האוניברסיטה העברית בירושלים
THE HEBREW UNIVERSITY OF JERUSALEM

המרכז למחקר בכלכלה חקלאית
The Center for Agricultural Economic Research

המחלקה לכלכלת סביבה וניהול
Department of Environmental Economics and Management

Discussion Paper No. 2.19

The Nexus of Irrigation-Water Salinity, Agricultural Policy and Long-Run Water Management: Lessons from the Case of Israel

By

Iddo Kan, Yehuda Slater, Ami Reznik, and Israel Finkelshtain

P.O. Box 12, Rehovot 76100, Israel

ת.ד. 12, רחובות 76100

הפקולטה לחקלאות | המחלקה לכלכלת סביבה וניהול
רחובות ת.ד 12
טל': 08-9489230 | פקס: 08-9466267 | miriguy@savion.huji.ac.il

The Nexus of Irrigation-Water Salinity, Agricultural Policy and Long-Run Water Management: Lessons from the Case of Israel

Iddo Kan[1,2], Yehuda Slater[1], Ami Reznik[3] and Israel Finkelshtain[1,2]

[1] Department of Environmental Economics and Management; The Robert H. Smith Faculty of Agriculture, Food and Environment; The Hebrew University of Jerusalem; P.O.B. 12, Rehovot 761001, Israel

[2] The Center for Agricultural Economics Research; P.O.B. 12, Rehovot 761001, Israel

[3] School of Public Policy; University of California, Riverside

The Nexus of Irrigation-Water Salinity, Agricultural Policy and Long-Run Water Management: Lessons from the Case of Israel

Abstract

This paper incorporates the detrimental agronomic effects of irrigation-water salinity into an empirical economy-wide dynamic hydro-economic model to study the interactions between salinity, agricultural policies and optimal long-run water-allocation policies and water-infrastructure plans. Application to the case of Israel indicates economic viability of large-scale delivery of desalinated water for agricultural irrigation during a 30-year period (2016–2045). We explain this finding by the large share of salinity-sensitive crops in the total irrigation-water consumption and production value of the Israeli vegetative-agriculture sector, which stems from the historical policy to protect local agriculture. On average, the annual damage caused by the presence of salts in Israel's water sources is evaluated at nearly $4,500 per hectare of arable land, comprising deadweight loss in both the water and agricultural economies. Overlooking salinity's agronomic effects in the design of water infrastructures entails an annual welfare loss of nearly $1,360/hectare, and income redistributions wherein the profits of water suppliers increase significantly at the expense of the economic surpluses of urban water users, farmers, and consumers of agricultural products.

Keywords: Irrigation, Salinity, Agriculture, Policy, Water, Economics, Model, Desalination, Natural Resources

JEL Classification: Q15, Q25, Q28

1. Introduction

Salts already affect about a third of the global arable areas, and a fifth of the worldwide irrigated lands (Nellemann et al., 2009; Qadir et al., 2014). Salinization is expected to expand further through a range of processes driven by population growth; these include a rise in irrigation with brackish water as a substitute for the freshwater amounts diverted to domestic use, agricultural reuse of the growing treated wastewater (TWW) volumes discharged from urban areas (Jimenez and Asano, 2008; Qadir et al., 2007), and salinization of freshwater sources due to seawater intrusion and deep percolation of salts from irrigated lands (Assouline et al., 2015). The integrative nature of these large-scale spatially and inter-temporally linked processes implies that salinity damages should be considered intrinsically in the design of agricultural and water policies, and long-run water-infrastructure plans. We develop a dynamic hydro-economic model that characterizes the optimal water-allocation policies, while accounting for irrigation-water salinity effects, and endogenizing water-infrastructure development and the quantities and prices of vegetative products supplied to different agricultural markets.

Given the long construction period of water infrastructures, designers of water-supply systems should foresee scarcities and decide ahead of time which, when, where, and to what extent infrastructural elements should be installed or expanded. While “no model solves all problems” (Darper et al., 2003, p 156), hydro-economic models are an efficient tool for integrating the economic, hydrological, engineering, agronomic and environmental aspects associated with water-infrastructure blueprints (Harou et al., 2009; Booker et al., 2012). However, to date, the incorporation of salinity in such models has been limited. For example, Tanaka et al. (2006), Lavee et al. (2011), Reznik et al. (2016)

and Housh et al. (2011) refer to desalination as a means of resolving water-quantity scarcities, but overlook the benefits of reducing irrigation-water salinity; Howitt et al. (2010) and Welle et al. (2017) employed the van Genuchten and Hoffman (1984) production function to internalize salinity's agronomic impacts into a static analysis of irrigation-water management in California. However, to fully account for irrigation-water salinity, one should allow the model to track the dynamics of water salinity throughout the water-delivery system during the planning horizon, to affect salinity by allowing the establishment of desalination capacities at various nodes in the distribution network, and to comprise the salinity damage caused to agriculture and the environment, as well as the adaptation measures employed by relevant stakeholders. Moreover, the model should enable simulating exogenous changes, such as population growth, and agricultural and environmental policies. The above attributes are incorporated herein in the analysis for the case of Israel.

Israel today could manifest the present and future of developed economies that are encountering increasing water scarcity, such as California, Australia and Spain. Israel's water economy faces three basic difficulties: first, while precipitation is relatively abundant in the north, the population is concentrated in the center, and most of the agricultural lands are located in the south; second, whereas rainfall events occur only in winter, summer is the main irrigation season; third, natural freshwater is scarce, successive drought years are common, and the water scarcity is steadily increasing due to population growth (Kislev, 2011). Israel has confronted these challenges by establishing a complex water-distribution network to connect the various water sources and users, which spans most of the country's regions, in particular based on a national water carrier that essentially turns almost the entire state into a single basin in terms of water management. That is, the network interlinks scarcities across points in time and space, such that water use at a particular place and time may have opportunity costs with respect to usage for various purposes at alternative locales and times (Fisher et al., 2005). In addition, according to the Israeli Water Law (IMNI 1959), the government centrally controls water consumption by administrative prices and quotas.

Severe water scarcity in Israel motivated the exploitation of brackish water and TWW for irrigation, and in recent years, extensive seawater desalination has also been used to supply freshwater to both the urban and agricultural sectors (Burn et al., 2015; Martinez-Alvarez et al., 2016). Figure 1 presents the evolution of water sources and uses (IWA, 2016), population growth and vegetative agricultural production (ICBS, 2017) in Israel during the period 1996–2016. Fresh groundwater extractions declined dramatically from 1,590 million m^3/year in 1996 to only 830 million m^3/year in 2016, and since 2005, have been partly substituted by seawater desalination, which increased gradually to 590 million m^3/year in 2016 (Figure 1a). At the same time, the population increased by 50% (Figure 1c), generating larger amounts of sewage, which was reclaimed (Figure 1a) and reused for irrigation (Figure 1b). Concomitantly, domestic and industrial freshwater consumption increased by 30%, whereas freshwater for irrigation was cut by more than 40%. Nevertheless, with the growing agricultural use of TWW and brackish water (Figure 1b), the total irrigation water did not vary much along that period. The population growth also increased the demand for food and, while the index of local vegetative agricultural production increased by 35% from 1996 until 2010, it stabilized thereafter.

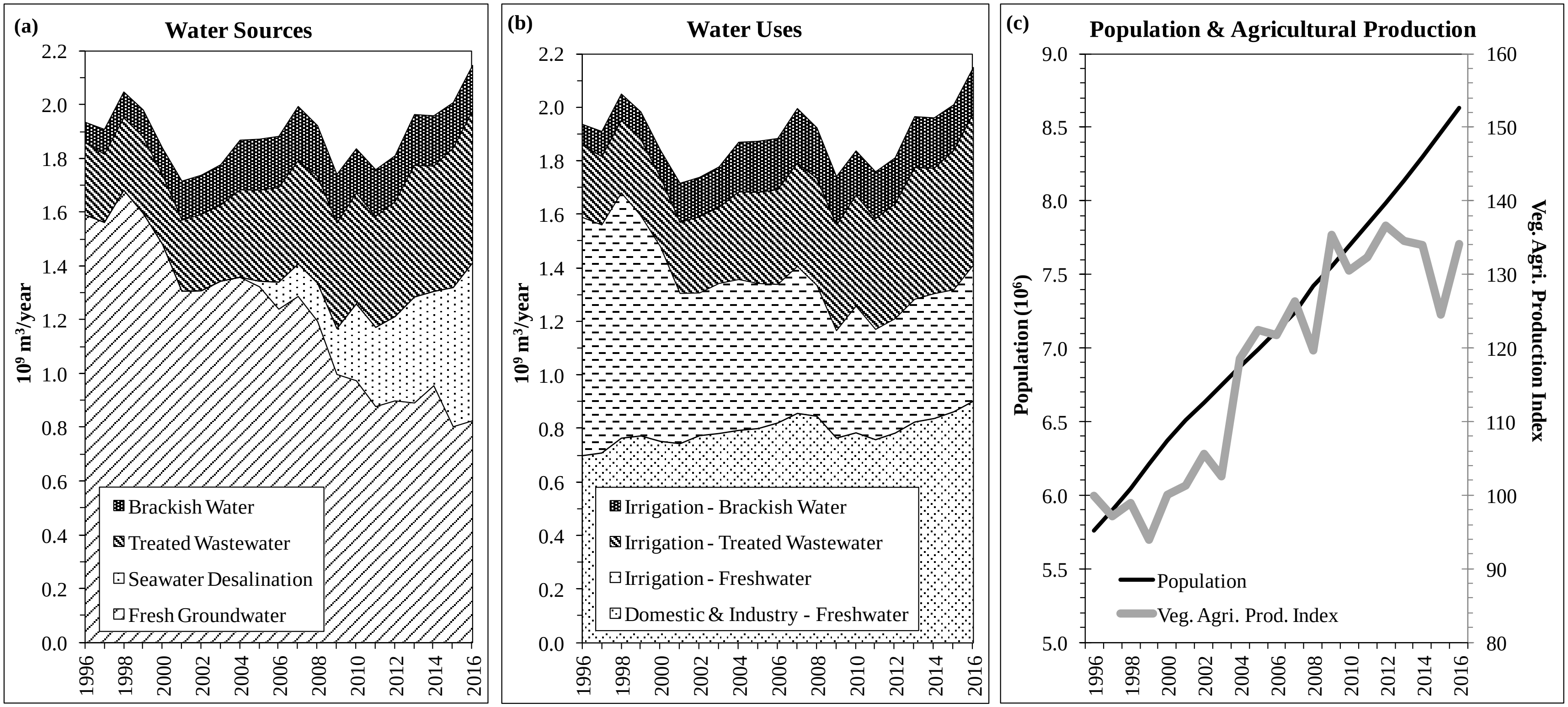

Figure 1. Trends of (a) water sources, (b) water uses and (c) population growth and vegetative-agriculture production in Israel during 1996–2016

The expectation of continued population growth prompts the question of how the Israeli water sector should be further developed. The objective of our analysis is to assess the effect of irrigation-water salinity on an optimal statewide infrastructure plan for Israel for a 30-year period (2016 to 2045), while accounting for potential changes in agricultural protection policies. To this end, we develop a statewide dynamic hydro-economic model that incorporates both water quantity and salinity. Specifically, we adopt and extend data and functional forms from the hydro-economic model MYWAS (Multi-Year Water Allocation System) (Reznik et al., 2017) and the positive mathematical programming (PMP) model VALUE (Vegetative Agriculture Land-Use Economics) (Kan and Rapaport-Rom, 2012). The salient conclusion of our simulated optimal solution is that the negative effect of irrigation-water salinity on agricultural production warrants large-scale desalination of water for irrigation. We attribute this result to the large share of high-value and salinity-sensitive crops in Israel's vegetative agriculture, motivated by Israeli agricultural protection policy.

By simulating a hypothetical scenario of zero salinity in Israel's water sources, we evaluate the statewide welfare loss caused by the presence of salts at about $1,340 million a year, which amounts to nearly $4,468 per hectare of arable land. This estimate, which accounts for the indirect salinity effects on domestic water supply and agricultural output prices, is an order of magnitude larger than the direct agricultural salinity damage of $440/hectare reported by Qadir et al. (2014). We further use the model to assess a case in which the designers of water infrastructures overlook the agronomic salinity effects; this scenario entails a welfare loss of more than $408 million a year compared to the optimal solution, which is about $1,360 per hectare of cultivable land. Moreover, ignoring salinity leads to higher water prices and lower water supply, which in turn increase water suppliers' surpluses at the expense of urban and agricultural water users, as well as of consumers of agricultural products.

The next section develops a general hydro-economic framework that incorporates both water quantity and salinity, and describes the empirical application specific to the Israeli water and agricultural economies. We then present an optimal water-management and infrastructure plan, and elicit the management and welfare implications of irrigation-water salinity by comparing the optimal solution to the case in which agronomic salinity effects are non-existent, or ignored; the sensitivities of the optimal solution to changes in agricultural policies and precipitation are assessed, and the final section concludes.

2. Hydro-Economic Model

In the appendix we present, in general terms, a spatial-dynamic hydro-economic model with multiple periods, regions, water sources with different salinities and availability constraints, extendable water treatments that are spatially interconnected, and may be delivered separately and/or mixed to various water consumers who differ in their demand sensitivity to salinity. Here, to obtain general insights, we explore the management and policy implications of irrigation-water salinity using a theoretical analysis based on a simplified static hydro-economic model. Then, we present the empirical specifications of the comprehensive framework for the case of Israel.

2.1 Theoretical Analysis

Consider a static hydro-economic model, encompassing two water sources with different salinities—groundwater and desalinated seawater—which are mixed and then delivered to two water consumers—urban and agricultural, with the latter's water demand being sensitive to salinity (Figure 2).

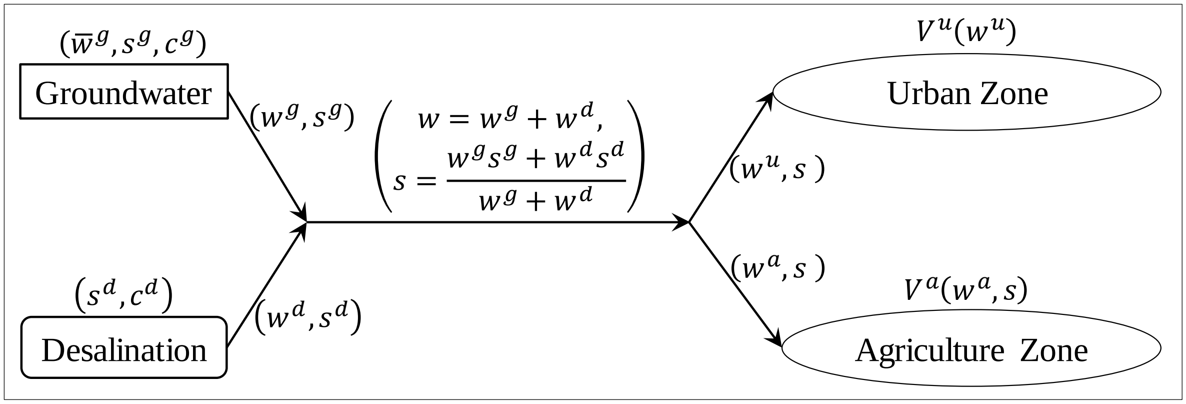

Figure 2. Scheme of a simplified static water-economy model

The groundwater salinity, denoted s^g, exceeds that of the desalinated water, s^d. We assume constant per-water-unit supply costs, where the per-water-unit cost of desalinated water, c^d, exceeds that of the groundwater, c^g. The extracted groundwater quantity, w^g, is limited by the availability constraint $\bar{w}^g$, whereas the delivery of desalinated water, w^d, is unlimited.

The total water supply, $w \equiv w^d + w^g$, is a mixture of the two sources, with average salinity $s = \frac{w^d s^d + w^g s^g}{w}$. The blended water is split such that the urban and agricultural sectors obtain amounts w^u and w^a, respectively, where $w^u + w^a = w$. Let the function $V^u(w^u)$ and $V^a(w^a, s)$ be, respectively, the value of water consumed by agents in the urban and agricultural zones gross of water-purchasing expenses, where water quantity is beneficial to both sectors (i.e., $V_w^u(w^u) > 0$ and $V_w^a(w^a, s) > 0$), and water salinity is harmful only to agriculture ($V_s^u(w^u) = 0$ and $V_s^a(w^a, s) < 0$).

A benevolent government manages water centrally by setting administrative water prices at the consumption points. We consider three scenarios: the first, termed *optimal solution* (OS), presumes that the water authority sets the water prices that maximize the economy's net benefits by solving the problem

$$\max_{w^d, w^g, w^u, w^a, s^d, s^g} V^u(w^u) + V^a(w^a, s) - c^d w^d - c^g w^g$$
$$s.t.\;\; w = w^u + w^a = w^d + w^g, \quad w^g \le \overline{w}^g, \quad s^d \ge \underline{s}^d, \quad s^g \ge \underline{s}^g, \quad s = \frac{w^d s^d + w^g s^g}{w}$$

where $\underline{s}^d$ and $\underline{s}^g$ are the salinity levels of the desalinated and groundwater sources, respectively. By substituting the balance constraint $w^u = w^d + w^g - w^a$ into $V^u(w^u)$, and optimizing with respect to w^a (assuming that the second-order conditions prevail), we get

$$V_w^u(w^u) = V_w^a(w^a, s) \tag{1}$$

implying that a similar optimal-solution administrative water price, denoted p^{OS}, should be set for both consumption sectors: $p^{\mathrm{OS}} = V_w^u(w^u) = V_w^a(w^a, s)$ (note that if urban water use entails the additional wastewater-treatment costs c^t, then, the efficient urban-water price will be higher than that of the agricultural sector by c^t). By employing this result, and optimizing with respect to w^g and w^d, we obtain

$$p^{\mathrm{OS}} = c^d - V_s^a(w^a, s)\frac{w^g(s^d - s^g)}{(w^d + w^g)^2} \tag{2}$$

$$p^{\mathrm{OS}} = c^g + V_s^a(w^a, s)\frac{w^d(s^d - s^g)}{(w^d + w^g)^2} + \lambda^g \tag{3}$$

where λ^g is the shadow price of the groundwater constraint $w^g \le \overline{w}^g$.

Equations 2 and 3 imply $c^d > p^{\mathrm{OS}} > c^g$. The term $V_s^a(w^a, s)\frac{w^g(s^d - s^g)}{(w^d + w^g)^2}$ in Eq. 2 reflects the marginal benefits accrued to the agricultural sector by salinity reduction through desalination, which warrants the supply of a desalinated-water quantity beyond the level that equates p^{OS} to c^d. On the other hand, the term $V_s^a(w^a, s)\frac{w^d(s^g - s^d)}{(w^d + w^g)^2}$ in Eq. 3 represents the marginal damage caused by groundwater due to increased salinity, which renders the optimal groundwater supply lower than that equating p^{OS} to c^g, even under an ineffective groundwater constraint (i.e., $\lambda^g = 0$).

From Eqs. 2 and 3, efficiency requires

$$\frac{V_s^a(w^a,s)}{w}(s^d - s^g) = c^d - (c^g + \lambda^g) \tag{4}$$

Thus, the per-water-unit agricultural benefits associated with reducing salinity from s^g to s^d should be equal to the difference between the per-water-unit supply (plus scarcity) costs of the two sources. Equation 4 also ensures the cost-recovery condition $p^{\mathrm{OS}}w \geq c^d w^d + c^g w^g$, which, based on Eq. 3, requires

$$\frac{V_s^a(w^a,s)}{w}(s^d - s^g) \geq c^d - \left(c^g + \lambda^g + \lambda^g \frac{w^g}{w^d}\right)$$

In view of Eq. 4, the profit of water suppliers is positive if $\lambda^g > 0$ and $\overline{w}^g > 0$.

The shadow values of the desalination and groundwater salinity constraints $s^d \geq \underline{s}^d$ and $s^g \geq \underline{s}^g$ are, respectively, $\lambda^{sd} = V_s^a(w^a, s)\frac{w^d}{w}$ and $\lambda^{sg} = V_s^a(w^a, s)\frac{w^g}{w}$, implying that the larger the share of a water source in the total water supply, the larger the negative agricultural impact associated with a marginal increase in the salinity of that source.

The objective of our second scenario, termed *zero salinity* (ZS), is to assess the economic impact of the presence of salts in the economy's water sources. Suppose that salinity vanishes from all water sources, including the sea, such that an unlimited amount of zero-salinity water is available at the cost of groundwater extraction, c^g (i.e., $s^d =$

$s^g = 0$). In this case, the optimal administrative water price, denoted p^{ZS}, is set such that $p^{\mathrm{ZS}} = c^g$ (i.e., $p^{\mathrm{OS}} > p^{\mathrm{ZS}}$), the water consumptions of the two sectors satisfy $c^g = V_w^u(w^u) = V_w^a(w^a, 0)$, and water-supply costs are exactly recovered. The salinity shadow values become $\lambda^{sd} = V_s^a(w^a, 0)\frac{w^d}{w}$ and $\lambda^{sg} = V_s^a(w^a, 0)\frac{w^g}{w}$, which may differ from those under the OS scenario due to both the different water volumes and the marginal effect of salinity on agricultural production at zero salinity, $V_s^a(w^a, 0)$ (our empirical crop-production functions imply $V_{ss}^a(\cdot) < 0$; i.e., the lower s, the more negative $V_s^a(w^a, s)$, and therefore the larger the marginal damage of salinity). To evaluate the overall damage caused by salinity, the economy's net benefits under the ZS and OS scenarios should be compared.

The purpose of the third scenario, termed *fixed salinity* (FS), is to compute the deadweight loss associated with overlooking the salinity-reduction effect of desalination. Suppose that the groundwater source is exhausted (i.e., $w^g = \bar{w}^g$), and additional desalination water is supplied ($w^d > 0$), where the water master refers to desalination as an instrument for resolving water scarcity, but ignores its impact on salinity; that is, she or he erroneously assumes that the salinity of the supplied blended water is fixed at s^g. Consequently, when determining the administrative water price, p^{FS}, according to Eqs. 2 and 3, the terms $V_s^a(w^a, s)\frac{w^g(s^d - s^g)}{(w^d + w^g)^2}$ and $V_s^a(w^a, s)\frac{w^d(s^d - s^g)}{(w^d + w^g)^2}$, respectively, are dropped; hence, $p^{\mathrm{FS}} = c^d$ (i.e., $p^{\mathrm{FS}} > p^{\mathrm{OS}}$), and Eq. 3 implies $\lambda^g = c^d - c^g$. Nevertheless, the actual salinity of the mixed water, s, will be lower than s^g due to the desalinated water. To compute the actual levels of w^d and w^a, Eqs. 1 and 2 and the water-balance constraint $w^u = w^d + \bar{w}^g - w^a$ are used to obtain the conditions

$$c^d = V_w^u(w^d + \bar{w}^g - w^a) = V_w^a(w^a, s) + V_s^a(w^a, s)\frac{\bar{w}^g(s^d - s^g)}{(w^d + \bar{w}^g)^2}$$

The difference between the economy's net benefits under FS and OS represents the welfare implications of ignoring the desalination's impact on irrigation-water salinity. Since the water price in the FS case exceeds that of the OS scenario, the consumer surpluses are lower than under OS, while the surplus for the water suppliers is positive, and equals $(c^d - c^g)\bar{w}^g$.

2.2 Empirical Application

Our objective is to evaluate the economic implications of irrigation-water salinity empirically by comparing the FS and ZS scenarios to the OS scenario in a large-scale spatial-dynamic framework, using Israel as a case study. Our empirical specification to the general hydro-economic framework in the appendix is based on integration of the MYWAS and VALUE models, whose elements (i.e., spatial structure, demand and production functions, costs, etc.) are detailed in Reznik et al. (2017) and Kan and Rapaport-Rom (2012), respectively. The integrated model (code and data) is available as online supplemental materials. Here we use the scheme in Figure 3 to briefly present the combined MYWAS–VALUE version, and the extensions specific to this study.

From MYWAS, we adopt the physical and economic setups of Israel's water economy and add, in specific nodes of the water network, optional infrastructures (e.g., a desalination facility), which may be constructed at an optimal timing and scope to be determined endogenously by the model. The investment criterion is maximization of the present value of the economic surplus created by the facility minus the investment cost. Precipitation-enriched freshwater sources include 16 natural water bodies (15 aquifers and the Lake of Galilee) and 3 groups of regional artificial reservoirs. There are 6 seawater-desalination plants (5 already exist in the center of Israel, and an additional optional one is assumed in the northwest region of Acre), and 4 brackish water-desalination plants, as well as 18 wastewater-treatment plants (WWTPs) applying secondary treatment, and 1 applying tertiary treatment to the sewage of the Tel Aviv metropolis. Each WWTP contains the option to introduce TWW-desalination activity (which necessitates tertiary pretreatment), where desalinated TWW can be delivered only to agricultural zones.

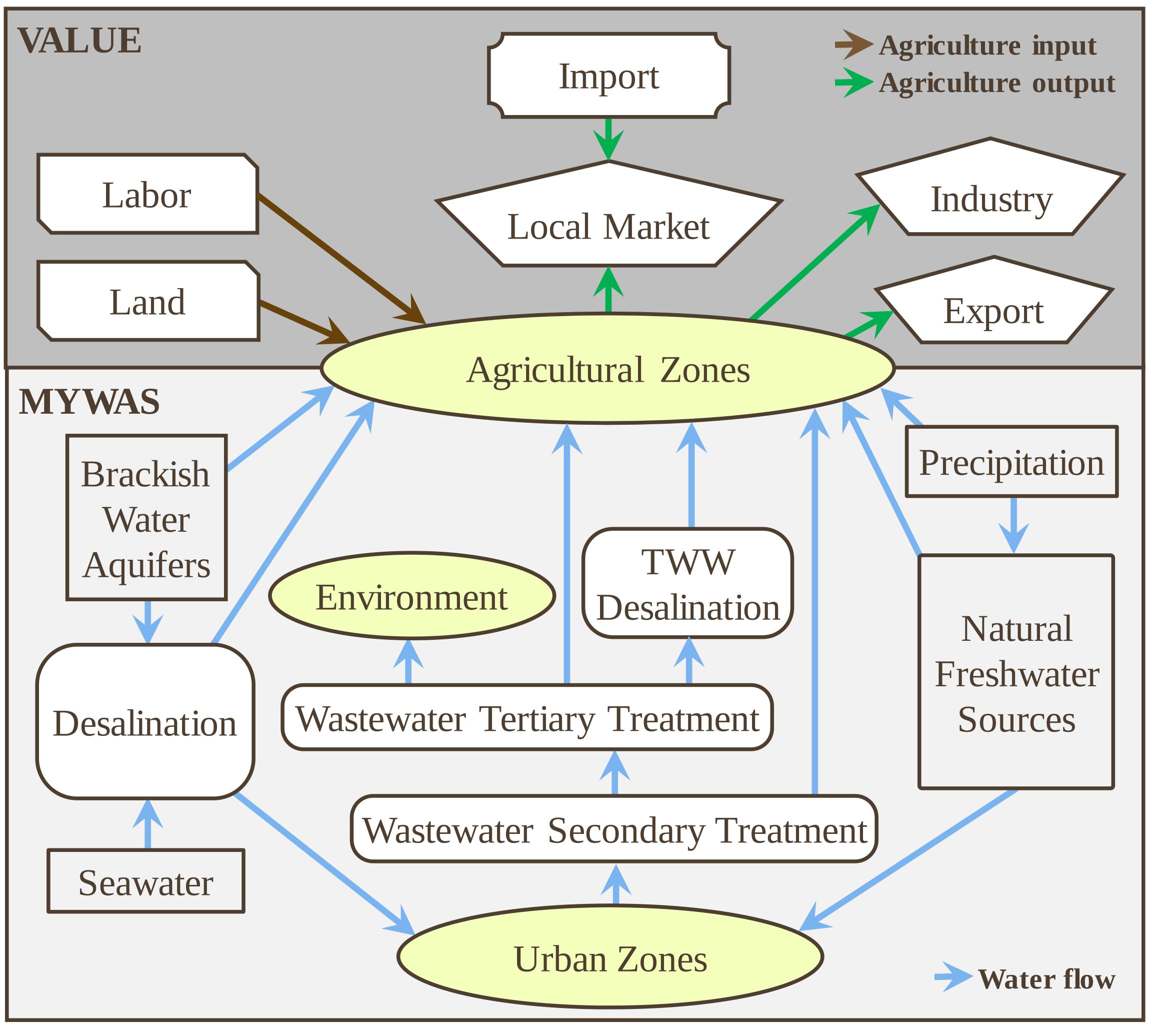

Figure 3. Scheme of the integrated MYWAS–VALUE hydro-economic model

Water users include 21 urban regions and 18 agricultural zones. Urban zones obtain freshwater for domestic and industrial uses from freshwater aquifers, desalinated seawater and desalinated brackish water, and generate sewage outflows, which undergo mandatory treatment at the WWTPs. The agricultural zones can consume irrigation water from all sources, including crop-specific direct precipitation inputs. Precipitation also enriches the freshwater aquifers and reservoirs. TWW can be discharged to the environment directly from WWTPs, conditional on being reclaimed to the level of tertiary treatment. The water-delivery system includes 183 freshwater pipelines and 58 pipelines for wastewater and brackish water. In addition, since in the base year desalinated seawater is supplied mainly to urban areas, we include 12 optional pipelines to connect seawater-desalination plants directly to agricultural areas near the Mediterranean shoreline.

Our MYWAS version digresses in two aspects from that of Reznik et al. (2017). First, the sets of inflow and outflow elements throughout the water-distribution network include the salinity concentrations as explicit decision variables, which are set subject to balance and exogenous constraints (see conditions (A2) and (A3) in the appendix). The salinity of desalination outflows (generated from either seawater, brackish water, or TWW) is 0.25 dS/m; the salinity of natural freshwater aquifers and reservoirs is 1 dS/m, and that of brackish-water aquifers is 4 dS/m. The salinity of freshwater pumped from the Sea of Galilee decreases linearly with the water stock of the lake. The salinity of non-desalinated TWW in a specific WWTP is determined by that of the plant's sewage inflows from the respective urban regions' outflows, where the salinity of each sewage outflow equals the salinity of the corresponding urban region's freshwater inflow, plus

0.7 dS/m—the salinity added through domestic uses (IWA, 2012). Pursuant to regulations, the salinity of urban inflows is constrained to not exceeding 1.4 dS/m. The irrigation-water salinity in an agricultural region is the average of the salinities of the inflows to the region, weighted by their respective volumes.

Second, the agricultural benefits of irrigation are computed explicitly by integrating the VALUE model into MYWAS. VALUE is a PMP model of Israeli vegetative agriculture. The model allocates the country's ~300,000 hectares of arable land to 55 crops in each of the 18 agricultural zones specified in MYWAS. Land allocations across crops are the farmers' decision variables in each region, where the per-hectare annual output of each crop in a given region depends on both the water available to the crop and the water salinity. The per-hectare water available for each crop equals the per-hectare irrigation water plus the rainfall during the crop's growing season, both assumed constant. The salinity of the water available to a crop equals the weighted average of the salinities of the regional irrigation water and the crop-specific annual rainfall, where rainfall salinity is 0.1 dS/m. Salinity affects agricultural outputs through the production function (Kan et al. (2002)):

$$y_{lt}^{i} = \frac{ymax_{l}^{i}}{1+\alpha_1\left(\alpha_2 s_{lt}^{i}+\alpha_3 {w_{lt}^{i}}^{\alpha_4}\right)^{\alpha_5}} \tag{5}$$

where y_{lt}^{i} is the actual output (ton/hectare per year) of crop i in location (region) l at time t; w_{lt}^{i} (m^3/hectare per year) and s_{lt}^{i} (dS/m) are, respectively, the annual amount and average salinity of the water available to the crop; $ymax_{l}^{i}$ (ton/hectare per year) is the maximum potential yield, representing the output under no water deficit and zero salinity; α_1 through α_5 are parameters.

To produce yield functions for the crops in the agricultural regions, we employ a four-stage meta-analysis procedure. First, we generate a dataset of plant-level relative yields (i.e., relative to $ymax_l^i$) under different combinations of water and salinity levels; to this end, we use the crop model developed by Shani et al. (2007), adopting crop salinity-tolerance parameters from Tanji (2002), and region-specific soil and climate parameters from Kan and Rapaport-Rom (2012); second, to account for intra-field spatial irrigation heterogeneity, we calculate field-level relative-yield levels by applying the log-normal distribution function (Knapp 1992), calibrated for a Christiansen uniformity coefficient of 85; third, we apply a non-linear regression to the generated dataset to estimate the parameters α_1 through α_5; finally, we calibrate $ymax_l^i$ using the observed yield, regional-water salinity and water application associated with each crop in each agricultural region in the base year. The resultant functions exhibit the expected properties of $\frac{\partial y}{\partial w} > 0$ and $\frac{\partial y}{\partial s} < 0$. In addition, we obtain $\frac{\partial^2 y}{\partial s^2} > 0$, meaning that the higher the salinity of the water applied to a crop, the lower the salinity's marginal damage; this implies that the lower the salinity in the water sources accessible by a water economy, the larger (in absolute terms) the shadow value of the salinity constraint associated with those sources.

Equation 5 incorporates the agronomic damage of irrigation-water salinity. Thus, lower salinity of a region's irrigation water increases crop production, changes the relative profitability of the crops, and thereby motivates changes in regional land allocation across crops. Moreover, the yields' salinity response may trigger a salinity reduction in the irrigation water by deliveries of desalinated water to agricultural zones.

In addition, the model can assess the direct agricultural impact of changes in rainfall, which affect both the per-hectare water available to each crop and the respective salinity.

Our VALUE version extends that of Kan and Rapaport-Rom (2012) by introducing the markets for vegetative agricultural products. Israel has a negligible impact on world food prices, and employs import tariffs which mostly protect local fresh fruit and vegetable products (Finkelshtain and Kachel 2009, Finkelshtain, Kachel and Rubin 2011). Thus, the local prices of agricultural outputs allocated to the processing food industry and export markets equal the (exogenous) world prices, whereas the prices of fresh vegetative products are determined in equilibrium subject to import tariffs. For the import price of each crop's output, we select the country from which the import cost (price plus transportation cost) is the lowest, and add the import tariff. We use constant-elasticity demand functions for fresh agricultural produce, with elasticity parameters adopted from Fuchs (2014). In addition, we account for Israel's immigration policy by calibrating the model with regional constraints on foreign labor, upon which the Israeli agriculture is heavily dependent (Kemp, 2010). These extensions enable us to study the impact of changes in import tariffs and foreign-labor constraints on the water and agricultural sectors.

We calibrate the integrated MYWAS–VALUE model to the base year, 2015. The objective function (Eq. A1 in the appendix) equals the benefits to residential and industrial urban water users plus the benefits to consumers of agricultural products (the cumulative area beneath the demand curves), minus all of the costs associated with water supply and the production and import of agricultural products throughout a 30-year planning period. We employ a *social* discount rate of 3.5%, as suggested by Nordhaus

(2007). Based on ICBS (2014) predictions, we assume an average annual population growth rate of 1.8%, which shifts the demands for urban water and agricultural products accordingly along the planning horizon. The average precipitation and annual enrichment of natural freshwater sources (taken from Weinberger et al. (2012)) are assumed for the entire simulated period. Initial monetary values are in 2015 US dollars, and scenario results are reported for an average year throughout the 30-year planning period, discounted to the 15th year (2030).

3. Economic Implications of Salinity

To evaluate the economic implications of salinity, we run the MYWAS–VALUE model under the OS, ZS and FS scenarios. An additional scenario, termed *environmental optimal solution* (EOS), is similar to the OS one, except that an environmental regulation prohibits discharge of tertiary-treated wastewater to natural waterways. The following three subsections describe the simulation results with the aid of Tables 1 through 5 and Figures 4 through 7, indicating links between the water-management patterns, salinity, shadow values, prices, agricultural outputs, and allocation of surpluses across the various agents in the economy. We then present sensitivity analyses of the optimal solution with respect to changes in the agricultural policies of import tariffs and foreign-worker quotas, and in precipitation levels.

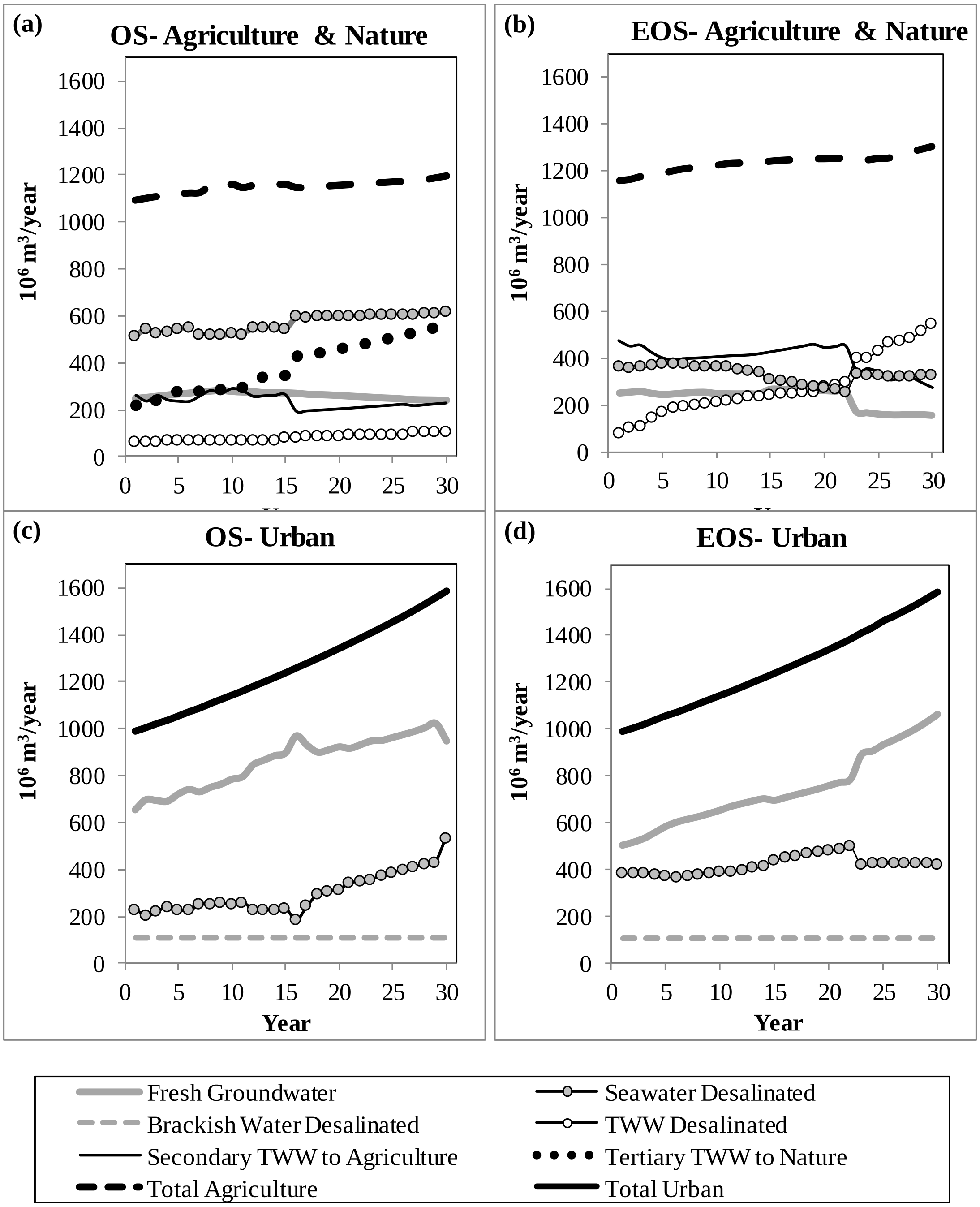

Figure 4. Trajectories of economy-wide annual water supplies to agriculture, nature and the urban sector under the OS and EOS scenarios

Table 1. Per-Annum Economy-Wide Average Water Volumes

	OS	EOS	ZS	FS
	All Water Consumers			
Water Supply (10^6 m^3/year)				
Fresh Groundwater	1,124	969	917	1,153
Brackish Groundwater	0	0	54	0
Brackish Groundwater Desalinated[a]	106	106	54	63
TWW	626	397	748	714
TWW Desalinated	81	276	0	0
Seawater Desalinated[a]	865	746	1,169	652
All	2,802	2,494	2,941	2,582
Water Consumption (10^6 m^3/year)				
Urban	1,263	1,262	1,291	1,242
Agriculture	1,149	1,231	1,314	998
Nature	390	0	336	342
All	2,802	2,494	2,941	2,582
	Agricultural Consumers			
Water Supply (10^6 m^3/year)				
Fresh Groundwater	262	229	329	270
Brackish Groundwater	0	0	54	0
Brackish Groundwater Desalinated	0	0	0	0
TWW	236	397	411	373
TWW Desalinated	81	276	0	0
Seawater Desalinated	569	329	520	355
All	1,149	1,231	1,314	998
Agricultural Labor (10^6 day/year)				
Foreign	9.5	9.5	9.5	9.4
Local	5.1	5.1	5.1	5.0

a. Under the Zero Salinity scenario, the sea and brackish-water aquifers are considered freshwater sources which need not be desalinated in order to be supplied to urban users.

Table 2. Per-Annum Economy-Wide Average Shadow Values of Water-Quantity and Water-Infrastructure-Capacity Constraints at Supply Points; Water-Consumption Prices at the Consumption Points ($\$/m^3$); Shadow Values of Land and Foreign Labor[a]

	OS	EOS	ZS	FS
	Water Quantity			
Water Supply Constraint ($\$/m^3$)				
Fresh Groundwater	0.27	0.19	0.07	0.87
Brackish Groundwater	0.11	0.06	0.04	0.00
Brackish Groundwater Desalinated	0.10	0.09	0.00	0.75
TWW	-0.09	-0.40	-0.15	0.14
TWW Desalinated	0.38	0.10	-0.15	0.82
Seawater Desalinated	0.00	0.00	0.00	0.00
Ban on TWW Discharge to Nature	-	0.10	-	-
Consumption Price ($\$/m^3$)				
Urban	1.54	1.55	1.11	1.77
Agriculture	0.41	0.22	0.16	0.72
All	0.86	0.89	0.56	1.13
	Water Infrastructure Capacity			
Capacity Constraint ($\$/m^3$)				
Fresh Groundwater	0.01	0.01	0.00	0.07
Brackish Groundwater	0.01	0.01	0.00	0.00
Brackish Groundwater Desalinated	0.47	0.43	0.07	0.25
TWW	0.05	0.05	0.04	0.15
TWW Desalinated	0.27	0.21	0.00	0.10
Seawater Desalinated	0.05	0.03	0.00	0.38
Water Conveyance	0.00	0.00	0.00	0.01
	Non-Water Agri. Production Factors			
Agricultural Land (\$/hectare)	5.12	6.32	7.35	2.49
Foreign Labor (\$/day)	59.3	65.4	83.5	45.7

a. Shadow values and prices were discounted to the 15th year, and then averaged.

Table 3. Per-Annum Economy-Wide Average Salinity Levels and Salinity Shadow Values at the Water Supply and Consumption Points

	OS	EOS	ZS	FS
	Salinity (dS/m)			
Supply[a]				
Fresh Groundwater	1.05	1.05	0.00	1.06
TWW	1.46	1.34	0.00	1.49
All	0.61	0.70	0.00	0.72
Consumption				
Urban	0.82	0.74	0.00	0.84
Agriculture	0.60	0.66	0.00	0.82
Nature	1.55	-	0.00	1.73
All	0.61	0.70	0.00	0.72
	Shadow Value ($\$10^6$ $(dS/m)^{-1}$ $year^{-1}$)[b]			
Supply				
Fresh Groundwater	-19.3	-18.0	-3.2	-31.3
Brackish Groundwater	0.0	0.0	-6.0	0.0
Brackish Groundwater Desalinated	-6.6	-8.8	0.0	-8.6
TWW	-17.5	-30.2	-43.8	-43.1
TWW Desalinated	-33.5	-59.2	0.0	0.0
Seawater Desalinated	-95.5	-66.4	-16.0	-98.0
Consumption				
Agriculture	-77.4	-85.3	-87.0	-68.1

a. The salinity of brackish water and desalinated water is 4 and 0.25 dS/m, respectively.

b. Shadow values were discounted to the 15th year, and then averaged.

Table 4. Per-Annum Economy-Wide Average Welfare Elements (\$$10^6$/year)[a]

	OS	EOS	ZS	FS
Agricultural Products				
Production Value	5,619	5,633	6,140	5,229
Import Value[b]	662	685	634	724
Variable Costs	3,999	4,105	4,321	3,765
Capital Costs	5,619	5,633	6,140	5,229
Water-Supply Costs				
Variable Costs	1,432	1,246	1,055	1,291
Capital and Operation Costs	472	788	435	421
Water to Nature	131	-	113	115
Total	1,903	2,035	1,490	1,712
Water Purchase Expenses				
Urban	1,855	1,865	1,375	2,100
Agriculture	728	255	195	655
Total	2,584	2,120	1,570	2,755
Surpluses				
Urban Water Consumers[c]	0	-18	907	-465
Agricultural Product Consumers[c]	0	-9	281	-241
Farming Profits	891	1,273	1,624	810
Water Suppliers	549	85	-32	929
Social Welfare	1,441	1,331	2,781	1,032

a. Annual welfare-element values were discounted to the 15th year, and then averaged.

b. The import value includes only imports of products associated with the 55 crops incorporated in the model.

c. Since the computation of the areas beneath the calibrated constant-elasticity demand functions of urban water and agricultural products involves extrapolations, we normalize the surpluses of urban-water and agricultural-product consumers to 0 under the OS scenario, and report their changes under each scenario in comparison to the OS.

Table 5. Laspeyres Quantity and Price Indices, and Value Index, of Agricultural Production by Salinity-Tolerance Crop Bundles, under the ZS and FS Scenarios, Expressed Relative to the OS Scenario (i.e., OS = 100)

	Quantity	Price	Value
Crop Bundle[a]		**ZS**	
Sensitive (36%)	116	95	111
Moderately Sensitive (47%)	112	97	108
Moderately Tolerant (14%)	103	100	103
Tolerant (3%)	126	99	124
All crops	113	97	110
		FS	
Sensitive (36%)	92	104	95
Moderately Sensitive (47%)	94	102	97
Moderately Tolerant (14%)	95	100	95
Tolerant (3%)	64	102	66
All crops	92	102	94

a. Values in parentheses indicate the share of the crop bundle in the production value under the OS scenario.

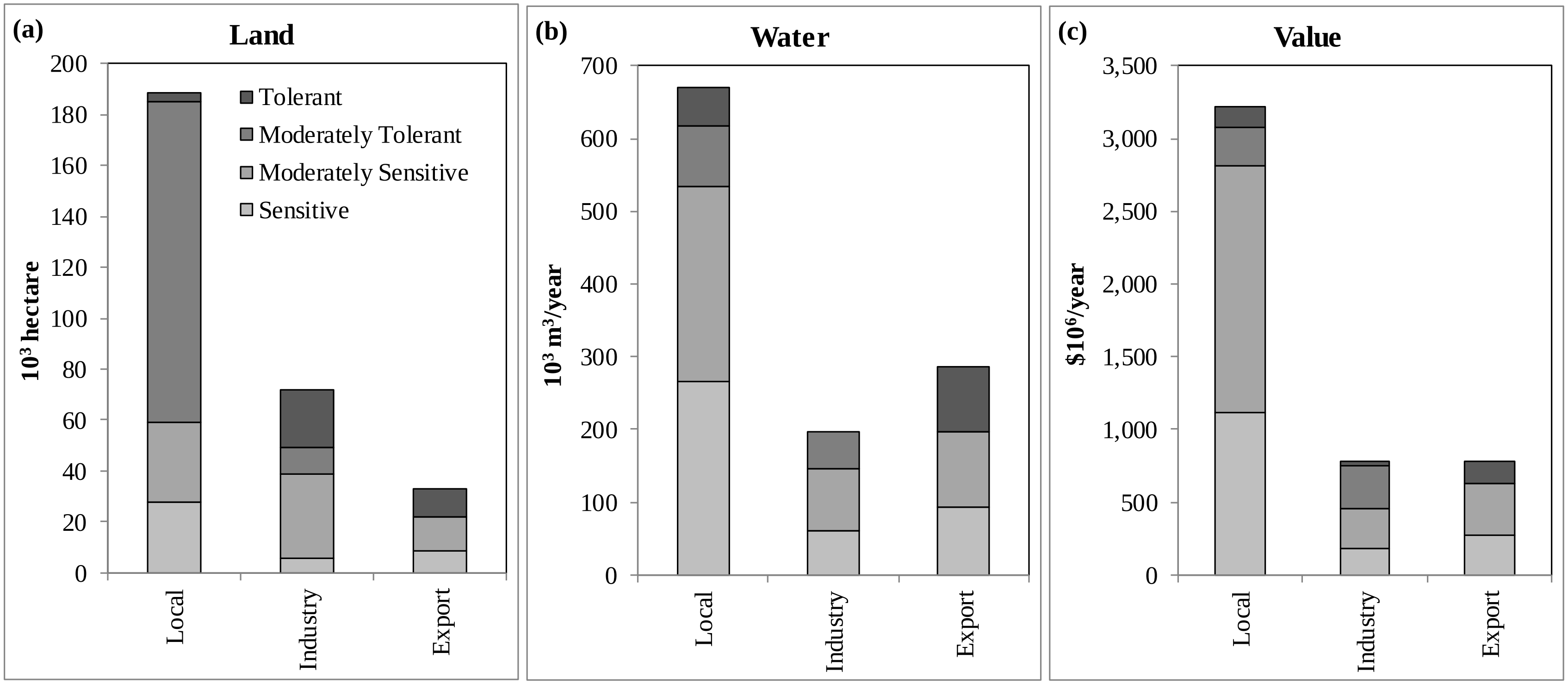

Figure 5. Base-year (2015) economy-wide allocation of (a) land use, (b) water consumption and (c) production value across four salinity-tolerance crop bundles, separated into production for local fresh produce, industry and export agricultural markets

3.1 Optimal Solution

The intriguing result of this paper is that, in order to cope with the agronomic damage of salinity in Israel, delivery of massive amounts of desalinated water to agricultural zones is warranted. Moreover, since the desalination cost of TWW (\$0.63/m^3) is higher than that of seawater (\$0.56/m^3), most of the desalinated irrigation water is produced in seawater-desalination plants, whereas TWW is discharged to the environment; desalination of TWW for agricultural use occurs only under the EOS scenario, where discharge to nature is prohibited.

The OS scenario suggests, for an average year throughout the planning period, a total desalination capacity of 1,052 million m^3/year (Table 1), of which 865 million m^3/year is desalinated seawater, and the rest—106 and 81 million m^3/year—is desalinated brackish water and TWW, respectively; the latter occurs mostly in areas remote from the Mediterranean shoreline. Of these desalinated volumes, agriculture obtains 650 million m^3/year, constituting 57% of the total irrigation water. As a result, the average salinity of the irrigation water (0.6 dS/m, Table 3) is quite close to that of desalinated water (0.25 dS/m). According to this plan, the share of desalinated seawater supplied directly to the agricultural regions is 77%, where eight out of the twelve optional direct pipeline connections between desalination plants and agricultural regions are realized, the optional seawater-desalination plant in the north of the country is built, and desalination capacities are extended with time to adjust for population growth. Figure 4a and 4c indicates, respectively, that the supply of desalinated seawater for irrigation is steady, and for urban use is growing with time. While the overall agricultural water consumption rises only moderately, that of the urban sector increases sharply, and generates increasing amounts of TWW, about half of which is disposed of by discharge to nature after tertiary treatment (Figure 4a), and the rest delivered to irrigation.

If TWW discharge to nature is forbidden (scenario EOS), the average TWW desalination increases by 195 million m^3/year compared to the OS scenario, replacing seawater desalination, which decreases by 119 million m^3/year (Table 1). Returning to OS, the total desalinated seawater remains at around 750 million m^3/year during half of the planning period, and then increases steadily, mostly through larger deliveries to urban users. The total agricultural consumption of desalinated water gradually increases throughout the period, from 581 to 724 million m^3/year.

Desalinating water for irrigation was found economically viable by Hadas et al. (2016) in a small-scale analysis for specific crops in the Arava region in southern Israel. Our analysis finds this strategy to be warranted at the economy-wide level. We attribute the motivation for the huge desalinated-water delivery to irrigation under the OS and EOS scenarios to the agricultural protection policy in Israel, which for decades has led to specific cropping specialization in the vegetative farming sector. Israel is a net importer of grains and other low-value water-intensive agricultural products, which are consumed mostly by the industrial sector free of import taxes (ICBS, 2017). On the other hand, as already noted, a tariff policy effectively protects fresh produce of fruit and vegetables from competing imports (OECD, 2018); hence, the prices of fresh products are generally determined in equilibrium in the local markets, and the demand for those products gradually increases via population growth. Apparently, these fresh-product crops require large volumes of irrigation water, and are also relatively sensitive to salinity. To elucidate, Figure 5 presents the agricultural land allocation, water use and production value of crops in Israel in the base year, separated into production for the local fresh-produce market, processing industry and export. In addition, each value is separated into four crop bundles that classify the crops according to their salinity tolerance (Maas and Hofmann 1977). Consider the salinity-sensitive and moderately sensitive crop bundles that are produced for the local market: while only 20% of the total arable land in the country is allocated to these two bundles (Figure 5a), their water consumption constitutes 46% of the total irrigation-water use (Figure 5b), and their production value amounts to 59% (Figure 5c). These patterns incentivize extensive allocation of desalinated water to the agricultural sector to reduce salinity, increase the yields and per-hectare profits of these crops, and in turn, the land allocated to them, and their total production value under the optimal solution.

We assume that prices in the economy are set efficiently, and use the models' demand functions and shadow values of water-balance constraints (see the appendix) to compute the water and agriculture-product prices associated with the optimal solution, and the corresponding surpluses allocated to the various sectors in the economy (Table 4) (where consumer surpluses are normalized to 0 under the OS scenario).

The computed shadow values and prices are instructive with respect to the relation between the alternative water uses and the water salinity. Consider the negative shadow value of the constraint requiring treatment of urban sewage at the WWTPs ($-0.09/$m^3$ under OS, Table 2), implying that farmers should be compensated to voluntarily consume non-desalinated TWW. This is because the salinity of TWW under OS (1.46 dS/m, Table 3) is higher than that of the irrigation water (0.6 dS/m) and therefore, farming use of TWW, which saves the tertiary-treatment cost needed to allow TWW discharge to nature, will reduce agricultural production. On the other hand, the salinity of desalinated TWW is lower than that of the irrigation water, implying a positive shadow value ($0.38/m^3). This also explains the shadow value of the capacity constraint (Table 2) associated with TWW desalination ($0.27/m^3) being larger than that of the WWTPs ($0.05/m^3).

Under the EOS scenario, TWW must be disposed of through irrigation; the statewide average shadow value of this constraint is $0.10/m^3 (Table 2). The social welfare is $110 million a year lower than that under the OS scenario (Table 4), implying that a ban on TWW discharge to nature is justified only if the associated (unknown) environmental damage is larger than this welfare difference. The shadow value of the constraint requiring sewage to be treated in WWTPs becomes $-0.40/m^3, representing the compensation to farmers for consuming TWW instead of desalinated TWW. In addition, while the salinity of the irrigation water changes only slightly under the EOS vs. OS scenario (Table 3), its total consumption rises (Table 1), which in turn reduces the irrigation-water price (i.e., its value of marginal production) from $0.41/m^3 under OS to $0.22/m^3 under EOS. Consequently, farming profits increase at the expense of those of the water suppliers (Table 4). Note that as irrigation-water supply increases under EOS, there is a rise in the shadow values of the constraints on agricultural land and foreign labor as supplemental irrigation-water production factors (Table 2).

Table 3 reports the shadow values associated with the presence of salinity in the various water sources. As shown theoretically, the more water consumed from a source, the more negative the shadow value of its salinity becomes; therefore, under the OS scenario, a marginal increase in the salinity of desalinated seawater is most destructive. In comparison, under EOS, the desalinated TWW replaces part of the desalinated seawater delivered to agriculture in the OS scenario, and therefore the salinity shadow values of desalinated seawater and TWW become closer under EOS.

3.2 Zero Salinity

For the ZS scenario in our dynamic framework, envisage a case in which all of Israel's water sources become pure freshwater (including the sea) exactly at the onset of the planning period. Technically, given the infrastructures and crop portfolios in the calibration year, MYWAS–VALUE searches for the optimal trajectories of water and agricultural managements, where per-hectare yields increase to their maximum levels under zero salinity (i.e., $ymax$), and the extraction cost of purely freshwater from all primary sources (i.e., the sea and sources of natural fresh and brackish waters) is similar to that of fresh groundwater (except that the water levels of the sea and brackish-water aquifers are fixed, whereas that of fresh groundwater varies with the stock (Reznik et al., 2017)). By comparing ZS to OS, we elicit the total welfare loss incurred by the presence of salts in Israel's water sources, while accounting for the agronomic impacts and the effects imposed on urban water users and agricultural product consumers.

The overall average annual welfare under the ZS scenario is $1,340 million higher than that of the OS case (Table 4). Without salinity, farmers' annual profits would be $733 million higher, consumers of agricultural products would gain an additional surplus of $281 million a year, the per-annum surplus of urban water users would increase by $907 million, and the yearly profit of water suppliers would decrease by $581 million. To put these findings in context, Qadir et al. (2014) estimated an average cost of $440 per hectare for salt-affected lands, amounting globally to $27 billion a year. As already noted, our analysis embeds direct and indirect salinity effects, and therefore obtains a much larger damage of $4,468/hectare to the economy as a whole, $2,444/hectare of which is farmers' losses.

In terms of water management, TWW desalination unsurprisingly vanishes, the overall water use in the economy increases slightly, less TWW is discharged to nature, and irrigation water use increases correspondingly (Table 1). Notice that the shadow value of irrigation-water salinity becomes more negative (from $-77 million per dS/m annually under OS to $-87 million per dS/m annually under ZS, Table 3), which is explained by the combination of lower salinity and the production-function property $\frac{\partial^2 y}{\partial s^2} > 0$. In addition, the irrigation-water price decreases (from $0.41 to $0.16/m^3, Table 2) as theoretically predicted earlier, and the shadow values of agricultural land and foreign labor increase accordingly.

The overall agricultural production value under ZS increases by 10% relative to OS (Table 4). To examine this change, we present Laspeyres quantity and price indices of the agricultural produce in Table 5 (using OS as the reference: OS = 100), separating the production into the four salinity-tolerance crop bundles (recall Figure 5). We get a 13% increase in the overall agricultural outputs (obtained through both per-hectare yield changes and reallocation of regional arable lands across crops) and a 3% reduction in average prices. Most of the effect of ZS is attributed to the increase in the outputs of crop bundles that are sensitive and moderately sensitive to salinity, which together comprise 82% of the total production value for all three markets in the base year (Figure 5).

3.3 Fixed Salinity

We conduct the FS scenario in two stages: first, we run the MYWAS–VALUE model while fixing the salinity concentrations in the agricultural regions at their base-year levels; the output of this simulation is an infrastructure-development plan that the planner considers optimal given her or his presumption of fixed irrigation-water salinity. However, as already noted in our theoretical analysis, salinity will change, and in turn affect crop production, agricultural output prices, farming profitability, land use, and surpluses of producers and consumers in the agricultural sector. Therefore, in the second stage, we rerun the model while allowing salinity to change, but at the same time forcing the development of infrastructures to follow the infrastructure plan resulting in the first-stage simulation. Thus, the FS scenario represents the expected evolution of the water and vegetative-agriculture economies under efficient prices, given a non-optimally designed water-infrastructure blueprint.

Compared to OS, under FS, the designer of the water-infrastructure system reduces the desalination of brackish water, TWW and seawater by 41%, 100% and 25%, respectively (Table 1). Instead, the reliance on fresh groundwater and secondary TWW increases, which in turn elevates the irrigation-water salinity from 0.6 to 0.82 dS/m (Table 3). In addition, water use by the agricultural sector decreases from 1,149 to 998 million m^3/year (Table 1), where the scarcity of seawater desalination and TWW capacities grow (Table 2). In accordance to the theory, the irrigation-water price increases from \$0.41 to \$0.72/m^3 (Table 2). Concomitantly, arable lands and foreign labor become less scarce, and the total value of agricultural production declines by \$389 million a year (7% compared to OS) (Table 4); the latter results from an 8% reduction in production quantity, in combination with a 2% increase in agricultural output prices (Table 5), where the shortage in local produce is partly covered by a 9% increase in the value of imported products (Table 4). These changes reduce the surplus of consumers of agricultural products and the farming profits by $241 and $81 million/year, respectively. In addition, water consumption by the urban sector decreases by 21 million m^3/year (2%) (Table 1), the associated urban-water price rises from $1.54 to $1.77/$m^3$ (Table 2), and urban water users face a surplus reduction of $465 million a year (Table 4). In line with the theory, the only gainers from these changes are the water suppliers, whose net income increases from $549 to $929 million a year. The overall annual deadweight loss compared to the OS scenario is $408 million, which is about $1,360 per hectare of arable land.

3.4 Sensitivity to Changes in Agricultural Policies

Israel's policies to protect its agricultural sector are continuously criticized economically (OECD 2018), and are subject to public and political debate (Hendel et al., 2017). Changes in Israeli agricultural policy can vary the irrigation-water demand, thereby affecting the optimal infrastructural development of the water system. We study the impact of three policy changes on the OS scenario; the first change assumes no import tariffs on vegetative products (termed NIT). In the second, we remove labor quotas to simulate an unlimited foreign labor (UFL) supply, where we assume no change in the regulated minimum wage of the foreign workers. Moreover, based on Kimhi (2016), we assume that foreign workers are not replacing local ones; hence, variations in foreign-labor employment do not affect the local workers' wages. The third scenario combines these two policy changes (termed NIT & UFL).

Figure 6 presents the changes in the optimal irrigation-water sources and welfare elements entailed by the three policy scenarios. The NIT policy only slightly affects the consumption of agricultural labor and irrigation water, and its welfare effects are attributed to a reduction in the agricultural price index (-8%) and quantity index (-4%)

relative to the OS scenario (not shown). The local agricultural production value and farmers' profit decrease, respectively, by $596 and $85 million a year (-11% and -10%); import value increases by $699 million a year (106%), consumers of agricultural products gain an additional surplus of $722 million a year, and the overall welfare in the economy increases by $361 million a year.

The UFL policy dramatically increases foreign (25%) and local (22%) labor, and also enlarges water consumption (11%), particularly through an additional 97 million m^3/year of desalinated seawater. The vegetative-agriculture quantity index increases 15%, but the price index declines 6% such that the production value rises only 8% ($430 million/year). However, since production costs increase to a larger extent, farmers lose $53 million a year from that policy, whereas consumers gain $475 million a year, and the overall welfare increases by $156 million a year. If, in addition to UFL, the NIT policy is applied, the welfare increase amounts to $284 million a year. To summarize, while the simulated labor reform has larger impacts on labor and water usage in the economy, the abolishment of import tariffs induces a larger overall welfare increase, shifting surpluses from farmers to agricultural-product consumers.

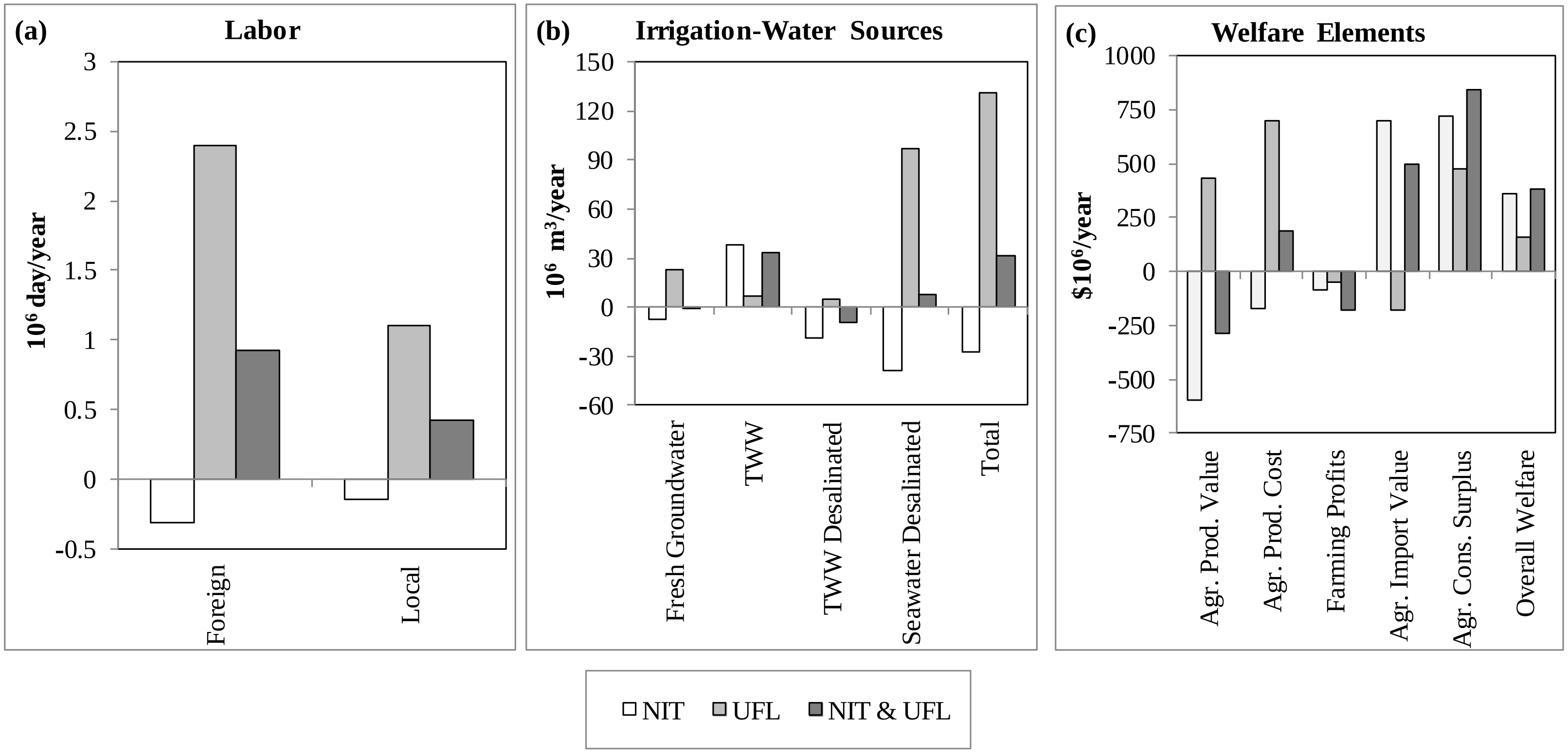

Figure 6. Changes in (a) labor, (b) the use of irrigation-water sources and (c) welfare elements, in response to the abolishment of agricultural product import tariffs (NIT), foreign-labor quotas (UFL), and both (NIT & UFL)

3.5 Sensitivity to Changes in Precipitation

Precipitation enriches aquifer- and surface-freshwater bodies, and directly waters agricultural fields. The average use of fresh groundwater under the OS scenario amounts to 1,124 million m^3/year (Table 1), of which the agricultural sector consumes 262 million m^3/year. In addition, based on land allocation in the base year, nearly 10^9 m^3/year of rainfall contributes to agricultural production.

Results from three general circulation models (Gent et al., 2011; Watanabe et al. 2010; Bentsen et al. 2013) predict precipitation-reduction rates of 10–30% in the coming decades for Israel. To assess the impact of precipitation change, we run the OS scenario under various levels of precipitation change, which we employ for every year throughout the planning period. Figure 7 plots the economy-wide average per-year water uses and water-supply sources versus the precipitation change in the range of -50% to +50% compared to the precipitation level under the OS scenario.

On average, the optimal response to precipitation reduction is to replace each cubic meter of reduced natural freshwater by an additional 0.67 m^3 of desalinated seawater and 0.03 m^3 of TWW for irrigation. In addition, irrigation-water consumption is to be reduced. In terms of social welfare, the average value of an additional 1% of precipitation is $27.3 million a year, of which $11.8 million per year are farmers' profits, $5.9 million a year are gained by agricultural product consumers, and $9.6 million a year are benefits to urban users.

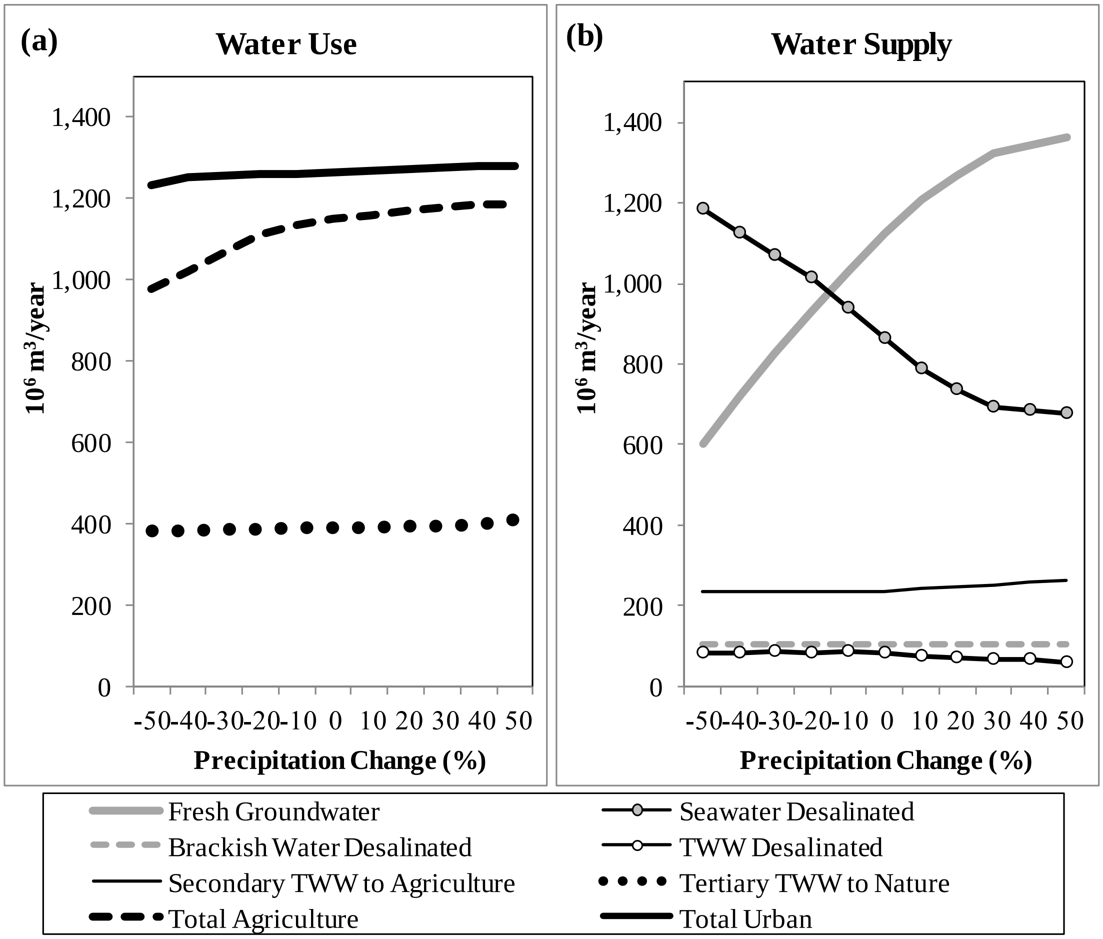

Figure 7. Effect of precipitation changes on economy-wide average annual (a) water use and (b) water supply under the optimal solution

5. Concluding Remarks

This study shows that, under the specific circumstances of Israel, irrigation-water salinity has a considerable impact on the optimal course of water-infrastructure development. Our ability to introduce salinity impacts on agricultural outputs into a large-scale economic-optimization framework builds on output from decades of theoretical and experimental agronomic research efforts, which enabled the development of crop-specific water-salinity production functions. Salinity, however, has additional economic implications through health effects on drinking water (Dasgupta et al., 2016), and other domestic and industrial water uses. Moreover, salinity is but one of a large set of water-quality elements that determine the economic value of water in various water uses, and therefore may affect optimal water-infrastructure plans. For example, desalinated-seawater content of Mg^{2+} and SO_4^{2-} is lower than recommended for drinking and irrigation water (Yermiyahu et al., 2007), and desalination removes nutrients that might otherwise save fertilization expenses (Ben-Gal et al., 2009; Dawson and Hilton, 2011). On the other hand, TWW desalination also removes various contaminants (Gur-Reznik and Dosoretz, 2015) that damage irrigation systems (Tarchitzky et al., 2013) and cause environmental and health risks (Fatta- Kasprzyk-Hordern et al., 2011; Kassinos et al., 2011) which, in turn, may affect the demand for agricultural products (Messer et al., 2017). The model developed in this study can be extended to assess the impact of such additional factors conditional on the availability of scientifically based empirical information on the related benefits and costs.

Appendix: General Hydro-Economic Framework

Consider a water economy encompassing activities of water treatments, deliveries and uses, operating at various locations throughout a given planning period T. Let $a \in \mathbf{a} = \{1, \dots, A\}$ indicate the type of activity, $l \in \mathbf{l} = \{1, \dots, L\}$ the location, and $t \in \mathbf{t} = \{1, \dots, T\}$ the time. Each activity has a set of input flows, denoted by $i \in \mathbf{i} = \{1, \dots, I\}$. For instance, an agricultural region may obtain water from freshwater and brackish-water aquifers, wastewater-treatment plants, seawater-desalination plants, etc. We define a vector of elements $\mathbf{q}_{lt}^{ai} = \left(w_{lt}^{ai}, s_{lt}^{ai}\right)$, specifying the delivered quantity w_{lt}^{ai} and salinity level s_{lt}^{ai} of the inflow i into activity a in location l at time t. (In general, water flow is characterized by the volume per time unit and a range of attributes, such as the concentration of various contaminants, elevation above sea level, temperature, pH, etc.; this study focuses on salinity as a single characteristic.) Accordingly, $\mathbf{q}_{lt}^{a\mathbf{i}} = (\mathbf{q}_{lt}^{a1}, \dots, \mathbf{q}_{lt}^{aI})$ is the activity's set of inflow elements. Similarly, each activity has a set of output flows, denoted $o \in \mathbf{o} = \{1, \dots, O\}$, with elements $\mathbf{q}_{lt}^{ao} = (w_{lt}^{ao}, s_{lt}^{ao})$, and accordingly $\mathbf{q}_{lt}^{a\mathbf{o}} = (\mathbf{q}_{lt}^{a1}, \dots, \mathbf{q}_{lt}^{aO})$. Note that locations of inflows and outflows of a particular activity may differ, as in the case of water-delivery activities. The vector $\mathbf{q}_{lt}^{a} = \left(\mathbf{q}_{lt}^{a\mathbf{i}}, \mathbf{q}_{lt}^{a\mathbf{o}}\right)$ comprises all of activity a's inflow and outflow elements.

Activities are interlinked spatially such that an outflow from a particular activity may constitute an inflow to another. The array of connected activities is represented by the matrix $\Gamma_{\boldsymbol{lt}}^{\boldsymbol{a}}$**,** which specifies for every pair of activities whether an outflow of one feeds the inflow of the other at time t, and vice versa. That is, in a pair of connected activities, each may serve as a source of outflow and as a destination for inflow; this is particularly relevant in pipes where the flow may switch direction.

Given the array matrix $\Gamma_{\boldsymbol{lt}}^{\boldsymbol{a}}$, balance constraints define the relations between the elements of connected inflows and outflows. For example, a wastewater-treatment plant may obtain sewage delivered from a few urban zones; hence, the inflow rate into the plant cannot exceed the sum of the outflow rates from the sewage-contributing zones, and the salinity of the plant's inflow cannot fall below the average salinities of the sewage deliveries, weighted by their respective amounts. Let $\mathbf{o} \to i = \{1 \to i, \dots, O \to i\}$ be the set of O outflows that feed inflow i. We denote by $\mathbf{b}_{lt}^{a\mathbf{o}\to i} \equiv \left(w_{lt}^{a\mathbf{o}\to i}, s_{lt}^{a\mathbf{o}\to i}\right) = b_{lt}^{a\mathbf{o}\to i}\left(w_{lt}^{a1\to i}, \dots, w_{lt}^{aO\to i}, s_{lt}^{a1\to i}, \dots, s_{lt}^{aO\to i}\right) = \mathbf{0}$ the set of balance constraints associated with inflow i, where $b_{lt}^{a\mathbf{o}\to i}(\cdot)$ is a water quantity and salinity balance function;

accordingly, $\mathbf{b}_{lt}^{a\mathbf{o}\to\mathbf{i}} = \mathbf{0}$ is the set of balancing constraints associated with activity *a*'s inputs.

Within each activity, the elements of a particular outflow o depend on those of the set of the activity's inflows: $\mathbf{b}_{lt}^{a\mathbf{i}\to o} \equiv \left(w_{lt}^{a\mathbf{i}\to o}, s_{lt}^{a\mathbf{i}\to o}\right) = b_{lt}^{a\mathbf{i}\to o}(w_{lt}^{a1\to o}, \dots, w_{lt}^{aI\to o}, s_{lt}^{a1\to o}, \dots, s_{lt}^{aI\to o})$. For instance, a desalination plant produces freshwater and brine, where the function $b_{lt}^{a\mathbf{i}\to o}(\cdot)$ specifies the dependence of one of the two outflows on the inflow elements, given the desalination technology. We denote by $\mathbf{b}_{lt}^{a\mathbf{i}\to\mathbf{o}} = \mathbf{0}$ intra-activity *a*'s entire set of input–output balance constraints.

An activity's inflow and outflow elements may also be subject to various capacity, regulatory, technological and feasibility constraints. For example, the amount of groundwater extraction is constrained by either the pumping capacity or the extractable groundwater stock; a typical reverse-osmosis desalination plant has, in addition to a maximum-inflow capacity, a minimum-inflow rate which is required to avoid damage to the membranes; for health reasons, the salinity of the water supplied for domestic use may not exceed an upper level; regulations may specify, respectively, minimum and maximum levels of water quantity and salinity to be allocated to the environment. We let $k \in \mathbf{k} = \{1, \dots, K\}$ indicate a type of constraint, and denote by $\overline{\mathbf{q}}_{lt}^{aik} = \left(\overline{w}_{lt}^{aik}, \bar{s}_{lt}^{aik}\right)$ and $\overline{\mathbf{q}}_{lt}^{aok} = \left(\overline{w}_{lt}^{aok}, \bar{s}_{lt}^{aok}\right)$, respectively, the upper levels of an activity's inflow and outflow elements associated with constraint k. Similarly, $\underline{\mathbf{q}}_{lt}^{aik}$ and $\underline{\mathbf{q}}_{lt}^{aok}$ are the sets of minimum (including non-negativity) constraints. We mark by $\overline{\mathbf{q}}_{lt}^{a\mathbf{k}}$ and $\underline{\mathbf{q}}_{lt}^{a\mathbf{k}}$ activity *a*'s set of maximum and minimum constraints, respectively, and $\mathbf{q}_{lt}^{a\mathbf{k}} = \left(\overline{\mathbf{q}}_{lt}^{a\mathbf{k}}, \underline{\mathbf{q}}_{lt}^{a\mathbf{k}}\right)$.

The inflow- and outflow-element constraints may change with time due to both exogenous and endogenous factors. For instance, the extractable stock of an aquifer at time t equals the extractable stock at $t-1$, plus the natural enrichment of the aquifer at $t-1$, minus the sum of the extractions from the aquifer at $t-1$. The salinity of the extractable stock also evolves in relation to the salinities of the groundwater, the enrichment, the extractions, and the hydrological streams within the aquifer. We denote by $\boldsymbol{\Delta}\mathbf{q}_{lt}^{a\mathbf{k}}$ the changes in the set of constraints $\mathbf{k}$; hence, given the initial level of constraints $\mathbf{q}_{l0}^{a\mathbf{k}}$, there is $\mathbf{q}_{lt}^{a\mathbf{k}} = \mathbf{q}_{l0}^{a\mathbf{k}} + \sum_{t=1}^{t} \boldsymbol{\Delta}\mathbf{q}_{lt-1}^{a\mathbf{k}}$ for every $t = 1, \dots, T$.

Of particular interest for water-economy design are the infrastructural constraints. Let $z \in \mathbf{z} = \{1, \dots, Z\}$ indicate a capacity or technological constraint, where $\mathbf{z}$ is a subset of $\mathbf{k}$ $(\mathbf{z} \subseteq \mathbf{k})$. The changes in infrastructures are limited by maximum and minimum constraints, denoted $\overline{\boldsymbol{\Delta}}\mathbf{q}_{lt}^{a\mathbf{k}}$ and $\underline{\boldsymbol{\Delta}}\mathbf{q}_{lt}^{a\mathbf{k}}$, respectively. The cost function $c_{lt}^{a}(\mathbf{q}_{lt}^{a\mathbf{z}})$ represents the capital and maintenance costs associated with $\mathbf{q}_{lt}^{a\mathbf{z}}$ at time t.

In addition, each activity is associated with benefits and costs, which depend on the set of inflow and outflow elements $\mathbf{q}_{lt}^{a}$; we denote by $v_{lt}^{a}(\mathbf{q}_{lt}^{a}, \mathbf{x}_{lt}^{a})$ the activity's net benefits, where $\mathbf{x}_{lt}^{a} = (x_{lt1}^{a}, \dots, x_{ltN}^{a})$ is a set of non-water variables which are to be established subject to a respective set of minimum, maximum and balance constraints, denoted generally $g_{lt}^{a}(\mathbf{x}_{lt}^{a}) = \mathbf{0}$. For example, in an agricultural zone, $\mathbf{x}_{lt}^{a}$ may represent land allocation to crops subject to the regional arable land and non-negativity constraints. In the case of a wastewater-desalination plant, the net benefits $v_{lt}^{a}(\mathbf{q}_{lt}^{a}, \mathbf{x}_{lt}^{a})$ are negative since this activity involves only costs, whereas the net benefits of water management in a city are expected to be positive because the willingness to pay for domestic water use is likely higher than the water-distribution costs. Note that the net benefits of urban water users may vary with time due to changes in various exogenous factors, such as population and income. Moreover, the net benefits of an activity located in some specific place l may be related to activities located elsewhere—this is the case of irrigation-water use by an agricultural sector in a small economy such as Israel's, which operates under import tariffs, such that vegetative-agriculture output prices are determined in equilibrium in the local markets; indeed, our empirical model captures these inter-regional links.

With the above settings, we are in a position to formulate the hydro-economic optimization problem. Let $\beta(t)$ be a discounting function. The decision variables are the sets of inflow and outflow elements $\mathbf{q}_{lt}^{a}$, the infrastructural expansions $\boldsymbol{\Delta}\mathbf{q}_{lt}^{a\mathbf{z}}$ and the non-water variables $\mathbf{x}_{lt}^{a}$ associated with each activity, location and time. Given the linkage matrix $\Gamma_{\boldsymbol{lt}}^{\boldsymbol{a}}$ and the initial level of constraints, $\mathbf{q}_{l0}^{a\mathbf{k}}$, the decision variables should be set so as to maximize the net benefits of the water economy:

$$\pi = \sum_T \beta(t) \sum_L \sum_A [v_{lt}^{a}(\mathbf{q}_{lt}^{a}, \mathbf{x}_{lt}^{a}) - c_{lt}^{a}(\mathbf{q}_{lt}^{a\mathbf{z}})] \qquad \text{(A1)}$$

subject to the sets of inter- and intra-activity balance constraints

$$\mathbf{b}_{lt}^{a\mathbf{o}\rightarrow\mathbf{i}} = \mathbf{0} \text{ and } \mathbf{b}_{lt}^{a\mathbf{i}\rightarrow\mathbf{o}} = \mathbf{0} \quad \forall\ a \in \mathbf{a},\ l \in \mathbf{l} \text{ and } t \in \mathbf{t}, \qquad \text{(A2)}$$

the set of exogenous constraints

$$\underline{\mathbf{q}}_{lt}^{a\mathbf{k}} \leq \mathbf{q}_{lt}^{a} \leq \overline{\mathbf{q}}_{lt}^{a\mathbf{k}} \text{ and } g_{lt}^{a}(\mathbf{x}_{lt}^{a}) = \mathbf{0} \quad \forall\ a \in \mathbf{a},\ l \in \mathbf{l} \text{ and } t \in \mathbf{t}, \qquad \text{(A3)}$$

the infrastructure capacity-expansion constraints

$$\underline{\Delta}\mathbf{q}_{lt}^{a\mathbf{z}} \leq \Delta\mathbf{q}_{lt}^{a\mathbf{z}} \leq \bar{\Delta}\mathbf{q}_{lt}^{a\mathbf{z}} \ \forall\ z \in \mathbf{z},\ a \in \mathbf{a},\ l \in \mathbf{l} \text{ and } t \in \mathbf{t}, \qquad \text{(A4)}$$

and non-negativity constraints associated with the respective variables.

Acknowledgments

This study was partly funded by the Israel Ministry of Agriculture and Rural Development (IMOARD, grant number 12-04-0011) and the Center for Agricultural Economics Research. The authors thank Yael Kachel from IMOARD for providing data. The empirical hydro-economic model developed in this study (code and data) is available as online supplemental materials to the paper.

References

Assouline, S., Russo, D., Silber, A., and Or, D. (2015). Balancing water scarcity and quality for sustainable irrigated agriculture. *Water Resources Research*, *51*(5): 3419-3436.

Ben-Gal, A., Yermiyahu, U., and Cohen, S. (2009). Fertilization and blending alternatives for irrigation with desalinated water. *Journal of Environmental Quality*, *38*(2): 529-536.

Bentsen, M., I. Bethke, Debernard, J.B., Iversen, T., Kirkevag, A., Seland, Ø., Drange, H., Roelandt, C., Seierstad, I.A., Hoose, C. and Kristjansson, J.E. (2013). The Norwegian earth system model, NorESM1-M - Part 1: description and basic evaluation of the physical climate. Geosci Model Dev 6:687–720.

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.

Burn, S., Hoang, M., Zarzo, D., Olewniak, F., Campos, E., Bolto, B., and Barron, O. (2015). Desalination techniques—a review of the opportunities for desalination in agriculture. *Desalination*, *364*: 2-16.

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*, *129*(3): 155-164.

Dasgupta, S., Huq, M., and Wheeler, D. (2016). Drinking water salinity and infant mortality in coastal Bangladesh. *Water Economics and Policy*, *2*(01): 1650003.

Dawson, C. J., and Hilton, J. (2011). Fertilizer availability in a resource-limited world: Production and recycling of nitrogen and phosphorus. *Food Policy*, *36*: S14-S22.

Fatta-Kassinos, D., Kalavrouziotis, I. K., Koukoulakis, P. H., and Vasquez, M. I. (2011). The risks associated with wastewater reuse and xenobiotics in the agroecological environment. *Science of the Total Environment*, *409*(19): 3555-3563.

Finkelshtain, I., and Y. Kachel. 2009. "The Organization of Agricultural Exports: Lessons from Reforms in Israel." In I. Kimhi, and I. Finkelshtain, eds. The

Economics of Natural and Human Resources in Agriculture. NY: Nova Science Publishers, pp. 237–264.

Finkelshtain, I., Y. Kachel, and O. Rubin. 2011. "Policy, Competitiveness and Agricultural Trade: An Application of the Policy Analysis Matrix (PAM) to Israeli Agriculture." Research report. Retrieved from https://www.uni-goettingen.de/de/document/download/837357d09971150feda783d42107fe45.pdf/Finkelshtain%20Kachel%20Rubin%20-%20Policy%20Analysis%20Matrix%20of%20Israeli%20Agriculture%202011%20(1).pdf

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.

Fuchs, H. (2014). The impact of reforms in import taxes of vegetative agricultural products on producers and consumers. Master thesis, The Hebrew University of Jerusalem. http://arad.mscc.huji.ac.il/dissertations/W/AGR/001978297.pdf. (Hebrew).

Gent, P. R., Danabasoglu, G., Donner, L. J., Holland, M. M., Hunke, E. C., Jayne, S. R., Lawrence, D. M., Neale, R. B., Rasch, P. J., Vertenstein, M., Worley, P. H., Yang, Z.-L., and Zhang, M. (2011). The Community Climate System Model Version 4, *J. Climate*, *24*: 4973–4991, 2011.

Gur-Reznik, S., and Dosoretz, C.G. (2015). Viability and reliability of dense membranes in removing trace organic contaminants for wastewater reclamation and purification: pros and cons, mechanisms, and trends. In Environmental Indicators (pp. 805-823). Springer, Dordrecht.

Hadas, E., Tripler, E., Dag, A. and Ben-Gal, A. (2016). The impact of water salinity of farmer's income. *Water and Irrigation*, *542* (Hebrew).

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.

Hendel, I., Lach, S., and Spiegel, Y. (2017). Consumers' activism: the cottage cheese boycott. *The RAND Journal of Economics*, *48*(4): 972-1003.

Housh, M., Ostfeld, A., and Shamir, U. (2011). Optimal multiyear management of a water supply system under uncertainty: Robust counterpart approach. *Water Resources Research*, *47*(10).

Howitt, R. E., MacEwan, D., Medellín-Azuara, J., and Lund, J. R. (2010). Economic modeling of agriculture and water in California using the statewide agricultural production model. *Davis, CA: University of California*.

ICBS (Israeli Central Bureau of Statistics) (2014). Statistical Abstract of Israel 2014, No. 65. The Israeli Central Bureau of Statistics, Jerusalem, Israel.

ICBS (Israel Central Bureau of Statistics) (2017). Food Balance for 2015. Online, accessed December 2018: http://www.cbs.gov.il/publications17/1689_food_balance_2015/pdf/h_print.pdf. (Hebrew).

IMNI (Israel Ministry of National Infrastructures) (1959). Water law. http://www.nevo.co.il/law_html/Law01/235_001.htm (Hebrew).

IWA (Israel Water Authority). (2012). Characterization of the contribution of the domestic sector to the sewage salinity. Online, accessed December 2018: http://www.water.gov.il/Hebrew/ProfessionalInfoAndData/Water-Quality/DocLib1/home-seage-salt.pdf. (Hebrew).

IWA (Israeli Water Authority) (2016). Water consumption by purpose 1998-2015. http://www.water.gov.il/Hebrew/ProfessionalInfoAndData/Allocation-Consumption-and-production/20156/matrot%20kolchin.pdf. (Hebrew).

Jimenez, B., and T. Asano (2008). Water Reuse: An international survey of current practice, issues and needs, Tech. Rep. 20, Int. Water Assoc., London, U. K.

Kan, I., and Rapaport-Rom, M. (2012). Regional blending of fresh and saline irrigation water: Is it efficient?. *Water Resources Research*, 48(7).

Kan, I., K.A., Schwabe and K.C. Knapp. (2002). Microeconomics of irrigation with saline water. *Journal of Agricultural and Resource Economics*, *27*(1): 16-39.

Kasprzyk-Hordern, B., Dinsdale, R. M., and Guwy, A. J. (2009). The removal of pharmaceuticals, personal care products, endocrine disruptors and illicit drugs during wastewater treatment and its impact on the quality of receiving waters. *Water Research*, *43*(2): 363-380.

Kemp, A. (2010). Reforming policies on foreign workers in Israel.. OECD Social, Employment and Migration Working Paper no. 103.

Kimhi, A. (2016). Expected consequences of the cancellation of the mechanism for foreign-workers allocation to agriculture. Israel Ministry of Agriculture and Rural Development. Policy Paper. https://www.moag.gov.il/yhidotmisrad/research_economy_strategy/publication/2016/Documents/ovdim%20zarim%202016.pdf. (Hebrew).

Kislev, Y. (2011). The water economy of Israel. Taub Center for Social Policy Studies in Israel. Policy Paper No. 2011.15. http://taubcenter.org.il/wp-content/files_mf/thewatereconomyofisrael.pdf.

Knapp, K. C. (1992). Irrigation management and investment under saline, limited drainage conditions: 1. Model formulation. *Water Resources Research*, *28*(12): 3085-3090.

Lavee, D., Ritov, M. and Becker, N. (2011). Is desalination the most sustainable alternative for water-shortage mitigation in Israel?. *International Journal of Sustainable Economy*, *3*(4): 410-424.

Maas, E.V. and G.J. Hoffman. (1977). Crop salt tolerance - current assessment. *Journal of the Irrigation and Drainage Division, American Society of Civil Engineers*, 103: 115-134.

Martinez-Alvarez, V., Martin-Gorriz, B., and Soto-García, M. (2016). Seawater desalination for crop irrigation—A review of current experiences and revealed key issues. *Desalination*, *381*: 58-70.

Messer, K.D., Costanigro, M. and Kaiser, H.M. (2017). Labeling food processes: The good, the bad and the ugly. *Applied Economic Perspectives and Policy*, *39*(3): 407-427.

Nellemann, C., M. M. Devette, and T. Manders (2009), The Environmental Food Crisis: The Environment's role in averting future food crisis: A UNEP rapid response assessment, U. N. Environ. Program. of the U. N., Nairobi, 101 pp.

Nordhaus, W.D. (2007). A review of the "Stern review on the economics of climate change". *Journal of Economic Literature*, *45*(3): 686-702.

OECD. (2018). Economic survey, Israel. Online, accessed December 2018: http://www.oecd.org/israel/economic-survey-israel.htm.

Qadir, M., E. Quill_erou, V. Nangia, G. Murtaza, M. Singh, R. J. Thomas, P. Drechsel, and Noble, A.D. (2014). Economics of salt-induced land degradation and restoration, Nat. Resour. Forum, 38, 282–295.

Qadir, M., D. Wichnels, L. Raschid-Sally, P. Singh Minhas, P. Drechsel, A. Bahri, and P. McCornick (2007), Agricultural use of marginal-quality water and challenges, in Water for Food; Water for Life. A Comprehensive Assessment of Water Management in Agriculture, edited by D. Molden, pp. 425, Earthscan, London, U. K.

Reznik, A., Feinerman, E., Finkelshtain, I., Fisher, F., Huber-Lee, A., Joyce, B., and Kan, I. (2017). Economic implications of agricultural reuse of treated wastewater in Israel: A statewide long-term perspective. *Ecological Economics*, *135*: 222-233.

Reznik, A., Feinerman, E., Finkelshtain, I., Kan, I., Fisher, F., Huber-Lee, A., and Joyce, B. (2016). The cost of covering costs: A nationwide model for water pricing. *Water Economics and Policy*, *2*(04): 1650024.

Shani, U., Ben-Gal, A., Tripler, E., and Dudley, L. M. (2007). Plant response to the soil environment: An analytical model integrating yield, water, soil type, and salinity. *Water Resources Research*, *43*(8).

Tanaka, S.K., Zhu, T.J., Lund, J.R., Howitt, R.E., Jenkins, M.W., Pulido, M.A., Tauber, M., Ritzema, R.S., Ferreira, I.C., (2006). Climate warming and water management adaptation for California. *Climate Change*, *76*: 361–387.

Tanji, K. K., & Kielen, N. C. (2002). Agricultural drainage water management in arid and semi-arid areas. FAO.

Tarchitzky, J., Rimon, A., Kenig, E., Dosoretz, C.G., and Chen, Y. (2013). Biological and chemical fouling in drip irrigation systems utilizing treated wastewater. *Irrigation Science*, *31*(6): 1277-1288.

Van Genuchten, M. T. and G. J. Hoffman. (1984). Analysis of crop salt tolerance data. In Soil Salinity under Irrigation, Processes and Management, eds. I. Shainberg and J. Shalhevet, 258-271 (Ecological Studies, 51). Berlin: Springer.

Watanabe, M., T. Suzuki, R. O'ishi, Y. Komuro, S. Watanabe, S. Emori, T. Takemura, M. Chikira, T. Ogura, M. Sekiguchi, K. Takata, D. Yamazaki, T. Yokohata, T. Nozawa, H. Hasumi, H. Tatebe, and M. Kimoto, (2010). Improved climate simulation by MIROC5: Mean states, variability, and climate sensitivity. *J. Climate*, *23*: 6312–6335.

Weinberger, G., Livshitz, Y., Givati, A., Zilberbrand, M., Tal, A., Weiss, M., and Zurieli, A. 2012. The natural water resources between the Mediterranean Sea and the Jordan River. Israel Hydrological Service, Jerusalem.

Welle, P. D., Medellín-Azuara, J., Viers, J. H., and Mauter, M. S. (2017). Economic and policy drivers of agricultural water desalination in California's central valley. *Agricultural Water Management*, *194*, 192-203.

Yermiyahu, U., Tal, A., Ben-Gal, A., Bar-Tal, A., Tarchitzky, J., and Lahav, O. (2007). Rethinking desalinated water quality and agriculture. *Science*, *318*(5852), 920-921.