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Household economic burden from seawater intrusion in coastal urban areas

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ABSTRACT

This study quantifies the direct and indirect household economic burden of saltwater intrusion in Beirut, Lebanon, which experiences chronic water shortages. Incurred burdens include water purchase, reduction in the lifespan of household appliances, and building-level water treatment systems. Due to salinity, median household expenditure on water exceeds 6.5% of income, significantly higher than worldwide averages. A majority of affected respondents are willing to pay for mitigation measures to reduce salinity. The reported willingness to pay increased with education, income, salinity, and household expenditure on alternative water sources.

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KEYWORDS

Seawater intrusion; household economic burden; willingness to pay; Beirut

Introduction

Population growth, rapid urbanization, and economic development have indirectly led to the improper management of water resources in many developing countries. Urban residents of coastal areas, where population hubs tend to concentrate, particularly in arid and semi-arid regions, are suffering from water shortages or are expected to suffer from them in the near future (McDonald et al., 2011; Vörösmarty, Green, Salisbury, & Lammers, 2000). In these regions, future climate change is expected to exacerbate an already difficult situation because of potential temperature rise, changes in precipitation amount or patterns, sea-level rise, and slower replenishment of renewable water resources (Dawadi & Ahmad, 2012; De Montety et al., 2008; Masciopinto, 2013; Nicholls et al., 2007; Pachauri et al., 2014), all of which are strong promoters of increased net water demand. In the face of these challenges, many coastal communities have resorted to unsustainable exploitation of groundwater resources in an effort to compensate for increased demand and dwindling surface water supplies (Ragan, Young, & Makela, 2000). Over-pumping from coastal aquifers invariably accelerates the process of seawater intrusion, increasing salinity, degrading groundwater quality, and reducing aquifer storage (Lashkaripour & Zivdar, 2005). In the absence of alternative water sources, many coastal communities have little choice but to continue to use salinized groundwater.

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 Supplemental data for this article can be accessed [here](#).

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Higher salinity affects the water's taste and can be associated with a series of socio-economic burdens, widely reported in the agriculture sector (Ayers & Westcot, 1985; Blaylock, 1994; Maas & Grattan, 1999; Maas & Hoffman, 1977; Munns & Gilliam, 2015) but seldom quantified at the household level. Deterioration of household appliances, leading to higher repair or replacement costs, installation of desalination units and the purchase of freshwater are additional expenses that urban users are incurring with increased salinity of groundwater aquifers (Michelsen, McGuckin, Sheng, Laceywell, & Creel, 2009; Ragan, Makela, & Young, 1993; Tihansky, 1974).

This study quantifies the economic burden associated with the use of brackish groundwater at the household level in Beirut, Lebanon, including adaptation-mitigation measures that are practised indirectly. Willingness to participate and pay models were developed to determine the main drivers/hurdles influencing the adoption of these measures with a cost-benefit analysis comparing current salinity-induced expenditures with costs of treatment at the household. The article concludes with an assessment of adaptation-mitigation measures towards minimizing future impacts associated with increasing salinization of coastal aquifers.

Materials and methods

Pilot area

The pilot area (Beirut, Lebanon) is located along the Eastern Mediterranean (Figure 1), with fractured limestone and heterogeneous aquifers underlying a highly urbanized metropolitan area facing water shortages and a high dependence on groundwater. The accessibility to and the high permeability of the formation with fractured limestone increases its vulnerability to seawater intrusion resulting largely from groundwater over-pumping. Population growth, rapid urbanization, and poor water management have all led to water shortages and poor water quality. Climate change predictions suggest that the study area will experience diminished flows during the spring season due to higher temperatures and early snowmelt, resulting in drier soils during the dry Mediterranean summers, with an estimated increase in the soil moisture deficit of 13–25 mm/y, depending on climatic scenarios (Bou-Zeid & El-Fadel, 2002). Moreover, the uneven temporal distribution of precipitation in a Mediterranean climate, where 90% of the total precipitation volume falls between November and April (Ministry of Environment, 2011), exacerbates water stress and complicates the management of the water sector. Currently, groundwater resources are facing significant stress due to over-exploitation (Ministry of Energy and Water, 2010; Ministry of Environment, 2011), with coastal aquifers being most vulnerable due to 150 MCM/y pumped beyond the safe yield (Ministry of Energy and Water and United Nations Development Programme 2014). In Beirut, annual water demand is estimated to be nearly twice the distribution capacity (Ministry of Environment, 2011), compelling severe rationing, with many areas receiving water only for 7–10 hours every other day (El-Fadel, Maroun, Semerjian, & Harajli, 2003). The imbalance between supply and demand forces the local population to purchase water and/or install unlicensed private wells, leading to the overexploitation of groundwater resources, which in turn promotes saltwater intrusion into coastal aquifers.

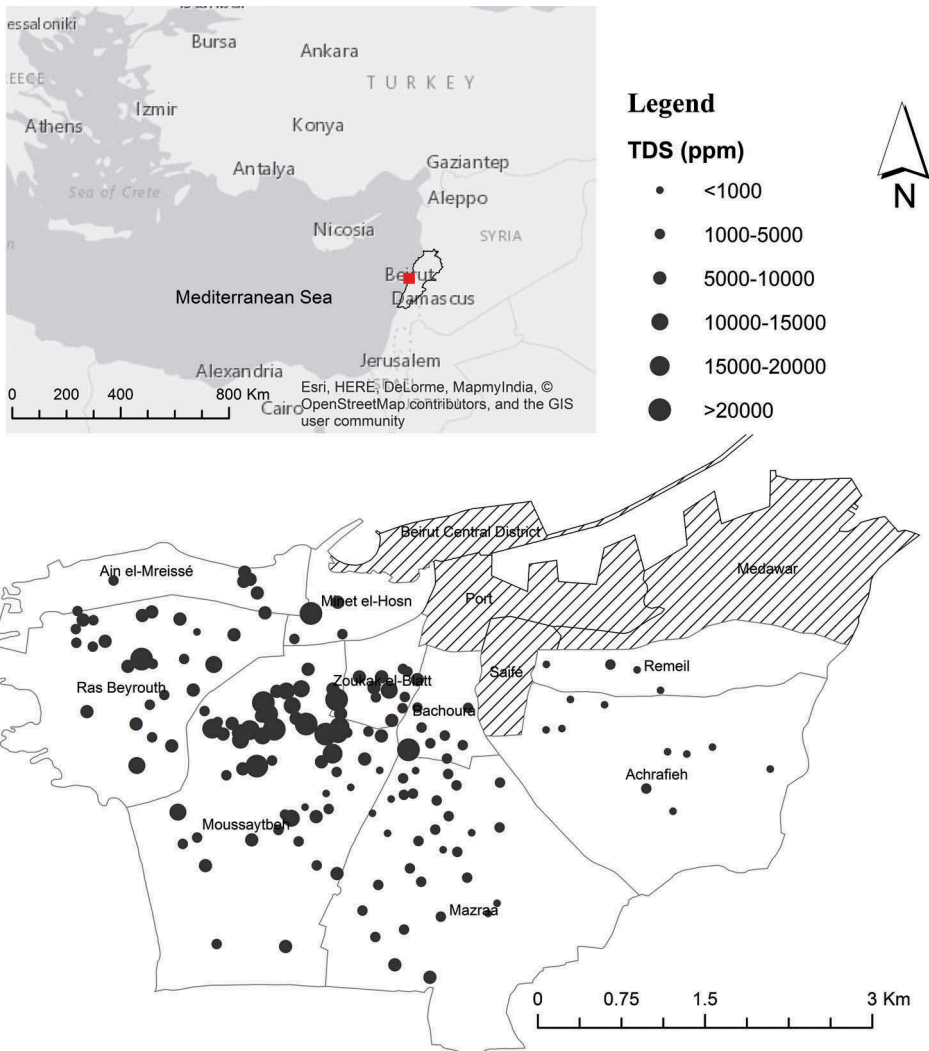


Figure 1. Pilot area, including locations of household surveys and groundwater sampling. (Colours represent well total dissolved solids [TDS] in ppm, measured in October 2013; shaded areas are largely non-residential districts.)

Field survey

The total population of administrative Beirut is estimated at 361,366 residents, for a population density of 18,068/km² (Central Administration of Statistics, 2008). This study targeted only those with access to a groundwater well. While accurate estimates of the number of wells are not directly available, as most are unlicensed, a World Bank study found that a maximum of 20,000 illegal wells were reportedly operating in the Greater Beirut Area, which has a population of 2 million (Kfoury, 2014). Assuming that the distribution of wells is proportional to population, then the target population was around 3600 buildings with illegal wells. Three hundred structured questionnaires were administered to residents who had access to a private well (refer to the online

supplementary material at <https://doi.org/10.1080/02508060.2017.1416441> for sample size determination). Stratified sampling was adopted to proportionally sample from each zone based on population. Random samples of digitized and georeferenced residential buildings were then selected from each zone. Commercial buildings were excluded. Household units, within a chosen residential building, were then randomly selected. In the event of a non-response, rejection, or inaccessibility, an adjacent building or household unit was selected. The questionnaire included targeted questions, and data were collected through face-to-face interviews. The survey targeted households using reverse osmosis or similar desalination units, as well as households that did not have access to any desalination technology but still had a well. The main topics of the questionnaire were the types of water sources used, the socio-economic status of the residents, the proliferation of desalination units, the direct and indirect water costs incurred by households, and the willingness of respondents to participate in programmes to control salinity (refer to online supplementary material). Survey rejection rate was 41%. In total 177 questionnaires out of the original 300 were completed. The corresponding spatial distribution of the survey across the city is shown in [Figure 1](#). Note that the eastern part of the city has better access to networked water and therefore has a lower density of private wells.

Groundwater characterization

Three rounds of groundwater sampling were conducted (June 2013, October 2013 and April 2014). Samples were also collected from the tap water of each surveyed unit. All samples were analyzed at the Environmental Engineering Research Center of the American University of Beirut for physical (temperature, conductivity, and total dissolved solids [TDS]), chemical (pH, dissolved oxygen, alkalinity, calcium and total hardness, nitrates, sulphides, chlorides), and microbiological parameters (total and faecal coliform) following standard analytical methods (refer to online supplementary material). Since the main focus of this study is the quantification of direct and indirect damages resulting from increased salinity, TDS was used as a surrogate indicator.

Economic assessment

Direct costs incurred by households

A typical household in the pilot area pays a fixed subscription fee for network water delivery in addition to the costs of installing a water treatment unit and/or purchasing bottled water and/or water cisterns. To estimate the true annual water cost incurred at the household level, respondents were asked for a breakdown of their water-related expenditures. Note that the capital costs for installing treatment units were amortized based on Equations (1) and (2).

$$\text{annual cost} = \text{total cost} \times \alpha \quad (1)$$

$$\alpha = r(1 + r)^Y / [(1 + r)^Y - 1] \quad (2)$$

where r is the discount rate (set at 5%) and Y is the lifespan of the treatment unit, assumed to be 10 years including its components.

Indirect costs incurred by households

Residents incur additional indirect costs resulting from the use of brackish water with high TDS levels, reported to reduce the lifespan of water-related appliances. The loss in the lifespan of appliances as a function of TDS levels was first developed by Tihansky (1974) and then adapted by Ragan et al. (2000) and Howitt, Medellín-Azuara, and MacEwan (2009):

$$Y_{Faucets}^{TDS} = 10.2 - 7 \times 10^{-4} TDS \quad (3)$$

$$Y_{Washer}^{TDS} = 5 + e^{1.8-7.9 \times 10^{-4} TDS} \quad (4)$$

$$Y_{Water Heater}^{TDS} = 5 + e^{2.4-1.4 \times 10^{-3} TDS} \quad (5)$$

where TDS represents the measured household TDS level and Y_i^{TDS} is the expected lifespan of appliance i at the measured TDS level. Attempts were made to build local relations between TDS levels and lost years of appliances using the data collected from the household survey, but they were clouded by inability to distinguish between TDS-based lost years and losses related to the daily power cuts in the pilot area. Given that the vast majority of appliances in the pilot area are imported, it is reasonable to rely on reported TDS–lifespan relations, as the latter are largely a function of the manufacturer. Moreover, the median lifespan predicted by the models under normal TDS of 500 ppm were checked with local vendors to ensure consistency. Thus, two scenarios were tested. The first assumed operation under normal salinity (TDS = 500 ppm), while the other predicted the lifespan at the levels of TDS measured at each household. The annual indirect cost incurred for each appliance was then calculated by amortizing the lost service years. The cost of appliances and fixtures was regionalized based on the local market value. A range of costs were applied, including the reported median, lower 25%, and upper 75% (Table 1). The annual damage is expressed as (Michelsen et al., 2009):

$$D_h = \sum_{i=1}^N P_i \times C_i (\alpha_i^{TDS} - \alpha_i^{500}) \quad (6)$$

Table 1. Range of appliance and fixture costs in the study area.

Appliances/fixtures	Costs (USD)			Expected lifespan @ TDS = 500 ppm (years)
	Median (50%)	Lower quartile (25%)	Upper quartile (75%)	
Clothes washer	500.0	450.0	654.0	9
Electric water heater	200.0	120.0	250.0	10
Kitchen faucet	50.0	21.0	100.0	10
Toilet faucet	50.0	26.0	100.0	10
Douche faucet	50.0	33.0	100.0	10
Pipe faucet	10.0	8.0	20.0	10
Showerhead	20.0	13.5	40.0	10

Where D_h is the annual damage, P_i is the percentage of households with appliance i , C_i is the cost of appliance i (median, lower 25% and upper 75%), N is the number of appliances and fixtures in a household, α_i^{TDS} is the amortization rate for appliance i operating under a given TDS, and α_i^{500} is the amortization rate for appliance i if operated under a TDS of 500 ppm.

The appliances and fixtures considered include water heaters, clothes washers, faucets in the kitchen, toilet and shower faucets, as well as showerheads and shower pipes. Dishwashers are not common in the pilot area.

People’s willingness to participate and willingness to pay

Logistic regression models (Equations (7) and (8)) were developed to predict people’s willingness to participate in a mitigation programme to reduce salinity at their households. The use of logistic regression in contingent valuation methods to quantify the willingness to participate in improving water services and quality is well established. The model estimates the probability of participation in a programme based on a set of independent variables. Moreover, the logarithm of the odds ratio for participation is expressed in Equation (9). Model results have been reported to be biased due to the tendency of respondents to please the interviewers (Alberini, 1995; Whittington et al., 1987). But properly administering the survey can reduce these biases (Briscoe, De Castro, Griffin, North, & Olsen, 1990).

A model was first developed to estimate respondents’ conviction that mitigation measures are needed in Beirut to reduce salinity. Two separate models were then developed to explore the main factors in respondents’ willingness to participate in a government-run programme versus one based at the household level. Independent predictor variables included building characterization, water sources, and demographics (Table 2). Round-two well water TDS values were used as they represent the end of the dry season.

$$\text{logit}(E|Y_i|X_i) = \text{logit}(p_i) = \beta X_i \tag{7}$$

$$p_i = \frac{e^{\beta X_i}}{e^{\beta X_i} + 1} \tag{8}$$

$$\log\left(\frac{P(\text{Participate})}{1 - P(\text{Participate})}\right) = \beta X \tag{9}$$

where Y_i is a binary variable that indicates whether a respondent is willing to participate (1) or not (0) in the mitigation programme, p_i is the probability of respondent i being satisfied with water quality, X is a matrix of predictors, and β is a vector of model coefficients. The change in the odds of participation is given by:

$$\text{odds}_1 = e^{\beta X_1} \tag{10a}$$

$$\text{odds}_2 = e^{\beta X_2} \tag{10b}$$

$$\text{odds}_2 / \text{odds}_1 = e^{\beta(X_2 - X_1)} \tag{10c}$$

Table 2. Predictor variables considered in the willingness to participate and pay analysis.

Category	Question	Variable type
Building characteristics	Number of stories	Continuous
	Number of apartments in the building	Continuous
	Age of the building	Continuous
	Time that the respondent has been residing in the building	Continuous
	Time since last major rehabilitation work, including changes to water pipes	Continuous
Water sources	Type of water sources used (public, bottled water, water tankers, wells)	Categorical
	Hours of public water delivery per week	Continuous
	Amount paid per year for the water establishment	Continuous
	Satisfaction with public water quality in summer	Binary
	Satisfaction with public water quality in winter	Binary
	Is water treated at the apartment level?	Binary
	Do you purchase bottled water?	Binary
	How much do you pay on average for bottled water per week?	Continuous
	Do you purchase water through water tankers?	Binary
	How much do you pay on average for bottled water per month?	Continuous
	Type of the water treatment in the apartment	Categorical
Since when have you started to feel that your water has become salty	Continuous	
Demographics	Number of residents in the apartment	Continuous
	Gender of respondent	Binary
	Highest educational level of the respondent	Categorical
	Apartment ownership	Binary (own/rent)
	Average monthly household expenditure	Continuous
	Average monthly household income	Continuous
	Number of cars in the household	Continuous
	Presence of skin and/or hair problems in the household	Binary

If a model coefficient is positive, this implies that an increase in the predictor variable will increase the odds of participation. Furthermore, multiple linear regression models (Equation 11) were developed to quantify how much each household was willing to pay per month to implement a governmental mitigation plan as compared to a building-level programme. The willingness to pay (WTP) analysis excluded respondents that stated that they did not want to participate in the programme, irrespective of their stated reason. Thus, both protest and true zeros were censored. True zero censorship is a direct result of assuming a log-linear relationship between WTP and the salient independent predictors (Barton, 2002). Censoring true zeroes may result in biased estimates. In total, 125 non-zero surveys (out of the 177) were used in the WTP analysis.

$$\log(C_i) = \beta_0 + \beta_1 x_1 + \dots + \beta_n x_n + \varepsilon \quad (11)$$

where C_i is the monthly cost household i is willing to pay, β_0 is the estimated intercept, $\beta_{1..n}$ are the estimated slopes, and x_1 to x_n are independent predictor variables (Table 2); ε is the residual error, assumed to follow a normal distribution norm $(0, \sigma^2)$. All analysis was conducted using the statistical software R (R Core Team, 2015).

Results and discussion

Existing conditions

Of the survey respondents, 77% were connected to the public water network and pay a flat rate based on the size of the apartment. While surveyed households reported paying

USD 160–200 per year, public water rates in the study area are based not on metering but on the area of the apartment. Large apartments can pay nearly USD 500 per year. Water delivery through the network occurs every other day, with water being supplied for seven hours on average. All respondents were using groundwater sources through private unlicensed wells at the building. The cost of tapping the groundwater was limited to electricity costs associated with pumping, as well as replacing the water pump on average every six years. The extracted groundwater was mostly used for household cleaning and bathing, with 6% of respondents indicating its use in cooking and only 2% drinking it. The majority of respondents purchase bottled water for drinking, cooking and bathing because of a prevailing poor perception of well water quality. Annual expenditure for bottled water ranged between USD 260 and USD 824 per year. A third of the respondents (32%) had installed treatment units at the building level, but most systems do not appear suitable for high salinity. Only 5% had installed brackish water reverse osmosis units, at a reported cost ranging between USD 204 and USD 1022 per household per year. These costs include regular operation and maintenance as well as amortization over 10 years. Also, 3.5% of respondents purchased water from unregulated water tankers to blend with the high-salinity water or to supplement the existing network supply. The cost of water delivered by tankers ranged between USD 3.75 and USD 7.5 USD per cubic metre, depending on the volume purchased and the quality (Constantine, Massoud, Alameddine, & El-Fadel, 2017).

The groundwater quality varied significantly across districts and over time (Figures 1 and 2). Samples collected over the three sampling rounds consistently exceeded the 500 ppm TDS level for domestic water, with two wells reaching >30,000 ppm. The tap water exhibited marginally better quality than samples collected directly from the well because tap water is often a blend of government and well water. Blending well water with network water and/or water delivered through tankers seemed to be a common adaptation measure practised in the study area. Many of the water tankers supply groundwater from aquifers outside the city. The median drop in TDS levels resulting from blending ranged from less than 2% to 18% across the three sampling rounds. The variations in the physical characteristics of the well and tap water across the three sampling rounds are summarized in Table 3.

Economic burden

The direct costs that a household pays include the annual subscription fee for the public water, the cost of bottled water, and the costs associated with investing in treatment units. The median direct cost was estimated at USD 624/y, with the largest contribution going towards the purchase of bottled water (Figure 3). On the other hand, salinity-induced indirect costs varied from less than USD 3 to USD 180 per year per apartment, depending on salinity and on appliances/fixtures installed. Figure 4 shows the indirect costs incurred as a function of salinity, while accounting for variations in the initial costs of appliances and fixtures. At relatively low TDS levels (2000 ppm), indirect costs per apartment ranged were USD 26–44/y, consistent with estimates reported by Ragan et al. (2000) for Colorado residents. At 10,000 ppm, indirect costs were USD 55–100 per apartment per year. At TDS levels between 20,000 and 30,000 ppm, indirect annual costs ranged between USD 60 and USD 180 per apartment.

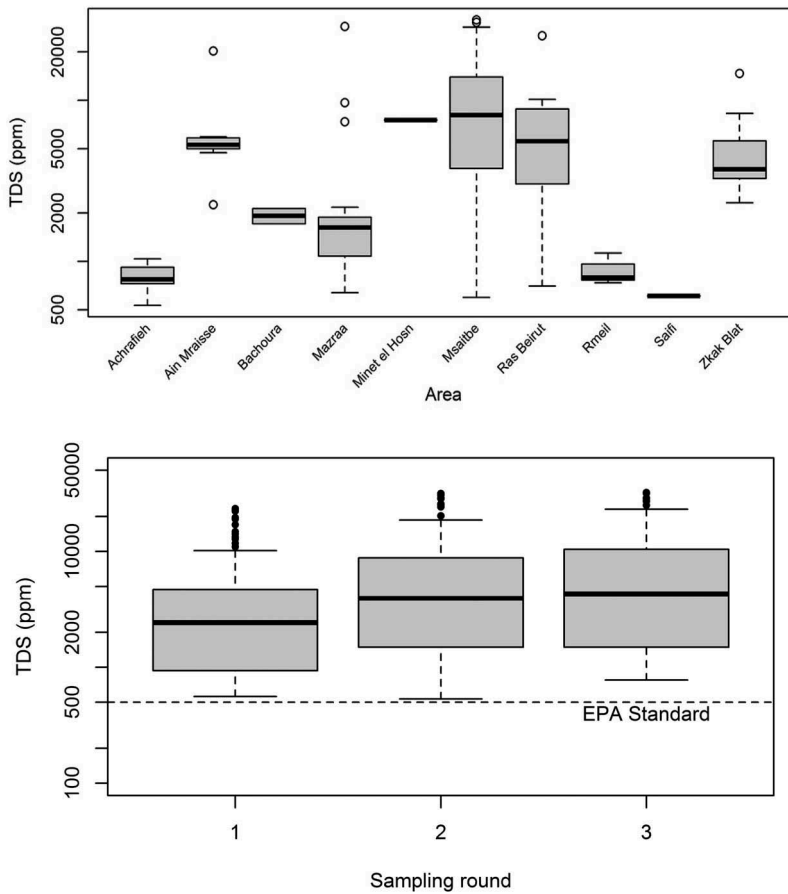


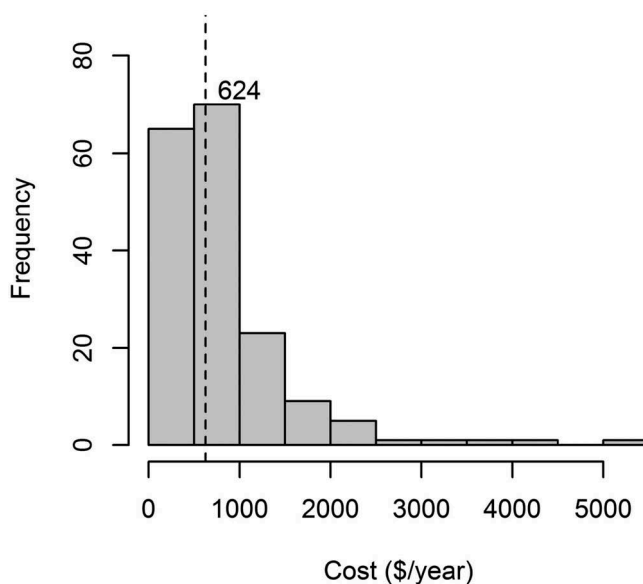
Figure 2. Spatio-temporal variability in well total dissolved solids (TDS) across the three sampling rounds.

Overall, the median annual total water cost incurred by the surveyed residents in the pilot area was around USD 850 (Figure 5), or about 6.5% of a median household's income. More than 85% of surveyed households pay more than twice what a typical US household pays for water (USD 300/y), although per capita consumption rates in Beirut (180–200 L/day – El-Fadel et al., 2003) are only about half of those reported for the US (378 L/day) and the median household income is a third of that reported for the US (Maupin et al., 2014; Phelps & Crabtree, 2013). Similar patterns emerge when comparing incurred costs with those reported across Europe (Germany, USD 91; France, USD 103; Austria, USD 74 per household per year – Biswas & Kirchherr, 2012; German Association of Energy and Water Industries, 2010), where per capita consumption is more comparable to rates within the study region.

Table 3. Summary of well (W) and tap (T) water quality variations for the three sampling rounds.

Sampling round	Parameter (ppm)	Laboratory analysis results			EPA standard ^a	Percentage over EPA standard
		Min.	Median	Max.		
June 2013	TDS: W	560	2,187	23,414	500	100
	T	101	1,315	22,224		87
	Cl: W	100	725	13,575	250	72.2
	T	5	450	13,080		67
	Ca hardness: W	28	560	3,400	-	
	T	12	450	3,400		
	Mg hardness: W	50	585	5,340	-	
	T	30	415	5,340		
	Total hardness: W	309	1,130	8,740	-	
	T	87	870	8,740		
October 2013	TDS: W	533	3,782	31,553	500	100
	T	102	2,009	28,704		92
	Cl: W	100	2,022.5	17,670	250	86
	T	20	1,235	17,330		81
	Ca hardness: W	80	520	3,700	-	
	T	16	400.5	3,550		
	Mg hardness: W	61	630	8,620	-	
	T	2	500	4,940		
	Total hardness: W	262	1,160	12,320	-	
	T	54	940	8,010		
April 2014	TDS: W	775	4,008.5	32,220	500	100
	T	422	2,351	33,385		99
	Cl: W	35.4	2,285	19,030	250	73
	T	7.8	1,212.5	19,750		70
	Ca hardness: W	36	459	2,810	-	
	T	20	326.5	1,740		
	Mg hardness: W	49	675	7,750	-	
	T	56	493	7,700		
	Total hardness: W	131	1,260	10,200	-	
	T	90	884	9,440		

^a United States Environmental Protection Agency (2014)

**Figure 3.** Distribution of direct costs incurred by households.

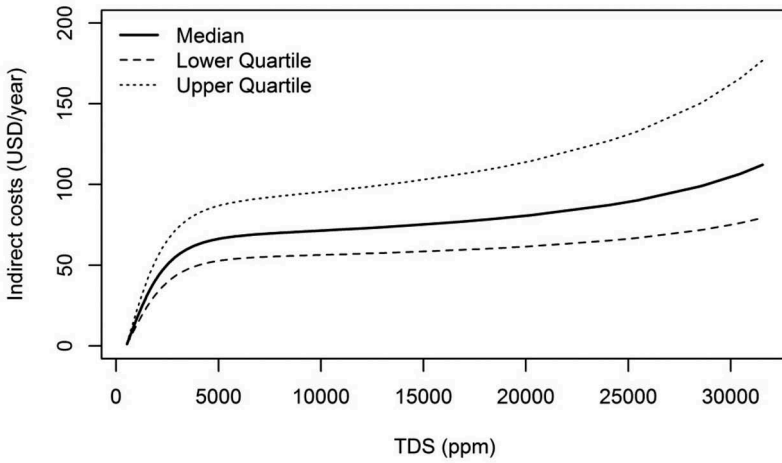


Figure 4. Estimated indirect costs as a function of salinity and appliances at the household level. (Variability in initial costs of appliances and fixtures is represented by the interquartile range at each salinity level.)

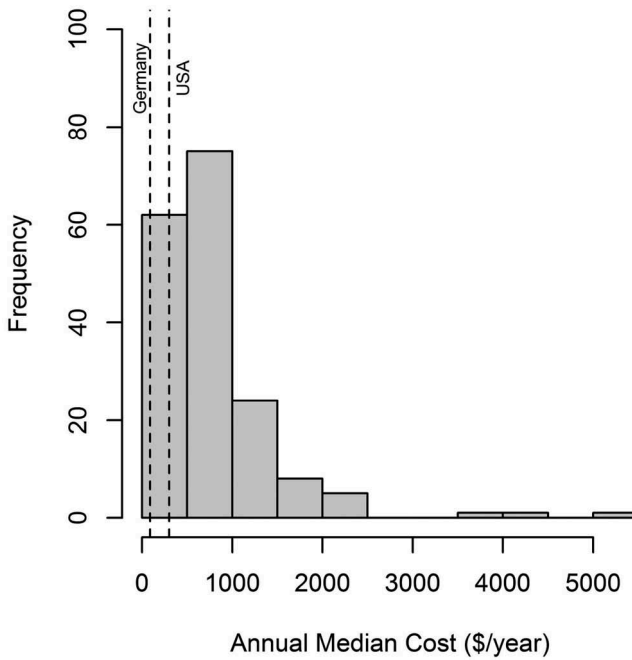


Figure 5. Distribution of total costs for water among surveyed residents.

Willingness to participate

Overall, 60% of respondents believed that there is a pressing need to enforce mitigation measures to reduce salinity. Less than 30% of respondents stated that they were not willing to contribute any financial resources towards a government-run plan to

safeguard the groundwater from saltwater intrusion and reduce salinity at the household level. The questionnaire probed for the reasons why a respondent opted not to participate in such a plan. Of those who refused, 7.5% stated that they could not afford the extra expense, and 63.5% that they do not trust the government. None stated that they did not believe in assigning a monetary value to water quality or did not believe that groundwater should be protected, and the remaining 28.8% did not specify the reason for their objection or chose a reason other than those specified in the questionnaire. When asked about their willingness to participate in a building-level plan to mitigate and/or reduce salinity in their wells, less than 30% of interviewees responded negatively. Of those, 70% stated that they see lack of cooperation within their building as a major impediment to implementation; 15% stated that they could not afford additional fees. Protest zeros (the equivalent of 18% of total respondents) were calculated as those who stated that they did not trust the government. True zeros were those who responded that they were unable to pay. Distinguishing between protest zeros and true zeros is an important step in the contingent valuation method (Meyerhoff & Liebe, 2006).

The logistic model results (Equation (12) and Table 4) indicate that the TDS levels of the well, education level of the primary household member, and the presence of a water treatment unit (reverse osmosis, water softener, UV, or chemical dosing) in the household were statistically significant predictors of people's belief in the need to endorse future mitigation/adaptation measures to reduce water salinity at the household level. Households with high TDS wells were more willing to adopt salinity-reducing mitigation/adaptation measures – an increase in TDS concentration from 500 to 5000 ppm increased the odds of accepting mitigation measures 3.8 times. Moreover, respondents with a higher education level were more receptive to mitigation measures than those with only primary education. The odds of acceptance were 2.6 times greater for households with a university degree as compared to those with only primary education. Households without a water treatment unit were more supportive of mitigation/adaptation measures. Their odds of participation were on average 3.8 times higher than their counterparts with a water treatment unit, which is expected, since the latter often have a false sense of security about their water supply. Note that income was as significant as education but was dropped from the final model due to its high correlation with education. The multivariate model accounted for 0.4 (McFadden's pseudo- R^2) and 0.6 (Cragg and Uhler's pseudo- R^2) of the total variability of the outcome, depending on the pseudo- R^2 equation used (Long, 1997).

Table 4. Model coefficients for the logistic model quantifying willingness to participate in measures to reduce salinity.

Coefficient	Estimate	Std. error	Z	Pr(> Z)
(Intercept)	-1.246	0.407	-3.064	0.00218
TDS/1000 (ppm)	0.298	0.067	4.418	9.94×10^{-6}
Education: university level	0.944	0.413	2.288	0.02214
Presence of a treatment unit	-1.323	0.583	-2.271	0.02317

$$\text{Logit}(p_i) = -1.24 + 0.298 \times \frac{\text{TDS}_i}{1000} + 0.944 \times \text{University EDU} - 1.323 \times \text{Treatment} \quad (12)$$

When asked about their willingness to participate in a government programme to reduce groundwater salinity, the probability of acceptance for a typical household with a TDS level of 5000 ppm was on average 70%. The probability of endorsing such a programme increased with well TDS levels. Households with a TDS of 5000 ppm were 1.3 times more likely to participate in the programme than households with access to a less affected well (500 ppm). The multivariate model (Equation (13) and Table 5) accounted for 0.2 (McFadden's pseudo- R^2) and 0.3 (Cragg and Uhler's pseudo- R^2) of the total variability of the outcome, depending on the pseudo- R^2 equation used (Long, 1997).

$$\text{Logit}(p_i) = 0.536 + 0.054 \times \frac{\text{TDS}_i}{1000} \quad (13)$$

When the same respondents were asked about their willingness to participate in a similar programme executed at the building level, the probability of acceptance for a university graduate who pays USD 10/month for bottled water and is using a well with a TDS level of 5000 ppm was around 80%. The probability increased with TDS levels – respondents exposed to well water with 5000 ppm were on average 1.4 times as likely to participate as those with access to a less affected well (500 ppm). The odds of participation were also three times higher on average for households with a university degree as compared to those with less education. Similarly, households that were incurring higher costs due to the purchase of bottled water were more accepting of the building-level programme. On average, as a household's expenditure on bottled water doubled from USD 10 to USD 20 per month, its odds of participation also doubled. The multivariate model (Equation (14) and Table 6) accounted for 0.3 (McFadden's pseudo- R^2) and 0.5 (Cragg and Uhler's pseudo- R^2) of the total variability of the outcome, depending on the pseudo- R^2 equation used (Long, 1997).

Table 5. Model coefficients for the logistic model quantifying willingness to participate in a government-initiated programme to mitigate saltwater intrusion.

Coefficient	Estimate	Std. error	Z	Pr(> Z)
(Intercept)	0.536	0.249	2.154	0.0312
TDS/1000 (ppm)	0.054	0.032	1.676	0.0937

Table 6. Model coefficients for the logistic model quantifying willingness to participate in a building-level programme to mitigate saltwater intrusion.

Coefficient	Estimate	Std. error	Z	Pr(> Z)
(Intercept)	-0.945	0.470	-2.011	0.04432
TDS/1000 (ppm)	0.065	0.036	1.834	0.06667
Education: university level	1.156	4.233×10^{-1}	2.731	0.00632
Monthly expenditure for bottled water (USD)	0.0792	0.0306	2.588	0.00966

$$\text{Logit}(p_i) = -0.945 + 0.065 \times \frac{\text{TDS}_i}{1000} + 1.156 \times \text{University EDU} + 0.079 \times \text{Monthly Cost BW} \quad (14)$$

Willingness to pay

The questionnaire elicited households' valuation of access to water with low salinity. Respondents were asked about their willingness to pay (WTP) per month for a programme to secure access to freshwater for both a government-run plan and a building-level plan. For a government plan, households with monthly income lower than USD 1500 were willing to pay on average USD 28/month, while households with monthly incomes over USD 1500 were willing to pay USD 51/month on average. TDS level was not a significant predictor of WTP in the case of the government plan. Note that while households with a higher income are willing to pay more, the relative expenditure of the two groups would be very similar (around 2% of their monthly income). For a building-level plan, well TDS level, educational level of the respondent, and the presence of a treatment unit were all statistically significant predictors (Table 7).

Holding all variables constant, people's WTP increased by 1.1% for every 10% increase in well TDS levels. Households with a higher education level were willing to pay on average 1.8 times more than respondents with only a basic education when both had access to similar-quality water and had no treatment units installed. Finally, households with existing treatment units were willing to pay 1.4 times more than households with no treatment. Thus, a typical household with access to a 5000 ppm TDS well, where its head has a university degree and has installed a treatment unit, is on average willing to pay around USD 100 per month. The rate drops to USD 30 per month for a household having access to a less affected well (TDS of 500 ppm), where the head has had only basic education, and there is no treatment unit installed.

While it is often expected that women are likely to pay more than men to improve water quality, given their higher involvement in household needs (Genius et al., 2008; Kayaga, Calvert, & Sansom, 2003), our results did not show a major role for gender in predicting WTP. The lack of a gender effect was also observed by Lee (2014) and Kaliba, Norman, and Chang (2003). Respondent's age, household tenure status, and family size were also not important predictors of stated WTP. The role of age in WTP appears to vary across studies. Kaliba et al. (2003) found that older respondents were less likely to support improved water utility services, while in the Mexican city of Parral, age had a nonlinear (quadratic) relation to respondents' WPT, with middle-aged respondents

Table 7. Model coefficients for willingness to pay for a salinity-mitigation programme executed at the building level.

Coefficient	Estimate	Std. error	<i>t</i>	Pr(> <i>t</i>)
(Intercept)	10.07	0.54	18.62	< 0.05
log(TDS)	0.11	0.06	1.66	0.01
Education: university level	0.57	0.15	3.89	< 0.05
Presence of a treatment unit	0.36	0.20	1.80	0.08

reporting the lowest rates (Vásquez, Mozumder, Hernández-Arce, & Berrens, 2009). Similar to our conclusions, Lee (2014) did not see a relation between WTP and age. Kaliba et al. (2003) reported that households with larger families were willing to pay more for improved water services. Yet the World Bank study did not find that family size was a significant factor in WTP for improved water services (World Bank Water Demand Research Team, 1993). Inconclusive results were also reported with regard to household ownership. Ownership status of the residential house was found to play an important role in predicting WTP of respondents in 11 major towns in Uganda (Kayaga et al., 2003). Similarly, Casey, Kahn, and Rivas (2006) reported a significant positive relation with home ownership when studying WTP in Manaus, Brazil. In contrast, home ownership in Mexico was not seen as a significant factor in predicting WTP for improved water quality (Vásquez et al., 2009).

Viability of reverse osmosis as a mitigation/adaptation measure

The economic viability of adopting reverse osmosis at the building level hinges on the degree that the aquifer is affected by saltwater intrusion as well as the size of the facility. The difference in expenditure between households with a reverse osmosis unit and those without it is limited to the net difference between the cost of installing and operating the unit and the savings a household is expected to accrue in terms of extending the life of appliances and fixtures. Thus, the breakeven point between the two options starts when well TDS exceeds 24,000 ppm, which renders the installation of reverse osmosis units at most buildings economically non-viable even when within respondents' WTP (USD 360–600/year) for a mitigation/adaptation programme to reduce salinity. This indicates that in addition to the indirect costs that people are incurring due to saltwater intrusion, other costs remain unquantified (damage to utensils, pumps, piping, health etc.) along with the discomfort associated with using brackish water for personal hygiene.

As salinity continues to increase, urban users will be pushed towards adopting building-level reverse osmosis units or incur higher indirect costs. If salinity increases from its current median of 4000 ppm to 9000 ppm (the current upper quartile), the median household total water bill is likely to increase by at least 28% (from USD 670 to USD 860 per year, assuming the household adopts reverse osmosis treatment with a cost of USD 260/year and saves USD 70 in indirect costs). If the current disengagement from regulating the installation and operation of building-level reverse osmosis continues, then over-exploitation of groundwater will worsen. The efficiency of single-stage brackish water reverse osmosis units is often between 50% and 65%; thus it is expected that pumping rates will double to overcome a given deficit in volume. The lack of regulations on proper brine discharge will also aggravate pollution, as brine will be either re-injected into the aquifer or discharged with municipal wastewater. The former will lead to increased salinization, while the latter may disrupt the operation of biological treatment units. It is thus imperative that the public sector intervene to alleviate the urban deficit and regulate groundwater pumping and the operation of building-level desalination units. Investing in large-scale desalination plants that can make use of seawater instead of brackish groundwater and adopt the more efficient

dual-cycle reverse osmosis system, thus preventing groundwater over-exploitation, could be one option.

Conclusion

The study highlighted the gravity of saltwater intrusion in a coastal urban area along the Eastern Mediterranean (Beirut, Lebanon) facing water shortage and unregulated groundwater exploitation typical of many developing coastal communities. Salinity as high as 30,000 ppm TDS was encountered, necessitating the control of unplanned pumping activities. In parallel, projections of climate change envisage rising net water demand (higher temperature coupled with population growth) and falling aquifer recharge (lower precipitation coupled with urbanization), which, along with the anticipated sea-level rise, will increase the rate of salinization. Naturally, overexploitation of vulnerable aquifers will persist without policies narrowing the gap between supply and demand.

On average, people reported a higher WTP for salinity-reducing mitigation measures at the building level as compared to a governmental city-wide plan. This is unsurprising given the lack of trust in governmental institutions due to inefficiencies and perceived corruption (Haddad, 2002). Higher education and income levels play a key role in promoting participation and/or WTP, consistent with earlier World Bank findings (World Bank Water Demand Research Team, 1993). Similar results were also reported by Kayaga et al. (2003), who examined WTP for improved water services in Uganda. Results from Greece, the Gaza Strip and South Korea also showed that income levels are significantly positively correlated with WTP (Al-Ghuraiz & Enshassi, 2005; Genius et al., 2008; Lee, 2014; Polyzou, Jones, Evangelinos, & Halvadakis, 2011). Yet, the manner in which education and income affect WTP is not universal. Vásquez et al. (2009) indicated that education played a marginal role in predicting WTP of respondents in the Mexican city of Parral, and other studies did not find that education is an important factor in predicting WTP for improved water services and quality (Kaliba et al., 2003; Lee, 2014; Polyzou et al., 2011). Similarly, even though it is generally expected that income is positively correlated with WTP, Casey et al. (2006) found that income did not have a significant role in Brazil, while Kaliba et al. (2003) found that in Tanzania, richer individuals were *less* likely to pay for water improvements. This may be because well-to-do Tanzanians tend to have access to additional sources of water.

In this study, the salinity of the well water was an important factor in people's WTP for improved water, with respondents exposed to higher TDS levels reporting higher WTP values. Similar to our findings, the World Bank study reported that respondents who perceived water quality to be poor showed a significantly higher WTP in comparison to those who perceived their water quality to be acceptable (World Bank Water Demand Research Team, 1993). In contrast, Raje, Dhobe, and Deshpande (2002) reported that water quality was only a marginally important factor in predicting WTP of respondents in Mumbai, India. The purchase of bottled water and/or investing in a water treatment unit, both of which are considered indicators for the use of alternative water sources, increased respondents' WTP, consistent with the hypothesis of Vásquez et al. (2009) for which no proof could be established when assessing WTP in Parral, Mexico.

Based on the survey results, an average household in Beirut pays over USD 850 per year for water (>6.5 % of median household income). Actual expenditure probably even higher, since health costs and pipe replacements were not accounted for in this study. Unfortunately, it is common for urban residents in developing countries to pay a significant percentage of their income for water (WaterAid, 2016). In the Mexican city of Parral, Vásquez et al. (2009) found that respondents were paying up to 7.6% of their reported household income, above and beyond their current water bill, in an effort to improve their water service. Casey et al. (2006) reported that based on a WTP study conducted in the city of Manaus, in Brazilian Amazonia, their respondents were willing to pay over USD 6 per month (~21% of a household's income) for improved water services. Combined, these results show that urban consumers in many developing countries are willing to pay higher rates if they are guaranteed improved water services and quality. While the Millennium Development Goals have made large strides towards ensuring access to safe drinking water through better infrastructure, guaranteeing access to sustainable sources remains weak. Consequently, the gains made in improving connectivity and upgrading water treatment plants are continuously eroding due to shortages and poor governance.

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References

- Alberini, A. (1995). Efficiency vs bias of willingness-to-pay estimates: Bivariate and interval-data models. *Journal of Environmental Economics and Management*, 29, 169–180. doi:10.1006/jeem.1995.1039
- Al-Ghuraiz, Y., & Enshassi, A. (2005). Ability and willingness to pay for water supply service in the Gaza Strip. *Building and Environment*, 40, 1093–1102. doi:10.1016/j.buildenv.2004.09.019
- Ayers, R. S., & Westcot, D. W. (1985). *Water quality for agriculture*. Rome, Italy: Food and Agriculture Organization of the United Nations.

- Barton, D. N. (2002). The transferability of benefit transfer: Contingent valuation of water quality improvements in Costa Rica. *Ecological Economics*, 42, 147–164. doi:10.1016/S0921-8009(02)00044-7
- Biswas, A., & Kirchherr, J. (2012). *Water prices in Europe need to rise substantially to encourage more sustainable water consumption*. London, UK: London School of Economics and Political Science.
- Blaylock, A. D. (1994). *Soil salinity, salt tolerance, and growth potential of horticultural and landscape plants*. Laramie, WY: University of Wyoming, Cooperative Extension Service, Department of Plant, Soil, and Insect Sciences, College of Agriculture.
- Bou-Zeid, E., & El-Fadel, M. (2002). Climate change and water resources in Lebanon and the Middle East. *Journal of Water Resources Planning and Management*, 128, 343–355. doi:10.1061/(ASCE)0733-9496(2002)128:5(343)
- Briscoe, J., De Castro, P. F., Griffin, C., North, J., & Olsen, O. (1990). Toward equitable and sustainable rural water supplies: A contingent valuation study in Brazil. *World Bank Economic Review*, 4, 115–134. doi:10.1093/wber/4.2.115
- Casey, J. F., Kahn, J. R., & Rivas, A. (2006). Willingness to pay for improved water service in Manaus, Amazonas, Brazil. *Ecological Economics*, 58, 365–372. doi:10.1016/j.ecolecon.2005.07.016
- Central Administration of Statistics. (2008). *Lebanon in Figures*. Beirut, Lebanon: Presidency of the Council of Ministers.
- Constantine, K., Massoud, M., Alameddine, I., & El-Fadel, M. (2017). The role of the water tankers market in water stressed semi-arid urban areas: Implications on water quality and economic burden. *Journal of Environmental Management*, 188, 85–94. doi:10.1016/j.jenvman.2016.11.065
- Dawadi, S., & Ahmad, S. (2012). Changing climatic conditions in the Colorado River Basin: Implications for water resources management. *Journal of Hydrology*, 430, 127–141. doi:10.1016/j.jhydrol.2012.02.010
- De Montety, V., Radakovitch, O., Vallet-Coulomb, C., Blavoux, B., Hermitte, D., & Valles, V. (2008). Origin of groundwater salinity and hydrogeochemical processes in a confined coastal aquifer: Case of the Rhône delta (Southern France). *Applied Geochemistry*, 23, 337–2349. doi:10.1016/j.apgeochem.2008.03.011
- El-Fadel, M., Maroun, R., Semerjian, L., & Harajli, H. (2003). A health-based socio-economic assessment of drinking water quality: The case of Lebanon. *Management of Environmental Quality: An International Journal*, 14, 53–368. doi:10.1108/14777830310479441
- Genius, M., Hatzaki, E., Kouromichelaki, E., Kouvakis, G., Nikiforaki, S., & Tsagarakis, K. (2008). Evaluating consumers' willingness to pay for improved potable water quality and quantity. *Water Resources Management*, 22, 825–1834. doi:10.1007/s11269-008-9255-7
- German Association of Energy and Water Industries. (2010). *Comparison of European water and wastewater prices*. Bonn, Germany: Wirtschafts und Verlagsgesellschaft Gas und Wasser mbH.
- Haddad, S. (2002). The relevance of political trust in Postwar Lebanon. *Citizenship Studies*, 6, 201–218. doi:10.1080/13621020220142978
- Howitt, R., Medellín-Azuara, J., & MacEwan, D. (2009). *Estimating the economic impacts of agricultural yield related changes for California*. Sacramento, CA: California Climate Change Center.
- Kaliba, A. R. M., Norman, D. W., & Chang, Y.-M. (2003). Willingness to pay to improve domestic water supply in rural areas of Central Tanzania: Implications for policy. *International Journal of Sustainable Development & World Ecology*, 10, 119–132. doi:10.1080/13504500309469791
- Kayaga, S., Calvert, J., & Sansom, K. (2003). Paying for water services: Effects of household characteristics. *Utilities Policy*, 11, 23–132. doi:10.1016/S0957-1787(03)00034-1
- Kfour, C. (2014). *Preserving Lebanon's water before the wells run dry*. Washington, DC: World Bank.
- Lashkaripour, G. R., & Zivdar, M. (2005). Desalination of brackish groundwater in Zahedan city in Iran. *Desalination*, 177, 1–5. doi:10.1016/j.desal.2004.12.002

- Lee, J.-S. (2014). Measuring the economic benefits of residential water quality improvement in Ulsan, Korea using a contingent valuation. *Urban Water Journal*, 11, 252–259. doi:10.1080/1573062X.2013.765490
- Long, S. J. (1997). *Regression models for categorical and limited dependent variables*. Thousands Oaks, CA: Sage Publications.
- Maas, E., & Grattan, S. (1999). Crop yields as affected by salinity. *Agronomy*, 38, 55–110.
- Maas, E. V., & Hoffman, G. J. (1977). Crop salt tolerance—current assessment. *Journal of the Irrigation and Drainage Division*, 103, 115–134.
- Masciopinto, C. (2013). Management of aquifer recharge in Lebanon by removing seawater intrusion from coastal aquifers. *Journal of Environmental Management*, 130, 306–312. doi:10.1016/j.jenvman.2013.08.021
- Maupin, M. A., Kenny, J. F., Hutson, S. S., Lovelace, J. K., Barber, N. L., & Linsey, K. S. (2014). *Estimated use of water in the United States in 2010* (pp. 2330–5703). Reston, VA: US Geological Survey.
- McDonald, R. I., Green, P., Balk, D., Fekete, B. M., Revenga, C., Todd, M., & Montgomery, M. (2011). Urban growth, climate change, and freshwater availability. *Proceedings of the National Academy of Sciences*, 108, 6312–6317. doi:10.1073/pnas.1011615108
- Meyerhoff, J., & Liebe, U. (2006). Protest beliefs in contingent valuation: Explaining their motivation. *Ecological Economics*, 57, 583–594. doi:10.1016/j.ecolecon.2005.04.021
- Michelsen, A., McGuckin, T., Sheng, Z., Lacewell, R., & Creel, B. (2009). *Rio Grande salinity management program: Preliminary economic impact assessment*. El Paso, TX: Rio Grande Salinity Management Coalition.
- Ministry of Energy and Water. (2010). *National water sector strategy*. Beirut, Lebanon: Author.
- Ministry of Energy and Water, and United Nations Development Programme. (2014). *Assessment of groundwater resources of Lebanon*. Beirut, Lebanon: Ministry of Energy and Water.
- Ministry of Environment. (2011). *State and Trends of the Lebanese Environment*. Beirut, Lebanon: Author.
- Munns, R., & Gilliam, M. (2015). Salinity tolerance of crops—What is the cost? *New Phytologist*, 208, 668–673. doi:10.1111/nph.13519
- Nicholls, R. J., Wong, P. P., Burkett, V., Codignotto, J., Hay, J., McLean, R., ... Arblaster, J. (2007). Coastal systems and low-lying areas. In M. Parry, O. Canziani, J. Palutikof, P. Van Der Linden, & C. Hanson (editors.), *Climate change 2007: Impacts, adaptation and vulnerability. contribution of working group II to the fourth assessment report of the Intergovernmental panel on climate change* (pp. 315–356). Cambridge, UK: Cambridge University Press.
- Pachauri, R. K., Allen, M. R., Barros, V. R., Broome, J., Cramer, W., Christ, R., ... Dasgupta, P. (2014). Climate change 2014: Synthesis report. Contribution of working groups I, II and III to the fifth assessment report of the intergovernmental panel on climate change. In R. Pachauri & L. Meyer (editors.), *Climate change 2014: Synthesis report* (pp. 151). Geneva: IPCC.
- Phelps, G., & Crabtree, S. (2013). *Worldwide, median household income about \$10,000*. Washington, DC: The Gallup Organization.
- Polyzou, E., Jones, N., Evangelinos, K. I., & Halvadakis, C. P. (2011). Willingness to pay for drinking water quality improvement and the influence of social capital. *The Journal of Socio-Economics*, 40, 74–80. doi:10.1016/j.socec.2010.06.010
- R Core Team. (2015). *R: A language and environment for statistical computing*. Vienna: R Foundation for Statistical Computing.
- Ragan, G. E., Makela, C., & Young, R. A. (1993). *Improved estimates of economic damages from residential use of mineralized water*. Fort Collins, CO: Colorado Water Resources Research Institute.
- Ragan, G. E., Young, R. A., & Makela, C. J. (2000). New evidence on the economic benefits of controlling salinity in domestic water supplies. *Water Resources Research*, 36, 1087–1095. doi:10.1029/1999WR900324

- Raje, D. V., Dhobe, P. S., & Deshpande, A. W. (2002). Consumer's willingness to pay more for municipal supplied water: A case study. *Ecological Economics*, 42, 391–400. doi:[10.1016/S0921-8009\(02\)00054-X](https://doi.org/10.1016/S0921-8009(02)00054-X)
- Tihansky, D. P. (1974). Economic damages from residential use of mineralized water supply. *Water Resources Research*, 10, 145–154. doi:[10.1029/WR010i002p00145](https://doi.org/10.1029/WR010i002p00145)
- United States Environmental Protection Agency. (2014). *Water quality standards handbook*. Washington, DC: United States Environmental Protection Agency: Office of Water.
- Vásquez, W. F., Mozumder, P., Hernández-Arce, J., & Berrens, R. P. (2009). Willingness to pay for safe drinking water: Evidence from Parral, Mexico. *Journal of Environmental Management*, 90, 391–3400. doi:[10.1016/j.jenvman.2009.05.009](https://doi.org/10.1016/j.jenvman.2009.05.009)
- Vörösmarty, C. J., Green, P., Salisbury, J., & Lammers, R. B. (2000). Global water resources: Vulnerability from climate change and population growth. *Science*, 289, 284–288. doi:[10.1126/science.289.5477.284](https://doi.org/10.1126/science.289.5477.284)
- WaterAid. (2016). *Water: At what cost? The State of the World's Water 2016*. London, UK: Author.
- Whittington, D., Briscoe, J., Mu, X., Barrón, W., Bourgeois, T., & Duval, J. M. 1987. Willingness to pay for water in rural areas: Methodological approaches and an application in Haiti. Wash Field Report. WASH Project.
- World Bank Water Demand Research Team. (1993). The demand for water in rural areas: Determinants and policy implications. *World Bank Research Observer*, 8, 47–70. doi:[10.1093/wbro/8.1.47](https://doi.org/10.1093/wbro/8.1.47)