JOURNAL OF THE AMERICAN WATER RESOURCES ASSOCIATION

Vol. 58, No. 6 AMERICAN WATER RESOURCES ASSOCIATION December 2022

Colorado Basin Incentive-Based Urban Water Policies: Review and Evaluation

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Research Impact Statement: Over the decades since the SSD Project, the major cities of the CRB have adopted a broad mix of programs to incentivize reduced per capita use and to protect watershed health for urban supply reliability. Per capita water use in major Colorado River Basin cities has decreased and, in many cities, overall urban use has declined despite significant population growth. Incentive-based demand reduction programs and watershed protection policies have been evaluated in varying ways, with cost-effectiveness as a primary recommended evaluation approach. Programs reviewed here have been funded and implemented by an innovative mix of city governments collaborating with state and federal agencies and NGOs.

ABSTRACT: Major cities located in the Colorado River Basin (CRB) rely on incentive-based policies to address water use and supply reliability challenges, through programs provided by cities themselves, by state and federal agencies, and by NGOs. This review examines water use trends across cities, the phenomena of declining per capita use, and finds that most large cities have adopted municipal rate structures designed to incentivize lower use. A number of urban areas provide incentives to use gray water and effluent for outdoor use and to harvest rainwater. Incentive-based programs to protect watershed health have become a water supply strategy implemented through programs and partnerships across the CRB. The paper concludes by reviewing the ways that incentive-based urban water policies are being evaluated, and by providing guidelines for designing and evaluating programs to reduce urban use and protect watersheds that provide urban supplies.

(KEYWORDS: urban water; per capita use; rate structures; watershed health.)

INTRODUCTION

In the time period of the Severe, Sustained Drought Study, policies to reduce urban water use and to protect forests and watersheds for improved urban supply reliability were rare in the Colorado River Basin (CRB). Much has changed in the intervening 25 years. This article describes a variety of policy instruments that rely on economic incentives to motivate urban water conservation and to address urban water supply reliability challenges. We draw upon our recent research on programs implemented by major cities located in six of the seven CRB states. Major cities considered are those that have high population in the context of their state, thus Phoenix, Arizona and Cheyenne, Wyoming both are included despite their vastly different populations (1.6 million and 65,000, respectively). The programs discussed here are not intended to provide a complete inventory of all incentive-based programs, but rather to give readers a sense of the breadth of types of programs. California provides such a rich array of incentive-based tools, it would necessitate its own article and is excluded in this review. We include not only programs developed by cities themselves, but also examples of state, federal, and foundation-funded programs which offer incentive-based policies to reduce municipal use and enhance supply reliability. Critical funding from the Walton Family Foundation (WFF) and the Water Funders' Initiative helped jumpstart an array of such initiatives in the basin (Jacobs 2019).

To provide context, we first compare demographic and water use trends across CRB cities. Later we reference these trends to discuss the effectiveness of the programs examined. We then discuss programs to

Paper No. JAWR-21-0031-P of the *Journal of the American Water Resources Association* (JAWR). Received March 3, 2021; accepted June 4, 2022. **Discussions are open until six months from issue publication**.

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Citation: Colby, B.G., and H. Hansen. 2022. "Colorado Basin Incentive-Based Urban Water Policies: Review and Evaluation." *JAWRA Journal of the American Water Resources Association* 58 (6): 1098–1115. https://doi.org/10.1111/1752-1688.13041.

reduce per capita or overall urban water use, such as water rate structures and incentives for residential turf retirement, xeriscape installation, urban stormwater capture, rainwater harvesting, and gray water and effluent reuse. We consider ways in which demand reduction incentives are being evaluated and provide guidelines for designing and evaluating programs to reduce urban use. We then examine municipal programs for forest and watershed health, evaluations of these programs, and offer guidelines for designing and evaluating these supply reliability programs.

Our review of demographic and water use trends in eight CRB cities finds that all experienced notable population growth over the last 20–30 years, while decreasing per capita use — and in some cases reducing total use (Salt Lake City Public Utilities 2020; City of Phoenix 2022). The Great Recession (the longest United States [U.S.] recession since World War II, 2007–2009) is credited with some of the observed decline in urban water use (Yoo et al. 2014; Bennett and Kochhar 2019). However, once cities began to rebound, water usage remained below pre-recession levels. This suggests that demand reduction policies may be contributing to lower per capita use.

Quantifying the contribution of policies to changes in use necessitates a completely different approach based on detailed empirical data and analysis. Empirical studies have been conducted for specific urban areas (reviewed under *Rate Structure Effects, Price Elasticities*) (Figueroa et al. 2013; Price et al. 2014; Stoker and Rothfeder 2014; Yoo et al. 2014; Fullerton and Cárdenas 2016; Brelsford and Abbott 2017; Clarke et al. 2017; Mayer 2017; Yoo and Perrings 2017; Brent 2018; Buchanan 2018; Luby et al. 2018; Garcia et al. 2019; Rupprecht et al. 2020). We are not aware of a study that provides comparative empirical analysis across cities of the CRB, though Bruno and Jessoe (2021) summarize elasticity estimates across prior studies and Richter et al. (2020) survey some CRB water utilities about factors that led to decreased per capita water use while populations grew.

Our review finds that programs to protect watersheds and water supply reliability have been implemented by many of the basin's cities. A variety of evaluation approaches are being utilized to assess effectiveness.

To focus on incentive-based water policy tools, it is useful to summarize how these are distinguished from conventional command and control (C&C) instruments. There are four types of distinctions: (1) incentive-based tools influence water use indirectly through economic signals, while C&C policies set explicit directives (Olmstead and Stavins 2009); (2) incentive-based tools allow water users' flexibility in adapting behavior and technologies (Stavins 2003); (3) incentives for innovation and adoption of new conservation tools are stronger when water users can create and/or choose lower cost approaches; and (4) marginal costs of conservation efforts move toward becoming equalized among similar types of water users, approaching the "equi-marginal" standard for economic efficiency (Stavins 2003). Under C&C policies, water users are held to the same standards regardless of their comparative marginal costs of reducing water use (Olmstead and Stavins 2009). While this article focuses on incentive-based policies, command-and-control policies play a key role where political and legal considerations make incentive-based tools difficult to implement or ineffective.

In addition to the policies discussed here, there are many other incentive-based initiatives in the CRB. These include innovative water trading agreements to share shortages and myriad incentives to reduce water use in agriculture, improve water quality, settle Native American water claims, and provide water for environmental needs. These other important incentive-based initiatives in the CRB are not covered in this article focused on urban water.

POPULATION AND WATER USE TRENDS

We begin by reviewing urban population and water use trends, to place in context the programs we later describe and evaluate. Many CRB cities have managed to decrease their per capita water usage in the midst of growing populations, as the following examples demonstrate. Table 1 shows the actual population in each city in 2000, 2010, and 2020 and the growth rate in that year. All the cities experienced positive growth over this period.

Tables 2 and 3 show trends in water use in eight CRB cities. Note the significant changes in per capita use (measured in gallons per capita per day [GPCD]) in all cities for various time periods. There is a wide range of increases and decreases in total use measured in acre-feet (AF) across cities and time periods, with a number of notable declines in total use during periods of significant population increase.

In Table 1, we see that all eight cities included in our review experienced population growth over the last 30 years. If per capita use were constant, total water usage in these cities must increase to account for a higher population served. However, Table 2 reveals that GPCD has decreased across the board. Where total use is not decreasing, it is at least not rising at the same rate as the population. Table 3 shows that to be the case for all cities except Phoenix in 2020. The rate of change for population in Phoenix is 11% while use rose by 14% over the 2010–2020

TABLE 1. Municipal population trends.

	2000		2010		2020	
Cities	Population	% Change	Population	% Change	Population	% Change
Albuquerque	448,627	16	545,559	22	564,559	3
Cheyenne	52,763	5	59,466	13	65,132	10
Denver	554,636	19	600,158	8	715,522	19
Las Vegas	478,868	85	583,756	22	641,903	10
Phoenix	1,320,994	34	1,445,632	9	1,608,139	11
Salt Lake City	181,456	13	186,440	3	199,723	7
Santa Fe	61,805	8	67,947	10	87,505	29
Tucson	486,591	16	520,116	7	542,629	4

Note: (1) Population data obtained through the United States Census Bureau (n.d.). (2) Census Bureau uses city boundaries to determine population. (3) Percentage change is calculated over the 10-year period preceding year noted at top of column.

TABLE 2. Municipal water utility gallons per capita per day use trends.

	2000		2010		2020	
Cities	Use (GPCD)	% Change	Use (GPCD)	% Change	Use (GPCD)	% Change
Albuquerque	216		157	−27	121	−23
Cheyenne	192		157	−18	134	−15
Denver	220		163	−26	144	−12
Las Vegas						
Phoenix	205		165	−20	155	−6
Salt Lake City	285		210	−26	206	−2
Santa Fe	137		104	−24	93	−11
Tucson	165		139	−16	119	−14

Note: There may be inconsistencies between the city boundaries and the utility service area, since many municipal utilities service populations adjacent to the city. (1) Albuquerque: GPCD as reported by the ABCWUA (n.d.). (2) Cheyenne: Total city use GPCD as reported by City of Cheyenne Board of Public Utilities (n.d.). Since the GPCD for 2000 was not available, the value reported under 2000 is from 2002. (3) Denver: GPCD as reported by Denver Water (n.d.). (4) Las Vegas: We were not able to obtain the GPCD values from Las Vegas Valley Water District. (5) Phoenix: Approximate GPCD values as reported in a graph by City of Phoenix (2022). (6) Salt Lake City: Total GPCD as reported by Salt Lake City (n.d.). Since the GPCD value for 2020 was not available, we report the 2018 in its place. (7) Santa Fe: Total water supplied GPCD as reported by the City of Santa Fe (n.d.). (8) Tucson: Total potable GPCD as reported by City of Tucson (2020).

period. This result may stem from inconsistencies in definitions of city borders for the population estimate and municipal water service areas. Phoenix still reduced their GPCD by 6% across this period.

INCENTIVES FOR REDUCED URBAN WATER USE

“Reduced use” and “water conservation” are broad terms in urban water management. Frequently, they refer to reduced application of water for a specific purpose — such as reduction in water applied to outdoor landscape irrigation or lower volumes of metered household water used for flushing toilets. This common way of characterizing urban water “conservation” is not satisfying when considering basin-wide water consumption because a portion of the water delivered to households and businesses returns to the local water supply. For example, potable water is delivered for household use and charges appear on water bills. However, once used and treated, this water can be used for other purposes — such as landscape irrigation and replenishing stream flows and aquifers. A portion of urban outdoor landscape use evapo-transpires (is consumed) and a portion returns to local groundwater or surface water.

Ideally, urban “conservation” programs would focus on reducing the consumptive use portion of urban water use — the portion of water delivered to households and businesses that does not return to the regional water supply. However, programs generally focus upon (and measure) water delivered rather than water consumed. This article necessarily follows that convention, while advocating for a consumptive use focus essential for considering a regional water balance.

We consider programs funded in a variety of ways. In addition to funds provided by municipal, state,

TABLE 3. Municipal water utility total use trends.

	2000		2010		2020	
Cities	Use (AF)	% Change	Use (AF)	% Change	Use (AF)	% Change
Albuquerque	101,502		91,913	−9	87,254	−5
Cheyenne	13,963		11,760	−16	11,607	−1
Denver	256,514		213,887	−17	214,942	0
Las Vegas			311,910		320,090	3
Phoenix	320,000		295,000	−8	335,000	14
Salt Lake City	89,138		75,755	−15	77,867	3
Santa Fe			9,086			
Tucson	122,500		104,000	−15		

Note: (1) Albuquerque: Total water billed as reported by ABCWUA (n.d.). (2) Cheyenne: Total water sold as reported by City of Cheyenne Board of Public Utilities (n.d.). Since the total water sales for 2000 was not available, the value reported under 2000 is from 2002. (3) Denver: Total water use as reported by Denver Water (n.d.). (4) Las Vegas: Total water billed as reported by LVVWD (n.d.). We use the 2012 value in place of 2010 and we are unable to obtain the 2000 value. (5) Phoenix: Approximate water produced values as reported in a graph by City of Phoenix (2022). (6) Salt Lake City: Total water sales as reported by Salt Lake City (n.d.). Since the GPCD value for 2020 was not available, we report the 2018 in its place. (7) Santa Fe: Total utility customer use as reported by the City of Santa Fe (n.d.). Only the 2010 value was reported. (8) Tucson: Total use as reported by City of Tucson (2020). We use the 2002–2007 average use for 2000 and the 2012 use for 2010. (9) All use values were converted to acre-feet (AF).

and federal governments, philanthropic foundations contribute toward urban water initiatives throughout the CRB. For example, Resource Central, a non-profit based in Boulder Colorado uses WFF grants to fund toilet replacement and turf replacement programs with plans to eventually scale to a statewide level (Runyon 2018; Larson 2020; Resource Central n.d.; Walton Family Foundation n.d.). The Nina Mason Pulliam Charitable Trust began a five-year campaign in 2020 to assist the Verde River in Arizona (Nina Mason Pulliam Charitable Trust n.d.). This \$19.5 M initiative, in partnership with The Nature Conservancy, provides funding for reducing municipal groundwater use and expanding effluent and stormwater use (Nina Mason Pulliam Charitable Trust n.d.).

The CRB cities reviewed here include a wide range of incentive-based programs to reduce water use. The section that follows covers rate structures and their effects, including price elasticities.

Municipal Rate Structures

Municipal Rate Structures Overview. The term rate structure refers to the overall manner in which a city charges for water, including different prices per unit at differing seasons and/or use levels. Municipal rate structures can be designed to address several different goals. Revenue generation to cover water provider costs is key. Conveying incentives to reduce water use (overall or seasonally, outdoors or indoors) now also is a goal of many urban water providers.

The municipal water rate structures of eight CRB cities are categorized in Table 4 by their units of measure for metered water, their rate structure, seasonal provisions, fixed monthly fee, and any other charges or fees. All of the cities included in this research meter water delivered to households and charge customers based on volume used. The Albuquerque Bernalillo County Water Utility Authority (ABCWUA) recently received a \$2 M award from the New Mexico Water Trust Board for an Advanced Metering Infrastructure (AMI) project to replace 16,000 water meters (New Mexico Finance Authority n.d.).

There are two standard units of measure used to charge residential customers for water delivered: 1,000 gallon units and 100 cubic feet (CCF) units (748 gallons). Cities can charge a flat fee for each metered unit of water or create tiered rate structures with higher prices for higher levels of use to encourage conservation. All cities reviewed charge a flat monthly service fee. Among the major cities reviewed, Phoenix alone includes a base water allotment in their monthly charge. Another water rate tool that municipalities use is seasonal rates for winter and summer when water needs are different. Many cities pass along fees for state regulations or environmental charges to customers, through additional charges on monthly water bills.

In Table 4, there are several units of measure for volumetric water charges. Nearly all of the rate structures are tiered. Phoenix and Albuquerque charge a single usage rate but include seasonal adjustments. Two other municipalities price water depending on the season. Salt Lake City discounts water in winter while Santa Fe charges more in summer. Four cities include miscellaneous fees in their rate structure. Some, like Albuquerque and Las Vegas, pass on fees from larger authorities such as regional water districts or state surcharges. The effect of these rate structures on

TABLE 4. Municipal rate structures.

Cities	Units of measure	Type of rate structure	Seasonal provisions	Fixed monthly fee	Other fees and charges
Albuquerque	100 cubic feet	Single usage rate	Yes, April–October discount for 100%–150% of winter mean water usage	Yes	$0.024 per unit Water-State Surcharge. Conservation surcharge for excess use from May–November
Cheyenne	1,000 gallons	Tiered rates	No	Yes	No
Denver	1,000 gallons	Tiered rates	No	Yes	No
Las Vegas	1,000 gallons	Tiered rates	No	Yes	Yes, fees levied by Southern Nevada Water Authority
Phoenix	100 cubic feet	Volumetric charge for units above monthly allocation	Yes, low, medium, and high season rates	Yes, includes 6 units October–May and 10 June-September	Environmental charge of $0.62 per unit
Salt Lake City	100 cubic feet	Tiered rates	Yes, flat rate November through March	Yes	No
Santa Fe	1,000 gallons	Tiered rates	Yes, first tier usage is greater May–August	Yes	No
Tucson	100 cubic feet	Tiered rates	No	Yes	Central Arizona Project Fee — $0.70/CCF and Conservation Fee — $0.10/CCF

water demand can best be understood through econometric studies that estimate own-price elasticity of demand in different seasons (Kenney et al. 2008; Klaiber et al. 2014; Yoo et al. 2014; Clarke et al. 2017). Findings from some of these types of studies in the CRB are presented later.

Throughout the CRB, cities are adopting "smart" metering technology that allows more frequent and accurate reads of water use and storage of the data (City of Aurora n.d.). Many Colorado cities including, Denver, Colorado Springs, and Aurora have completed or are in the process of upgrading to AMI (McCurley 2009; City of Aurora n.d.; Denver Water n.d.). Santa Fe began their own AMI project in 2015 to replace meters for 34,000 customers, in their first full meter exchange in 30 years (Water Conservation Staff 2015).

While metering is now a widespread practice for major cities, there are some regions where some types of municipal service are unmetered. "Secondary water," as it is called in Utah, is non-potable water used for irrigation. The original water delivery system of canals built in the early days of Utah's settlement provides unmetered secondary water to 61% of residential customers of the state's urban water suppliers (Weiser 2018). Rates are typically $10–15 dollar a month regardless of the water volume used. This water supply may only be used for outdoor watering (Weiser 2018). The Salt River Project (SRP) in central Arizona manages a similar system under which residential customers receive water through canals, unmetered. Customers order water in five-minute increments up to their maximum allotment (determined by acreage of land) and pay a flat fee for water delivery and upkeep of facilities (SRP n.d.). Brent (2018) reports that residential SRP customers can pay an annual fee of $60 for non-metered flood irrigation service.

Luby et al. (2018) examine water pricing schemes of 35 American cities including three in the CRB; Phoenix, Denver, and Las Vegas. They find that cities facing greater water scarcity, like the ones in the CRB, actually had lower water prices than cities without scarcity. Two vital measurements are the average price paid for essential water use by a household (8 CCF) and for additional use (8–16 CCF). The authors present increasing block rate structures as a solution to address scarcity and equity issues. Utilities would be expected to charge higher prices for the additional use category water. The essential use average water bill for the first 8 CCF of water is $45.79 for Denver, $43.02 for Las Vegas, and $26.55 for Phoenix. The additional 8 CCF of water above essential use are charged at $41.78 for Denver, $14.44 for Las Vegas, and $41.20 for Phoenix (Luby et al. 2018).

Table 5 provides specific rate structure information for the eight cities examined. For comparison, all units and rates have been converted to 1,000 gallons. Unique provisions and attributes in certain cities are further discussed in the following sections.

Of the rates examined and converted for comparison, Santa Fe has the highest Tier 1 rate by far — at $6.06 per 1,000 gallons. In fact, this Tier 1 rate is greater than five of the cities' highest tiered rates. Las Vegas has the lowest Tier 1 price of $1.34 per 1,000 gallons. However, their first Tier also includes the smallest volume of the cities compared.

TABLE 5. Municipal pricing (prices per unit volume).

	Tier 1		Tier 2		Tier 3		Tier 4	
Cities	Volume (gallons)	Price (per 1,000 gallons)	Volume (gallons)	Price (per 1,000 gallons)	Volume (gallons)	Price (per 1,000 gallons)	Volume (gallons)	Price (per 1,000 gallons)
Albuquerque	All units	$2.70						
Cheyenne	0–6,000	$4.42	6,000-24,000	$5.46	24,000-42,000	$6.78	42,000+	$8.44
Denver	0–average winter consumption (AWC)	$2.36	AWC + 15,000	$4.25	Greater than AWC + 15,000	$5.66		
Las Vegas	0–167	$1.34	167–334	$2.39	334–667	$3.55	667+	$5.27
Phoenix	0–4,488 October–May and 0–7,480 June–September	$1.04 October–May and $0.62 June–September	All usage above Tier 1	$4.29 December–March, $5 April, May, October, November, and $5.48 June–September				
Salt Lake City	748–7,480	$1.84	8,228–22,440	$2.51	23,118–44,880	$3.47	44,880+	$3.70
Santa Fe	0–7,000 September–April and 0–10,000 May–August	$6.06	All usage above Tier 1	$21.72				
Tucson	748–5,236	$2.77	5,984–11,220	$5.11	11,968–22,440	$11.10	23,188	$17.32

Note: (1) AWC is the average monthly water consumption for January, February, and March. Minimum is 5,000 gallons and maximum is 15,000. (2) Las Vegas Valley thresholds are different for meter size. The thresholds included in the table are for a 5/8″ m, (3) Phoenix includes a base allotment of water as part of the Monthly Service Charge which varies based on meter size. The prices included are for a 5/8″ m, and (4) All Salt Lake City usage is charged at Tier 1 November thru March.

Rate Structure Effects and Price Elasticities

Increasing block rates charge a higher price/unit as monthly use rises, for tiers above a base level. Luby et al. (2018) note that Santa Fe has reduced per capita consumption by 50% since implementing increasing block rate pricing in 1997. Santa Fe charges $4.43 per CCF for a seasonally adjusted amount of essential use, increasing the rate to $16.25 for additional usage (Luby et al. 2018).

In principle, an efficient water price equals the long-run marginal cost (LRMC) of supply (Olmstead and Stavins 2009). However, given uncertainties over supply costs, this can be difficult to calculate, and urban water prices are often well below the LRMC (Olmstead and Stavins 2009). Since LRMC is an elusive pricing goal, Olmstead and Stavins present more achievable approaches, noting that programs to reduce use can yield net benefits by minimizing long-run costs. Flexibility given to customers from price increases is more cost-effective at reducing demand than non-price demand management programs and higher water prices also incentivize reduced use through technological adoption (Olmstead and Stavins 2009).

Price elasticity of water demand is a key consideration in evaluating the effectiveness of water rates in reducing use. Price elasticity of demand is a numerical measure of how the level of water use responds to a change in water price. For example, a price elasticity of -1.30 indicates that a price increase of 10% decreases water use by 13%. As with the studies summarized here, price elasticity can be measured from data on water use and price for an urban area.

The Great Recession needs to be considered to understand water use trends in CRB cities. Per capita water use is typically lower in periods of economic recession, a result that Garcia et al. (2019) discuss in their review of urban water sustainability in Las Vegas, Los Angeles, and Miami. Yoo et al. (2014) explain that slowed growth during a recession can account for some of the reduction in water use. After the recession, water usage in Las Vegas rebounded — but not to pre-recession levels — indicating water use reduction cannot be attributable to the recession (Garcia et al. 2019). Rupprecht et al. (2020) credit water conservation policies with a 23.3% reduction in water production in Tucson from 2005 to 2015, and they recognize that the recession likely enhanced the effects of these policies.

The City of Tucson over the past two decades has achieved a 37% reduction in GPCD (Making Action Possible n.d.). Tucson's highest GPCD was 121 in 1996 and was 79 GPCD in 2019 (Making Action Possible n.d.). By comparison, the State of Arizona's 2015 average water use was much higher at 146 GPCD (Making Action Possible n.d.). Mayer (2017) notes that between 2005 and 2015, overall water use in Tucson decreased by 23% and that Tucson Water customers pay 11.7% lower rates than if the city had not achieved cost-saving usage reductions from conservation (Mayer 2017).

Clarke et al. (2017) estimate household water demand in Tucson using a Stone-Geary function to estimate price elasticity. While residential water demand is generally price inelastic, demand is highly seasonal in arid regions with notable peaks in summer (Clarke et al. 2017). Mean price elasticity estimates by Clarke, Colby, and Thompson for Tucson customers in 2010–2011 ranged from −0.20 in January to −0.12 in July. The median price elasticity estimates for the same period range from −0.14 to −0.08 indicating an even smaller price response (Clarke et al. 2017). Clarke, Colby, and Thompson's results indicate that outdoor summer water use may not be as responsive to prices as previously thought.

Even with growing population, the City of Phoenix has reduced total water use and per capita consumption (City of Phoenix 2022). In 2014, Phoenix used 180 GPCD, a 29% reduction from 1990 (City of Phoenix 2022). Yoo et al. (2014) estimate the price elasticity of water demand in Phoenix, Arizona using 2000–2008 residential water demand data. Short-run price elasticity from 2000 to 2002 was estimated at −0.661 while the long-run price elasticity was −1.155 (Yoo et al. 2014). Price elasticity is smaller in the short-run because water users have fewer options to adapt to price changes in this shorter time horizon.

Bruno and Jessoe (2021) synthesize price elasticity of water demand estimates in the CRB since 2003 for both urban and agricultural water users. They find a great range of price elasticities, useful to inform policymakers on the role prices can play in reducing use. Elasticity estimates compiled by Bruno and Jessoe (2021) range from −0.10 to −0.76. Dalhuisen et al. (2003) completed a meta-analysis on price elasticity for urban water, finding a mean elasticity of −0.41.

Klaiber et al. (2010) estimate price elasticity of demand for Phoenix, with its increasing block pricing structures. Seasonal price changes for Phoenix are also considered in their study. Klaiber et al. control for housing attributes, external water usage, and socio-economic characteristics using percentiles of water consumption to construct their sample. Price responses are estimated for water consumption in the 10, 25, 50, 75, and 90 percentiles, by month. The study finds that larger water users are more demand inelastic and that dry weather conditions lead to less price responsiveness (Klaiber et al. 2010).

Brent (2018) estimates price elasticity and interaction between landscape choices and residential responses to price changes. He uses monthly metering records for single-family homes in Phoenix from 1998 to 2009 to create a dataset with almost 25 million observations. Brent (2018) finds that higher water rates lead to increased water-efficient landscape adoption. Dry landscape households are less responsive to price compared to the general population. Brent calculates both the intensive margin elasticity estimates where landscape is held constant, and the extensive margin elasticity estimates where residential users may convert their landscape. At the intensive margin, the focus is on short-run water use since irrigators are prevented from making landscape conversion decisions and elasticity effects are direct (Moore et al. 1994; Brent 2018). The extensive margin represents an indirect effect through landscape reallocation since that decision is allowed in the estimate (Moore et al. 1994; Brent 2018). Brent (2018) explains that the intensive margin acts as an upper bound and the extensive margin a lower bound. Households with "wet" landscapes have a demand elasticity of −0.25 and "dry" landscape households have a demand elasticity of −0.20 (Brent 2018). The long-run extensive margin elasticity estimated by Brent (2018) ranges from −0.06 to −0.09. Landscape effects become stronger in the long run.

Fullerton Jr and Cárdenas (2016) forecast Phoenix water usage using monthly data from 2008 to 2014 and distinguish single-family and multi-family residential demand. Price, weather, and income are all important determinants of demand (Fullerton and Cárdenas 2016). Multi-family demand is more price inelastic since renters often do not have their own meter and do not face price signals. Owners of multi-family units respond to increased water price with high-efficiency appliance and fixture installations (Fullerton and Cárdenas 2016). Price elasticity of demand was estimated to be −0.36 for single-family and −0.31 for multi-family use (Fullerton and Cárdenas 2016).

Larson and Perrings (2013) measure willingness to pay for water-related environmental attributes in the Phoenix housing market, to understand the value Phoenix residents place on attributes landscape type and proximity to lakes, streams, and golf courses. Based on housing sales data for the year 2000, Phoenix residents are willing to pay more to purchase a home with greater levels of vegetation (Larson and

Perrings 2013). Overall, residents display a preference for environmental and locational characteristics that require substantial water resources (Larson and Perrings 2013).

Yoo and Perrings (2017) evaluate the economic impact of short-run responses to water availability changes in Phoenix. Responses are more constrained in the short-run so a shift in water availability can have high costs (Yoo and Perrings 2017). They use 2005 U.S. Geological Survey estimates for Phoenix water use and match to IMPLAN Model economic data. Yoo and Perrings evaluate two scenarios of water change and response. In the first, surface water supply is reduced by 1% but water users cannot supplement with other sources of supply — so users relying on surface water must reduce their output. This scenario leads to an estimated loss of \$166.8 M in value added and 2,571 jobs lost (Yoo and Perrings 2017). In the second scenario, groundwater can be substituted for lost surface water, at a cost (Yoo and Perrings 2017). Water users then have a choice in how they respond to cutbacks. Yoo and Perrings estimate the implicit value of surface water as \$10.54/acre-foot and of groundwater as \$14.88/acre-foot.

The City of Albuquerque has achieved reductions in per capita water usage since implementing demand reduction programs in the 1990s (ABCWUA 2016). In 1995, GPCD was 251, falling to a low of 127 GPCD in 2016, a reduction of nearly 50% (ABCWUA 2016). Even with population growth, total water usage decreased 16% from 1996 to 2009 (Price et al. 2014). Residential water use in Albuquerque makes up 61% of all water deliveries (Price et al. 2014).

The City of Santa Fe has had a steady decline in GPCD, to an all-time low of 90 GPCD in 2015 (Water Resources Staff 2015). Demand reduction programs influenced this trend, along with rezoning of utility boundaries and high precipitation (Water Resources Staff 2015).

Salt Lake City has experienced a 27.7% reduction in total water usage since 2001 (Salt Lake City Public Utilities 2020). Per capita residential use declined notably in the early 2000s and has fluctuated since, then trending upward since 2015 (Salt Lake City Public Utilities 2020). Coleman (2009) examines water use data from Salt Lake City between 1999 and 2002 to determine the effects of price and non-price policies. During the study period, water rates were higher in the summer months and increased from year to year (Coleman 2009). Water use was 52% greater in summer months than in winter months. The mean long-run price elasticity for household water demand was −0.485. The short-run mean was less elastic at −0.391 (Coleman 2009). Price elasticity in summer was much higher at −1.445, meaning there was more responsiveness to price during these months (Coleman 2009). The non-price policy evaluated by Coleman was an information campaign, "Slow the Flow, Save H_2O". Coleman estimates the campaign reduced water consumption by 6.673%, but only 1.269% over the long run.

Even though Utah has one of the driest climates, it ranked second highest in the U.S. for per capita water use in 2010 (Stoker and Rothfeder 2014). Stoker and Rothfeder (2014) develop water demand models for Salt Lake City to identify factors that influence urban water use. Stoker and Rothfeder found that parcels with more bedrooms, kitchens, bathrooms, turf, and trees had higher water use. Average water use varied seasonally, highest in summer (Stoker and Rothfeder 2014).

Buchanan (2018) examines Denver water deliveries to evaluate demand reduction efforts in 2000, finding reductions in both overall water usage and per capita use between 2000 and 2016.

The City of Cheyenne has achieved per capita water use reductions of 12% since the implementation of demand reduction programs in 2003, even with a 10% population increase (Figueroa et al. 2013).

Garcia and Islam (2018) attribute Las Vegas' reduced per capita water use of 92 GPCD from 1990 to 2012 to technological changes and increased appliance efficiency. Other factors contributing to declines are increasing marginal prices, and consumer response to water stress in the region (Garcia and Islam 2018). In 1993, Nevada set efficiency standards for residential and commercial indoor plumbing fixtures. Then in 2004, the City of Las Vegas limited turf grass in new residential construction. The population of the city was growing during this time and new residential areas were being built to use less water (Garcia and Islam 2018). Brelsford and Abbott (2017) completed a similar analysis over a shorter time period from 1996 to 2007 to determine drivers of a 55% reduction in Las Vegas water consumption.

The Southern Nevada Water Authority (SNWA) created the Water Conservation Coalition (WCC) in 1995 to foster collaboration with public and private entities (Southern Nevada Water Authority n.d.). The WCC's purpose is for the advancement of water-efficient technology and practices in Southern Nevada businesses, especially those in the hospitality industry (Southern Nevada Water Authority n.d.). SNWA and WCC offer incentives for capital improvements to business through their Water Efficient Technology program (Southern Nevada Water Authority n.d.). The WCC also encourages Las Vegas area businesses to reduce water consumption through the Linen Exchange program, under which hotels change linens and towels less frequently during a guest's stay, and the Water Upon Request program, where restaurants only offer water to customers upon request (Southern Nevada Water Authority n.d.).

Richter et al. (2020) conduct a more general analysis of urban water usage trends in the American West looking at major cities both within and outside the CRB. This study surveyed 20 water utilities about policy drivers and incentives that led to decreased per capita use while populations grew. Cities needed to show both population growth and a decrease in per capita water use to be eligible for study. Across the 20 cities examined, population grew by an average of 21% but water use decreased by 19%. In the CRB, Albuquerque, Denver, Flagstaff, Fort Collins, Las Vegas, Los Angeles, Phoenix, San Diego, and Tucson were studied. The authors found direct connections between residential demand reduction and overall urban savings. Efficiency improvements such as low flow appliances and turf removal are contributing examples (Richter et al. 2020).

Luby et al. (2018) discuss equity issues in water pricing, noting that pricing policies to reduce use can place a disproportionate burden on lower income customers. This is demonstrated by higher price sensitivity in low-income households, which show the greatest reductions with price increases. Luby et al. argue that policymakers should ensure that all residents can afford adequate water, a view further articulated by Klaiber et al. (2014). Klaiber et al. (2014) report that residential water users who consume greater quantities are more price inelastic.

Implications of Demand and Elasticity Studies for Colorado Basin

Our overview of demand studies and trends leads to several observations linked to CRB urban water policies. In the past few decades, CRB cities generally have succeeded in reducing both total and per capita urban use. Policies and programs designed to reduce per capita use have enabled cities to use their water more efficiently in the midst of consistent population growth.

The data available and used in the studies we review do not allow establishing a definitive causal link between policies and changes in per capita demand. However, Luby et al. (2018), Clarke et al. (2017), and other rate structure studies observe that implementing increasing block rates for municipal water reduce per capita demand. The Great Recession is also a contributing factor in reduced demand, with slowed economic activity in the study period for many of the empirical analyses reviewed.

The studies clarify how weather patterns and seasonality impact price responsiveness in urban use. Clarke et al. (2017) present the smaller price responses for water use in the summer and Klaiber et al. (2010) show the same for dry weather. Typically, overall urban water use is higher in summer months as shown by Coleman (2009) and Stoker and Rothfeder (2014). Technological improvements such as high efficiency appliances and their adoption have been one of the bigger drivers in the reduced per capita use (Garcia and Islam 2018; Richter et al. 2020).

The studies reviewed suggest that CRB cities have been able to use water price and other incentives effectively over the past decades to reduce per capita use. However, consumer responsiveness to price and other incentives could be smaller in the future, in cities where much of the easy-to-cut water use has already been trimmed. Elasticity estimates from past studies are not necessarily indicative of future consumer responsiveness to water price. It is necessary to re-evaluate price elasticity regularly over time, using new studies that analyze data on recent response to changes in water price.

Incentives to Reduce Potable Water Use and Outdoor Use

Many CRB cities have implemented programs to reduce potable water use, and to specifically reduce outdoor water use. Policies that reduce potable use save not only water, but also the substantial costs and energy resources embedded in treating water to potable standards and conveying that water within a potable delivery system to households and businesses.

Gray Water Reuse. Residential gray water usage is legal in all CRB states with varying incentives and regulations. Santa Fe, New Mexico is one of the only municipalities examined that offers a rebate for the installation of a Laundry to Landscape Gray Water System to divert water from a residential clothes washer to the home's landscape (City of Santa Fe n.d.). The City of Tucson in Arizona also offers a gray water rebate for residents (City of Tucson 2020). New residential construction in Tucson must configure plumbing to allow for the installation of a gray water system in the future (City of Tucson 2020). None of the other cities examined offer incentives encouraging the usage of gray water. In a review of two CRB cities, Tucson and Denver, Neale et al. (2020) estimate a household gray water system for single residence irrigation can cost $2,300 per household and delivers a minimal reduction in water usage. The authors use the annualized life-cycle cost of water demand reduction strategies using data found from literature or their own estimates of implementation and operational costs. Their total annualized cost of implementing a gray water system is $4.62/kilogallon (Neale et al. 2020).

Effluent Use. Effluent use is common among the larger cities in the CRB. Distribution of treated wastewater can be cost-effective so many utilities are willing to sell effluent for outdoor use at a reduced rate (Wilhelmi and Tucker 2015). Nearly all cities use recycled treated wastewater for irrigation of large turf areas such as golf courses, parks, or sports fields. Golf courses in Phoenix obtain 21% of their irrigation water from effluent, while in Tucson that figure is 54% (Kelley 2018). Tucson provides reclaimed water service to schools, parks, and golf courses. The city estimates that an 18-hole golf course saves $415,000 a year using reclaimed water (City of Tucson 2020). The Palo Verde Nuclear Generating Station purchases reclaimed water in central Arizona to cool the plant (Tenney 2018). The Palo Verde Generating Station is examining using groundwater to reduce their reliance on the treated wastewater by up to 20% (Staten 2020). Switching to groundwater may be cheaper than paying for treated wastewater (Staten 2020). Palo Verde has the funds for the project and is awaiting state and local approval (Staten 2020). Phoenix reuses wastewater for irrigation, and to recharge groundwater aquifers (Tenney 2018).

Pricing is a key aspect of wastewater reuse programs. In a survey of 35 Arizona water utilities, the median volumetric rate for reclaimed water was 77 cents per thousand gallons. This is lower than typical potable water rates, providing an inducement to use reclaimed water (Wilhelmi and Tucker 2015).

Scottsdale (near Phoenix) has agreements with 23 golf courses who provide funding for the Reclaimed Water Distribution System (RWDS) (City of Scottsdale n.d.). Over $50 M was dedicated to the infrastructure necessary to treat and deliver up to 20 million gallons a day for turf irrigation. The city owns and operates the RWDS, and member golf clubs pay for their reclaimed water in addition to maintenance, operation, and capital costs of the project (City of Scottsdale n.d.).

Santa Fe relies on reclaimed wastewater especially in summer for irrigating golf courses and parks (City of Santa Fe n.d.). Santa Fe′s effluent charges are $2.05 per 1,000 gallons or 50% of the potable water rate, whichever is greater (City of Santa Fe n.d.). Discounts are offered to effluent contractors who host municipal recreational programs hosted within their facility (City of Santa Fe n.d.). Santa Fe residents can purchase reclaimed water for $3.37 per 1,000 gallons and must pick up the water from a central facility to store in secure tanks on their own property, with a permit required (City of Santa Fe n.d.).

Albuquerque is completing three separate wastewater reuse projects with a grant from the New Mexico Environmental Department. The projects supply irrigation water to parks, golf courses and schools, recharge the Rio Grande in the winter, and provide drinking water supply (Davis 2020).

Denver Water operates the largest recycled water system in Colorado and is expanding to provide more reclaimed water for irrigation, industrial use, and lakes (Buchanan 2018; Denver Water n.d.). Non-potable water offered by Denver Water comes as either raw water or recycled water. Pricing depends on customer location. Raw water is cheaper than recycled water within the service area, $0.81 per 1,000 gallons vs. $0.99 per 1,000 gallons for recycled water. Outside the service area, recycled water costs $1.17 per 1,000 gallons and raw water costs $1.20 per 1,000 gallons. For comparison, Denver Water charges irrigation only business customers inside the City of Denver $1.40 per 1,000 gallons for treated water in the winter and $5.60 per 1,000 gallons in the summer and outside those rates rise to $2 per 1,000 gallons in the winter and $8 per 1,000 gallons in the summer (Denver Water n.d.).

The Las Vegas Valley Water District partners with the Clark County Water Reclamation District (CCWRD) and the City of Las Vegas to supply treated wastewater to golf courses and parks (LVVWD n.d.). Forty percent of Las Vegas's water use is treated wastewater (BOR 2015). Reclaimed water customers must cover the costs of installation and maintenance of equipment and infrastructure for reclaimed water service (CCWRD 2020). CCWRD charges $1.05 per 1,000 gallons of reclaimed water delivered (CCWRD 2020).

Cheyenne, Wyoming uses treated wastewater for irrigation and is expanding their water reuse facility (Western States Water Council 2011; Figueroa et al. 2013). The City of Cheyenne used recycled water to irrigate 300 acres of parks and athletic fields in 2013 (BOR 2015). Treated wastewater is viewed as a supplemental water source so the region can meet its growing demand (Figueroa et al. 2013).

Salt Lake City is building a new and larger Water Reclamation Facility to replace their old plant (Salt Lake City n.d.). The Environmental Protection Agency awarded Salt Lake City a $350 M loan to help develop the new facility (O'Donoghue 2020).

Tucson and Santa Fe are unique among cities studied in offering residential customers reclaimed water for irrigation. In 2011, Tucson Water provided the service to 704 homes (Campbell and Scott 2011). Tucson residents can connect to the reclaimed water service if they live within one half mile of reclaimed distribution pipes (Campbell and Scott 2011). Reclaimed Water Service costs $2.13 per CCF plus a fixed service fee. The lowest residential tier charged for potable water is $2.07 per CCF for the first 8 CCF, then increasing to $3.82 per CCF and continuing to rise with usage. Pricing for reclaimed water makes it cost-effective for high volume water users (Campbell and Scott 2011). The City of Tucson keeps reclaimed water rates lower than actual costs to incentivize usage by recovering funds with the drinking water rates (City of Tucson 2020).

Campbell and Scott surveyed residential customers connected to the Tucson reclaimed water system and found that 75% of the respondents believe the benefits of the reclaimed water service outweigh the costs. Nearly 90% were satisfied with their use of the reclaimed water (Campbell and Scott 2011). Many Tucson reclaimed water users said they derived benefits from the service because "it's good for the environment" (Campbell and Scott 2011).

Across the cities included in this review, effluent usage is more prevalent than gray water reuse. Effluent reuse can also be implemented on a much larger municipal scale compared to gray water which has mostly been employed on the residential level.

Turf Retirement and Xeriscape Incentives. Denver, Salt Lake City, Phoenix, and Tucson do not offer incentives for residents to switch from traditional turf to Xeriscape but offer information and workshops for residents (City of Phoenix 2020; City of Tucson 2020; Denver Water n.d.; Salt Lake City n.d.). In New Mexico, Albuquerque offers a Xeriscape rebate while Santa Fe does not (ABCWUA n.d.; Save Water Santa Fe 2022). Residents of Albuquerque can apply for the rebate prior to their Xeriscape conversion, for a minimum 500 square feet of turf converted and a rebate of up to $1 per square foot converted (ABCWUA n.d.). Six cities in Colorado and eight Arizona municipalities offered a landscape conversion rebate in 2019 (H_2O Radio 2019). Neale et al. (2020) find xeriscape conversion was the most effective end use efficiency strategy for both Denver and Tucson, reducing GPCD use by 22% and 12%. However, total costs were estimated to be $109 M and $46 M. Neale et al. (2020) rated programs based on their total costs (regardless of who bears the cost) and ability to reduce average annual water demand, as measured by GPCD.

Urban Stormwater Capture. Nearly all the CRB cities reviewed manage and direct stormwater to flow through drainage systems that discharge the water in manmade or natural channels or lakes. The City of Tucson and Pima County take a more active approach to stormwater management. In partnership with the U.S. Army Corps of Engineers, Pima County constructed ponds that capture stormwater for later use by the city (Davis 2019). Pima County collects and stores 250–400 AF of rainfall each year. The Project provides an alternative, cheaper water source for irrigation of City athletic fields (Davis 2014). The City of Las Vegas takes similar approach to stormwater capture with the Las Vegas Wash. This 12-mile channel is the main path water travels to reach Lake Mead and back. Stormwater is one of the four sources of excess water in the wash (Buranen 2018). As the water moves to Lake Mead, it flows through the Clark County Wetlands Park which acts a natural filter to clean the water and improve its quality. The Wetlands provide habitat to local wildlife (Las Vegas Wash Coordination Committee n.d.). According to Neale et al. (2020), urban stormwater capture systems total costs can range from $1,200 to $6,000 per acre-foot for treatment and usage. Large-scale urban systems could reduce demand for traditional water supplies by 10%–20% in Denver and 10%–17% in Tucson (Neale et al. 2020).

Rainwater Harvesting. While rainwater harvesting is legal in all the CRB states examined in this article, some states place more restrictions on collection of rainwater. Colorado and Utah have the most regulations for collecting rainwater (Castelo 2020). Colorado water rights laws can be interpreted to discourage rainwater collection, as harvesting prevents rainwater from flowing downstream — potentially decreasing water available to a right holder (Rochat 2020).

Rainwater harvesting takes two forms: active harvesting (collection of rain for storage and future use) and passive harvesting (directing and retain water within a landscape for natural irrigation) (Gardner 2017). Tucson offers a rebate to city water users for both active and passive harvesting, with completion of the Rainwater Harvesting Incentives Program Workshop (City of Tucson 2020). Both Albuquerque and Santa Fe have programs to reimburse water customers for rainwater catchments (ABCWUA n.d.; Save Water Santa Fe 2022). Bernalillo County offers residents free or discounted rain barrels (Bernalillo County n.d.). Salt Lake City sells rain barrels at cost as part of their Rain Barrel Initiative launched in 2015 (Salt Lake City n.d.). Phoenix, Las Vegas, Denver, and Cheyenne do not offer incentives for rainwater collection.

Many of the cities reviewed allow for and encourage turf retirement, stormwater capture, and rainwater harvesting, but there is still opportunity for further adoptions throughout these communities. Turf replacement and rainwater harvesting typically occur at the residential scale, but cities may explore opportunities to implement these conservation activities on public lands.

Evaluating Programs to Reduce Urban Use

It is important for any city (or other organization) sponsoring programs intended to reduce urban use to periodically evaluate these programs. Here we review a number of approaches to evaluating programs and provide suggested criteria and guidelines.

Several evaluative studies of the Las Vegas Water Smart Landscapes are summarized here. Baker (2021) and Brelsford and Abbott (2021) found the program reduced water use among participants by 19%–21%. Each study also examines program costs. Baker estimates program costs by adding administrative costs and rebate outlays, with these costs ranging from $2.65/kilogallon to $3.31/kilogallon, less than the cost of added water supplies. Baker calculates the cost of added water supplies based on agricultural to urban water sales (Baker 2021). Brelsford and Abbott (2021) find a consistent program cost of $3.37/kgal across a 10-year timescale, calculating this cost using dollars spent on the program. Net benefits per square foot of converted land equaled $2.35–$2.88/square foot (Baker 2021). Baker estimated net benefits by subtracting the administrative and conversion costs from the sum of the direct effect of conversion and the value of scarce water.

The two studies found that water savings were sustained over time. Baker (2021) found that a 4% price increase across all LVVWD residents would have achieved the same aggregate savings. Both studies note that further research is needed to determine the proportion of program participants who were essentially free-riders. Free-riders are those participants who would have converted landscape regardless of the rebate. Removing these participants from the calculations would increase the estimated costs per unit of water saved.

Brelsford and Abbott (2021) provide four metrics for evaluating program efficacy. They measure the success of a program by (1) the overall water savings, (2) the durability of water savings, (3) cost-effectiveness for the individual homeowner, and (4) cost-effectiveness for the subsidizing institution (Brelsford and Abbott 2021). Cost-effectiveness is a preferred measure, according to Brelsford and Abbott (2021), due to the difficulties of estimate the diverse social benefits of reducing water use — which are needed to complete a traditional benefit–cost analysis.

Radonic (2019) and Holland-Stergar (2018) evaluated the Tucson Water rebate for residential rainwater harvesting (RWH), a policy developed to relieve pressure on the potable water supply. They found program participation was good but found no indication of reduced potable water use. RWH rebate recipients added more vegetation to their landscape and did not cut back on potable water use (Holland-Stergar 2018). Radonic (2019) found that 72% of participants planted new vegetation and 28% installed the RWH system as part of a large landscaping project. The new plantings were not necessarily water intensive, but households did not wean themselves off potable irrigation systems (Radonic 2019). Participants perceived environmental benefits as more important than financial benefits (Radonic 2019).

Holland-Stergar (2018) reports that rainwater harvesting rebates are the least cost-effective conservation program offered by the City of Tucson, providing negligible water savings and costing the city $327,145 (Holland-Stergar 2018). Holland-Stergar evaluated programs on their ability to achieve two objectives: (1) encourage adoption and use of technologies and (2) result in decreased reliance on more traditional water sources (2018). The author notes that overall program goals are important to consider when evaluating programs. If customer outreach and education is viewed more important than water demand reductions, the Tucson Water RWH rebate can be seen as successful in outreach and education (Holland-Stergar 2018).

Neale et al. (2020), in a study of Denver, Colorado, Miami, Florida, and Tucson, Arizona, assess a combination of water conservation strategies. The authors estimate the cost for a variety of strategies including high efficiency appliances, advanced irrigation systems, xeriscape, gray water use, and rain runoff. For costs, they include lifetime capital, maintenance, and operations costs for each strategy. Neale et al. find that household rainwater harvesting systems (RHSs) for toilet flushing and irrigation cost between $1,500 and $1,600 and only deliver minimal demand reductions. Neale et al. evaluate strategies by assessing the trade-off between total cost and average annual water demand reductions for the strategies. The best solutions are those that provide demand reduction without a steep rise in costs (Neale et al. 2020).

Based on our review of evaluation studies for CRB urban water conservation programs, cost-effectiveness emerges as the most useful and implementable criterion when evaluating programs to reduce urban use. Cost-effectiveness has been employed in many other evaluations (Baker 2021; Brelsford and Abbott 2021). Total cost per unit of reduced water use is a useful way to measure cost-effectiveness. This number can then be weighed against other use reduction approaches and against the unit cost of acquiring additional water supplies for urban use. Benefit–cost analyses are another way to evaluate programs. However, these are more difficult to implement because they require estimation of benefits that may be difficult to capture in monetary terms — such as ecosystem benefits or community social and cultural values (Brelsford and Abbott 2021). If conducting a benefit–cost analysis, the LRMC of acquiring additional water supplies for city use is a sound metric to use when considering the benefits of reduced use.

Programs can also be evaluated using nonmonetary metrics, such as community acceptance and adoption rates. Municipal water users may derive benefits from believing they are helping the environment through their conservation efforts, beyond any financial benefits to the household. Radonic (2019) found this to be the case for many Tucson Water RWH rebate recipients.

Public support is important to consider before designing a program aimed at reducing urban water use. Holland-Stergar (2018) highlights how Tucson residents were instrumental in the implementation of the RHS standards and rebates. Greater public acceptability can aid in the eventual effectiveness of a program. Along with other considerations, members of the public may be concerned with fairness and how the cost/burden of water policies are spread across different neighborhoods, income classes, and racial/ethnic groups.

It is also important to consider monitoring and enforcement costs when designing programs. The City of Tucson mandated installation of RWH systems in all commercial buildings constructed after 2008. Builders must submit permits to the city before construction and site inspection occurs in the overall construction approval process, so RWH compliance can be assessed as part of that inspection with little added effort (Holland-Stergar 2018). On the other hand, the residential RWH program in Tucson requires added staff and household oversight and monitoring costs that can dampen adoption of residential RWH (Holland-Stergar 2018).

FOREST AND WATERSHED HEALTH PROGRAMS TO PROTECT URBAN SUPPLIES

A number of CRB urban water providers, and partner organizations, have programs to protect and restore forest and watershed conditions in the areas from which they receive their water supplies. Cities, in partnership with federal agencies and NGOs, seek to incentivize actions that prevent catastrophic fires. Such fires impair water quality, reduce storage of moisture in soils, and diminish reservoirs' capacity to store water. In this review of forest health programs intended to improve urban supply reliability in the CRB, we include available evaluations of effectiveness and public perceptions of programs.

Denver Water collaborates with the U.S. Forest Service, Colorado State Forest Service, and U.S. Department of Agriculture Natural Resources Conservation Service as part of the Forests to Faucet program established in 2010. The program protects water delivered to Denver Water customers and identifies zones to target for fire management practices. In 2017, the parties committed to invest \$33 M for the restoration of 40,000 acres of forestland (Denver Water n.d.). Denver Water invests in fire prevention measures in the Upper South Platte River watershed, with 80% of their water passes through the area before delivery to Denver. Two evaluation studies are summarized below.

Arizona cities also invest in programs for forest and watershed health. The Nature Conservancy partners with the U.S. Forest Service in the Future Forests program to thin dense forests in Arizona (Devoid 2018). SRP (a wholesale key provider and manager of water in greater Phoenix metro area) has contributed substantively to aid efforts in the Verde River watershed (Business News 2017). The Four Forest Restoration Initiative (4FRI) backed by the Northern Arizona Forest Fund (NAFF) is restoring extensive forest areas to reduce the risk of large forest fires (National Forest Foundation 2020; Forest Service n.d.). The NAFF has financed and completed several dozen restoration projects in partnership with cities and state and federal agencies since 2015 (National Forest Foundation 2020). One economic assessment of the 4FRI program is provided below.

New Mexico's Rio Grande Water Fund (RGWF), established in 2014, restores forests in northern New Mexico in partnership with urban water providers, NGOs, counties, and state and federal agencies (The Nature Conservancy 2020). Since the program's launch, over 140,000 acres have been treated and a half-dozen stream and wetland projects completed (The Nature Conservancy 2020). An evaluation of the RGWF initiative is summarized below. The New Mexico Forest and Watershed Health Plan to combat overly dense forests in Northern New Mexico (New Mexico Forest and Watershed Health Planning Committee 2004; New Mexico State Forestry n.d.). The Greater Santa Fe Fireshed Coalition (GSFFC) works with cities and public agencies in the region to improve watershed health using prescribed burns and other measures (GSFFC n.d.).

Federal and NGO funding plays a key role in watershed health initiatives. The U.S. National Resources Conservation Service (NRCS) Regional Conservation Partnership Program (RCPP) fosters partnerships between federal agencies and public and private entities, with a number of programs active in the CRB (Natural Resources Conservation Service n.d.). For example, the Upper Verde River Watershed Protection Coalition (UVRWPC), a collaboration between local governments in Yavapai County, Arizona, leads a project to recharge water in the Upper Verde River Watershed (UVRWPC 2020; Natural Resources Conservation Service n.d.). Prescott Valley and the NRCS provide funding to optimize groundwater recharge with new technologies (Natural Resources Conservation Service n.d.). Another example of an RCPP grant in the CRB is the $7.4 M award to the Uintah Water Conservancy District (UWCD) in Utah to improve water quantity and quality in the area (Uintah Water Conservancy District 2020; Natural Resources Conservation Service n.d.). Philanthropic foundations provide support for many of the watershed health initiatives in the CRB (Water Funders Initiative n.d.).

Evaluating Programs to Protect Urban Supplies through Healthy Forests and Watersheds

Some of the same criteria used to design and evaluate programs for reduced urban use can also be applied to forest and watershed health programs intended to improve supply reliability. This section briefly summarizes the evaluation of forest and watershed health initiatives and provides guidance for future evaluation studies.

Two studies, four years apart, evaluate the specific components of the Colorado Forests to Faucet program. Jones et al. (2017) use a return on investment (ROI) analysis to evaluate fuel treatment interventions in the Upper South Platte River watershed near Denver, for watershed protection from wildfire. The study finds that fire mitigation treatments have a positive ROI after large storm events. Denver Water expended over $26 M on water quality remediation following the Buffalo Creek and Hayman fires, and one goal of watershed protection interventions is to reduce future fires and water quality impacts. Jones et al. (2017) simulated scenarios of fuel mitigation treatments from 5% to 100% coverage. Benefits increase as more area is treated, but financial returns begin to decline after 80% of the area is treated. ROI becomes positive after only 17% on the area is treated and peaks when about 50% of the area is treated (Jones et al. 2017). Treating 50% of the area (about 1,000 hectares) costs an estimated $2.6 M.

Jones et al. (2021) more recently evaluated Colorado's Forest to Faucet program using a cost–benefit analysis rather than ROI. The authors find the program has benefits that outweigh its costs, when considering both the source water protection benefits and additional benefits such as reduced property loss, forest recovery and rehabilitation costs, fire suppression costs, and recreation and endangered species values. Fuel treatment costs were estimated at $1,000 per acre treated. Benefits to the values at risk (sources water and additional benefits) were measured using estimated avoided costs. The economic value of protecting source water is $4–42 M and the value of the additional benefits is between $24 and 100 M depending on the probability of fire and time horizons (Jones et al. 2021). Accounting for the probability of wildfire occurrence led to lower benefit–cost ratios (Jones et al. 2021). Jones et al. (2021) were able to incorporate some non-market values for water quality and supply reliability in their evaluation.

Hjerpe and Mottek-Lucas (2018) provide a regional economic analysis of Arizona's 4FRI for the year 2017. The impact zone includes five counties in northwest Arizona where forest restoration is targeted. Hjerpe and Mottek-Lucas examine program expenditures and employment and find that 4FRI contributed nearly $100 M in direct output and nearly $150 M in total output, including multiplier effects (Hjerpe and Mottek-Lucas 2018).

Hartwell et al. (2016) estimate the ROI for the RGWF in new Mexico's San Juan-Chama headwaters. Simulating possible fire scenarios, the authors compare estimated damages between the no treatment and RGWF treatment plan. The difference between the two is the benefit from investment. These benefits serve as a lower-bound baseline, since Hartwell, Kruse, and Buckley include only those financial values they can easily quantify. They estimate the cost of fire treatment using data on current costs and treatment area plans. ROI is calculated as the difference between the benefits and costs divided by the costs. Hartwell et al. (2016) find that the ROI for treatment ranges from 246% to 375%, varying by basin.

A different kind of economic study, the Contingent Valuation Method (CVM), estimates Willingness to Pay to quantify values held by the public for supply reliability and improved watershed health. CVM studies are useful as part of evaluating watershed protection programs. Mueller (2014) estimates the willingness to pay (WTP) for forest restoration in the Lake Mary and Upper Rio de Flag watersheds as part of the 4FRI program. Survey participants were asked how much they would be willing to pay in the form of a monthly fee on their water bill to monitor and maintain forest health of the Lake Mary and Upper Rio de Flag watersheds. The mean WTP for the 4FRI program is $4.89 per month (Mueller 2014). Mueller (2014) also finds that respondents who had a greater awareness of the connection between forest restoration and watershed health and this specific program had higher WTP responses.

Adhikari et al. (2017) use a contingent valuation study to analyze the level of public support by New Mexico households for forest restoration for watershed health. Over 900 Albuquerque households responded to the CV survey through mail or Internet in 2013 (Adhikari et al. 2017). The study finds a mean annual household WTP of $64.44 and a median WTP of $37.76 to reduce wildlife risk and provide water source protection (Adhikari et al. 2017). In a

CVM study for Santa Fe in 2011, survey respondents were willing to pay $7.80 per year to protect against wildfire to support water supply. This response prompted initiation of a Payment for Ecosystem Services (PES) program by the Santa Fe City Council that year (Adhikari et al. 2017). CVM studies, such as the two summarized, provide important insights on public values. Findings from these studies can inform development of future programs, as well as assessment of watershed protection programs already underway.

Different evaluation tools used for supply reliability programs include a ROI analysis — used by Jones et al. (2017) and Hartwell et al. (2016). Hjerpe and Mottek-Lucas (2018) use a regional economic contribution analysis in their evaluation of the Four Forests Restoration Initiative. Community and cultural values also generate relevant criteria for these types of programs and the public derives recreational benefits from healthy forests and reservoirs. Both Mueller (2014) and Adhikari et al. (2017) use CVM as a way to evaluate public values and support for forest restoration, increased watershed health and the benefits these provide.

CONCLUSIONS

This article examines incentive-based tools used to reduce urban water use and to protect watersheds that supply cities in the CRB, excluding California. Most major cities have adopted municipal rate structures that have multiple rate tiers with higher rates for higher monthly volumes of use. Some cities have seasonal shifts in water rates designed to reduce summer outdoor use (see Tables 4 and 5). Smart meters are being more widely adopted and these open up a new generation of tools that can use time-of-day and seasonal pricing to reduce use.

All major CRB cities have decreased per capita water use over the past 20–30 years. Each of the eight cities examined in this article have experienced significant population growth since 1990, with the highest growth rates in Las Vegas and Phoenix (see Table 1). Technological efficiency improvements and policies incentivizing adoption of these improvements have contributed to declines in per capita use in all cities. Some cities have had increases in overall water use, but at rates far less than their population growth.

A number of CRB cities provide incentives to use gray water and effluent for outdoor use and to harvest rainwater. The importance to urban water supplies of protecting forests and watershed health is being more widely recognized through programs and partnerships across the CRB.

Our review of studies that evaluate programs intended to reduce urban use revealed a need for additional robust, data-based evaluation studies for some types of programs. For several of the CRB's largest cities, there is a rich literature evaluating the effects of water rate using statistical analyses that estimate responsiveness to price. These studies find that price elasticities vary depending on location, season, and indoor vs. outdoor use. There are fewer studies that employ a data-based empirical approach to estimate the effectiveness of programs other than water rate structures. However, studies that weigh program costs against program achievements have been conducted for some programs in some CRB locations. We identified and reviewed evaluation studies on rainwater harvesting, turf retirement, and xeriscape programs. We found very few studies that compare efficacy and costs per unit of reduced use across different types of CRB demand reduction programs. These kinds of comparative studies would be highly valuable to urban water managers.

We also reviewed CRB programs to improve urban supply reliability through protecting and restoring forest and water shed health. For these programs, a range of return of evaluation approaches also is being employed. These include ROI analyses (Hartwell et al. 2016; Jones et al. 2017), regional economic contribution analyses (Hjerpe and Mottek-Lucas 2018), and CVM estimation of public values (Mueller 2014; Adhikari et al. 2017).

Given the need for more studies that evaluate and compare different approaches to reducing urban demand and protecting watersheds for supply reliability, we offer recommendations for future evaluation studies. Economic, environmental, and social criteria may all be relevant, depending on program objectives. Policymakers should look to create multi-pronged programs relying on different tools that address different criteria. Achieving environmental criteria is especially important for programs to improve forest and watershed health because these aim to safeguard water supply reliability and water quality. Obtaining public support can lead to more successful programs in terms of adoption and compliance. Demand reduction programs and policies can be evaluated using a range of economic tools such as cost-effectiveness, cost–benefit analysis, WTP studies, and ROI.

Although much has changed in the CRB over 25 years, agriculture remains the dominant consumptive use of water. The vast majority of the population resides in major urban areas, as do the bulk of economic activities that provide jobs and regional economic activity. The demand reduction and supply reliability programs discussed here were non-existent or in their infancy at the time of the Severe Sustained Drought Project. They now provide an important set of tools for cities to cope with extended drought and increasing regional aridity. While the policies reviewed involve relatively small volumes of water, overall, they represent an important development in urban water management and resilience — given the ongoing pattern of severe, sustained drought that characterizes the CRB.

DATA AVAILABILITY STATEMENT

Data sharing is not applicable to this article as no original data are analyzed in this article.

ACKNOWLEDGMENTS

The authors acknowledge helpful input from water managers throughout the CRB, as well as the invaluable suggestions provided by anonymous reviewers and editors.

AUTHOR CONTRIBUTIONS

Bonnie Colby: Conceptualization; data curation; formal analysis; funding acquisition; investigation; methodology; project administration; resources; supervision; writing – original draft. Hannah Hansen: Data curation; investigation; writing – review and editing.

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