Economic implications of agricultural reuse of treated wastewater in Israel: A statewide long-term perspective

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Abstract

We develop an Israeli version of the Multi-Year Water Allocation System (MYWAS) mathematical programming model to conduct statewide, long-term analyses of three topics associated with agricultural reuse of wastewater. We find that: (1) enabling agricultural irrigation with treated wastewater significantly reduces the optimal capacity levels of seawater and brackish-water desalination over the simulated 3-decade period, and increases Israel’s welfare by 3.3 billion USD in terms of present values; (2) a policy requiring desalination of treated wastewater pre-agricultural reuse, as a method to prevent long-run damage to the soil and groundwater, reduces welfare by 2.7 billion USD; hence, such a policy is warranted only if the avoided damages exceed this welfare loss; (3) desalination of treated wastewater in order to increase freshwater availability for agricultural irrigation is not optimal, since the costs overwhelm the generated agricultural benefits. We also find the results associated with these three topics to be sensitive to the natural recharge of Israel’s freshwater aquifers, and to the rate at which domestic-water demand evolves due to population and income growth.

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1. Introduction

Population growth has increased urban demand for freshwater and the need for sewage disposal, both of which motivate the agricultural reuse of wastewater. Wastewater is therefore perceived as a renewable resource for agricultural irrigation (Rutkowski et al., 2007), and its use is becoming common worldwide (Qadir et al., 2007). For example, in Israel, more than 85% of treated wastewater (TWW) is used for crop irrigation (IWA, 2012); in Spain, nearly 71% (Iglesias et al., 2010); and in California, about 20% of reclaimed wastewater is utilized in agriculture (Angelakis and Snyder, 2015). Thus, agricultural reuse of TWW substitutes for scarce freshwater sources, saves on fertilizer and energy costs through reuse of plant nutrients and trace elements (Dawson and Hilton, 2011), and its stable and drought-proof supply carries valuable agricultural benefits (Feinerman and Tsur, 2014). However, wastewater reuse is also associated with detrimental environmental and social implications (Hanjra et al., 2012), as well as negative health effects due to the presence of pathogens (Kazmia et al., 2008), heavy metals (Li et al., 2009), pharmaceuticals and other synthetic compounds (Ratola et al., 2012). As TWW differs from freshwater in salinity, pH, and concentrations of suspended solids and dissolved organic matter, TWW irrigation can change the soil’s physical, biological and chemical characteristics (Lado et al., 2012). An increase in soil salinity can reduce plant growth (Dinar et al., 1986; Kan et al., 2002), and accumulation of chloride, sodium and boron may be toxic to the plants (Bresler et al., 1982). Long-term irrigation with TWW might increase soil sodicity, which in turn reduces soil-structure stability (Feigin et al., 1991; Levy et al., 2014).

Given these pros and cons, TWW reuse requires long-term planning and investments that affect water economies at the basin, aquifer and even statewide levels. It requires setting sewage-reclamation quality standards and agricultural application constraints that will account for health and food safety and long-run processes of soil and groundwater contamination; it further necessitates the continuous development of infrastructures for collecting and reclaiming sewage, and distributing the TWW outputs to farming areas. Accordingly, a pricing scheme that incentivizes the efficient use of TWW should be implemented, taking into account both the productivity and supply costs of TWW relative to its alternatives—freshwater and brackish-water sources.
Being located at the boundary of a desert, and facing rapid and steady population growth, Israel's natural freshwater sources have fallen short of meeting the growing demand, particularly for domestic uses. In response, desalination plants have been installed to enhance and stabilize the freshwater supply, and freshwater allotments for agriculture have been cut, and replaced by TWW quotas (Kislev, 2011). Consequently, Israel is the country with the largest agricultural reliance on TWW, constituting about 40% of total agricultural water use (IWA, 2012); in comparison, TWW makes up 17% and 6% of the irrigation water in Spain and California, respectively (Goldstein et al., 2014). Moreover, Israel's water system connects, via the National Water Carrier, almost all of its major water resources into one operational system, which supplies water to almost all of its regions. This national water system essentially turns the country into one basin; that is, the net benefits associated with consuming a water unit at a particular location should be weighed against those derived by consuming this unit in a different place. These attributes make Israel a case of interest for many regions throughout the world that are facing growing water scarcity.

Our objective in this paper is to assess water-management and welfare implications of agricultural reuse of TWW in Israel from a state-wide, long-term perspective. Specifically, we are interested in three particular topics: first, we assess the welfare contribution of agricultural reuse of TWW by evaluating the welfare loss that would occur if TWW irrigation were not available for agricultural applications. The second topic focuses on the assessment of potential long-term damage caused by TWW irrigation to soil properties and groundwater quality. To this end, we evaluate an upper value for these damages by computing the costs of avoiding them altogether through desalination of TWW. That is, TWW desalination is considered a mandatory pre-reuse treatment. Such a policy not only prevents the long-term damage, but also increases the amount of freshwater available for the agricultural sector. Thus, regardless of the long-term damage, desalination of TWW as a method for merely increasing agricultural production comprises our third topic: we search for the optimal level of TWW desalination for agricultural use, where the desalination costs are weighed against the agricultural benefits obtained by turning TWW into freshwater for irrigation.

Our analytical tool is the Multi-Year Water Allocation System (MYWAS) model (Fisher and Huber-Lee, 2011). The MYWAS is the extended multi-year version of the 1-year steady-state WAS (Water Allocation System) model (Fisher et al., 2005). It is a dynamic discrete-time non-linear mathematical programming model that searches for optimal water allocation and infrastructural investments along time and space, while taking into account a range of economic data, physical factors and constraints. Thus, our approach follows the growing number of studies that have adopted hydro-economic modeling to explore efficient water management (see reviews by Harou et al., 2009 and Booker et al., 2012). Such models aim to solve the complex problem of water management while integrating different areas of expertise into a coherent unified framework, including hydrology, engineering, economics, environmental effects and geography. For instance, Xu et al. (2001), Haruvy et al. (2008) and Rosenberg et al. (2008) have included wastewater reuse in hydro-economic models. A prominent example in terms of spatial scope, detail and complexity is the CALVIN (California Value Integrated Network) model (Draper et al., 2003; Jenkins et al., 2004). Similarly, the Israeli version of the MYWAS encompasses a detailed water-allocation network on a national scale, incorporates demand functions for domestic, industrial and agricultural uses, and enables agricultural reuse of TWW. Given the extensive agricultural use of brackish water and TWW in Israel, we incorporate into the model the impact of water quality on agricultural production, and capture the substitution between freshwater, brackish water and TWW.

We study the above three topics by comparing a baseline scenario and three variations of this scenario in relation to the topics under consideration, where under each scenario, the MYWAS searches for the optimal management for a period of three decades while accounting for forecasted changes in water demand and natural enrichment of groundwater stocks. The baseline scenario consists of TWW reuse, and no infrastructures for TWW desalination; our simulation suggests continuing the on-going policies in Israel of increasing seawater desalination and the substitution of agricultural freshwater by TWW. Compared to the baseline, the absence of TWW for agricultural reuse (first topic) exacerbates the reliance on seawater desalination for both urban and agricultural sectors’ freshwater supply, causing a welfare loss of nearly 3.3 billion USD in terms of present values over the simulated 30-year time horizon.

As to the second topic, the upper bound of welfare loss associated with avoiding soil and groundwater damage through TWW desalination amounts to 2.7 billion USD; that is, a policy of mandatory TWW desalination is warranted only if the value of the avoided damage exceeds that welfare-loss valuation.

For the third topic, we conclude that under the current TWW-desalination technology and the agricultural substitutability between freshwater and TWW, increasing freshwater availability for agricultural irrigation through TWW desalination is too costly relative to the generated agricultural benefits. All of these results are found to be highly sensitive to the natural recharge level of freshwater aquifers. Thus, forecasts for drier climatic conditions in Israel (e.g., Krichak et al., 2011) are expected to increase Israel’s reliance on TWW, and to entail negative welfare effects.

The next section briefly describes the MYWAS model; the third section details the scenarios and discusses the results; the fourth section provides key conclusions.

2. MYWAS – The Israeli Version

A complete formal description of the Israeli version of the MYWAS appears in Appendix A, and detailed topology and data are available in Reznik et al. (2016). In this section, we outline the main properties of the model.

Fig. 1 presents a stylized topology of the MYWAS tool. The water consumers are located in urban and agricultural zones. Urban zones obtain only freshwater for domestic and industrial uses, delivered either from renewable freshwater aquifers, or from plants that desalinate seawater and/or brackish water extracted from non-renewable saline aquifers. These freshwater sources also provide freshwater to the agricultural consumers, who may also consume brackish water directly from the saline aquifers, or TWW from wastewater-treatment plants that treat the sewage generated by the urban sector. Freshwater aquifers are enriched by natural recharge (precipitation), which also affects the demand for agricultural water use.

The model’s detailed topology specifies the water sources (surface water, fresh and saline groundwater aquifers, desalination plants and wastewater-treatment plants), the regions of demand for agricultural
and urban (domestic and industrial) water uses, and the connecting lines between the water sources and the demand zones. Calibrated to Israel’s water economy situation in 2014, the model incorporates 48 water sources—16 underground aquifers, 19 wastewater-treatment plants, 3 surface reservoirs, 6 seawater-desalination plants, and 4 desalination plants for saline water—and 183 pipelines for freshwater and 58 pipelines for non-freshwater. For each water source, the data include annual recharge, maximum hydrological and technical extraction capacities, detailed cost data and linkages to demand regions. The data were obtained from TAHAL Consulting Engineers Ltd., with approval from the Israeli Water Authority (IWA).

The model maximizes the present value of net social welfare over a specified time period.\(^1\) The MYWAS uses the WEAP (Water Evaluation and Planning) system as an interface. The WEAP system is linked to the optimization software GAMS through the program Python, which feeds the data from the WEAP system into GAMS, runs the optimization process and introduces the optimization results back into the WEAP system.

3. Simulations

As already mentioned, to study the three topics we apply the MYWAS for three decades. In all cases, the demand functions for urban water use shift to the right by 2.14% a year, accounting for an annual population growth rate of 1.8% (CBS, 2014) plus 0.34% for the effect of income increase on water demand.\(^2\) Water is a relatively minor intermediate input in Israel’s industry (Kislev and Zaban, 2013); we assume the current trend of 1% annual increase in the industrial demand (Kislev and Zaban, 2013) to prevail along the entire simulation period. Agricultural demand is expected to vary over time with changes in agricultural output prices and technological improvements (Schoengold et al., 2006); however, based on a meta-analysis of demand estimations, Scheierling et al. (2006) found insignificant responsiveness of agricultural demand to technological availability. Moreover, despite considerable technological improvements and price fluctuations, the overall agricultural water consumption in Israeli agriculture has remained quite stable in the last four decades (Kislev and Zaban, 2013). We therefore fixed the agricultural demand functions throughout the simulated period.

We run the four scenarios assuming average annual recharge of aquifers and surface water sources throughout the entire simulation period, where the statewide aggregated natural recharge amounts to 1200 \(\times 10^9\) m\(^3\)/year (Weinberger et al., 2012). In addition, we assess the sensitivity of the results to drought conditions by incorporating, throughout the entire period, a natural recharge level as low as 60% of the annual average (Luckmann et al., 2016). Reduced precipitation is expected to shift the demand for water. We account for the effect on agricultural irrigation using the elasticity of water demand with respect to rainfall, which was estimated by Bar-Shira et al. (2006) to be −0.047. Regarding the urban sector, the economic literature provides evidence for positive (Bar-Shira et al., 2007), negative (Olmscheid et al., 2007) and insignificant (Schleich and Hillenbrand, 2009) impacts of rainfall on the elasticity of water demand. Using sensitivity analyses, we find the simulation results to be quite robust to changes in urban water demand elasticity within the range of estimated rainfall impacts. Hence, the results for the case of no change in urban demand elasticity are used.

Appendix B reports the results in detail, where Figs. 2, 3 and 4 present the optimal trajectories of selected measures under the BL, NW and

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\(^1\) The notion of production effectiveness is adopted from Caswell and Zilberman (1986).

\(^2\) Forecast capability can be assessed by comparing the model and observed outcomes.

Fisher et al. (2005) used the WAS model (the earlier static version of the MYWAS tool) to predict Israel's water consumptions for 2010 based on 1995 data. Their projections for the consumptions by urban, industrial and agricultural sectors varied from the actual ones by −0.2%, −15.5% and 11.7%, respectively.

\(^3\) The MYWAS can also examine pricing schemes by searching for minimal-cost solutions where consumption is set a priori based on a system of prices and/or quotas. In the current study, we conduct only net-benefit optimizations, implying that prices are endogenously determined.

\(^4\) Bar-Shira et al. (2007) estimate a water-demand income elasticity of 0.17 for Israel. Multiplying by a conservative long-term income growth rate of 2% (World Bank, 2016), we obtain 0.34%. 

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MD scenarios; the outcomes of the OD scenario are not shown, as they were found similar to the BL.

Figs. 2 and 3 present, respectively, trends of agricultural water consumption and water-supply patterns for the two levels of natural recharge. We describe the main findings, starting with the case of average recharge (charts a, c, e and g in Figs. 2 and 3). Water use is characterized by a steady increase in freshwater consumption in the urban sector due to the effect of population and income growth, with minor differences between the scenarios (Appendix B); this process produces growing amounts of TWW, which under the BL scenario is used by the agricultural sector (Fig. 2e) as a substitute for freshwater irrigation (Fig. 2c). This is an intuitive process: as agricultural irrigation has both larger water-demand elasticity than that of the urban sector and the possibility to substitute TWW for freshwater, the above trend constitutes an efficient solution to both the growing scarcity of freshwater for domestic consumption and the need to dispose of the growing amounts of TWW generated as a productive byproduct of the domestic freshwater consumption.

Consider the NW scenario in comparison to the BL: the consumption of TWW under the NW scenario is (by constraint) zeroed (Fig. 2e), that of brackish water is stable (Fig. 2g), and agricultural consumption of freshwater increases significantly, by about $400 \times 10^6$ m$^3$/year (Fig. 2c), but not to the extent that it completely compensates for the absence of TWW; hence, the total consumption of irrigation water is considerably reduced (Fig. 2a). Forbidding wastewater recycling under the NW scenario entails faster extension of the seawater-desalination capacities compared to the BL (Fig. 3c) and provision of larger amounts of desalinated seawater (Fig. 3e), while generally keeping groundwater extraction (Fig. 3a) and brackish water desalination (Appendix B) unchanged.

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**Fig. 2.** Statewide agricultural water consumption.
The MD scenario resembles the NW one in the sense that irrigation with TWW is forbidden (Fig. 2e), but desalinated TWW can be used for agricultural production, thereby alleviating the irrigation water shortage compared to the NW scenario. Consequently, seawater-desalination capacities (Fig. 3c) and desalination levels (Fig. 3e) are lower than in the NW scenario, but since TWW desalination increases with time (Fig. 3g), the overall irrigation amounts are kept stable, and exceed those of the NW scenario (Fig. 2a). As already noted, the optional desalination alternative (OD) appeared to be similar to the BL, indicating that TWW-desalination costs overwhelm the agricultural benefits associated with the water-quality improvement.

Compared to the average-recharge cases, the low-recharge ones exhibited a considerable reduction in groundwater extraction (Fig. 3b versus a), and increases in seawater-desalination capacity (Fig. 3d versus c), seawater desalination (Fig. 3f versus e), and TWW desalination under the MD scenario (Fig. 3h versus g).

If policy-makers chose to induce the simulated optimal solutions by using water prices, then tariffs should be set so as to reflect the consumers’ marginal utility (in monetary terms) for the domestic water consumers, and the water’s value of marginal product (VMP) for the industrial and agricultural sectors. In addition, the water suppliers should be charged for groundwater-extraction fees based on the scarcity level of the water in the aquifers. These optimal prices should be both region-specific and updated on an annual basis. Fig. 4 presents statewide time paths of weighted-average domestic-water marginal utilities, VMPs of agricultural irrigation water, and shadow values of minimal groundwater-stock constraints; the latter reflects the levels of aquifer-water scarcity.

The fluctuations in the domestic consumers’ marginal utility (Fig. 4a) correspond to the timing of extended increases in seawater desalination levels (Fig. 3c): marginal utility gradually increases with time, and sharply declines each time the seawater desalination level is...
increased (e.g., in the years 2029, 2035 and 2039 under BL). The agricultural VMP of freshwater (Fig. 4c) and TWW (Fig. 4e) is stable, and so are the groundwater scarcity rents (Fig. 4g). Apparently, low recharge has only a minor impact, except under the NW scenario, under which optimal prices of groundwater and agricultural freshwater and TWW considerably increased (Fig. 4d, f and h).

We conducted sensitivity analyses with respect to selected parameters of the model; the impacts of three of them are noteworthy. First, following the literature debate on social discounting (Nordhaus, 2007), we ran the model under a lower annual discount rate of 1.4% (Stern, 2007) instead of 5.0%. Consequently, the seawater-desalination capacities were expanded earlier in time, which in turn enabled reducing extractions from natural freshwater aquifers so as to leave larger stocks at the end of the planning horizon. In the second analysis, we considered the effectiveness of TWW relative to freshwater in agricultural production. As already noted, while TWW irrigation can harm yields through salinity effects and reduce profits due to application constraints, TWW has the advantage of being a more reliable water source than freshwater and having higher nutrient contents (Toze, 2006; Hamilton et al., 2007). Thus, it may well be that TWW’s productivity is equivalent to that of freshwater, implying that the factor used to convert TWW to freshwater equivalents should be 1.0 instead of 0.83. In fact, this change causes only a slight reduction in the agricultural use of freshwater. The third analysis relates to the rate at which domestic-water demand grows through time; this resulted in a dramatic impact on the optimal trajectories. For example, an increase from an annual growth of 2.14% to 3% entails a 40% increase in the amount of seawater desalinated during the simulated period, where at the same time, total agricultural freshwater use decreases by 9% and TWW reuse increases by 15%. Thus, the evolution of domestic-water demand is a major driving force in the course of optimal water management.

**Fig. 4.** Statewide average marginal utility (a and b), agricultural value of marginal product of freshwater (c and d) and of TWW (e and f), and groundwater-scarcity shadow values (g and h).
What are the welfare implications of the TWW policies and climate conditions? Table 1 presents various differences in the objective functions obtained under the BL, NW and MD scenarios. The upper section of the table presents the impacts of the two TWW policies versus BL. Under average recharge, prohibiting irrigation by wastewater (the NW scenario) reduces welfare by more than 3 billion USD over the simulated period; the contribution of wastewater reuse increases by 40% under the low-recharge case. On average, throughout the simulated period, the amounts of TWW reused in the BL scenario (about 18.3 billion m³) are worth 18 ¢/m³ and 25 ¢/m³ under average and low recharge, respectively. Note, however, that these calculations overlook a wide range of costs and benefits associated with the implementation of alternative TWW-disposal methods, such as discharge to waterways and ultimately to the Mediterranean Sea.

If irrigation by TWW is allowed conditional on desalination (the MD scenario), welfare decreases by 2.7 and 3.2 billion USD under average and low recharge, respectively. Recall that this welfare loss is warranted provided that it does not exceed the damage caused to soil and groundwater by TWW irrigation under the BL option, which on average amounts to 15 ¢/m³ and 17 ¢/m³ under average and low recharge, respectively.

The bottom row of Table 1 presents welfare effects stemming from the low natural recharge. We evaluate a damage of more than 3.5 billion USD under the BL scenario, with a minor impact of the two TWW policies.

4. Key Conclusions

About 330 km² of municipal wastewater are generated annually throughout the world (Hernández et al., 2015). TWW reuse is steadily increasing, and this trend is expected to continue; for instance, reuse of TWW increased in California from approximately 400 million m³ in 1989 to over 820 million m³ in 2009. Indeed, California has classified reuse as one of the major means to reduce its water use, and estimates suggest a potential addition of 1850 million m³ (NWRI, 2012). The findings of this study based on the case of Israel can thus be of interest for countries facing intensifying water scarcity.

We conclude that agricultural reuse of TWW is an optimal water-management strategy under water shortage, mainly as an economically effective way to transfer freshwater from the farming sector to the urban sector; that is, from freshwater users with high demand elasticity to those with low demand elasticity. Moreover, the welfare gains associated with agricultural TWW reuse dramatically increase as freshwater sources become scarcer due to supply and demand shifters such as climate change, population growth and income increase, and thereby increase the attractiveness of substituting freshwater with TWW in farming production. The Israeli experience follows this practice: during the last two decades, agricultural use of freshwater has decreased by 25 million m³ a year, whereas TWW irrigation has increased by 17 million m³ a year, implying a TWW/freshwater exchange ratio of 0.67; with the expected continuation of population and income rise, our analysis suggests an exchange ratio of 1.5 for the next 30 years, climbing to 2.0 under permanent drought conditions.

Water management is associated with market failures, which economically justify intensive governmental intervention. Hence, the optimal state-wide long-run strategies described above should be based on a governmental policy program that accounts for a range of aspects; one of them is an infrastructure investment plan. As we show, TWW-reuse policy affects the course of the development of a water economy as a whole; for example, avoiding TWW reuse may make the construction of seawater-desalination capacity optimal that otherwise would be unwarranted. Another challenge is to design a pricing scheme that signals to consumers the correct marginal costs of supplying water with heterogeneous qualities, and at the same time covers costs (Dinar and Pochat, 2015; Reznik et al., 2016). This requires coordination and agreement among the various players, including municipalities, farming associations and water suppliers (Feinerman et al., 2001).

The Israeli version of the MYWAS developed in this study is a powerful tool for designing and analyzing long-run water-management policies from an economic perspective, while taking into account hydrologic, agronomic, engineering, climatic and economic aspects. However, the MYWAS does not cover the entire range of influential factors; we mention a few potential avenues for enriching the analysis.

While the MYWAS model takes into account the effect of water quality on agricultural production, and the substitution between qualitatively different water sources, the quality of each source is considered constant throughout the simulation period. Therefore, the analysis overlooks the effects of desalination on the salinity of freshwater supplied to the domestic, industrial and agricultural sectors, as well as on the salinity of TWW. Consequently, our study assigns credits to desalination through the enlargement of freshwater quantity only, but ignores the associated salinity improvements. Furthermore, salinity is one of many water-quality indicators with welfare implications. One may account for these impacts by adding state variables that measure the concentration of selected materials throughout the water-supply system, and refer to their influence on demands and treatment costs. Our study implicitly refers to the agronomic risks associated with intensive TWW irrigation, by assigning them an upper level through evaluating their prevention costs. A deeper understanding of the damages is needed to enable introducing them explicitly into the model, and thereby to search for the optimal level of treatment.

Rather than using agricultural demand functions, one may incorporate agricultural water use into the model by linking it to a detailed agricultural model, thus enabling explicit farming responses such as crop- acreage selection and water applications. Another potential extension refers to hydrological processes with respect to both water quantity and quality. Groundwater stock and its salinity level depend on deep percolations from agricultural areas (Knapp and Baerenklau, 2006), which in turn depend on the patterns of agricultural water use. An aquifer’s water-stock level affects leakage to other groundwater bodies, and seawater intrusion into coastal aquifers (Kan et al., 2010). These processes should be accounted for in a dynamic model so as to obtain an accurate optimal management course.

Finally, our analysis is deterministic. Accounting for stochastic and uncertain processes (e.g., Tsur and Zemel, 2004) may vary the results.

Supplementary data to this article can be found online at http://dx.doi.org/10.1016/j.ecolecon.2016.11.018.

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Appendix A. Formal Description of the MYWAS

Consider a small open economy with natural sources for freshwater and brackish water, urban regions with demands for domestic and industrial uses of freshwater, agricultural regions that demand irrigation water of various qualities, and an infrastructural system incorporating wastewater-treatment plants, desalination plants for seawater and treated wastewater, pumping stations and pipelines. Long-run efficient management of such a water economy is the objective of the MYWAS.

The MYWAS is a discrete-time dynamic optimization model. Let us denote by the subscript indices \( a = 1, \ldots, A \) an agricultural region, \( u (u = 1, \ldots, U) \) an urban region, \( f (f = 1, \ldots, F) \) a source for freshwater, \( b (b = 1, \ldots, B) \) a source for brackish water, and \( h (h = 1, \ldots, H) \) a wastewater-treatment plant. We use the superscripts \( b, \beta, l \) and \( w \) to indicate freshwater, brackish water, sewaged, and treated wastewater, respectively, and \( d \) and \( i \) to indicate domestic and industrial uses of freshwater in the urban regions. Time is denoted by \( t, t = 1, \ldots, T \), where \( T \) is the optimization planning horizon. Water quantities consumed in demand regions are represented by \( Q \); for instance, \( Q^{ft}_{ut} \) is the amount of freshwater consumed by the domestic sector in urban region \( u \) during time \( t \). \( E \) stands for extractions from sources; for example, \( E^{bf}_{bt} \) and \( E^{bf}_{wt} \) are, respectively, the amounts of freshwater and brackish water extracted at time \( t \) from source \( b \) and \( h \), and \( E^{bf}_{wt} \) and \( E^{bf}_{wt} \) are the quantities of treated, and treated-and-desalinated wastewater produced by plant \( h \) during time \( t \), respectively. Water transfers between spatial points are denoted by \( G \); thus, \( G^{bf}_{wt} \) is the amount of brackish water delivered during time \( t \) from brackish-water source \( b \) to agricultural region \( a \). Accordingly, the vectors \( Q, E \) and \( G \) incorporate all of the water quantities consumed in the various regions, extracted from sources, and transferred between points, respectively; these are all optimization decision variables.

Water extractions \((E)\) and transfers \((G)\) are constrained by infrastructural capacities; the latter can be extended periodically, where the levels of the extension constitute additional decision variables. We symbolize capacity expansions by the letter \( N \); for instance, \( EN^{bf}_{bt} \) is the increase in the capacity of extraction of brackish water from source \( b \) during time \( t \), and \( GM^{bf}_{wt} \) represents the increase in transfer capacity of sewage water produced in urban region \( u \) to wastewater-treatment plant \( h \) at time \( t \). The vectors \( EN \) and \( GN \) incorporate all of the increases in extraction and transfer capacity, respectively.

We further define state variables that represent the cumulative increases in infrastructural capacities; these variables are denoted by \( EM \) and \( GM \) for extraction and transfer capacities, respectively. For example, \( EM^{bf}_{bt} = \sum_{\tau=0}^{t-1} EN^{bf}_{bt} \) is the cumulative increase in the extraction capacity of brackish-water source \( b \) until time \( t \). In addition, we define the starting level of capacities; for instance, \( GM^{bf}_{wt} \) is the capacity of sewage-water transfer from urban region \( u \) to wastewater-treatment plant \( h \) at the beginning of the planning time. Accordingly, \( GM^{bf}_{wt} + GM^{bf}_{wt} \) is the overall transfer capacity at this link; hence, \( c^{bf}_{wt} \leq GM^{bf}_{wt} + GM^{bf}_{wt} \).

Additional state variables represent the extractable amounts of water stocks stored in the various natural freshwater and brackish-water sources. For some freshwater source \( f \), the amount of water available for extraction at time \( t \) is denoted \( V^{ft}_{t} \), and it is physically restricted by \( V^{ft}_{t} \) from above and by \( V^{ft}_{t} \) from below. We also introduce \( k^{ft}_{t} \) as the unit value of each cubic meter left in aquifer storage after time \( T \), representing the marginal welfare contribution from water to future generations.\(^5\) Alternatively, an end-point minimum constraint \( V^{ft}_{t} + V^{ft}_{t} \geq V^{ft}_{t} \), can be additionally introduced to reflect the same welfare considerations beyond the planning horizon \( T \). The extractable stock at some time \( t \) is given by: \( V^{ft}_{t} = V^{ft}_{t} + \sum_{\tau=1}^{t} R^{ft}_{p} - \sum_{\tau=1}^{t} (E^{bf}_{bt} + L^{ft}_{t}) \), where \( V^{ft}_{t} \) is the initial extractable content of the source, \( R^{ft}_{p} \) is the natural recharge during time \( t \), and \( L^{ft}_{t} \) is the spillover from the source during time \( t \), where by definition \( L^{ft}_{t} = \max(0, V^{ft}_{t} - V^{ft}_{t}) \). A minimum overflow level \( L^{ft}_{t} \) may be assigned to each freshwater source \((e., \text{to reflect “water for nature” regulations}); the vectors \( L \) and \( L \) are defined accordingly.

Under the above specifications, desalination plants can be considered freshwater sources with annual stock of zero; i.e. \( V^{ft}_{t} = 0 \). However, for notational simplicity we set \( V^{ft}_{t} = EM^{bf}_{bt} + GM^{bf}_{wt} \), such that only the plants’ capacity constraints can be effective. Also note that in brackish-water sources \( R^{ft}_{t} = 0 \), and therefore \( L^{ft}_{t} = 0 \).

We use \( C \) to denote variable costs per volumetric unit, where these costs incorporate energy, as well as variable operational and maintenance costs. For instance, \( C^{bf}_{bt} \) is the per-unit transfer cost of treated wastewater from wastewater-treatment plant \( h \) to agricultural region \( a \) at time \( t \), and \( C^{bf}_{bt} \) stands for the cost of extracting one unit of freshwater from source \( f \) at time \( t \) in the latter, the cost depends on the source’s stock \( V^{ft}_{t} \), where lower stocks entail larger extraction costs (note that for wastewater-treatment plants, the stock is irrelevant).

Costs of capacity increase, which mostly comprise capital investments, are represented by \( S \); for example, \( S^{bf}_{bt} \) is the per-time-unit cost of increasing the capacity of wastewater-treatment plant \( h \) by one unit at time \( t \). Likewise, \( S^{bf}_{bt} \) is the per-time-unit cost associated with increasing the capacity of freshwater transfer from source \( f \) to agricultural region \( a \) by one unit at time \( t \). These costs are expressed in terms of per-time-unit payments for a loan taken to finance the capacity increase, computed based on a constant interest rate and constant payments during the entire lifetime of the respective infrastructure. We further assume that increased infrastructural capacities are rebuilt at the end of the infrastructure’s lifetime, and therefore the per-time-unit payments prevail forever; this assumption eliminates impacts of the planning horizon \( T \) on the optimal course of capacity increases.

The benefits associated with water consumption are based on constant-elasticity demand functions. The function \( (\mu + 1)^{-1} V^{fl}_{t} (Q^{fl}_{t})^{\mu+1} \) is the total willingness to pay for the amount of freshwater consumed by the domestic sector in region \( u \) at time \( t \), \( Q^{fl}_{t} [\text{i.e., the area below the inverse-demand function } V^{fl}_{t} (Q^{fl}_{t})^{\mu}] \), where \( V^{fl}_{t} \) and \( \mu \) are parameters, and \( 1/\mu \) is the urban-demand elasticity. Similarly, the industrial sector’s total willingness to pay is \( (\mu + 1)^{-1} V^{fl}_{t} (Q^{fl}_{t})^{\mu+1} \).

To represent benefits in the agricultural sector, we apply the function \( (\eta + 1)^{-1} A^{fl}_{a} (Q^{fl}_{a})^{\eta+1} \) \( \text{as the associated with the consumption of freshwater, treated wastewater and brackish water (see Feinerman et al., 2001). In this case, if either } Q^{ft}_{t} < 0 \text{ or } Q^{ft}_{t} > 0, \text{ the demand elasticity does not equal } 1/\eta, \text{ and therefore it is not constant, and depends on the overall water consumption from the various water types. The parameters } \gamma \text{ and } \delta \text{ translate treated wastewater and brackish water to freshwater in terms of their value of marginal product (VMP). That is, the freshwater VMP equals } \alpha (Q^{ft} + \delta Q^{bt} + \gamma Q^{wa}) \text{, and the treated-wastewater VMP is } \alpha (Q^{ft} + \delta Q^{bt} + \gamma Q^{wa}) \text{, hence, } \delta = \frac{V^{wa}}{V^{wa}}, \text{ in other words, } \delta \text{ is the increase in the regional consumption of freshwater required to maintain the regional VMP unchanged in response to a reduction of one unit in the regional consumption of treated wastewater.} \)
Additional benefits, or costs, are associated with spillovers from freshwater storage. Let $\psi_{bt}$ represent the net benefits per unit of overflow, where benefits may be related to environmental contributions of surface streams, and costs to damages, such as floods.

Given the interest rate $r$, the initial levels of the state variables, and the levels assigned to the exogenous factors throughout the planning horizon (e.g., aquifer-recharge levels, expansion of demands that are introduced through changes in parameters $t_{11}, t_{12}$ and $a_{13}$, etc.), the problem solved by the MYWAS is:

$$
\max \sum_{\text{Q.E.}} \frac{1}{1+r} \sum_{t=1}^{T} \left[ \sum_{u=1}^{H} \alpha_u (Q_u - 0)^+ \tau - \sum_{d=1}^{A} b_d - \sum_{h=1}^{B} W_h - \sum_{a=1}^{C} C_a \right] + \sum_{t=1}^{T} \left[ \sum_{u=1}^{H} \alpha_u (Q_u - 0)^+ \tau - \sum_{d=1}^{A} b_d - \sum_{h=1}^{B} W_h - \sum_{a=1}^{C} C_a \right] + \sum_{t=1}^{T} \left[ \sum_{u=1}^{H} \alpha_u (Q_u - 0)^+ \tau - \sum_{d=1}^{A} b_d - \sum_{h=1}^{B} W_h - \sum_{a=1}^{C} C_a \right] + \sum_{t=1}^{T} \left[ \sum_{u=1}^{H} \alpha_u (Q_u - 0)^+ \tau - \sum_{d=1}^{A} b_d - \sum_{h=1}^{B} W_h - \sum_{a=1}^{C} C_a \right]
$$

subject to:

$$Q_{ut}^d \leq \sum_{f=1}^{F} C_{jut}^d \forall u, t;$$

$$Q_{ut}^d + Q_{ut}^d \leq \sum_{f=1}^{F} C_{jut}^d \forall u, t;$$

$$Q_{ut}^d \leq \sum_{b=1}^{B} C_{bat} \forall u, t;$$

$$Q_{ut}^d \leq \sum_{b=1}^{B} C_{bat} \forall u, t;$$

$$\sum_{d=1}^{A} G_{jut}^d + \sum_{d=1}^{A} G_{jut}^d \leq E_{jut}^d \forall f, t;$$

$$\sum_{b=1}^{B} G_{bat} \leq E_{bat}^d \forall b, t;$$

$$\sum_{h=1}^{H} G_{hat} \leq \theta(Q_{ut}^d + Q_{ut}^d) \forall u, t;$$

$$E_{hat}^d \leq \rho \sum_{u=1}^{U} C_{hat} \forall h, t;$$

$$\sum_{a=1}^{A} G_{hat} \leq E_{hat} - E_{hat} \forall h, t;$$

$$\sum_{a=1}^{A} G_{hat} \leq E_{hat} \forall h, t;$$

$$E_{hat}^d \leq E_{hat}^d \forall h, t;$$

$$\sum_{t=1}^{T} G_{jut}^d \forall f, a, t;$$

$$\sum_{t=1}^{T} G_{jut}^d \forall f, u, t;$$

$$\sum_{t=1}^{T} G_{jut}^d \forall b, a, t;$$
\[ GM_{lht} = \sum_{r=1}^{t-1} GN_{lht}, \forall u, h, t; \] (15)

\[ GM_{w} = \sum_{r=1}^{t-1} GN_{w}, \forall h, a, t; \] (16)

\[ GM_{h} = \sum_{r=1}^{t-1} GN_{h}, \forall h, a, t; \] (17)

\[ EM_{f}^{h} = \sum_{r=1}^{t-1} EN_{f}^{h}, \forall f, t; \] (18)

\[ EM_{b}^{h} = \sum_{r=1}^{t-1} EN_{b}^{h}, \forall b, t; \] (19)

\[ EM_{w}^{h} = \sum_{r=1}^{t-1} EN_{w}^{h}, \forall h, t; \] (20)

\[ EM_{l}^{h} = \sum_{r=1}^{t-1} EN_{l}^{h}, \forall h, t; \] (21)

\[ C_{f}^{h} \leq GM_{f}^{h} + GM_{l}^{h}, \forall f, u, t; \] (22)

\[ C_{l}^{h} \leq GM_{l}^{h} + GM_{f}^{h}, \forall f, a, t; \] (23)

\[ C_{b}^{h} \leq GM_{b}^{h} + GM_{l}^{h}, \forall b, a, t; \] (24)

\[ C_{l}^{h} \leq GM_{l}^{h} + GM_{l}^{h}, \forall u, h, t; \] (25)

\[ C_{w}^{h} \leq GM_{w}^{h} + GM_{l}^{h}, \forall h, a, t; \] (26)

\[ C_{h}^{h} \leq GM_{h}^{h} + GM_{w}^{h}, \forall h, a, t; \] (27)

\[ E_{f}^{h} \leq EM_{f}^{h} + EM_{f}^{h}, \forall f, t; \] (28)

\[ E_{b}^{h} \leq EM_{b}^{h} + EM_{b}^{h}, \forall b, t; \] (29)

\[ E_{l}^{h} \leq EM_{l}^{h} + EM_{l}^{h}, \forall h, t; \] (30)

\[ E_{w}^{h} \leq EM_{w}^{h} + EM_{w}^{h}, \forall h, t; \] (31)

\[ EM_{f}^{h} + EM_{b}^{h} \leq EM_{l}^{h} + EM_{w}^{h}, \forall h, t; \] (32)

\[ V_{f} = V_{f0} + \sum_{r=1}^{t} R_{f} - \sum_{r=1}^{t-1} \left( E_{f}^{h} - L_{f} \right), \forall f, t; \] (33)

\[ V_{b} = V_{b0} - \sum_{r=1}^{t} E_{b}^{h}, \forall b, t; \] (34)

\[ E_{f}^{h} \leq V_{f} + V_{f}, \forall f, t; \] (35)

\[ E_{b}^{h} \leq V_{b} + V_{b}, \forall b, t; \] (36)

\[ V_{f} \geq V_{f}, \forall f; \] (37)

\[ V_{b} \geq V_{b}, \forall b; \] (38)

\[ L_{f} = \max(0, V_{f} - V_{f}), \forall f, t; \] (39)

\[ Q, E, G, EN, GN \geq 0, \] (40)

\[ I \geq \frac{1}{2}. \]
The objective function includes two aggregate elements: one is associated with the sets of water variables \(Q, E, G, \) covering the period \(t = 1, \ldots, T\), the other refers to the sets of capacity expansions \(EN\) and \(CN\), ranging from \(t = 1\) to \(T - 1\).

The set of constraints (1) through (4) ensures that, for the water type (\(\phi, \beta\) or \(w\)), the amount consumed in each region will not exceed the amounts delivered to that region from all sources. Constraints (5) and (6) guarantee that the amounts delivered to demand regions from freshwater and brackish-water sources will not exceed the amounts extracted from those sources. The set of limits (7) constrains the aggregated sewage amounts delivered to wastewater-treatment plants from each urban region such that it will not exceed the amount of sewage produced in that region, where the parameter \(\theta\) is the sewage/freshwater production rate. According to (8), production of treated wastewater at each wastewater-treatment plant will not exceed the amount of sewage transferred to the plant from all urban regions, where \(\rho\) stands for the wastewater/sea-wastage-conversion rate. The constraints in (9) ensure, for each wastewater-treatment plant, that the total amount of wastewater delivered to agricultural regions from the plant will not exceed the plant’s wastewater-desalination and the sewage produced in that region.\(\rho\) The parameter \(\eta\) is the sewage/freshwater production rate. According to (8), production of treated wastewater at each wastewater-treatment plant will not exceed the amount of sewage transferred to the plant from all urban regions, where \(\rho\) stands for the wastewater/sea-wastage-conversion rate. The constraints in (9) ensure, for each wastewater-treatment plant, that the total amount of wastewater delivered to agricultural regions from the plant will not exceed the plant’s wastewater-desalination plant.

(10) limits the deliveries of desalinated wastewater to agricultural districts to the amount produced at each wastewater-desalination plant, and Eq. (11) ensures that desalination of treated wastewater will not exceed wastewater production in each plant. Eqs. (12) through (21) define the cumulative state variables, and the limits (22) through (31) restrain the extractions and transfers by their corresponding capacities. Eq. (32) confines the capacity of wastewater desalination to not exceed its corresponding wastewater-treatment plant. Eqs. (33) and (34) define the extractable stocks in freshwater and brackish-water sources, respectively, where (35) and (36) use these stocks as upper limits for the corresponding extractions, and (37) and (38) impose endpoint minimum stocks. Eq. (39) defines the spillover from freshwater sources, and (40) gives the non-negativity and minimal spillover constraints.

A1. Calibrating Demand Functions

For the domestic and industrial urban inverse-demand functions, \(v_{td}^{QL}(Q_{td}^{QL})^\phi, v_{td}^{QI}(Q_{td}^{QI})^\phi, v_{td}^{QI}(Q_{td}^{QI})^\phi, v_{td}^{QI}(Q_{td}^{QI})^\phi,\) the elasticity parameter \(\mu\) is taken from Bar-Shira et al. (2007). The \(v_{td}^{QL}\) and \(v_{td}^{QI}\) parameters are calibrated for each region based on regional consumption and the water price observed in the base period \(t = 1\): \(v_{td}^{QL} = \frac{\partial Q_{td}^{QL}}{\partial Q_{td}}\mu^\phi, v_{td}^{QI} = \frac{\partial Q_{td}^{QI}}{\partial Q_{td}}\mu^\phi,\) where \(p_{t}\) and \(p_{t}\) are the corresponding freshwater prices for domestic and industrial uses. Then, \(v_{td}^{QL}\) and \(v_{td}^{QI}\) for each time \(t\) may change due to population and income growth.

Regarding the agricultural inverse-demand function \(\alpha_{a}Q_{a}^{QL} + \alpha_{a}Q_{a}^{QI} + \gamma Q_{a}^{QI}\), the parameters \(\alpha_{a}\) and \(\gamma\) are determined based on historic exchange rates used by MOAG in the replacement of freshwater quotas by treated-wastewater and brackish-water allotments. The parameter \(\alpha_{a}\) is adopted from Bar-Shira et al. (2006). The parameter \(\alpha_{a}\) is calibrated for every region \(a\) based on the consumption level and the agricultural freshwater price observed in the base period, \(p_{t}\), such that \(\alpha_{a} = \frac{\partial Q_{a}^{QL}}{\partial Q_{a}}\mu^\phi, \frac{\partial Q_{a}^{QI}}{\partial Q_{a}}\mu^\phi,\gamma Q_{a}^{QI}\). In later periods throughout the simulations, \(\alpha_{a}\) may change due to external factors.

References


