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Economic Implications of Agricultural Reuse of Treated Wastewater in Israel: A Statewide Long-Term Perspective

by

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Abstract

We apply the Multi-Year Water Allocation System (MYWAS) mathematical programming model to Israel's water economy 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 2 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 1.5 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.

Key words: wastewater reuse, agriculture, water management, economics, modeling, desalination

JEL: Q25, Q15

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I. Introduction

Population growth has increased urban demand for freshwater and the need for sewage disposal, both motivating 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 46% of reclaimed wastewater is utilized in agriculture (Sato et al., 2013). 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), and pharmaceutical 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 plant (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 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 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, where TWW constitutes 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 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 statewide, 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). MYWAS is the extended multi-year version of the one-year steady-state WAS (Water Allocation System) model (Fisher et al., 2005). It is a dynamic 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 MYWAS encompasses a detailed water-allocation network at the 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, MYWAS also accounts for the impact of water quality on agricultural production, and captures the substitution between freshwater, brackish water and treated wastewater.

We study these three topics by comparing a baseline scenario and three variations of this scenario in relation to the topics under consideration, where under each scenario MYWAS searches for the optimal management for a period of 3 decades. The baseline scenario comprises TWW reuse, and no infrastructures for TWW desalination. 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 more than 2 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 1.5 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, increasing freshwater availability for agricultural irrigation through TWW desalination is too costly relative to the generated agricultural benefits. Finally, 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 concludes.

II. MYWAS – The Israeli Version

In this section, we outline the main properties of the Israeli version of MYWAS. A complete formal description of the model appears in the Appendix, and detailed topology and data are available in Reznik et al. (2014).

The model's topology specifies the water sources (groundwater aquifers, natural surface water, desalination plants and wastewater-treatment plants), the regions of demand for agricultural and non-agricultural (domestic and industrial) water uses, and the connecting lines between the sources and the demand zones. Calibrated to Israel's water economy situation in 2010, the model incorporates 46 water sources, including 16 underground aquifers, 19 wastewater-treatment plants, 3 surface reservoirs, 4 seawater-desalination plants, 4 desalination plants for saline water, 183 pipelines for freshwater and 58 pipelines for marginal water. 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 Inc., with approval from the Israel Water Authority (IWA).

Demand functions for the urban sector are based on estimates by Bar-Shira et al. (2007). Using updated data on average incomes and water prices and water consumption, we calculated the demand elasticity for each urban-demand zone specified in the topology, and calibrated the MYWAS's constant-elasticity urban-demand function accordingly (see Appendix). For the agricultural zones, we calibrated a demand function that incorporates the different water qualities supplied to the region. The function represents the value of the marginal product of freshwater, where non-freshwater sources (treated wastewater and brackish water) have factors applied to convert to units of "freshwater equivalents." The conversion factors are calibrated based on the administrative conversion factors applied by the IWA and the Israeli Ministry of Agriculture and Rural Development (MOAG) to convert freshwater quotas to quotas of non-freshwater sources (see Appendix).

To treat endogenously the extension of infrastructure throughout long-run simulations, we specified the level of investment required to extend the capacity of each infrastructure element, the lifetime of each element, and the interest rate. We assume that investments in expansions of infrastructural capacities increase linearly with increasing capacity.

The model is run so as to maximize the present value of net social welfare over a specified time period.[1] 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 WEAP into GAMS, runs the optimization process and finally, introduces the optimization results back into WEAP.

[1] 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.

III. Simulations

As already mentioned, to study the three topics, we apply MYWAS under four scenarios. The first, termed Baseline (BL), reflects the current status of TWW in Israel, where TWW is used for irrigation and there is no TWW desalination. In the second scenario, termed No Wastewater (NW), we assume that TWW is not allowed for irrigation; by comparing this set-up to the BL scenario, we evaluate the welfare contribution of TWW reuse. In the third scenario, called Mandatory Desalination (MD), we allow irrigation by TWW provided that it is desalinated beforehand so as to prevent long-term damage to soil properties and groundwater quality; comparison to the BL scenario enables evaluation of these damages. Under the fourth scenario, termed Optional Desalination (OD), TWW is desalinated only if the benefits associated with increasing the freshwater amounts available for agricultural irrigation exceed the desalination costs.

The four scenarios (BL, NW, MD and OD) are run by MYWAS for 3 decades. In all cases, the demand functions for urban use shift to the right according to an annual population growth rate of 1.8% (CBS, 2014). We run the scenarios under the average statewide annual recharge of 1,200 x 10^6 m^3/year throughout the entire simulation period. In addition, we assess the sensitivity of the results to the natural recharge of aquifers by incorporating the lowest recharge level ever recorded in Israel—750 x 10^6 m^3/year—throughout the entire period.

The optimal time paths of agricultural water consumption and water supply are presented, respectively, in Figures 1 and 2.[2] We describe the main findings, starting with average recharge (charts a, c, e and g in Figs. 1 and 2).

Water use under the BL scenario is characterized by a steady increase in freshwater consumption in the urban sector (not shown) due to population growth, thereby producing larger amounts of TWW that is used by the agricultural sector (Fig. 1e), which in turn reduces freshwater consumption by the agricultural sector (Fig. 1c). Consider the NW scenario. In comparison to BL, the consumption of TWW under the NW scenario is (by constraint) zeroed (Fig. 1e), that of brackish water is stable (Fig. 1g), and agricultural consumption of freshwater increases significantly (Fig. 1c), but not to the extent that it compensates for the absence of TWW; hence, the total consumption of irrigation water is considerably reduced (Fig 1a). Fig. 2c indicates that forbidding wastewater recycling under the NW scenario entails additional seawater desalination compared to BL, while generally keeping groundwater extraction (Fig. 2a) and brackish water desalination (Fig. 2e) unchanged.

The mandatory desalination (MD) scenario is similar to the NW case in the sense that irrigation with TWW is forbidden (Fig. 1e), but here TWW desalination constitutes an alternative to seawater desalination; nevertheless, the desalinated TWW can only be used for agricultural irrigation. Consequently, seawater desalination decreases slightly

[2] Not shown is freshwater consumption by the urban and industrial sectors, which exhibits relatively minor differences between the four scenarios.

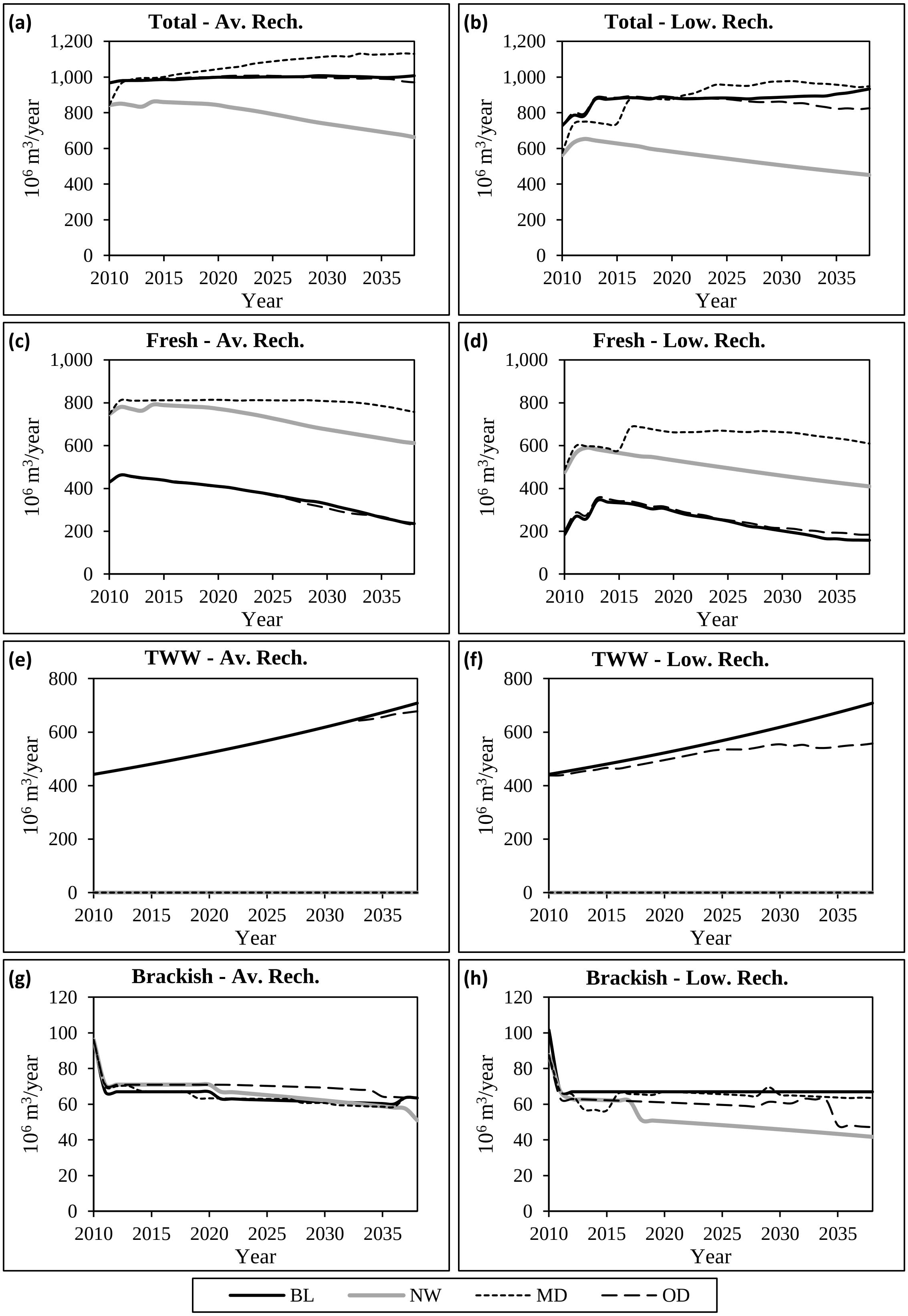

Figure 1 – Statewide agricultural water consumption

compared to BL (Fig. 2c), but since TWW desalination increases with time (Fig. 2g), the overall irrigation amounts grow and exceed those of BL during most of the simulated period (Fig. 1a).

The optional desalination alternative (OD) appears to be almost the same as BL, indicating that TWW-desalination costs overwhelm the associated agricultural benefits. This prevails only under the case of average recharge, whereas desalination becomes optimal when recharge decreases to the lowest recorded level (Fig. 2h), even though only a small fraction of the TWW is desalinated—in only 3 out of the 19 wastewater-treatment plants, all of which are located in northern Israel.

Other impacts of the lowest-recharge cases compared to the average-recharge ones are a considerable reduction in groundwater extraction (Fig. 2b versus 2a), and increases in desalination of seawater (Fig. 2d versus 2c) and of brackish water for agricultural use under the NW scenario (Fig. 2f versus 2e).

If policy-makers would opt to induce the simulated optimal solutions by using water prices, then tariffs should reflect the consumers' marginal utility (in monetary terms) for the urban sector, 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. These optimal prices should be both region-specific and updated on an annual basis. Figure 3 presents statewide time paths of weighted-average urban-water marginal utilities, VMPs of agricultural irrigation water, and shadow values of minimal groundwater-stock constraints, the latter reflecting levels of aquifer-water scarcity.

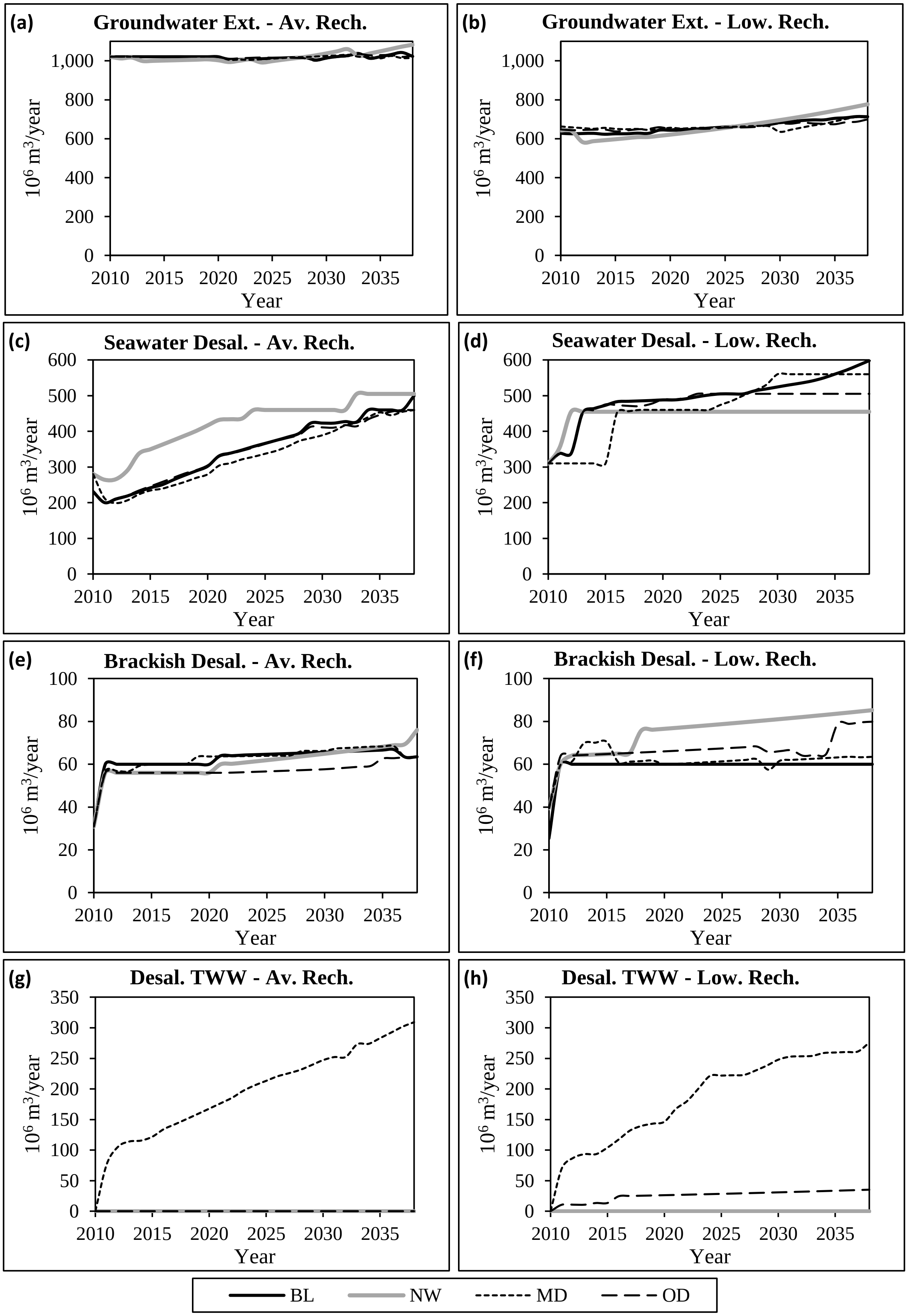

Figure 2 – Statewide water supply

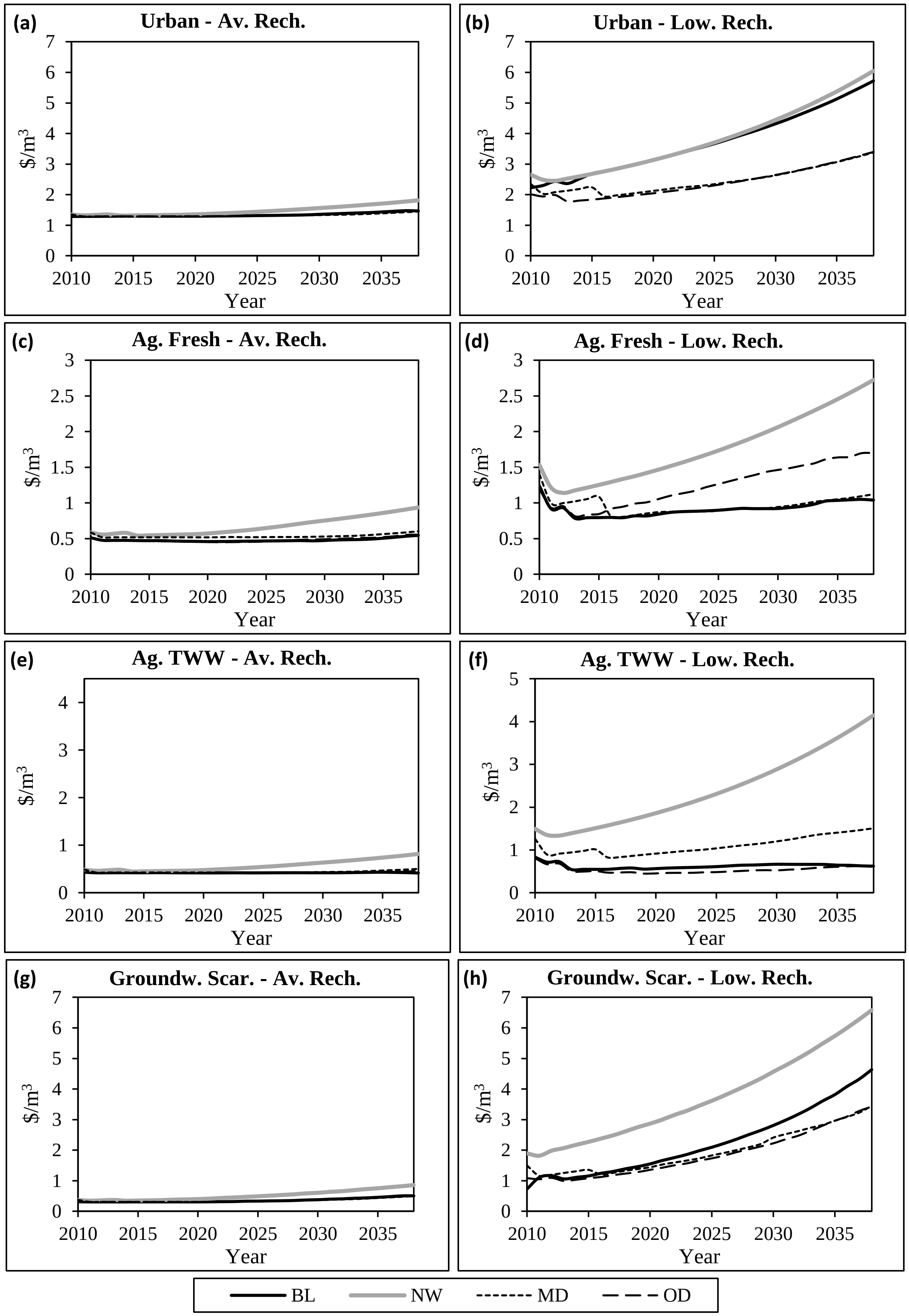

Figure 3 – Statewide average marginal utility (a and b), agricultural value of marginal product (c, d, e, and f) and groundwater scarcity shadow values (g and h).

The differences between the four TWW scenarios under average natural recharge are minor (Fig. 3a, c, e and g). On the other hand, there are significant differences between the scenarios when recharge is low (Fig. 3b, d, f and h), and these gaps increase with time as the demand for urban freshwater grows. Considering the BL scenario, marginal utilities in the urban sector increase dramatically (Fig. 3b). The NW scenario exhibits a trend similar to that of BL in the urban sector, and differs in the others. The relatively lower total consumption of agricultural water under the NW scenario (Fig. 1b) implies higher agricultural VMPs of all water sources, including freshwater (Fig. 3d), TWW (Fig. 3f), and brackish water (not shown). Groundwater scarcity increases as well (Fig. 3h), and is higher relative to the other three scenarios in which TWW use is allowed, with or without desalination.

Comparing the TWW-desalination scenarios (MD and OD) to the BL scenario reveals a considerable impact on the marginal utility of urban consumers. Apparently, even the small amount of desalinated TWW under the OD scenario (Fig. 1h) releases enough freshwater for urban use so as to induce a large reduction in marginal utility. Yet, in the agricultural sector, freshwater consumption under the OD scenario is considerably lower than that under MD (Fig. 1d); therefore, freshwater's agricultural VMP is higher in the OD case (Fig. 3d).

What are the welfare implications of the TWW policies and climate conditions? Table 1 presents various differences in the objective functions obtained under the four scenarios. The upper section of the table presents the impacts of the three TWW policies versus BL. Under average recharge, prohibiting irrigation by wastewater (the NW scenario) reduces welfare by more than 2 billion USD over the simulated period; this evaluated contribution of wastewater reuse is more than doubled under the lowest-recharge case. 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.

Table 1 – Welfare Analyses (Present Values, 5% Discount Rate, Million USD)

	Base-Line (BL)	**No Wastewater for irrigation (NW)**	**Mandatory treated-wastewater Desalination (MD)**	**Optional treated-wastewater Desalination (OD)**
Difference versus the BL scenario:				
Average recharge	0	-2,145	-1,789	41
Lowest recharge	0	-4,765	-578	1,586
Lowest recharge versus average recharge	-6,196	-8,815	-4,985	-4,650

If irrigation by TWW is allowed conditional on desalination, welfare decreases by 1.79 and 0.58 billion USD under average and lowest 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.

According to the OD scenario, the net benefits associated with desalinating TWW for agricultural use emerge only under the case of low natural recharge, and amount to 1.59 billion USD. This should be an underestimated evaluation, as it ignores an expected rise in the demand for irrigation water under drought conditions.

The bottom row of Table 1 presents welfare effects stemming from the lowest natural recharge. We evaluate a damage of more than 6 billion USD under the BL scenario. This effect is highly sensitive to TWW policies, and can increase by about 40% if TWW reuse is totally forbidden (NW), or decrease by about 20% under the two TWW-desalination scenarios.

IV. Concluding Remarks

Wastewater reuse is constantly increasing worldwide, and this trend is expected to continue. In California, for instance, reuse of TWW has increased from approximately 325,000 acre-feet (ac-ft) in 1989 to over 670,000 ac-ft in 2009, and estimates suggest a potential addition of 1.5 million ac-ft (NWRI, 2012). Indeed, California has classified reuse as one of the major means to reduce its water use. This paper assesses policies for agricultural reuse of TWW from a statewide long-term perspective, while taking into account hydrologic, agronomic, engineering, climatic and economic aspects. However, our study 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 urban, industrial and agricultural sectors, as well as on the salinity of the TWW. Consequently, our study assigns credits to desalination through the enlargement of freshwater quantity only, but ignores the associated quality improvements.

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.

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References

1. Bar-Shira, Z., Kislev, Y. and Cohen, N. 2007. The demand for water in the municipalities. Economic Quarterly 54: 179–203.
2. Booker, J.F., Howitt, R.E., Michelsen, A.M. and Young, R.A. 2012. Economics and the modeling of water resource and policies. Natural Resource Modeling 25(1): 168–218.
3. Bresler, B., McNeal, B.L. and Carter, D.L. 1982. Saline and Sodic Soils: Principles, Dynamics, Modeling. Advanced Series in Agricultural Sciences (Vol. 10). Springer-Verlag, New York.
4. CBS (Israeli Central Bureau of Statistics) 2014. Statistical Abstract of Israel 2014, No. 65. The Israeli Central Bureau of Statistics, Jerusalem, Israel.
5. Dawson, C.J. and Hilton, J. 2011. Fertilizer availability in a resource-limited world: production and recycling of nitrogen and phosphorus. Food Policy 36(S1): S14–S22.
6. Dinar, A., Letey, J. and Vaux H.J. 1986. Optimal ratios of saline and nonsaline irrigation waters for crop production. Soil Science Society of America Journal 50: 440–443.
7. Draper, A.J., Jenkins, M.W., Kirby, K.W., Lund, J.R. and Howitt, R.E. 2003. Economic-engineering optimization for California water management. Journal of Water Resources Planning and Management June: 155–164.
8. Feigin, A., Ravina, I. and Shalhevet, J. 1991. Irrigation with sewage effluent. In Advanced Series in Agricultural Sciences (Vol. 17). Springer-Verlag, Berlin.
9. Feinerman, E. Plessner, Y. and Eshel, D.M. 2001. Recycled Effluent : Should the Polluter Pay ? American Journal of Agricultural Economics 83(4):958–971.

10. Feinerman, E. and Tsur, Y. 2014. Perennial crops under stochastic water supply. Agricultural Economics 45(6): 757–766.
11. Fisher, F.M. and Huber-Lee, A.T. 2011. The value of water: optimizing models for sustainable management, infrastructure planning, and conflict resolution. Desalination and Water Treatment 31: 1–23.
12. Fisher, F.M., Huber-Lee, A.T., Arlosoroff, S., Eckstein, Z., Haddadin, M., Hamati, S.G., Jarrar, A., Jayyousi, A., Shamir U. and Wesseling, H. (with a special contribution by Salman, A.Z. and Al-Karablieh, E.K.) 2005. Liquid Assets: An Economic Approach for Water Management and Conflict Resolution in the Middle East and Beyond. RFF Press, Washington DC.
13. Goldstein, M., Shenker, M. and Chefetz, B. 2014. Insights into the uptake processes of wastewater-borne pharmaceuticals by vegetables. Environmental Science and Technology 48: 5593–5600.
14. Hanjra, M.A., Blackwell, J., Carr, G., Zhang, F. and Jackson, T.M. 2012. Wastewater irrigation and environmental health: implications for water governance and public policy. International Journal of Hygiene and Environmental Health 215(3): 255–269.
15. Harou, J.J., Pulido-Velazquez, M., Rosenberg, D.E., Medellín-Azuara, J., Lund, J.R. and Howitt, R.E. 2009. Hydro-economic models: concepts, design, applications, and future prospects. Journal of Hydrology 375(3-4): 627–643.
16. Haruvy, N., Shalhevet, S. and Bachmat, Y. 2008. A model for integrated water resources management in water-scarce regions: irrigation with wastewater combined with desalination processes. International Journal of Water 4(1): 25–40.
17. Iglesias, R., Ortega, E., Batanero, G. and Quintas, L. 2010. Water reuse in Spain: data overview and costs estimation of suitable treatment trains. Desalination 263: 1–10.
18. IWA (Israeli Water Authority) 2012. Long-term National Master Plan for the Water Sector: Policy Document, [online] available at: http://www.water.gov.il/Hebrew/Planning-and-Development/Planning/MasterPlan/DocLib4/MasterPlan-en-v.4.pdf
19. Jenkins, M.W., Lund, J.R., Howitt, R.E., Draper, A.J., Msangi, S.M., Tanaka, S.K., Ritzema, R.S. and Marques, G.F. 2004. Optimization of California's water supply system: results and insights. Journal of Water Resources Planning and Management August: 271–280.
20. Kan, I., Leizarowitz, A. and Tsur, Y. 2010. Dynamic-spatial management of coastal aquifers. Optimal Control Application and Methods 31: 29–41.
21. Kan, I., Schwabe, K.A. and Knapp, K.C. 2002. Microeconomics of irrigation with saline water. Journal of Agricultural and Resource Economics 27(1): 16–39.
22. Kazmia, A.A., Tyagia, K., Trivedi, R.C. and Kumar, A. 2008. Coliforms removal in full scale activated sludge plants in India. Journal of Environmental Management 87: 415–419.
23. Kislev, Y. 2011. Water pricing in Israel in theory and practice. http://departments.agri.huji.ac.il/economics/teachers/kislev_yoav/Kislev-Acc-6-11-10.pdf.
24. Knapp, K.C. and Baerenklau, K.A. 2006. Ground water quantity and quality management: agricultural production and aquifer salinization over long time scales. Journal of Agricultural and Resource Economics 31(3): 616–641.

25. Krichak, S.O., Breitgand, J.S., Samuels, R. and Alpert, P. 2011. A double-resolution transient RCM climate change simulation experiment for near-coastal eastern zone of the Eastern Mediterranean region. Theoretical and Applied Climatology 103(1–2): 167–195.
26. Lado, M., Bar-Tal, A., Azenkot, A., Assouline, S., Ravina, I., Erner, Y., Fine, P., Dasberg, S. and Ben-Hur, M. 2012. Changes in chemical properties of semiarid soils under long-term secondary treated wastewater irrigation. Soil Science Society of America Journal 76: 1358–1369.
27. Levy, G.J., Fine, P., Goldstain, D., Azenkot, A., Zilberman, A., Chazan, A. and Grinhut, T. 2014. Long term irrigation with treated wastewater (TWW) and soil sodification, Biosystems Engineering (in press), http://dx.doi.org/10.1016/j.biosystemseng.2014.05.004
28. Li, P., Wang, X., Allinson, G., Li, X. and Xiong, X. 2009. Risk assessment of heavy metals in soil previously irrigated with industrial wastewater in Shenyang, China. Journal of Hazardous Materials 161: 516–521.
29. NWRI (National Water Research Institute) 2012. Review of California's Water Recycling Criteria for Agricultural Irrigation. Final Report. Prepared for the California Department of Public Health. Fountain Valley, CA.
30. Qadir, M., Sharma, B.R., Bruggeman, A., Choukr-Allah, R. and Karajeh, F. 2007. Non-conventional water resources and opportunities for water augmentation to achieve food security in water scarce countries. Agricultural Water Management 87: 2–22.
31. Ratola, N., Cincinelli, A., Alves, A. and Katsoyiannis, A. 2012. Occurrence of organic microcontaminants in the wastewater treatment process. A mini review. Journal of Hazardous Materials 239: 1–18.
32. Reznik, A., Huber-Lee, A., Joyce, B., Feinerman, E., Finkelshtain, I., Kan, I. and Fisher, F. 2014. Understanding the Economics of Water in Israel: the Multi-Year Water Allocation System Model. The Center for Agricultural Economic Research, Discussion Paper No. 11.14. http://departments.agri.huji.ac.il/economics/en/publications/discussion_papers/2014
33. Rosenberg, D.E., Howitt, R.E. and Lund J.R. 2008. Water management with water conservation, infrastructure expansions, and source variability in Jordan. Water Resources Research 44: W11402.
34. Rutkowski, T., Raschid-Sally, L. and Buechler, S. 2007. Wastewater irrigation in the developing world – two case studies from the Kathmandu Valley in Nepal. Agricultural Water Management 88: 83–91.
35. Sato, T., Qadir, M., Yamamoto, S., Endo, T. and Zahoor, A. 2013. Global, regional, and country level need for data on wastewater generation, treatment, and use. Agricultural Water Management 130: 1−13.
36. Tsur, Y. and Zemel, A. 2004. Endangered aquifers: groundwater management under threats of catastrophic events. Water Resources Research 40(6): 1–10.
37. Xu, P., Valette, F., Brissaud, F., Fazio, A. and Lazarova, V. 2001. Technical-economic modelling of integrated water management: wastewater reuse in a French island. Water Science and Technology 43(10): 67–74.

Appendix – Formal Description of the MYWAS model

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 MYWAS.

MYWAS is a discrete-time dynamic optimization model. Let us denote by the subscript indices *a* ($a = 1,...,A$) an agricultural region, *u* ($u = 1,...,U$) an urban region, *f* ($f = 1,...,F$) a source for freshwater, *b* ($b = 1,...,B$) a source for brackish water, and *h* ($h = 1,...,H$) a wastewater-treatment plant. We use the superscripts ϕ, β, *l* and *w* to indicate freshwater, brackish water, sewage, and treated wastewater, respectively. Time is denoted by *t*, $t = 1,...,T$, where *T* is the optimization planning horizon. Water quantities consumed in demand regions are represented by *Q*; for instance, Q_{ut}^{ϕ} is the amount of freshwater consumed in region *u* during time *t*. *E* stands for extractions from sources; for example, E_{ft}^{ϕ} and E_{bt}^{β} are, respectively, the amounts of freshwater and brackish water extracted at time *t* from sources *f* and *b*, and E_{ht}^{w} and E_{ht}^{ϕ} are the quantities of treated, and treated-and-desalinated wastewater produced by plant *h* during *t*, respectively. Water transfers between spatial points are denoted by *G*; thus, G_{bat}^{β} 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_{bt}^{β} is the increase in the capacity of extraction of brackish water from source *b* during time *t*, and GN_{uht}^{l} 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, $EM_{bt}^{\beta} = \sum_{\tau=1}^{t-1} EN_{b\tau}^{\beta}$ 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_{uh0}^{l} is the capacity of sewage-water transfer from urban region *u* to wastewater-treatment plant *h* at the beginning of the planning problem. Accordingly, $GM_{uh0}^{l} + GM_{uht}^{l}$ is the overall transfer capacity at this link; hence, $G_{uht}^{l} \le GM_{uh0}^{l} + GM_{uht}^{l}$.

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}, and it is physically restricted by $\bar{V}_f$ from above and by $\underline{V}_f$ from below. We also introduce k_{fT} 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.[3] Alternatively, an end-point minimum constraint $\underline{V}_{fT}$, $\bar{V}_f \geq \underline{V}_{fT} \geq \underline{V}_f$, 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: $V_{ft} = V_{f0} + \sum_{\tau=1}^{t} R_{f\tau} - \sum_{\tau=1}^{t-1} \left(E_{f\tau}^{\phi} + L_{f\tau} \right)$ where V_{f0} is the initial extractable content of the source, R_{ft} is the natural recharge during time t, and L_{ft} is the spillover from the source during time t, where by definition $L_{ft} = \max\left(0, V_{ft} - \bar{V}_f\right)$. A minimum overflow level $\underline{L}_{ft}$ may be assigned to each freshwater source (e.g., to reflect "water for nature" regulations); the vectors $\mathbf{L}$ and $\underline{\mathbf{L}}$ are defined accordingly.

Under the above specifications, desalination plants can be considered as freshwater sources with annual stock of zero i.e. $V_{ft} = 0$. However, for notational simplicity we set $V_{ft} \equiv EM_{f0}^{\phi} + EM_{ft}^{\phi}$, such that only the plants' capacity constraints can be effective. Also note that in brackish-water sources $R_{bt} = 0$, and therefore $L_{bt} = 0$.

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_{hat}^{w} is the per-unit transfer cost of treated wastewater from wastewater-treatment plant h to agricultural region a at time t, and $C_{ft}^{\phi}\left(V_{ft}\right)$ stands for the cost of extracting one unit of

[3] We assume this value to be constant across regions and equal to the marginal cost of water desalination at \$0.4/CM.

freshwater from source *f* at time *t*; in the latter, the cost depends on the source's stock V_{ft}, where lower stocks entail larger extraction costs (note that for wastewater-treatment plants, stock is irrelevant).

Costs of capacity increase, which mostly comprise capital investments, are represented by *S*; for example, S_{ht}^{w} is the per-time-unit cost of increasing the capacity of wastewater-treatment plant *h* by one unit at time *t*. Likewise, S_{fat}^{ϕ} is the per-time-unit cost associated with increasing by one unit the capacity of freshwater transfer from source *f* to agricultural region *a* 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 $\left(\mu+1\right)^{-1}\upsilon_{ut}\left(Q_{ut}^{\phi}\right)^{\mu+1}$ is the total willingness to pay for the amount of freshwater consumed in region *u* at time *t*, Q_{ut}^{ϕ} [i.e., the area below the inverse-demand function $\upsilon_{ut}\cdot\left(Q_{ut}^{\phi}\right)^{\mu}$], where υ_{ut} and μ are parameters, and $1/\mu$ is the urban-demand elasticity.

To represent benefits in the agricultural sector, we apply the function $\left(\eta+1\right)^{-1}\alpha_{at}\cdot\left(Q_{at}^{\phi}+\delta Q_{at}^{w}+\gamma Q_{at}^{\beta}\right)^{\eta+1}$, which represents the value of production (*VP*) associated with the consumption of freshwater, treated wastewater and brackish water (see Feinerman et al., 2001). In this case, if either $Q_{a1}^{w} > 0$ or $Q_{a1}^{\beta} > 0$, the demand elasticity does not equal $1/\eta$, and therefore it is not constant, and depends on the overall water consumption from the various water types. The parameters γ and δ translate treated wastewater and brackish water to freshwater in terms of their value of marginal product (*VMP*). That is, the freshwater *VMP* equals $\alpha \cdot \left(Q^{\phi} + \delta Q^{w} + \gamma Q^{\beta}\right)^{\eta}$, and the treated-wastewater *VMP* is $\delta\alpha \cdot \left(Q^{\phi} + \delta Q^{w} + \gamma Q^{\beta}\right)^{\eta}$; hence, $\delta = \dfrac{vmp^{w}}{vmp^{\phi}}$. In other words, δ is the increase in the regional consumption of freshwater required to maintain the regional *VP* 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, **L**. We let ψ_{ft} 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 υ_{ut} and α_{at}, etc.), the problem solved by MYWAS is

$$\max_{\mathbf{Q,E,G,EN,GN}} \sum_{t=1}^{T} \frac{1}{(1+r)^t} \left[\begin{array}{l} \sum_{u=1}^{U} \frac{\upsilon_{ut} \cdot \left(Q_{ut}^{\phi}\right)^{\mu+1}}{\mu+1} + \sum_{a=1}^{A} \frac{\alpha_{at} \cdot \left(Q_{at}^{\phi} + \delta Q_{at}^{w} + \gamma Q_{at}^{\beta}\right)^{\eta+1}}{\eta+1} + \psi_{ft} L_{ft} \\ - \sum_{f=1}^{F}\sum_{u=1}^{U} G_{fut}^{\phi} \cdot C_{fut}^{\phi} - \sum_{f=1}^{F}\sum_{a=1}^{A} G_{fat}^{\phi} \cdot C_{fat}^{\phi} - \sum_{f=1}^{F} E_{ft}^{\phi} \cdot C_{ft}^{\phi}\left(V_{ft}\right) - \sum_{b=1}^{B}\sum_{a=1}^{A} G_{bat}^{\beta} \cdot C_{bat}^{\beta} \\ - \sum_{b=1}^{B} E_{bt}^{\beta} \cdot C_{bt}^{\beta}\left(V_{bt}\right) - \sum_{u=1}^{U}\sum_{h=1}^{H} G_{uht}^{l} \cdot C_{uht}^{l} - \sum_{h=1}^{H}\sum_{a=1}^{A}\left(G_{hat}^{w} \cdot C_{hat}^{w} + G_{hat}^{\phi} \cdot C_{hat}^{\phi}\right) \\ - \sum_{h=1}^{H}\left(E_{ht}^{w} \cdot C_{ht}^{w} + E_{ht}^{\phi} \cdot C_{ht}^{\phi}\right) + \sum_{f}^{F} k_{fT} \cdot V_{fT} \end{array} \right]$$

$$- \sum_{t=1}^{T-1} \frac{1}{(1+r)^t} \left[\begin{array}{l} \sum_{f=1}^{F}\sum_{u=1}^{U} GM_{fut}^{\phi} \cdot S_{fu}^{\phi} + \sum_{f=1}^{F}\sum_{a=1}^{A} GM_{fat}^{\phi} \cdot S_{fa}^{\phi} + \sum_{f=1}^{F} EM_{ft}^{\phi} \cdot S_{f}^{\phi} + \sum_{b=1}^{B}\sum_{a=1}^{A} GM_{bat}^{\beta} \cdot S_{ba}^{\beta} \\ + \sum_{b=1}^{B} EM_{bt}^{\beta} \cdot S_{b}^{\beta} + \sum_{u=1}^{U}\sum_{h=1}^{H} GM_{uht}^{l} \cdot S_{uh}^{l} + \sum_{h=1}^{H}\sum_{a=1}^{A}\left(GM_{hat}^{w} \cdot S_{ha}^{w} + GM_{hat}^{\phi} \cdot S_{ha}^{\phi}\right) \\ \sum_{h=1}^{H}\left(EM_{ht}^{w} \cdot S_{h}^{w} + EM_{ht}^{\phi} \cdot S_{h}^{\phi}\right) \end{array} \right]$$

Subject to:

(1) $Q_{at}^{\phi} \le \sum_{f=1}^{F} G_{fat}^{\phi} + \sum_{h=1}^{H} G_{hat}^{\phi}\ \forall a,t;$ (2) $Q_{ut}^{\phi} \le \sum_{f=1}^{F} G_{fut}^{\phi}\ \forall u,t;$ (3) $Q_{at}^{\beta} \le \sum_{b=1}^{B} G_{bat}^{\beta}\ \forall a,t;$

(4) $Q_{at}^{w} \le \sum_{h=1}^{H} G_{hat}^{w}\ \forall a,t;$ (5) $\sum_{u=1}^{U} G_{fut}^{\phi} + \sum_{a=1}^{A} G_{fat}^{\phi} \le E_{ft}^{\phi}\ \forall f,t;$ (6) $\sum_{b=1}^{B} G_{bat}^{\beta} \le E_{bt}^{\beta}\ \forall b,t;$

(7) $\sum_{h=1}^{H} G_{uht}^{l} \le \theta Q_{ut}^{\phi}\ \forall u,t;$ (8) $E_{ht}^{w} \le \rho \sum_{u=1}^{U} G_{uht}^{l}\ \forall h,t;$ (9) $\sum_{a=1}^{A} G_{hat}^{w} \le E_{ht}^{w} - E_{ht}^{\phi}\ \forall h,t;$

(10) $\sum_{a=1}^{A} G_{hat}^{\phi} \le E_{ht}^{\phi}\ \forall h,t;$ (11) $E_{ht}^{\phi} \le E_{ht}^{w}\ \forall h,t;$ (12) $GM_{fat}^{\phi} = \sum_{\tau=1}^{t-1} GN_{fat}^{\phi}\ \forall f,a,t;$

(13) $GM_{fut}^{\phi} = \sum_{\tau=1}^{t-1} GN_{fut}^{\phi}\ \forall f,u,t;$ (14) $GM_{bat}^{\beta} = \sum_{\tau=1}^{t-1} GN_{bat}^{\beta}\ \forall b,a,t;$

(15) $GM_{uht}^{l} = \sum_{\tau=1}^{t-1} GN_{uht}^{l}\ \forall u,h,t;$ (16) $GM_{hat}^{w} = \sum_{\tau=1}^{t-1} GN_{hat}^{w}\ \forall h,a,t;$

(17) $GM_{hat}^{\phi} = \sum_{\tau=1}^{t-1} GN_{hat}^{\phi}\ \forall h,a,t;$ (18) $EM_{ft}^{\phi} = \sum_{\tau=1}^{t-1} EN_{ft}^{\phi}\ \forall f,t;$ (19) $EM_{bt}^{\beta} = \sum_{\tau=1}^{t-1} EN_{bt}^{\beta}\ \forall b,t;$

(20) $EM_{ht}^{w} = \sum_{\tau=1}^{t-1} EN_{ht}^{w}\ \forall h,t;$ (21) $EM_{ht}^{\phi} = \sum_{\tau=1}^{t-1} EN_{ht}^{\phi}\ \forall h,t;$

(22) $G_{fut}^{\phi} \le GM_{fu0}^{\phi} + GM_{fut}^{\phi}\ \forall f,u,t;$ (23) $G_{fat}^{\phi} \le GM_{fa0}^{\phi} + GM_{fat}^{\phi}\ \forall f,a,t;$

$$(24)\ G^{\beta}_{bat} \le GM^{\beta}_{ba0} + GM^{\beta}_{bat}\ \ \forall b,a,t;\ (25)\ G^{l}_{uht} \le GM^{l}_{uh0} + GM^{l}_{uht}\ \ \forall u,h,t;$$

$$(26)\ G^{w}_{hat} \le GM^{w}_{ha0} + GM^{w}_{hat}\ \forall h,a,t;\ (27)\ G^{\phi}_{hat} \le GM^{\phi}_{ha0} + GM^{\phi}_{hat}\ \forall h,a,t;$$

$$(28)\ E^{\phi}_{ft} \le EM^{\phi}_{f0} + EM^{\phi}_{ft}\ \forall f,t;\ (29)\ E^{\beta}_{bt} \le EM^{\beta}_{b0} + EM^{\beta}_{bt}\ \forall b,t;$$

$$(30)\ E^{\phi}_{ht} \le EM^{\phi}_{h0} + EM^{\phi}_{ht}\ \forall h,t;\ (31)\ E^{w}_{ht} \le EM^{w}_{h0} + EM^{w}_{ht}\ \forall h,t;$$

$$(32)\ EM^{\phi}_{h0} + EM^{\phi}_{ht} \le EM^{w}_{h0} + EM^{w}_{ht}\ \forall h,t;\ (33)\ V_{ft} = V_{f0} + \sum_{\tau=1}^{t} R_{ft} - \sum_{\tau=1}^{t-1}\left(E^{\phi}_{ft} - L_{ft}\right)\forall f,t;$$

$$(34)\ V_{bt} = V_{b0} - \sum_{\tau=1}^{t} E^{\beta}_{bt}\ \forall b,t;\ (35)\ E^{\phi}_{ft} \le V_{ft}\ \ \forall f,t; (36)\ E^{\beta}_{bt} \le V_{bt}\ \forall b,t;$$

$$(37)\ V_{fT} \ge \underline{V}_{fT}\ \ \forall f;\ (38)\ V_{bT} \ge \underline{V}_{bT}\ \forall b; (39)\ L_{ft} = \max\left(0, V_{ft} - \overline{V}_{f}\right)\forall f,t;$$

$$(40)\ \mathbf{Q, E, G, EN, GN} \ge 0, \mathbf{L} \ge \underline{\mathbf{L}}$$

The objective function includes two aggregative elements: one is associated with the sets of water variables **Q**, **E**, **G**, covering the period $t = 1,...,T$; the other refers to the sets of capacity expansions **EN** and **GN**, ranging from $t = 1$ to $t = T - 1$.

The set of constraints (1) through (4) ensures that, for the water type (ϕ, β 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 θ 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 ρ stands for the wastewater/sewage-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 production that has not been desalinated. Equation (10) limits the deliveries of desalinated wastewater to agricultural districts to the amount produced at each wastewater-desalination plant, and equation (11) ensures that desalination of treated wastewater will not exceed wastewater production in each plant. Equations (12) through (21) define the cumulative capacity state variables, and the limits (22) through (31) restrain the extractions and transfers by their corresponding capacities. Equation (32) confines the capacity of wastewater desalination to not exceed its corresponding wastewater-treatment plant. Equations (33) and (34) define the extractable stocks in freshwater and brackish-water sources, respectively, where (35) and (36) use these stocks as upper limits to the corresponding extractions, and (37) and (38) impose endpoint minimum stocks. Equation (39) defines the spillover from freshwater sources, and (40) are non-negativity and minimal spillover constraints.

<u>Calibrating Demand Functions</u>

For the urban inverse-demand functions, $\upsilon_{ut} \cdot \left(Q_{ut}^{\phi}\right)^{\mu}$, the elasticity parameter μ is taken from Bar-Shira et al., (2005). The υ_{ut} parameter is calibrated for each region based on the regional consumption and the water price observed in the base-period ($t=1$): $\upsilon_{u1} = p_{u1}^{\phi} \cdot \left(Q_{u1}^{\phi}\right)^{-\mu}$, where p_{u1}^{ϕ} is the corresponding freshwater price for urban use. Then, υ_{ut} for each time *t* may change due to population growth and other external factors.

Regarding the agricultural inverse-demand functions $\alpha_{at} \cdot \left(Q_{at}^{\phi} + \delta Q_{at}^{w} + \gamma Q_{at}^{\beta}\right)^{\eta}$, the parameters δ and γ 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 α_{at} is calibrated for every region *a* based on the consumption level and the agricultural freshwater price observed in the base period, p_{a1}^{ϕ}, such that $\alpha_{a1} = p_{a1}^{\phi} \cdot \left(Q_{a1}^{\phi} + \delta Q_{a1}^{w} + \gamma Q_{a1}^{\beta} \right)^{-\eta}$. In later periods throughout the simulations, α_{at} may change due to external effects.