs. Further, water management strategies create more stable municipal costs over longer timescales. As households conserve municipal costs stabilize and become less susceptible to increases from future uncertainty. As such, the adoption of water management strategies creates a water supply planning safety factor. This water supply planning factor can hedge municipal investments in future water supplies from extreme weather events, infrastructure cost uncertainty, and population growth.

Several critical parameters remain consistently important across Scenarios A and B for municipal and rural success indicators. Irrigated acreage and municipal costs in 2050 for both scenarios are highly sensitive to changes in infrastructure costs. Such changes in infrastructure costs by more than 25% can lead to changes municipal costs by $6.5 (Scenario A) and $11 billion (Scenario B). Accurately planning and estimating future water supply infrastructure projects is imperative in regards municipal costs, rural expenditures, and irrigated acreage. Xeriscaping yields are the second-most critical parameter for municipal costs and irrigated acreage. As such, the adoption of water management practices are key to the health of rural and municipal communities.

Water supplies and future demands can be largely affected by variations in climate and policy, particularly for in semiarid regions. Although changes in future water supplies are largely out of the control of municipal water planners, policies remain a key tool for influencing municipal costs. Given decreases in urban water demand through household conservation, municipalities can control their future water supply costs by influencing raw water requirements. However, using raw water requirements as a policy tool is only feasible when these costs are low. Once costs rise above a certain threshold they become driven by variations in water supplies. Municipalities should exercise their control over raw water requirements and promote urban water conservation. Doing so allows for lower municipal costs in the long-run, an increased water supply planning factor, and a more sustainable future for water supply planning in semiarid regions.

Uncertainty in future climate, institutional settings, technology, and population growth create great challenges for managing our increasingly-scarce water resources. In this study we coupled an integrated social, ecological, and technological framework with future uncertainty to quantify the effects of uncertainty on water supply planning. However, every study has limitations. Future work should be conducted to fully quantify the effects of urban growth on rural economies, particularly at the urban fringe. Additional uncertainties pertaining to channel conveyance efficiencies can be included. Lastly, the inclusion of other water management strategies, policies, and market institutions should be analyzed.

CHAPTER 4: CONCLUSIONS

Water scarcity driven by population growth, changes in climate, and growing urban water demands is one of the greatest challenges facing our society today. An integrated modeling approach to water resources management is a practical pathway to resolve these challenges. This research uses an integrated modeling framework to evaluate the allocation of water rights in a representative semiarid river basin. Additionally, the effectiveness of water management practices, water markets, and regional policy at combating water scarcity impacts are quantified. All modeled outcomes are evaluated under climatic, institutional, and socioeconomic uncertainty. This research adds to the existing literature to help urban planners prepare for population growth and better manage their freshwater resources.

As growing population growth and subsequent urban demands require increased rural-to-urban water transfers, the adoption of water management strategies are imperative to sustaining rural communities. The adoption of new supply systems, urban indoor and outdoor conservation, and agricultural irrigation technology help to offset growth in urban demands. Xeriscaping yields the greatest tradeoffs when not available for adoption and is considered the most important water management strategy. Lowering these demands through water management strategies helps to decrease the magnitude of rural water transfers and their impacts on local, rural economies. Further, water management strategies cost less to municipal water supply planners.

Water markets affecting water rights transfers are better suited to "free" the market of constraints to allow for a more optimal transfer of water. Although goal-oriented markets increase the magnitude of irrigated acreage, they create massive municipal costs tradeoffs. As such, we recommend a market driven approach to managing water transfers and their required market institutions. Institutionally, the removal of buy-and-dry constraints helps to facilitate the effectiveness of a free water market. With additional options and more flexibility in their use of water, producers can fully optimize their individual use of water. This in turns helps to sustain irrigated acreage and rural communities without large increases to municipal costs. However, irrigated acreage seems to inevitably decline as population growth overwhelms the adoption of water management practices. Coinciding with a population of six million in the SPRB, this signal gives planners time to prepare for such acreage declines.

Uncertainty of model drivers were evaluated using CSaws to assess critical parameters and quantify where thresholds of success indicators occur. When water management strategies are available for adoption parameters effecting the efficiency and maximum available adoption of xeriscaping are the most critical to changes in municipal costs and irrigated acreage. Comparable with Chapter 1, xeriscaping is considered one of the most important parameters for sustaining irrigated acreage and lowering municipal costs due to its high sensitivity to success indicator outcomes. Uncertainty in infrastructure costs, a main indicator of the price of water, can cause municipal costs to differ by almost a factor of two. As such, the accurate estimation of infrastructure costs is imperative for water supply planning portfolios.

Using raw water requirements municipalities can control their future costs in water supply acquisition against changes in climate. By keeping costs low in the short run through feasible decreases in raw water requirements, municipalities can maintain their control over future expected costs. As such, decreasing current raw water requirements can create a water supply planning safety factor. This safety factor can help protect municipalities against future fluctuations in infrastructure costs, population growth, and changes in climate.

Further research regarding integrative approaches to water resources should include a fully-coupled interaction between water rights and water supplies. This may include combining several surface water, groundwater, and hydrologic models to better assess intra and inter annual fluctuations in water supplies. Additionally, the translation of municipal, agricultural, and environmental water rights to drive the water allocation model should be considered. To better represent the impacts of water transfers on individual communities a fine resolution model is important. Further, a robust economic analysis of the effects of water transfers, particularly at the urban fringe, is key to analyze the effects of future institutions and policy. Lastly, additional water management strategies (e.g. green infrastructure, greywater reuse, stormwater capture, etc.), water market institutions, and policies should be evaluated at a fine resolution.

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