Discussion Paper

The Cost of Water in a Desert

How Much Can Saudi Arabia Save From Water Loss Reduction?

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Abstract

Saudi Arabia is an arid nation experiencing a severe shortage of freshwater, which necessitates reliance on costly, energy-intensive seawater desalination to meet rapidly growing residential demand. Per-capita domestic water use is very high by international standards, reflecting both climatic conditions and the effects of low tariffs and high system losses. The Saudi government has launched an investment program aimed at reducing water system losses from 36% in 2024 to 15% by 2031. This paper estimates the program's impact on domestic water demand, desalination costs, and fiscal liabilities. Using an econometric model based on the Stone-Geary utility function, we estimate that the program would help meet projected water demand – distinguishing between subsistence and variable water consumption – while avoiding the need for 2 billion m^3/year of desalination capacity and 7.6 TWh of electricity demand by 2050 (equivalent to about half of current electricity consumption in Madinah). We also calculate that water tariff revenues (US$0.65* billion in 2025) cover only about 20% of the levelized cost of desalination ($3.2 billion in 2025), leaving a cost recovery gap – or fiscal liability – rising from $2.6 billion in 2025 to $4.6 billion per year by 2050, even before accounting for water transmission and distribution system costs. Without the government program, the total fiscal liability accumulated from subsidized desalination over the 2025-2050 period could reach $91.4 billion. Implementation of the program would reduce desalination subsidy requirements by $0.5 billion in 2031 and nearly $1 billion in 2050. Over the 2025-2050 period, cumulative fiscal savings would reach $17.5 billion.

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*All values in U.S. dollars.

Introduction

01

As an arid nation, Saudi Arabia faces a severe shortage of freshwater resources, with water demand consistently rising due to population growth, urbanization, and rapid economic development. The Kingdom depends heavily on costly and energy-intensive desalination processes to meet its increasing municipal water demand, making it the largest producer of desalinated water in the world (USSBC 2022).[1]

Saudi Arabia's water sector faces high transmission and distribution losses (before the meter), significant water losses within housing units (after the meter), inefficiently high per-capita water consumption (SWPC 2024), and a large share of water produced that is either unbilled or unpaid (Ordem dos Engenheiros 2015). Low water tariffs cover only a fraction of the high and rising costs of water supply, requiring the government to absorb most of the expense. As a result, consumers have little incentive to conserve water or adopt water-saving technologies, while public budgets bear an increasing burden of fiscal liabilities, which this study quantifies.

Government policies in the water sector have primarily focused on expanding water supply through investment in desalination plants, rather than implementing comprehensive demand-side management strategies (Al-Zubari et al. 2017). A unique combination of limited water availability and abundant energy resources has accelerated the adoption of modern desalination technologies. However, reliance on desalination requires intensive energy use – not only for production but also for transporting water from coastal desalination plants across the country to end users, who are sometimes hundreds of kilometers away in the desert.

While supply-side policies have been able to meet growing demand, water-use efficiency and demand-side management have lagged, largely due to subsidized water tariffs (Al-Zubari et al. 2017). Furthermore, according to the SWPC (2024), water losses – defined as the gap between water produced and water billed to consumers, known as unaccounted-for water (UFW) – remain high, at approximately 36% in 2024, consistent with earlier World Bank (2005) estimates of 30%-40%.

This study has three key objectives. The first is to estimate current and simulate future domestic water demand for Saudi Arabia, distinguishing between subsistence and variable water use. A subsistence level represents essential or basic water needs; it is a fixed component of water demand, insensitive to changes in price and income. The variable demand component rises with income and decreases with price.

The second objective is to estimate the energy needed for desalinated water production to satisfy future water demands. The final goal is to assess the cost of water desalination, recovery revenue gaps from existing tariffs, and fiscal cost savings from the Ministry of Environment, Water and Agriculture (MEWA) program of water losses reduction.

Literature Review

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The literature review is organized into two distinct sections. The first examines methodological approaches to analyzing domestic water demand, focusing on econometric and utility-based models used to estimate household water demand in response to factors such as income, price, household size, and climatic conditions. This discussion offers insights into model selection and parameterization relevant to the Saudi context. The second section reviews research on energy for water, emphasizing the interdependencies between water production – particularly through desalination – and energy consumption. It explores how rising water demand intensifies energy use, especially in arid regions reliant on desalination, and assesses studies that evaluate the economic and environmental implications of this coupling.

2.1 Water Demand Literature

Water demand modeling literature focuses on capturing consumer behavior, price responsiveness, and income effects. The Cobb-Douglas function is one of the most widely used functional forms, as it assumes a constant price elasticity across all consumption levels regardless of changes in marginal prices. However, this assumption can be unrealistic in the context of an increasing block tariff structure, in which the price per unit of water rises with higher usage levels. Consumers facing different marginal prices are likely to exhibit varying sensitivity to price changes. Consequently, the elasticity of water demand should vary across blocks.

Another limitation of the standard approach is its inability to differentiate between subsistence water use and discretionary consumption, which is crucial in water demand analysis, particularly in regions experiencing water scarcity. Subsistence or threshold water level is fundamental for health, dignity, and survival. International public-health literature provides normative benchmarks. For example, Gleick (1996) suggests that 50 liters per person per day of clean water should be considered a fundamental human right, while Howard et al. (2020) report 100 liters per person per day as an optimal level for drinking, cooking, and hygiene to maintain a low health concern level.

To address this limitation, recent studies have adopted the Stone-Geary utility function. This approach explicitly incorporates a subsistence level of consumption, below which usage cannot fall, regardless of changes in income or prices (see Stone 1954; Deaton and Muellbauer 1980). The Stone-Geary model decomposes total water demand into two components:

(1) a non-discretionary price-inelastic subsistence component, and (2) a discretionary component that varies with economic conditions. Literature estimates of subsistence water demand vary between 0.64 cubic meters per capita per month for Sri Lanka and 13 cubic meters per capita for Texas, indicating its strong dependency on climatic conditions.

For example, Gaudin, Griffin, and Sickles (2001) compared Stone-Geary and generalized Cobb-Douglas functional forms for Texas municipalities and confirmed that including the subsistence level of improved out-of-sample forecasting and prevented implausible extrapolations at low consumption levels. They estimated that subsistence water consumption was around 13 cubic meters per capita per month.

In Seville, Spain, Martínez-Espiñeira and Nauges (2004) estimated a two-part model in which the short-run subsistence quantity changes between 2.6 and 4.7 cubic meters per capita per month. By allowing subsistence to evolve in response to past consumption, they showed that appliance ownership and habits gradually shift the threshold upward, highlighting the dynamic nature of “basic” demand.

In a Canadian panel, Renzetti, Dupont, and Chitsinde (2015) incorporated a household-specific fixed component and estimated the subsistence level to be 10.96 cubic meters per capita per month. Applying a Stone-Geary utility framework to data from 301 municipalities in Andalusia for 2005, García-Valiñas, Martínez-Espiñeira, and González-Gómez (2010) estimated that the basic water consumption threshold was approximately 11 cubic meters per household per month.

Dalmas and Reynaud (2004) analyzed residential water demand in Slovakia using data from 71 municipalities between 1999 and 2001 using linear, log-log, and Stone-Geary model types. They found that water demand was price inelastic across all specifications and estimated a minimum consumption threshold of 31.5 cubic meters per person annually in 2001.

Studies in developing countries report slightly lower subsistence levels. For example, Dharmaratna and Harris (2012) estimated that Sri Lanka’s subsistence level ranges from 0.64 to 1.06 cubic meters per capita per month. Al-Qunaibet and Johnston (1985) were among the first to operationalize this idea for an arid Gulf economy. Using monthly data for Kuwait (1973-1981), they estimated the subsistence level at approximately 42 liters per capita per day (LPCd) after accounting for weather variables; the term was statistically significant and, by construction, price-inelastic.

No study estimates the subsistence level of water consumption for the GCC countries except Kuwait. However, actual per-capita municipal water use is high, ranging from 300 to 750 liters per person per day, depending on the country (Sherif et al. 2023; Shevah 2017; Barau and Al Hosani 2015; Manawi et al. 2014). Sherif et al. (2023) estimate that the average per-capita water consumption in the GCC is approximately 550 liters per person per day.

2.2 Energy for Water

Water is essential at various stages of energy production, while energy is indispensable for providing water services. Consequently, constraints or inefficiencies in one resource can significantly affect the availability and sustainability of the other.

Research on the water-energy nexus (WEN) began attracting academic attention in the mid-1990s with the seminal work of Gleick (1994), who emphasized the critical need for integrated water and energy policies. Subsequent contributions by Hansen (1996) and deMonsabbert and Liner (1998) expanded this perspective by quantitatively analyzing the effects of energy prices on water demand and examining integrated conservation models.

Since then, the WEN field has expanded significantly in terms of scope, geographical focus, and methodological diversity. Fayiah et al. (2020) highlight this rapid growth, noting that more than 120 peer-reviewed studies and 23 case studies have been published recently. These studies span a variety of sectors – including urban infrastructure, industrial processes, agriculture, and national policy – and deploy a wide range of methodological approaches. For instance, Duan and Chen (2020) examined WEN drivers in China, while Al-Masri, Chenoweth, and Murphy (2019) analyzed policy frameworks governing the nexus in Jordan. Similarly, Hong et al. (2019) assessed nexus efficiency in China’s construction sector using provincial-level data, demonstrating how WEN research can inform both operational and policy-level decision-making.

The WEN has gained prominence in the literature over the past decades as a framework for integrated resource planning.

Al-Mutrafi et al. (2018) calculated the energy used for pumping and treating water, particularly for seawater desalination and transporting water over long distances, as well as the water demand for energy production, such as cooling power plants, extracting oil and gas, and producing biofuels.

Saudi Arabia's status as the world's largest producer of desalinated water has substantial implications for energy use and environmental outcomes. Several studies have emphasized Saudi Arabia's unique nexus challenges, in which the energy cost of water provision – particularly for seawater desalination and long-distance water transfer – is among the highest globally (e.g., Al-Mutrafi et al. 2018). Research shows that desalination alone consumed up to 13% of the Eastern Province's total electricity generation capacity in 2013, accounting for approximately 5% of the Kingdom's total energy generation capacity, highlighting the substantial burden on the national energy system (Al-Mutrafi et al. 2018).

Traditionally, the country has relied on underground aquifers; however, decades of intensive groundwater use, mainly for agriculture, have severely depleted nonrenewable resource reserves (Rambo et al. 2017). Siddiqi and Anadon (2011) provide evidence that water abstraction and production processes in the MENA region are highly energy-intensive, with their study estimating that up to 9% of Saudi Arabia's annual electricity consumption is devoted to obtaining fresh water. Energy demand for seawater desalination in the Middle East is expected to increase from 40.4 million tonnes of oil equivalent (Mtoe) in 2023 to 71.2 Mtoe in 2030 (Bredariol, Lim, and Staas 2024).[2] In Saudi Arabia, approximately one-third of national oil production is already allocated to meeting domestic water desalination and energy needs, significantly limiting the availability of fossil fuels for export (Rambo et al. 2017).

Modeling Domestic Water Demand: Methodology and Data

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In this study, domestic water demand is modeled using an equation derived from the Stone-Geary utility function, which allows for modeling a portion of consumption that is unresponsive to price changes and another portion that can adapt instantaneously to price variations.

The Stone-Geary function has been widely employed in empirical water demand studies due to its flexibility in capturing essential (minimum) water use levels alongside discretionary use. Its application spans diverse geographic and socioeconomic contexts, including Kuwait (Al-Quanibet and Johnston 1985); Spain (Gaudin, Griffin, and Sickles 2001; Martínez-Espiñeira and Nauges 2004); Sri Lanka (Dharmaratna and Harris 2012), and Canada (Renzetti, Dupont, and Chitsinde 2015).

The Stone-Geary utility function is a classic model in consumer theory that represents preferences when a minimum level of consumption (subsistence) is necessary before discretionary spending occurs. The Stone-Geary utility function can be written as follows:

$$U = \beta^w \ln(D^w - \gamma^w) + \beta^z ln(D^z - \gamma^z) \tag{1}$$

Where U is total utility, D^w and D^z are the domestic demand for water and all other goods, respectively, γ^w and γ^z are the subsistence level water use and other goods, respectively. β^w and β^z represent the marginal budget share allocated to water and the other goods. The following budget constraint, subject to the utility function, is used by assuming that the price of other aggregated goods is normalized to 1 ($P^z = 1$). So, the budget constraint is

$$Y = D^w P^w + D^Z \tag{2}$$

Y is income, P^w and P^z are the per-unit prices for water and other aggregate goods, respectively.

Maximizing the utility function (1) subject to the budget constraint (2) produces the following water-demand equation:

$$D^w = \frac{\beta^w (Y - P^w \gamma^w - \gamma^z)}{P^w} + \gamma^w \tag{3}$$

By rearranging the water-demand equation, we can derive the following relation:

$$\beta^w = \frac{P^w D^w - P^w \gamma^w}{Y - P^w \gamma^w - \gamma^z} \tag{4}$$

The parameters β^w and β^z represent the fixed proportions of the supernumerary income, that is, the portion of total income remaining after the subsistence quantities of water (γ^w) and other goods (γ^z) have been acquired. These parameters indicate how the leftover income is allocated across goods once basic consumption needs are met, with β^w reflecting the marginal budget share spent on water and β^z on all other goods. The detailed derivation of the Stone-Geary utility function is provided in the Appendix. After assuming the $\gamma^z = 0$, the final demand equation for water can be written as follows:

$$D^w = (1 - \beta^w)\gamma^w + \beta^w \frac{Y}{P^w} \tag{5}$$

The Stone-Geary utility function has gained increasing attention in water demand modeling due to its ability to distinguish between essential and discretionary water use explicitly. Unlike standard log-log specifications, which assume constant elasticity across all consumption levels, the Stone-Geary approach allows for the modeling of a fixed, subsistence level of demand that is unresponsive to price, while discretionary water usage adjusts to price and income changes (Martínez-Espiñeira and Nauges 2004). This feature is particularly relevant in the context of residential water use, where basic needs such as drinking, cooking, and hygiene are expected to be inelastic, whereas discretionary activities like lawn watering are more price-sensitive. The model's structure – relying on only two parameters per good, namely the subsistence quantity γ and the marginal budget share β offers a parsimonious yet economically interpretable framework (Deaton and Muellbauer 1980).

The Stone-Geary utility function makes several assumptions that are especially useful for modeling consumer behavior related to water demand. These assumptions include a positive marginal propensity to consume, a positive subsistence level of water use, and strong separability between water and other goods. The assumption of strong separability is frequently used in empirical studies estimating water demand with a single-equation model, as it isolates water consumption behavior without detailing a complete demand system for all goods (Espey, Espey, and Shaw 1997; Worthington and Hoffman 2008; Dharmaratna and Harris 2012).

$$D^w = (1 - \beta^w)\gamma^w + \beta^w \frac{Y}{P^w} + \alpha_1 POP_t + \alpha_2 TEMP_t + \varepsilon_t \tag{6}$$

In equation (6), β^w and γ^w are parameters representing the share of water use in supernumerary income and the subsistence level, respectively. POP_t is the population and $TEMP_t$ is the average annual temperature; α_1 and α_2 are coefficients for population and temperature, respectively. The water demand equation (6) considers population, temperature, income, and price variables.

Population is a fundamental determinant of domestic water demand. With an increase in population, the demand for water rises due to domestic needs (e.g., drinking, sanitation, and hygiene), as well as institutional water demand (e.g., schools, hospitals, and other departments). Including population as a control variable ensures that the model accurately reflects scale effects and avoids attributing population-driven changes to price or income effects. Temperature is a critical climatic factor influencing domestic water demand, especially in arid regions. Higher temperatures lead to increased water use for personal cooling (e.g., showers, air-conditioning systems that use water), and outdoor activities such as gardening and car washing. This is particularly relevant for regions like Saudi Arabia, where residential water use is responsive to temperature extremes and dry climate conditions. Incorporating population and temperature variables in water demand analysis improves the robustness of the demand specification and aligns with empirical practices in the literature on residential water demand modeling. These variables are commonly treated as exogenous controls in demand estimation frameworks, helping improve model fit and isolate the effects of economic variables such as price and income (see e.g., Olmstead Hanemann, and Stavins 2007; Nauges and Whittington 2010; Martínez-Espiñeira and Nauges 2004; Arbués, Garcıa-Valiñas, and Martınez-Espiñeira 2003; Worthington and Hoffman 2008; Javid 2025).

We include a dummy variable in the model to account for the policy intervention introduced by Saudi Arabia's water price reform in 2016. We create a dummy variable using the partition approach with interaction terms,[3] defining a variable, DUM 2016, which equals 1 for the years 2016-2023, the period after the reform was implemented, and 0 otherwise. Conversely, the term (1-DUM2016) equals 1 for the pre-reform period from 1995 to 2015, and 0 otherwise. Equation (6) can be written with an interactive term of this dummy variable.

$$\begin{aligned} D^w = (1 - \beta^w)\gamma^w + \beta^{w1} \frac{Y}{P^w} D2026 \\ + \beta^{w2} \frac{Y}{P^w}\ (1 - DUM2016) \\ + \alpha_1 POP_t + \alpha_2 TEMP_t + \varepsilon_t \end{aligned} \tag{7}$$

As Equation (7) specifies, the Stone-Geary utility-based water demand model is estimated using the fully modified ordinary least squares (FMOLS) procedure. The FMOLS technique is employed to correct endogeneity bias that may arise from the presence of a price variable on the right-hand side of the equation. Water price is often endogenous in demand estimation because it may be correlated with unobserved factors affecting water consumption or due to simultaneity, where prices are adjusted in response to changes in demand.

The FMOLS approach addresses endogeneity issues by applying a semiparametric correction to the OLS estimator, ensuring that the long-run elasticities obtained from the estimated equation are consistent and unbiased, even when explanatory variables such as price are endogenous (see Phillips and Hansen 1990). The estimated coefficients are used to calculate the price and income elasticities. In the Stone-Geary utility-based model, the price and income elasticities have equal magnitude but opposite signs. These elasticities are estimated using the following formula:

$$\xi_p = -\beta^w \frac{Y}{P.D^w} = \xi_Y \tag{8}$$

Where, ξ_p and ξ_Y represent the price and income elasticity of water demand, respectively, D^w is water use; P is the price of water and Y is household income. The positive sign elasticity represents the income elasticity, and the negative sign represents the price elasticity of water demand.

Many of the applied studies we reviewed in Section 2 also used a log-log, or double-log functional form of the water demand. In this form, the logarithmic transformation of water demand is regressed on the logarithmic expressions of the price of water and the income of households, in addition to other explanatory factors, which can be expressed in the following econometrically estimable equation:

$$wd_t = b_0 + b_1 wp_t + b_2 y_t + c_i X_{it} + u_t \tag{9}$$

Where, wdu, y and wp are the logarithmic transformations of water demand, household income and water price; X is the vector of *i* number of control variables, such as urban population, temperature; u is the error term; subscript t indicates time.

3.1 Data

This study used the annual data from 1994 to 2023 for empirical analysis. The water pricing system in Saudi Arabia uses an increasing block tariff (IBT). The government implemented a five-tiered system in which the price per cubic meter of water increases with higher consumption levels.

Table 1 displays the domestic water prices before and after 2015. We calculated the average price of water supplied for domestic use as average water bills divided by total water use, following Opaluch (1982) and Ruijs, Zimmermann, and van den Berg (2008). Nominal water prices are given in (\$/m^3), converted to SAR/m^3, and divided by the consumer price index (CPI) to transform them into real prices. (A detailed explanation of the water price calculation methodology is provided in Appendix B.) For the projected values of water price, this study assumes that the future domestic water price will remain at the 2023 level.

Table 1. Block tariff rates for domestic water before and after price reform.

Water tariff rates before 2015			Water tariff rates after 2015		
Block	Monthly consumption (m^3)	Price (\$/m3)	Block	Monthly consumption(m^3)	Price (\$/m^3)
1	1-50	0.027	1	1-15	0.027
2	51-100	0.04	2	16-30	0.27
3	101-200	0.53	3	31-45	0.80
4	201-300	1.07	4	46-60	1.067
5	301+	1.60	5	61+	1.60

Source: McIlwaine and Ouda (2020), citing MEWA (2017).

The model's dependent variable is annual domestic water demand, measured in million m^3. Data is sourced from the Ministry of Environment, Water and Agriculture (MEWA) and various statistical yearbooks published by the General Authority of Statistics (GASTAT). Household income, expressed in millions of Saudi riyals (SAR), serves as a proxy for the income variable, with data obtained from the World Development Indicators (WDI) online database. To derive real income values, nominal household income figures are adjusted by dividing them by the consumer price index. The projected values of household consumption up to 2050 are based on the Oxford Economics forecasted growth rate of GDP.

The urban population, measured by the number of residents, is also taken from WDI. The projected values of the urban population up to 2050 are based on the United Nations' World Population Prospects, revised in 2024 (UN 2024). This variable captures demographic demand drivers, since a larger urban population directly increases water requirements for drinking, hygiene, and other needs. TEMP is the annual temperature (°C), reflecting the climatic influence on water use. In a hot, arid country like the Kingdom of Saudi Arabia (KSA), higher temperatures are expected to boost water use (for cooling, irrigation, and personal use). The temperature data are drawn from the World Bank (2025) Climate Change Knowledge Portal. For temperature data, the SSP5-8.5 10-90th percentile range was used, which refers to the Shared Socioeconomic Pathways (SSP) framework applied in the IPCC Sixth Assessment Report. Descriptive statistics are given below in Table 2.

Table 2. Descriptive statistics.

	Variables				
Statistic	Municipal water use (m^3/year)	Temperature (Celsius)	Urban population (million)	Income (SAR million)	Price (real SAR/m^3)
Mean	3,129.88	26.23	19.12	639,143	0.62
Median	2,280.50	26.15	19.35	601,624	0.14
Maximum	5,285.20	27.14	28.26	1,208,477	2.13
Minimum	1,707.00	25.46	10.00	319,821	0.10
Std. Dev.	1,292.93	0.46	5.89	300,554	0.84

Source: McIlwaine and Ouda (2020), citing MEWA (2017).

Results and Discussion

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4.1 Estimation Results

We began our empirical analysis by examining the time series features of the data. The results of the unit tests are reported in Appendix C. The coefficient estimates from the Stone-Geary model, presented in Table 3, align with prior expectations concerning income, water price, weather, and urban population. The results indicate that the marginal budget share of income spent on water is statistically significant and positive, although the magnitude of the coefficient is small. However, a notable difference exists between the periods before and after 2016.

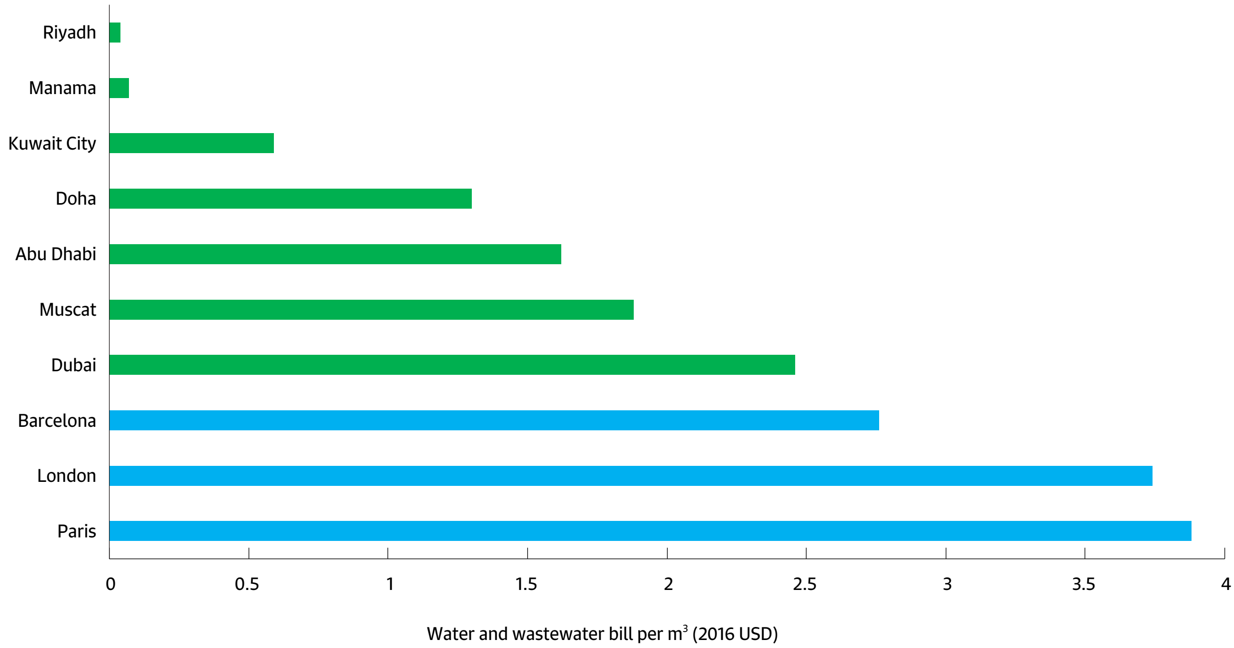
Table 3. Water demand model estimates.

Variable	Coefficient	t-Statistic	Prob
$\frac{Y}{P} * DUM2016$	0.00214	3.624	0.002
$\frac{Y}{P} * (1 - DUM2016)$	0.00010	3.393	0.003
POP	0.0001	2.753	0.012
$TEMP$	420.045	2.703	0.013
C	-12031.42	-3.016	0.006
$DSH2012$	1669.759	25.006	0.000
$DUM1922$	323.157	5.866	0.000
$T11997$	160.885	5.726	0.000
$T12015$	-108.809	-4.976	0.000

The share of income spent on water increased after the price reform: from an average of 0.06% during 1994-2015 to 0.66% during 2016-2023, well below the World Bank affordability benchmark of 3%-5% of household income/expenditure in developing countries. The estimated coefficient for the marginal budget share (β_w) to water is relatively low (0.002), since water expenses comprise only a small portion of household expenditures compared to international standards (see Figure 1).

Households in the GCC pay less for water compared to households in Paris, Barcelona, and London. Within the GCC, households in Abu Dhabi pay approximately 50 times more than those in Riyadh (World Bank Group 2017). The estimates reported for Riyadh are based on data before Saudi Arabia's water price reform. After the reform, the weighted average price of water increased about six times, raising the expected water bill in Riyadh to approximately \$0.24 per m^3.

Figure 1. Water and wastewater bill per m^3 in selected cities.



Source: World Bank Group (2017).

The urban population and temperature contribute positively to water demand, with statistically significant coefficients. The coefficient of urban population is 0.0001, indicating that as the population increases, total urban water demand rises proportionally. This magnitude suggests each additional urban resident contributes roughly 100 cubic meters of extra water use per year (since 0.0001 in the model units corresponds to 0.0001 million m^3 per person, i.e., $100m^3$). This equates to about 274 liters per person per day, which is reasonable given historically high per-capita water use in Saudi Arabia. The positive sign confirms that population and urbanization growth are major drivers of an increase in aggregate water demand.

A dummy variable DSH2012 is used to account for a structural break or a one-time shock around 2012. This captures an abrupt change in water demand in 2012 or a permanent increase from 2012 onward that other factors do not explain. DSH2012 = 1 for 2012-2023 and 0 otherwise. DUM1922 is a dummy for 2019-2022: this dummy equals 1 for 2019 through 2022, covering an unusual period in the data. It likely captures anomalies during these years – for example, changes in usage patterns during the COVID-19 pandemic when residential water use spiked as people stayed home.

Since our model explicitly incorporates a subsistence level of water use, denoted by (γ), we proceed to quantify this threshold by employing the estimated coefficients, specifically (β), in conjunction with the mean values of the explanatory variables. This calculation uses the following equation, which reflects the underlying relationship specified in the model and allows for a data-driven estimation of the minimum required water use level consistent with observed behavior.

$$\gamma = (1-\beta)(\alpha_0 + \alpha_1 POP + \alpha_2 TEMP + \alpha_3 DSH12012 + \alpha_4 D191922 + \alpha_5 T1997 + \alpha_6 T12015)$$

The estimated subsistence level of water use in Saudi Arabia is approximately 406 cubic meters (m^3) per household per year, with the national average household size of 3.7 persons. This corresponds to a monthly subsistence requirement of around 9.1m^3 per person. The estimated value of threshold water use indicates that approximately 55% of water use in Saudi Arabia is not responsive to price change. This benchmark reflects the minimum volume of water deemed necessary to meet basic needs and maintain essential living standards. It is a critical parameter for understanding baseline consumption behavior and designing equitable water policies.

The estimated subsistence-level water use in this study closely matches Renzetti, Dupont, and Chitsinde (2015) for British Columbia and García-Valiñas, Martínez-Espiñeira, and González-Gómez (2010) for Spain, but is generally higher than the estimates of Dharmaratna and Harris (2012) for Sri Lanka and Dalmas and Reynaud (2004) for Slovakia. In contrast, it is lower than Gaudin, Griffin, and Sickles's (2001) estimates for Texas. These cross-country differences reflect variations in climate, pricing, household structure, and access to alternative water sources, emphasizing the importance of interpreting subsistence water thresholds within local economic, climatic, and institutional contexts.

We also estimated the price elasticity of water demand using the Stone-Geary functional form, as specified in Equation (8). In this model, the estimated price elasticity of water demand is not constant but varies with changes in income and water prices. By applying average values for household income, real water prices, and water use over the period 2016-2023,[4] we calculated an average price elasticity of -0.25. This estimate is slightly more elastic than the -0.20 value reported by Javid (2025) for Saudi Arabia. It is important to note that Javid's (2025) estimate was derived using time series data spanning from 1994 to 2023 and does not explicitly differentiate between the pre- and post-water price reform periods. In contrast, our estimate is based solely on data from the post-reform period, which may partly explain the difference in elasticity values.

Our estimated elasticity falls within the range documented in the literature. For example, Gaudin, Griffin, and Sickles (2001) reported price elasticities between -0.19 and -0.28, while Renzetti, Dupont, and Chitsinde (2015) reported a broader range of -0.09 to -0.41. Thus, our estimate of -0.25 falls comfortably within these bounds, suggesting that the behavioral response of Saudi water use under the new pricing structure is broadly consistent with international findings.

As noted earlier, and in line with previous empirical studies, we employed a log-log functional form specified in Equation (9) to estimate the income and price elasticities of water demand. The estimation was conducted using *Autometrics* – a machine learning based model selection algorithm – applied with super saturation to an autoregressive distributed lag (ARDL) model for the period 1994-2023. Details of the estimation process and diagnostic and cointegration testing are provided in Appendix D.

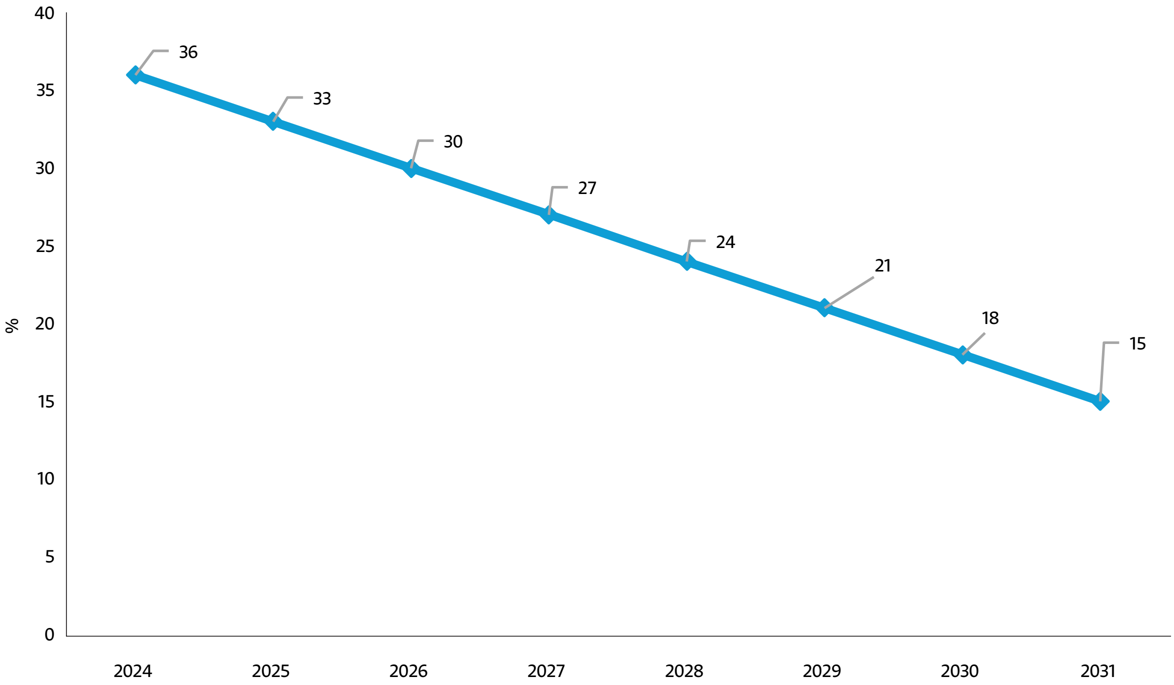
Our findings indicate that the long-run income and price elasticities of water demand are 1.16 and -0.09, respectively. These estimates are broadly consistent with those reported in other empirical studies. The observed differences between the elasticities derived from the log-log specification and those obtained using the Stone-Geary functional form are primarily attributable to differences in model structure and in the methods used to estimate/calculate elasticities.

05

Water Demand Simulations

According to the Saudi Water Partnership Company (SWPC 2024), the persistently high demand for water in Saudi Arabia can be attributed to water losses occurring during transmission and distribution (before the meter), and inefficient use and losses within housing units (after the meter). The MEWA has outlined a comprehensive plan to enhance water use efficiency and reduce water system losses before the meter from the current level of approximately 36% in 2024 to around 15% by 2031 and thereafter (Figure 2).

Figure 2. MEWA strategy on reduction of water losses (% annually).



Source: SWPC (2024).

In Saudi Arabia, estimating the total amount of water that is produced and enters a water distribution system but is lost or unaccounted for before it reaches paying customers – so-called "non-revenue water (NRW)" – is challenging. NRW broadly includes real (physical) losses, such as leaks and bursts (focus of MEWA program), and apparent (commercial) losses, including water theft, unauthorized use, and metering inaccuracies, which are more difficult to quantify and address with investments alone. Data for Riyadh shows that 34% of water is non-revenue. It comprises 21% physical and 13% commercial losses arising from under-metering, illegal connections, and authorized unbilled consumption (for free public use). Additionally, about 24% of billed water is unpaid, which is a collection issue (Ordem dos Engenheiros 2015).

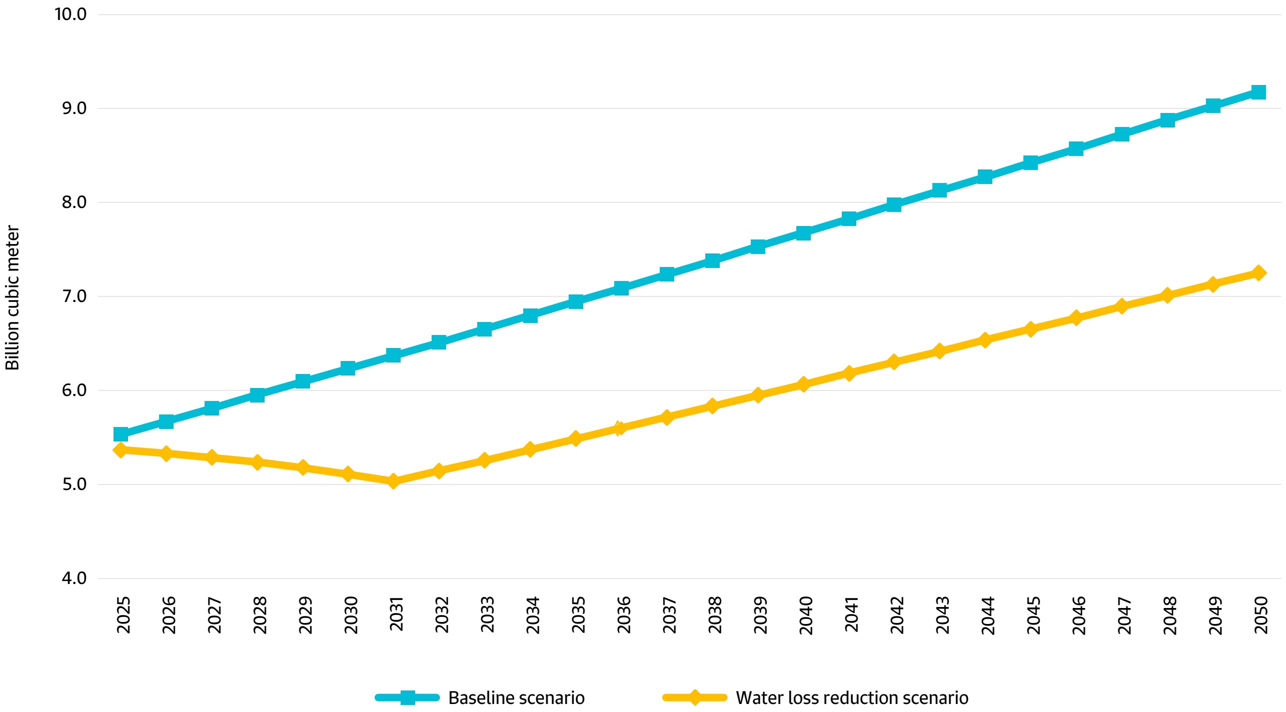
We simulate two scenarios of water demand using the water demand model described in the previous section.

- **Scenario 1:** Baseline scenario, assuming no reduction of water losses.

- **Scenario 2:** Water-loss reduction, assuming implementation of MEWA's strategy for reducing water losses from 36% in 2024 to 15% by 2031 and no further policy action thereafter. All other variables, such as water tariffs, are the same as in the baseline scenario.

Simulation results are illustrated in Figure 3.

Figure 3. Urban water demand scenarios.



Source: Authors' calculations.

In the baseline scenario, KSA urban water demand would increase by 67%, from approximately 5.4 billion cubic meters per year (m^3/year) in 2025 to around 9.2 billion m^3/year by 2050 (see Figure 3),[5] reflecting the impact of rapid urbanization, economic diversification, mega infrastructure development projects, and the expansion of the tourism sector. Unless new water infrastructure projects are implemented on time, a structural water supply deficit would emerge around 2035, with the currently available and committed facilities.

Suppose MEWA's planned water losses reduction program is implemented. In that case, water demand is expected to decline to 5 billion m^3/year in 2031 and then gradually increase in line with population growth and economic development, to 7.2 billion m^3/year by 2050, in the absence of further water efficiency improvements – about 2 billion m^3/year of avoided desalination capacity in 2050. Under this scenario, the available and committed water supply capacity will be sufficient to meet demand until 2045.

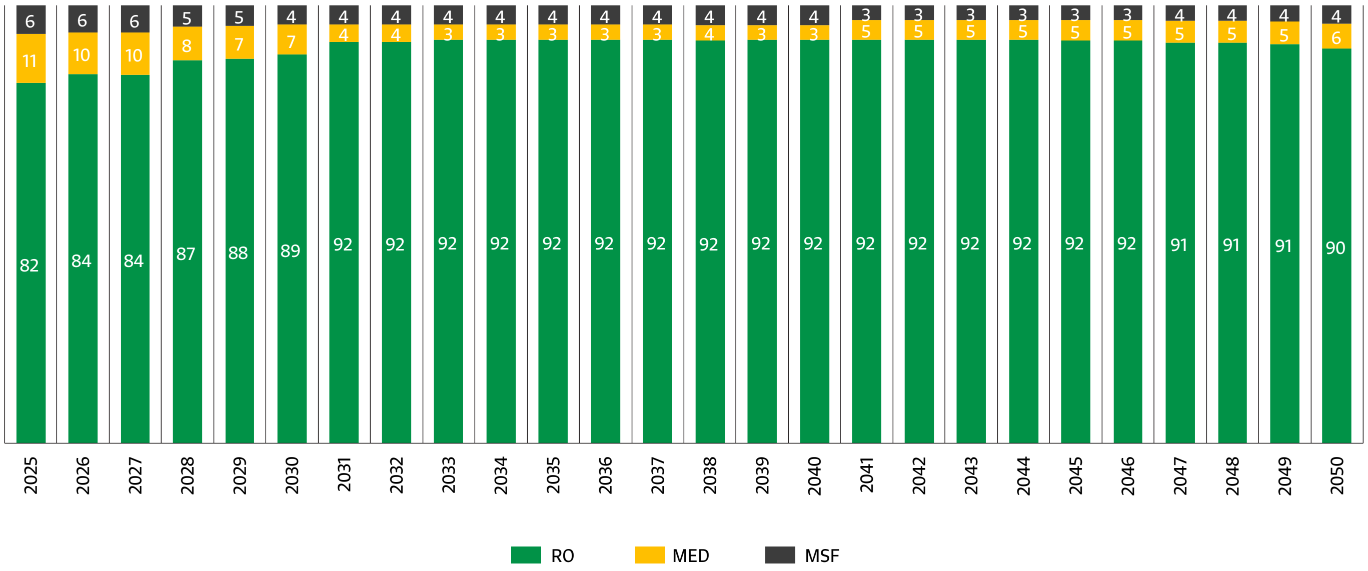
The projection data for groundwater and surface water used for municipal purposes are sourced from the Fourth National Communication of the Kingdom of Saudi Arabia, submitted to the United Nations Framework Convention on Climate Change in March 2022. According to this report, groundwater and surface water together account for 16% of total water demand in 2025, gradually declining to 10% by 2050. We subtracted the projected groundwater and surface water components from the total municipal water demand to estimate the demand for desalinated water for municipal use.

5.1 Energy for Desalination in Saudi Arabia

This section projects the energy demand for desalinated water production in Saudi Arabia across the three primary desalination technologies employed in the Kingdom: reverse osmosis (RO), multi-stage flash distillation (MSF), and multi-effect distillation (MED).

As of 2025, RO is the dominant desalination technology in Saudi Arabia, accounting for approximately 82% of total desalinated water production. It is followed by MED at roughly 11% and MSF at around 6% (see Figure 4). The mix of technologies is evolving rapidly, and by 2031, RO's share is projected to increase to 92%. This shift is driven by the fact that all new desalination plants now rely exclusively on RO technology, owing to its lower energy consumption, modular scalability, and reduced capital costs compared with thermal methods. After 2031, RO is expected to maintain its dominant position, as older MSF and MED plants will gradually be phased out upon reaching the end of their operational lifespans or replaced by more cost-effective and energy-efficient RO facilities.

Figure 4. RO, MED, and MSF share (%) of desalination water.

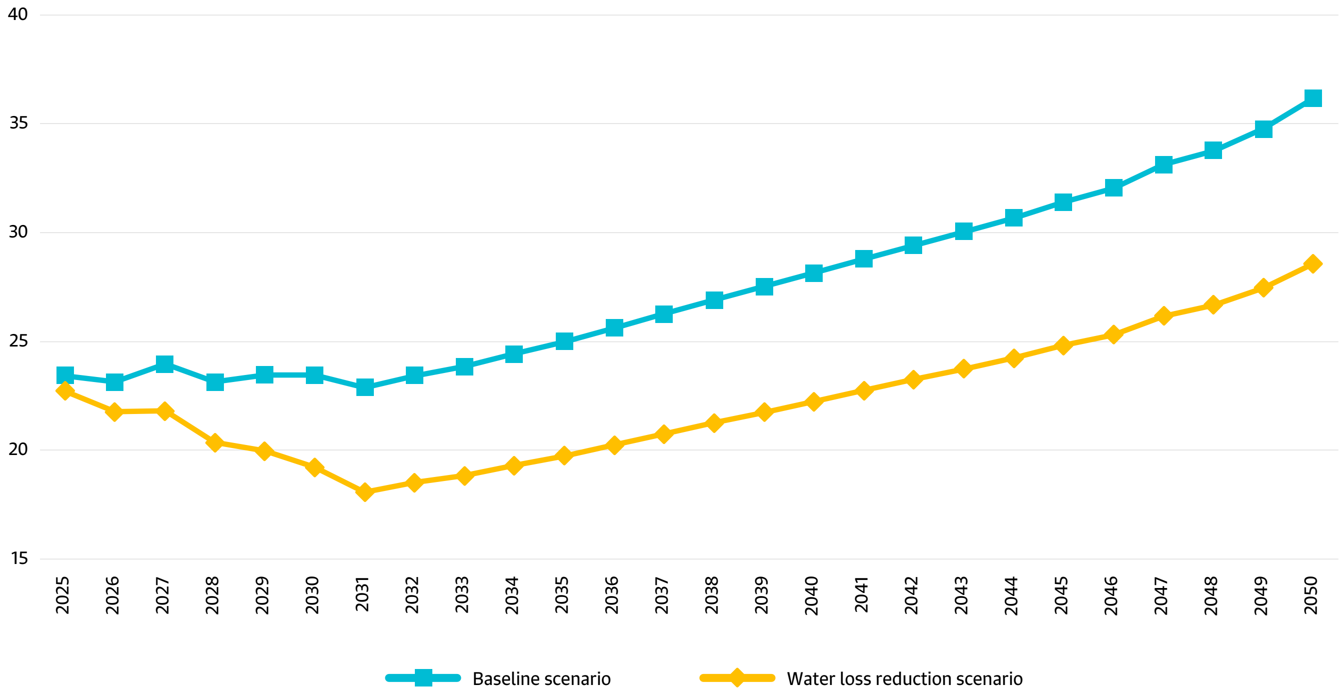


Source: Author's calculation based on SWPC (2024) data.

RO is increasingly favored for its lower specific energy consumption and modular scalability. RO systems rely on high-pressure pumps to force seawater through semi-permeable membranes, typically consuming between 3-5 kWh per cubic meter (kWh/m^3) of water produced. According to the Saudi Water Authority (SWA 2024),[6] the total plant energy use for an average RO system is approximately 3.4 kWh/m^3. This value was adopted for projecting desalinated water production from RO technology. The Ras Al Khair MSF plant has achieved outstanding performance, producing 740,000 cubic meters of water per day while consuming just 14.6 kWh of energy per cubic meter. It is the only MSF plant in Saudi Arabia expected to remain operational until 2052 (SWA 2024). For the energy demand projection of the MSF plant, we used 14.6 kWh/m^3. As reported by (SWA 2024), MED desalination plants are more energy-efficient than MSF, using between 6 and 22.5 kWh/m^3, with an average of 11.9 kWh/m^3 to treat 1 cubic meter of seawater.

Figure 5 shows the simulated energy demand for desalinated water production under two water demand scenarios: baseline and loss reduction. In the baseline scenario, where no improvements are made and current water losses (36%) persist, energy demand is projected to rise significantly from 23.42 TWh in 2025 to 36.17 TWh by 2050. The loss reduction scenario assumes a decrease in water losses by 21% (from 36% to 15%) relative to the baseline by 2031, resulting in a lower projected energy use by desalination plants from 22.7 TWh in 2025 to 18.1 TWh in 2031 – before increasing again to 28.58 TWh by 2050. The difference of 4.6 TWh in 2031 grows to approximately 7.6 TWh by 2050 – equivalent to nearly half of the current power consumption of the city of Madinah.

Figure 5. Energy demand for desalinated water production in Saudi Arabia.



Source: Authors' calculations.

Desalination Water Production Cost in Saudi Arabia

06

The three desalination technologies are capital- and energy-intensive, requiring significant upfront investments and continuous energy consumption, particularly in thermal and membrane-based processes.

In Saudi Arabia, the full financial cost of water desalination is assessed using the levelized cost of water (LCOW) metric. LCOW represents the total cost of water produced over the entire lifecycle of desalination plants, encompassing annualized capital expenditures, annual operating costs, and the plants' lifespan. This formula serves as an initial approximation of the minimum revenue needed to cover all supply costs, including a reasonable return on capital, thereby ensuring sustainable and commercially viable investments, maintenance, and operation of the facilities. By dividing the total annual LCOW by the volume of water produced, one can estimate the average tariff per cubic meter of water at the gate of desalination plants. The mathematical equation for calculating the levelized cost is as follows:

$$LCOW = \frac{(capital\ cost \times CRF) + annual\ O\&\ M\ cost}{Q}$$

Where $CRF = \frac{r(1+r)^n}{[(1+r)^n]-1}$ n= useful life of the plant in years, r= discount rate, Q = annual water production.

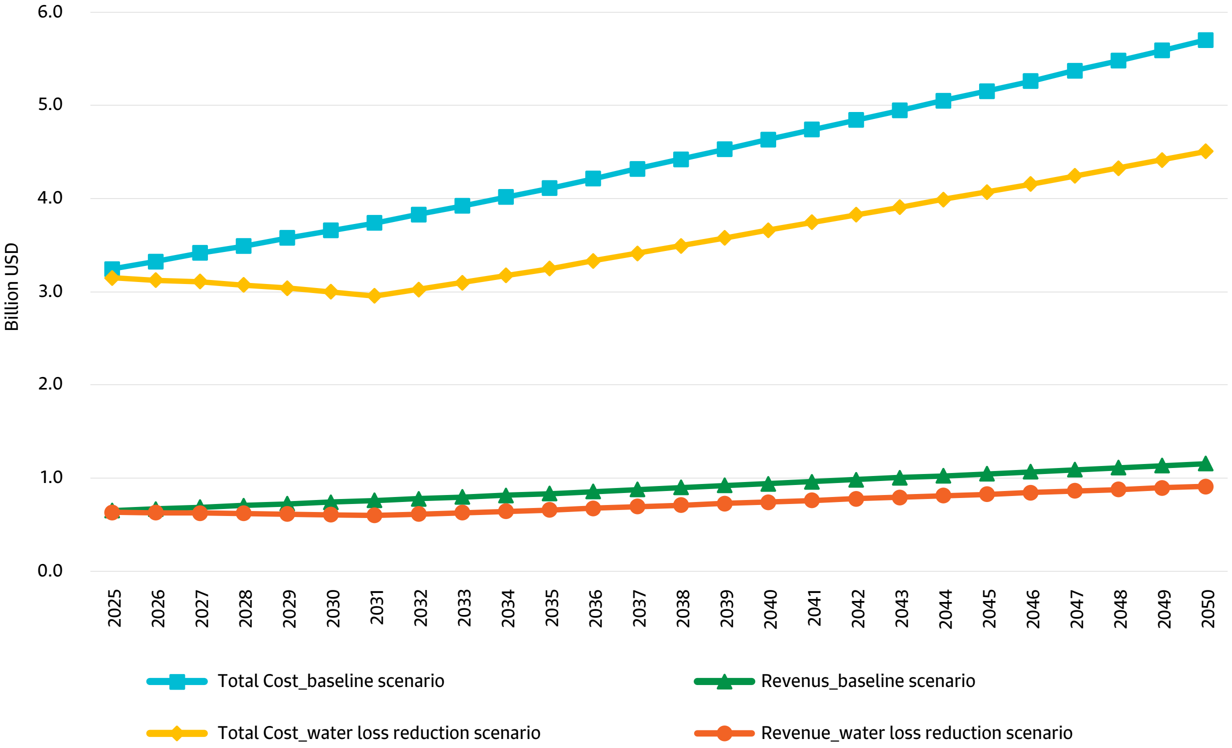
To estimate the cost of reverse osmosis desalination in Saudi Arabia, we incorporate data from three categories of plants: those currently operational, those under construction, and those in the planning phase. For baseline capital and operating cost estimates, we rely on data provided by the World Bank (2019), which reports capital expenditure (CAPEX) and operational expenditure (OPEX) for six RO desalination plants with varying production capacities across different regions of the Kingdom. Using this dataset, we calculated weighted average CAPEX and OPEX values to derive representative per-unit costs for producing cubic meters of water. CAPEX and OPEX data for selected desalination plants in Saudi Arabia are presented in Appendix E.

We considered the actual operational lifespan of different plants, which ranged from 20 to 35 years, based on verified commissioning and decommissioning dates. Instead of using a single 12% interest rate (as in World Bank 2019), we applied specific discount rates aligned with the plants' commissioning dates to better reflect changing financing conditions and capital costs over time.

The average levelized cost of water production from RO in Saudi Arabia is approximately $0.68 per cubic meter. Costs vary by region. For example, the Tabuk region reports one of the lowest production costs at $0.65/m3, while the Madinah region sees a slightly higher cost of around $0.72/m3, reflecting differences in local infrastructure, energy supply, and operational efficiency.

For the MSF desalination plant, we consider the Ras Al Khair desalination plant one of the world's largest and most technologically advanced plants. The total capacity is 1,025,000m^3/day, including 727,000 m^3/day from MSF units and 307,000 m^3/d from RO units (World Nuclear Association 2024). The cost of water production from the Ras Al Khair MSF component is approximately $0.83 per cubic meter. According to SWPC (2024), Ras Al Khair is the only MSF desalination plant in Saudi Arabia expected to operate beyond 2025, with an anticipated lifespan until 2052. All other MSF plants in the country are scheduled for decommissioning by the end of 2025, as Saudi Arabia transitions to more cost-effective and energy-efficient RO technologies.

Figure 6. Cost of desalination water production and sale revenue in Saudi Arabia.



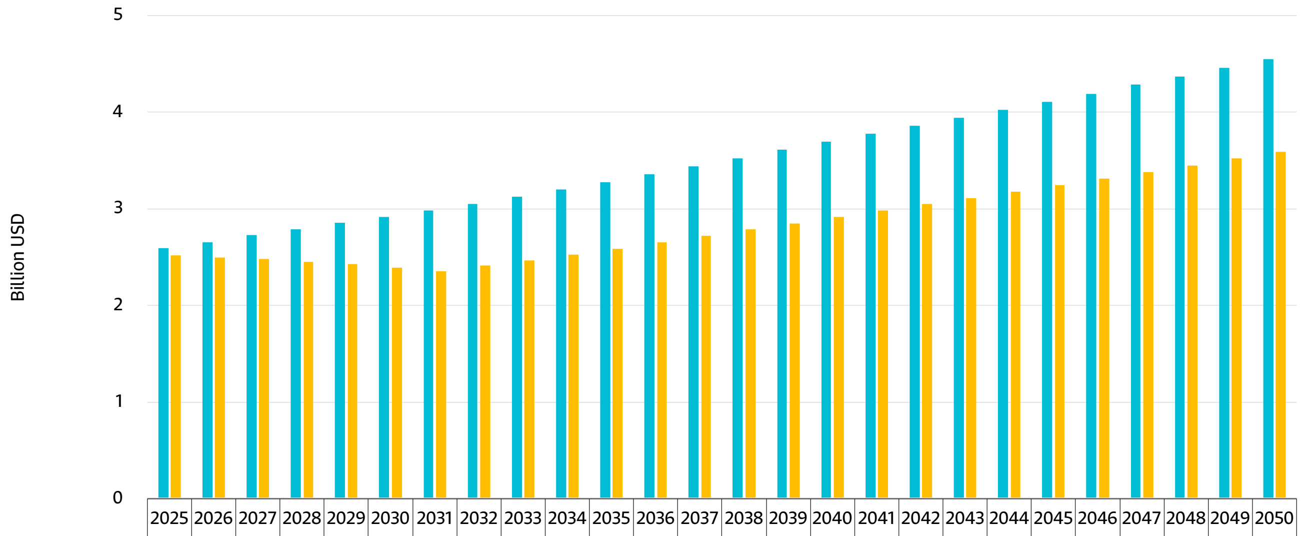
Source: Authors' calculations.

Figure 6 presents the projected costs of desalinated water production and sales revenue under two demand scenarios. In the baseline scenario, which assumes no reduction in water losses, the total levelized cost of desalinated water production is projected to increase from approximately $3.1 billion in 2025 to $3.7 billion in 2031 and $5.7 billion by 2050, while sales revenue is expected to rise from $0.65 billion in 2025 to about $1.2 billion by 2050.

In the water loss reduction scenario, where water losses decrease from 36% in 2024 to 15% by 2031, the total desalination cost initially declines from $3.1 billion in 2025 to around $2.9 billion in 2031. However, after 2031, costs begin to rise again, reaching $4.5 billion by 2050, driven primarily by increasing demand despite improved loss control.

According to GASTAT (2025), the total production of desalinated water in 2023 was 3,019 million m^3 per year, which is assumed to represent the total desalinated water consumed by households. The per-household monthly water consumption was estimated using an average household size of 3.7 persons. Based on this assumption, the per-household monthly desalinated water consumption was approximately 28 m^3, placing it within the first and second tariff blocks. Accordingly, the price of desalinated water was determined by dividing the total water bill by the monthly household consumption of desalinated water. Thus, the calculated average water price of desalinated water is $0.14/m3. We estimate water sale revenue under the two demand scenarios (Figure 7). Despite a price increase in 2016, water sales revenue in Saudi Arabia remains relatively low and covers only about 20% of the desalinated water production cost (excluding transmission and distribution).

Figure 7. Cost recovery gap for desalination water production in Saudi Arabia.



	2025	2026	2027	2028	2029	2030	2031	2032	2033	2034	2035	2036	2037	2038	2039	2040	2041	2042	2043	2044	2045	2046	2047	2048	2049	2050
🟦 Subsidy_baseline scenario	2.6	2.6	2.7	2.7	2.8	2.9	2.9	3.0	3.1	3.2	3.2	3.3	3.4	3.5	3.6	3.6	3.7	3.8	3.9	4.0	4.1	4.1	4.2	4.3	4.4	4.5
🟨 Subsidy_water loss reduction scenario	2.5	2.5	2.4	2.4	2.4	2.4	2.4	2.4	2.5	2.5	2.6	2.7	2.7	2.8	2.9	2.9	2.9	3	3.1	3.2	3.2	3.3	3.4	3.5	3.5	3.5

Source: Authors' calculations.

The remaining 80% of total production costs not recovered through water sales becomes a fiscal liability. Ouda (2013) reported that Saudi consumers paid less than 5% of water production costs, while the government covered the remaining 95% through subsidies; however, these findings are based on data collected before the water price reforms.

Figure 7 illustrates the projected cost-recovery gap in desalination water production estimated in this study across the two water demand scenarios. The blue bars (subsidy baseline) scenario represent the baseline, in which the government must cover most of the shortfall between production costs and water sale revenues. The green bars (subsidy water-loss reduction) scenario reflect a water-loss reduction scenario, where losses are reduced by 21% (from 36% to 15%) by 2031.

Under the baseline demand scenario, the annual subsidy requirement – defined as the difference between total production cost and water sales revenue – is projected to range from approximately $2.6 billion per year in 2025 to $4.6 billion in 2050. The total undiscounted fiscal expenditures to subsidize water production during 2025-2050 would amount to $91.4 billion without the government program.

Under a water-loss reduction scenario, the cost recovery gap would narrow, yielding fiscal savings of about $0.2 billion in 2026, gradually rising to $0.5 billion in 2031 and nearly $1 billion per year by 2050. This does not even account for the lower fiscal costs of maintaining the water transmission and distribution system. Over the 2025-2050 period, cumulative fiscal savings would total $17.5 billion.

Conclusion and Policy Recommendation

07

This study examines several critical challenges that influence the long-term sustainability of the urban water sector in Saudi Arabia: the subsistence level of urban water use, future water demand projections, the importance of reducing water losses, and the economic and energy implications of desalination.

First, this study employs the Stone-Geary utility-based demand model to estimate Saudi Arabia's domestic water demand. The empirical results indicate that approximately 55% of the urban water use occurs at the subsistence level of consumption. Policymakers can use this insight to design conservation programs and a tariff structure that ensure a minimum lifeline supply for every household while discouraging excessive or wasteful use above that subsistence level. The findings support the development of an equitable tariff system in which the first two blocks of consumption (covering subsistence needs) remain affordable, and higher usage blocks are priced progressively to incentivize efficiency.

Second, based on the Stone-Geary demand function, the study estimates that urban water demand might increase by 66% by 2050 under a business-as-usual scenario. This growth could surpass planned supply expansions and jeopardize supply reliability.

Third, the study quantifies the impact of water loss reduction, showing that reducing water losses from 36% in 2024 to 15% by 2031, in line with MEWA's national strategy, can significantly narrow the gap between demand and supply, meeting projected water demand, while avoiding the need for 2 billion m^3/year of desalination capacity and 7.6 TWh of electricity demand by 2050 (equivalent to about half of current electricity consumption in Madinah).

The key conclusion is that the Kingdom can significantly reduce its fiscal liabilities for desalination by reducing water losses.

The estimated water tariff revenues ($0.65 billion in 2025) cover only about 20% of the levelized cost of desalination ($3.2 billion in 2025), leaving a cost recovery gap – or fiscal liability – of $2.6 billion in 2025. Without the MEWA program, this could raise to $4.6 billion per year by 2050, even before accounting for water transmission and distribution system costs.

Without the MEWA program, the total fiscal liability accumulated by the Saudi government from subsidized desalination over the period 2025-2050 could reach $91.4 billion. Implementation of the program would reduce desalination subsidy requirements by $0.5 billion in 2031 and nearly $1 billion in 2050. Over the 2025-2050 period, cumulative fiscal savings would reach $17.5 billion.

The results highlight the economic and energy implications of Saudi Arabia's water management. Our analysis demonstrates the strong water-energy nexus by quantifying the amount of desalinated water required to meet projected demand and the electricity needed to produce it. This has significant implications for national planning: as desalination capacity expands, it will depend heavily on energy resources, which carry both financial costs and environmental impacts. The study’s cost-recovery assessment objectively evaluates the fiscal burden associated with current water supply practices.

The findings indicate that, under existing tariff levels, government subsidies would need to increase substantially to offset rising water production costs. These findings can help national stakeholders, including government ministries, regulators, and utility companies, make informed decisions about infrastructure investments and water loss reduction initiatives.

This study does not assess the economic and fiscal feasibility of other opportunities for more integrated urban water management policies that would encourage conservation and efficiency, such as tariff reforms or behavioral incentives. Such reforms could further improve the operational efficiency, reliability, and sustainability of water and energy systems, while at the same time reducing fiscal liabilities and freeing up fiscal space to advance other goals of Saudi Arabia’s Vision 2030.

Endnotes

[1] Unless otherwise specified, the terms “municipal water demand” and “domestic water demand” are used interchangeably in this study.

[2] Bredariol et al. (2024) data on energy demand for desalination in the Middle East are expressed in petajoules (PJ). We then converted PJ to million tonnes of oil equivalent (mtoe) using the IEA conversion factor: 1 PJ = 0.0238846 mtoe.

[3] See Vermeulen and Barkema (2001) and Yip and Tsang (2007) for more details of partition.

[4] The period of analysis spans from 1994 to 2023, as indicated in the data section. Due to the water price reform implemented in 2016, a structural break occurred in the data. Therefore, the period from 2016 to 2023 was used to estimate the price elasticity of water demand.

[5] The Ministry of Environment, Water, and Agriculture (MEWA) revised historical domestic water demand data, resulting in much higher values than previously reported. Consequently, the domestic water demand projections in this study are significantly higher than those in Javid (2025). For example, the earlier estimate for domestic water demand in 2023 was 3.6 billion m^3 annually, but updated figures from MEWA for 2023 now indicate 5.3 billion m^3 per year.

[6] *Desalination Energy Consumption: A Comprehensive Report on Energy Consumption in the Water Desalination Sector* (SWA 2024).

References

Al-Masri, Raya A., Jonathan Chenoweth, and Richard J. Murphy. 2019. "Exploring the Status Quo of Water-Energy Nexus Policies and Governance in Jordan." *Environmental Science & Policy* 100: 192–204. https://doi.org/10.1016/J.ENVSCI.2019.06.012.

Al-Mutrafi, Homoud, Waleed Al-Zubari, Alaa El-Sadek, and Ibrahim Abdel Gelil. 2018. "Assessment of the Water-Energy Nexus in the Municipal Water Sector in Eastern Province, Saudi Arabia." *Computational Water, Energy, and Environmental Engineering* 7: 1-26. https://doi.org/10.4236/cweee.2018.71001.

Al-Qunaibet, Mohammad H., and Richard S. Johnston. 1985. "Municipal Demand for Water in Kuwait: Methodological Issues and Empirical Results." *Water Resources Research* 21, no. 4: 433–38. https://doi.org/10.1029/WR021i004p00433.

Al-Zubari, Waleed, Abdulaziz Al-Turbak, Walid Zahid, Khalid Al-Ruwis, Ali Al-Tkhais, Ibrahim Al-Muataz, Ahmed Abdelwahab, Ahmed Murad, Meshari Al-Harbi, and Zaher Al-Sulaymani. 2017. "An Overview of the GCC Unified Water Strategy (2016–2035)." *Desalination and Water Treatment* 81: 1–18. https://doi.org/10.5004/dwt.2017.20864.

Arbués, Fernando, María Ángeles García-Valiñas, and Roberto Martínez-Espiñeira. 2003. "Estimation of Residential Water Demand: A State-of-the-Art Review." *The Journal of Socio-Economics* 32, no. 1: 81–102. https://doi.org/10.1016/S1053-5357(03)00005-2.

Barau, Aliyu Salisu, and Naeema Al Hosani. 2015. "Prospects of Environmental Governance in Addressing Sustainability Challenges of Seawater Desalination Industry in the Arabian Gulf." *Environmental Science & Policy* 50: 145–154. https://doi.org/10.1016/j.envsci.2015.02.008.

Bredariol, Tomás de Oliveira, Jinsun Lim, and Leonie Staas. 2024. "Energy Is Vital to a Well-Functioning Water Sector." IEA Commentary, March 22. https://www.iea.org/commentaries/energy-is-vital-to-a-well-functioning-water-sector.

Castle, Jennifer L., Jurgen A. Doornik, and David F. Hendry. 2021. "Modelling Non-stationary 'Big Data'." International Journal of Forecasting 37, no. 4: 1556–1575. https://doi.org/10.1016/j.ijforecast.2020.08.002.

Dalmas, Laurent, and Arnaud Reynaud. 2004. "Residential Water Demand in the Slovak Republic." In *Econometrics Informing Natural Resources Management: Selected Empirical Analyses*, edited by Phoebe Koundouri, 83–99 (chap. 4). Cheltenham, UK and Northampton, MA: Edward Elgar. https://doi.org/10.4337/9781845424657.00010.

Deaton, Angus, and John Muellbauer. 1980. *Economics and Consumer Behavior*. Cambridge: Cambridge University Press.

deMonsabert, Sharon, and Barry L. Liner. 1998. "Integrated Energy and Water Conservation Modeling." *Journal of Energy Engineering* 124, no. 1: 1–19. https://doi.org/10.1061/(ASCE)0733-9402(1998)124:1(1).

Dharmaratna, Dinusha, and Edwyna Harris. 2012. "Estimating Residential Water Demand Using the Stone-Geary Functional Form: The Case of Sri Lanka." *Water Resources Management* 26, no. 8: 2283–2299. https://doi.org/10.1007/s11269-012-0017-1.

Doornik, Jürgen A. 2009. "Autometrics." In *The Methodology and Practice of Econometrics: A Festschrift in Honour of David F. Hendry*, edited by Jennifer L. Castle and Neil Shephard, 88–121. Oxford: Oxford University Press. https://doi.org/10.1093/acprof:oso/9780199237197.003.0004.

Doornik, Jürgen A., and David F. Hendry. 2013. "Model Selection Using Autometrics." In *Handbook of Research Methods and Applications in Empirical Macroeconomics*, edited by N. Hashimzade and M. Thornton, 72-95. Edward Elgar.

Doornik, Jürgen A., and David F. Hendry. 2024. *Modelling Dynamic Systems Using PcGive 15 (OxMetrics Documentation)*. Oxford: Timberlake Consultants.

Duan, Cuncun, and Bin Chen. 2020. "Driving Factors of Water–Energy Nexus in China." *Applied Energy* 257: 113984. https://doi.org/10.1016/j.apenergy.2019.113984.

Ericsson, Neil R., and James G. MacKinnon. 2002. "Distributions of Error Correction Tests for Cointegration." *The Econometrics Journal* 5, no. 2: 285–318. https://doi.org/10.1111/1368-423X.00085.

Espey, Molly, James Espey, and W. Douglass Shaw. 1997. "Price Elasticity of Residential Demand for Water: A Meta-Analysis." *Water Resources Research* 33, no. 6: 1369–1374. https://doi.org/10.1029/97WR00571.

Fayiah, Moses, ShiKui Dong, Shailendra Singh, and E. A. Kwaku. 2020. "A Review of Water–Energy Nexus Trend, Methods, Challenges and Future Prospects." *International Journal of Energy and Water Resources* 4: 107–118. https://doi.org/10.1007/s42108-020-00057-6.

García-Valiñas, Maria A., Roberto Martínez-Espiñeira, and Francisco González-Gómez. 2010. "Affordability of Residential Water Tariffs: Alternative Measurement and Explanatory Factors in Southern Spain." *Journal of Environmental Management* 91, no. 12: 2696–2706. https://doi.org/10.1016/j.jenvman.2010.07.029.

Gaudin, Sylvestre, Ronald C. Griffin, and Robin C. Sickles. 2001. "Demand Specification for Municipal Water Management: Evaluation of the Stone-Geary Form." *Land Economics* 77, no. 3: 399–422. https://doi.org/10.2307/3147133.

General Authority for Statistics (GASTAT). 2025. Government of Saudi Arabia. https://www.stats.gov.sa/en.

Gleick, Peter H. 1994. "Water and Energy." *Annual Review of the Environment and Resources* 19: 267–299. https://doi.org/10.1146/annurev.eg.19.110194.001411.

Gleick, Peter H. 1996. "Basic Water Requirements for Human Activities: Meeting Basic Needs." *Water International* 21, no. 2: 83–92. https://doi.org/10.1080/02508069608686494.

Hansen, Lars Gårn. 1996. "Water and Energy Price Impacts on Residential Water Demand in Copenhagen." *Land Economics* 72, no. 1: 66–79. https://doi.org/10.2307/3147158.

Hong, Jingke, Xiaoyang Zhong, Shan Guo, Guiwen Liu, Geoffrey Qiping Shen, and Tao Yu. 2019. "Water-Energy Nexus and Its Efficiency in China's Construction Industry: Evidence from Province-Level Data." *Sustainable Cities and Society* 48: 101557. https://doi.org/10.1016/j.scs.2019.101557.

Howard, Guy, Jamie Bartram, Ashley Williams, Alycia Overbo, and Jo-Anne Geere. 2020. *Domestic Water Quantity, Service Level and Health* (2nd ed.). Geneva: World Health Organization. https://apps.who.int/iris/handle/10665/338044.

Javid, Muhammad. 2025. "Modeling the Impact of Price and Usage Efficiency on Domestic Water Demand in Saudi Arabia." *Utilities Policy* 96: 101971. https://doi.org/10.1016/j.jup.2025.101971.

Manawi, Yehia M., Majeda A. M. M. Khraisheh, Ahmad Kayvani Fard, Farid Benyahia, and Samer Adham. 2014. "A Predictive Model for the Assessment of the Temperature Polarization Effect in Direct Contact Membrane Distillation Desalination of High Salinity Feed." *Desalination* 341: 38–49. https://doi.org/10.1016/j.desal.2014.02.028.

Martínez-Espiñeira, Roberto, and Céline Nauges. 2004. "Is All Domestic Water Consumption Sensitive to Price Control?" *Applied Economics* 36, no. 15: 1697–1703. https://doi.org/10.1080/0003684042000218570.

McIlwaine, Stephen J., and Omar K. M. Ouda. 2020. "Drivers and Challenges to Water Tariff Reform in Saudi Arabia." *International Journal of Water Resources Development* 36, no. 6: 1014–1030. https://doi.org/10.1080/07900627.2020.1720621.

Ministry of Environment, Water, and Agriculture (MEWA). 2017. Saudi National Water Strategy 2030. https://www.mewa.gov.sa/ar/Ministry/Agencies/TheWaterAgency/Topics/Pages/Strategy.aspx.

Nauges, Céline, and Dale Whittington. 2010. "Estimation of Water Demand in Developing Countries: An Overview." *The World Bank Research Observer* 25, no. 2: 263–294. https://doi.org/10.1596/4441.

Olmstead, Sheila M., W. Michael Hanemann, and Robert N. Stavins. 2007. "Water Demand under Alternative Price Structures." *Journal of Environmental Economics and Management* 54, no. 2: 181–198. https://doi.org/10.1016/j.jeem.2007.03.002.

Opaluch, James J. 1982. "Urban Residential Demand for Water in the United States: Further Discussion." *Land Economics* 58, no. 2: 225–227. https://doi.org/10.2307/3145896.

Ordem dos Engenheiros. 2015. *Water Supply and Sanitation in Saudi Arabia.* September 24. https://www.ordemengenheiros.pt/fotos/editor2/areainternacional/20150924.pdf.

Ouda, Omar K. M. 2013. "Review of Saudi Arabia Municipal Water Tariff." *World Environment* 3, no. 2: 66–70. https://www.researchgate.net/publication/262202288_Review_of_Saudi_Arabia_Municipal_Water_Tariff.

Phillips, Peter C. B. and Bruce E. Hansen. 1990. "Statistical Inference in Instrumental Variables Regression with I(1) Processes." *The Review of Economic Studies* 57, no. 1: 99–125. https://doi.org/10.2307/2297545.

Rambo, Khulood A., David M. Warsinger, Santosh J. Shanbhogue, and Ahmed F. Ghoniem. 2017. "Water-Energy Nexus in Saudi Arabia." *Energy Procedia* 105: 3837–3843. https://doi.org/10.1016/j.egypro.2017.03.782.

Renzetti, Steven, Diane P. Dupont, and Tina Chitsinde. 2015. "An Empirical Examination of the Distributional Impacts of Water Pricing Reforms." *Utilities Policy* 34: 63–69. https://doi.org/10.1016/j.jup.2014.12.004.

Ruijs, Arjan, Alexandra Zimmermann, and Marrit van den Berg. 2008. "Demand and Distributional Effects of Water Pricing Policies." *Ecological Economics* 66, nos. 2–3: 506–516. https://doi.org/10.1016/j.ecolecon.2007.10.015.

Saudi Water Authority (SWA). 2024. *Desalination Energy Consumption: A Comprehensive Report on Energy Consumption in the Water Desalination Sector*. Riyadh: Saudi Water Authority. https://www.scribd.com/document/808587066/SWCC-report.

Saudi Water Partnership Company (SWPC). 2024. *7-Year Statement (2024–2030)*. https://www.swpc.sa/wp-content/uploads/2024/09/SWPC_7-Years-Statement-2024_2030.pdf.

Sherif, Mohsen, Muhammad Usman Liaqat, Faisal Baig, and Mohammad Al-Rashed. 2023. "Water Resources Availability, Sustainability and Challenges in the GCC Countries: An Overview." *Heliyon* 9, no. 10: e20543. https://doi.org/10.1016/j.heliyon.2023.e20543.

Shevah, Yehuda. 2017. "Challenges and Solutions to Water Problems in the Middle East." In *Chemistry and Water*, 207–258. Oxford: Elsevier. https://doi.org/10.1016/B978-0-12-809330-6.00006-4.

Siddiqi, Afreen, and Laura Diaz Anadon. 2011. "The Water–Energy Nexus in Middle East and North Africa." *Energy Policy* 39, no. 8: 4529–4540. https://doi.org/10.1016/j.enpol.2011.04.023.

Stone, Richard. 1954. "Linear Expenditure Systems and Demand Analysis: An Application to the Pattern of British Demand." *The Economic Journal* 64, no. 255: 511–527. https://doi.org/10.2307/2227743.

United Nations (UN). 2024. *World Population Prospects 2024*. https://population.un.org/wpp.

U.S.-Saudi Business Council (USSBC). 2022. *Saudi Arabia's Water Sector: Economic Brief*. February. https://www.ussaudi.org/wp-content/uploads/2022/02/Water-2022-Economic-Brief.pdf.

Whittington, Dale, John Boland, and Vivien Foster. 2002. *Water Tariffs and Subsidies in South Asia: Understanding the Basics*. Water and Sanitation Program Paper 2. Washington, DC: World Bank. https://sswm.info/sites/default/files/reference_attachments/BOLAND%20and%20WHITTINGTON%202000%20Water%20Tariff%20Design%20in%20Developing%20Countries.pdf.

Vermeulen, Freek A. M., and Harry G. Barkema. 2001. "Learning Through Acquisitions." *Academy of Management Journal* 44, no. 3: 457–476. https://doi.org/10.5465/3069364.

World Bank Group. 2005. *A Water Sector Assessment Report on the Countries of the Cooperation Council of the Arab States of the Gulf*. http://documents.worldbank.org/curated/en/415761468052156362.

World Bank Group. 2017. *Water for Prosperity and Development: Risks and Opportunities for the Gulf Cooperation Council Countries*. https://documents.worldbank.org/en/publication/documents-reports/documentdetail/920341516598399529.

World Bank Group. 2019. *The Role of Desalination in an Increasingly Water-Scarce World*. Water Global Practice Technical Paper. https://documents.worldbank.org/en/publication/documents-reports/documentdetail/476041552622967264.

World Bank Group. 2025. “Climate Change Knowledge Portal (CCKP): Temperature Data.” https://climateknowledgeportal.worldbank.org.

World Nuclear Association 2024. “Nuclear Power in Saudi Arabia.” Information Library. https://world-nuclear.org/information-library/country-profiles/countries-o-s/saudi-arabia.

Worthington, Andrew C., and Mark Hoffmann. 2008. “An Empirical Survey of Residential Water Demand Modelling.” *Journal of Economic Surveys* 22, no. 5: 842–871. https://doi.org/10.1111/j.1467-6419.2008.00551.x.

Yip, Paul S. L., and Eric W. K. Tsang. 2007. “Interpreting Dummy Variables and Their Interaction Effects in Strategy Research.” *Strategic Organization* 5, no. 1: 13–30. https://doi.org/10.1177/1476127006073512.

Appendix A

$$L = \beta^w \ln\left(D^w - \gamma^w\right) + \beta^z ln\left(D^z - \gamma^z\right) + \lambda\left(Y - D^w P^w - D^Z\right) \tag{A1}$$

Taking the partial derivative of (A1) with respect to D^w , D^z and λ

$$\frac{\partial L}{\partial D^w} = \frac{\beta^w}{(D^w - \gamma^w)} - \lambda P^w = 0 \rightarrow \frac{\beta^w}{(D^w - \gamma^w)} = \lambda P^w \tag{A2}$$

$$\frac{\partial L}{\partial D^z} = \frac{\beta^z}{(D^z - \gamma^z)} - \lambda = 0 \rightarrow \frac{\beta^z}{(D^z - \gamma^z)} = \lambda \tag{A3}$$

$$\frac{\partial L}{\partial \lambda} = Y - D^w P^w + D^Z = 0 \rightarrow Y = D^w P^w + D^Z \tag{A4}$$

It is assumed that $\beta^w + \beta^z = 1$

By solving equation (A1) and (A2)

$$D^z = \frac{\beta^z P^w (D^w - \gamma^w) + \gamma^z \beta^w}{\beta^w} \tag{A5}$$

By substituting the value of D^z from equation (A5) into equation (A4), we have

$$Y = D^w P^w + \frac{\beta^z P^w (D^w - \gamma^w) + \gamma^z \beta^w}{\beta^w} \tag{A6}$$

$$Y\beta^w = \beta^w D^w P^w + \beta^z P^w (D^w - \gamma^w) + \gamma^z \beta^w \tag{A7}$$

$$Y\beta^w = \beta^w D^w P^w + \beta^z P^w D^w - \beta^z P^w \gamma^w + \gamma^z \beta^w \tag{A8}$$

$$Y\beta^w = D^w P^w (\beta^w + \beta^z) - \beta^z P^w \gamma^w + \gamma^z \beta^w \tag{A9}$$

$$D^w P^w = Y\beta^w + \beta^z P^w \gamma^w - \gamma^z \beta^w \tag{A10}$$

Solving for D^w

$$D^w = \frac{Y\beta^w + \beta^z P^w \gamma^w - \gamma^z \beta^w}{P^w}$$

$$D^w = \frac{Y\beta^w + (1 - \beta^w) P^w \gamma^w - \gamma^z \beta^w}{P^w}$$

$$D^w = \frac{Y\beta^w + P^w \gamma^w - \beta^w P^w \gamma^w - \gamma^z \beta^w}{P^w}$$

$$D^w = \frac{\beta^w (Y - P^w \gamma^w - \gamma^z)}{P^w} + \gamma^w \tag{A11}$$

Solving equation (A11) for B^w

$$B^w = \frac{P^w D^w - P^w \gamma^w}{Y - P^w \gamma^w - \gamma^z}$$

Appendix B: Price Calculation

The average water price is calculated by dividing the total water cost by the total amount of water used. For an increasing-block tariff, the water bill is calculated as follows (see Whittington, Boland, and Foster 2002 for details):

Let Q^* = amount of water used by a specific consumer

Q_1 = maximum amount of water used in the first block at the price P_1

Q_2 = maximum amount of water used in the second block at the price P_2

Q_3 = maximum amount of water used in the third block at the price P_3

If $Q^* < Q_1$, then the consumer's water used bill will be equal to $Q_1 P_1$

If, $Q_1 < Q^* <$ Q2 then the consumer's water used bill will equal $Q_1 P_1 + Q_2 P_2$

If $Q_1 + Q_2 < Q^* < Q_3$ then the consumer's water used bill will equal

$Q_1 P_1 + Q_2 P_2 + [Q^* - (Q_1 + Q_2)] P_3$ and so on for all blocks of the tariff

Following Opaluch (1982) and Ruijs, Zimmermann, and van den Berg (2008), the average price of water used is calculated as average water bills divided by total water used as follows:

$$P_t = \frac{Q_1P_1 + Q_2P_2 + Q_2P_3 + Q_4\,P_4 + [Q^* - (Q_1 + Q_2 + Q_3 + Q_4)]\,P_5}{Q}$$

Nominal water prices are given in (\$/m^3), converted to SAR/m^3, and divided by the consumer price index (CPI) to transform them into real prices.

Appendix C: Unit Root Test

We started our empirical analysis by examining the data's time series features. In time-series analysis, unit root tests are used to examine the stationarity of variables. A stationary series has a constant mean and variance over time, which is a crucial assumption in econometric modeling. Regression results may be spurious if variables are non-stationary and used without transformation. Augmented Dickey-Fuller (ADF) and Phillips-Perron (PP) tests are used to test the time-series properties of the variables included in this analysis, and the results are reported in Table A1.

Table A1. Unit root test results.

Panel a. ADF unit root test						
	Level				First difference	
Variables	t-stat	C	T	k	t-stat	C
w_t	-1.732		x	2	-4.669a	x
y_t	-2.530		x	0	-4.415a	x
pop_t	-2.568	x		0	-4.213a	x
p_t	-1.724		x	0	-5.204a	x
tem_t	-4.421b		x	0	-5.546a	
Panel b. PP unit root test						
w_t	-1.733		x	3	-4.651a	x
y_t	-2.527		x	3	-4.486a	x
pop_t	-2.569	x		0	-4.243a	x
p_t	-1.724	x		3	-5.203a	x
tem_t	-4.528a		x	2	-15.161a	

Notes: The optimal lag order k is selected based on the Schwarz criterion. a, b, and c indicate that the unit root null hypothesis is rejected at the 1%, 5%, and 10% level, respectively. C indicates constant, and T indicates constant and trend.

ADF and PP tests show that all variables except temperature are non-stationary at the level and are stationary at their first difference. This implies that all variables are integrated into order 1. This validates using cointegration techniques such as FMOLS in our Stone-Geary model estimation, since the variables are non-stationary but likely to have a long-run equilibrium relationship.

Appendix D: Log-Log Specification Results

We estimate the log-log specification of the urban water demand using the autoregressive distributed lag (ADL) method. The ADL in the case of our variables can be expressed as follows:

$$wd_t = b_0 + \sum_{i=1}^{p} b_{1i} wd_{t-i} + \sum_{i=0}^{q1} b_{2i} y_{t-i} + \sum_{i=0}^{q2} b_{2i} wp_{t-i} + \sum_{i=0}^{q3} b_{2i} popu_{t-i} + \sum_{i=0}^{q4} b_{2i} temp_{t-i} + b_3 T + e_t \quad \text{(D1)}$$

Where, *wd* is the natural logarithmic expression of demand for water in the urban areas; y and wp are the natural logarithmic expressions of the household income and water price, both in real prices of 2010; *popu* and *temp* are the natural logarithmic expressions of the urban population and temperature; e is the error term; *p* and *q1* - *q4* are the maximum lar orders for the dependent variable and explanatory variables, respectively; T stands for liner time trend; Subscript t indicates time, that is, year.

To provide robust estimations, we employ Autometrics with super-saturation. *Autometrics* is a machine-learning modeling algorithm method (Castle, Doornik, and Hendry 2021; Doornik 2009; Doornik and Hendry 2013, 2024). Also, Doornik and Hendry (2024) provide a practical reference for running *Autometrics* in the current OxMetrics environment.

Super-saturation, in our case, covers impulse indicator saturation (IIS), change in impulse-indicator saturation (DIIS), step-indicator saturation (SIS), and trend-indicator saturation (TIS). All these indicators are included in a model simultaneously to capture any types of structural breaks, outliers, and other anomalies that might occur in the data.

We consider (9) without the temperature variable as the initial general unrestricted model (GUM) and estimate it with one lag order of the variables, as we have only 30 annual observations. The reason we dropped the temperature is that although we estimated various options of *Autometrics* with different target sizes for the GUM and final specification selections by retaining this variable, none of these estimation results were sensible (these intermediate results are available from the authors upon request).

Table A2 presents the selected specification by *Autometrics*, which can be considered as a congruent GUM ADL specification.

Table A2. Final ADL specification and test results.

Panel A: Initial GUM ADL specification		
Regressor	**Coefficient**	**Standard error**
wd_{t-1}	0.908***	0.012
b_0	-1.541***	0.955
y_t	-0.160***	0.047
y_{t-1}	0.267***	0.045
w_{pt}	-0.008***	0.002
$popu_{t-1}$	0.052	0.032
Selected indicators by Autometrics		
DBL_2005	0.033***	0.007
DBL _2006	0.057***	0.010
DBL _2007	0.081***	0.011
DBL _2008	0.085***	0.010
DBL _2009	0.074***	0.008
DP_2012	0.478***	0.009
DSH_2000	0.077***	0.008
Panel B: Residual diagnostics and misspecification test results		
Statistic	**Sample value**	**Probability**
F_{AR} (2,15)	1.900	0.184
F_{ARCH} (1,28)	1.538	0.225
F_{HETR} (17,11)	1.962	0.129
$\chi^2_{Wald}(4)$	4.063	0.131
F_{Reset} (2,15)	4.046	0.039**
Panel C: Long-run coefficients and cointegration test results		
Regressor	**Coefficient**	**Standard error**
C	-16.687***	3.305
y_t	1.158***	0.234
wp_t	-0.092***	0.027
$popu_t$	0.559	0.336
Statistic	**Sample value**	**Probability**
$\chi^2_{Wald}(4)$	674.219	0.000
t_{ECM}	-7.876	0.000

Notes: wdu_t is the dependent variable in the estimations; F_{AR}, F_{ARCH}, F_{HETR}, and F_{Reset} denote F statistics to test the null hypotheses of no autocorrelation, no autoregressive conditioned heteroscedasticity, no heteroscedasticity in the residuals, and no functional form misspecification, respectively; χ^2_N and χ^2_{Wald} indicate the Chi-squared statistics to test the null hypotheses of normal distribution of the residuals and joint significance of the long-run coefficients from the Wald test, respectively; t_{ECM} is the t-statistic from the ECM-based cointegration test; *, **, *** indicate statistical significance at the 1%, 5%, and 10% levels, respectively. indicates the Chi-squared statistics to test the null hypothesis of normal distribution of the residuals; *** indicates statistical significance at the 1% level. Estimation period: 1994-2023.

Source: Authors' construction.

Panel A of Table A2 shows that the congruent GUM specification is reduced to a final specification, where only the current terms of income and price, along with the autoregressive term, the first lag of income and urban population, and dummy indicators, are retained. Note that although we included the urban population as part of the final specification, its statistical significance is 0.130, which is slightly insignificant, not strongly. The other retained variables are hugely statistically significant.

Panel B reports that the final conditional ADL specification successfully passes all the residual diagnostics. It also passes the misspecification tests at the 1% significance level only. This suggests that the relationship between urban water demand and its determinants is non-linear, which opens an avenue for future research to investigate.

The cointegration test results presented in Panel C indicate that the urban water demand establishes a long-run relationship with water price, income, and urban population, as the t-statistic from the ECM-based cointegration test is greater than the critical value at the 1% significance level (see Ericsson and MacKinnon 2002). Individually, the estimated long-run coefficients are statistically significant at different significance levels. Jointly, they are statistically significant at the high level, as the Wald test indicates.

Discussion of the Results

Short Run Dynamics

Panel A of Table 4 shows that urban water demand in Saudi Arabia is highly persistent – the coefficient on lagged consumption is 0.91, implying that more than 90%of a shock to the water demand carries over to the following year. Current household income has a small negative short-run effect, -0.16, but last year’s income enters with a positive coefficient of 0.27. The sign reversal is typical in ADL specifications that capture adjustment costs: when income rises, households initially conserve because of tariff blocks and billing delays, then expand demand the following year once higher discretionary spending filters through. The net effect of income becomes positive, which is consistent with demand theories. The contemporaneous own price elasticity is -0.008, statistically significant yet economically small. It suggests that annual tariff changes have little immediate effect. The urban population with a lag increases urban water demand by the elasticity of 0.052, and it is slightly insignificant at conventional levels. This might be explained by the fact that demographic dynamics in the year-to-year framework are usually not variable enough, suggesting that considering five-year dynamics may make the population effect statistically significant.

Long Run Relationship

The ECM t-statistic and the Wald test strongly reject the null hypothesis of no cointegration, confirming a stable long-run relationship among urban water demand, income, price, and urban population. The error correction-based long run solution yields an income elasticity of 1.16, suggesting that a 10% rise in real household income eventually lifts urban water use by about 16%. The long-run own price elasticity of urban water demand is only -0.09, highlighting the limited deterrent effect of existing tariffs. In other words, it shows that a 100% rise in the water price (i.e., doubling of the tariffs) leads to only 9% decline in demand for water in the urban areas. This suggests that tariffs may still not be a significant cost component for households, despite their substantial increase since 2016. This might also suggest that there is room for raising tariffs to make water price cost-covering, as Saudi Arabia heavily subsidizes it, and if the present level of water use in the urban areas is not efficient and leads to waste. Population is important, with an elasticity of 0.56, which is less than unity.

Appendix E

Table A3. Capital and operating and maintenance costs of desalination plants in Saudi Arabia.

Plant name and location	Operation year	Size (MLD)	Capital cost (million $/MLD)		O&M cost ($/m3)	
			Total	Per MLD	Total	Per MLD
RO						
Al Jubail (4)	2014	100	169.00	1.69	19.90	0.20
Yanbu,	2016	30	67.70	2.26	9.82	0.33
KAUST	2017	40	82.00	2.05	14.20	0.36
Shuaibah (3) Extension	2011	150	273.80	1.83	30.00	0.20
Shuqaiq	2010	212	285.00	1.34	34.40	0.16
Jeddah 3	2013	240	322.60	1.34	36.80	0.15
MED						
Rabigh, KSA	2005	25	58.40	2.34	2.34	0.09
Marafiq Jubail IWPP	2009	800	1115.00	1.39	44.80	0.06
MFD						
Yanbu Ph3,	2016	550	1000.00	1.82	52.20	0.09
Shuaibah 3 IWPP	2010	880	1640.00	1.86	68.90	0.08

Source: World Bank Group (2019).

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About the Project

This study was conducted as part of the project titled “Water-Energy Nexus in Saudi Arabia.” The rising demand for domestic water consumption in Saudi Arabia is driven by low tariffs, high system losses, and extreme climatic conditions. The Kingdom’s water sector poses significant fiscal and energy burdens. These factors contribute to unsustainably high per capita water consumption and strain the national economy, largely due to the country’s heavy reliance on energy-intensive seawater desalination. In response, the government has launched a national investment program to reduce water system losses. This project evaluates the potential impact of that program on water demand, desalination costs, energy use, and fiscal liabilities.

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