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Large-Scale Desalination and the External Impact on Irrigation-Water Salinity: Economic Analysis for the Case of Israel

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Key Points:

- Subsidizing desalination is socially warranted because the associated irrigation-water salinity reduction is an external effect of water consumption by all sectors
- Irrigation with desalinated water is found optimal for Israel due to the large share of water-intensive salinity-sensitive high-value crops
- Ignoring the damage entailed by irrigation-water salinity benefits water suppliers at the expense of consumers of water and agricultural products
Large-Scale Desalination and the External Impact on Irrigation-Water Salinity: Economic Analysis for the Case of Israel

Abstract

Recent agro-economic studies in water-scarce countries such as Spain, Australia, Saudi Arabia, and Israel have revealed the economic viability of irrigating high-value crops with desalinated water. However, the worldwide growth of large-scale desalination capacities is primarily designed to resolve urban-water scarcity, disregarding the impact of desalination on irrigation-water salinity. We develop a dynamic hydro-economic programming model where infrastructure capacities and allocations of water quantities and salinities in a regional water-distribution network are endogenous. We show that subsidizing desalination is socially warranted because the associated reduction in irrigation-water salinity is an external effect of water consumption by all water users. Empirical application to the entire state of Israel indicates that large-scale desalination of seawater and treated wastewater for irrigation is optimal. This result stems from the large share of irrigation-intensive salinity-sensitive high-value crops, motivated by Israel’s policies to support local agriculture. Ignoring salinity results in a 45% reduction in desalinated irrigation water, a 29% reduction in farming profits, a 250% increase in water-suppliers’ profits, and an average deadweight loss of $1,200 a year per hectare of arable land.

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

JEL Classification: Q15, Q25, Q28
1. Introduction

Salts already affect about a third of the world’s arable areas, and a fifth of the world’s 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 & 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 spatially and intertemporally linked processes implies that salinity damage should be considered intrinsic to the design of agricultural and water policies, as well as to the long-run development of water infrastructures at the regional and state levels, in particular desalination.

Worldwide desalination capacity is continuously growing, and is expected to be nearly double by 2050 relative to the 2015 level of 97 × 10^6 (m^3 day^-1) (Darre & Toor, 2018). Nevertheless, due to high supply costs, adverse environmental effects, and agronomic constraints, large-scale desalination is generally perceived as a technology for relieving urban-water scarcity rather than agricultural water shortages and salinity damage (Burn et al., 2015; Martínez-Alvarez et al, 2016; Sepehr et al., 2017). However, recent field- and farm-level analyses have indicated that desalination can provide an economically viable water source for irrigating high-value crops in water-scarce countries such as Spain (Reca et al., 2018; Zarzo et al., 2013), Australia (Barron et al., 2015), Saudi
Arabia (Multsch et al., 2017), and Israel (Hadas et al., 2016; Kaner et al., 2017). Thus, the tendency to overlook the potential contribution of desalination to agricultural productivity in the design of policies and water infrastructures may entail suboptimal water management and deadweight loss.

Water economies at the basin, regional or state levels may comprise multiple water sources with diverse qualities and supply costs (e.g., naturally recharged surface water and groundwater, brackish water, desalinated seawater, and TWW), various consumers (e.g., domestic, industrial, and agricultural users) with different demands for water quantity and quality, and an ensemble of infrastructures that constitute a complex water-distribution network to treat and convey water from sources to consumers, separated or blended. Efficient management of such networks should account for the linkage of scarcities across points in time and space, meaning that water use at a particular place and time may have opportunity costs with respect to its usage for various purposes at alternative locales and times (Fisher et al., 2005). Moreover, population growth, climate change, and the long construction period for water infrastructures imply that water-system designers should foresee scarcities and decide ahead of time which, when, where, and to what extent infrastructural elements should be installed or extended.

Hydro-economic models are an efficient tool for integrating the economic, hydrological, engineering, agronomic, and environmental aspects associated with water policies and water-infrastructure blueprints of large-scale water systems (Booker et al., 2012; Harou et al., 2009). However, despite the vast economic literature on irrigation-water salinity (Connor et al., 2012; Feinerman & Yaron, 1983; Knapp, 1992; Letty & Dinar, 1986; and Schwabe et al., 2006 are but a few examples for topics and modeling
methods developed in the last decades), to date, the incorporation of salinity in hydro-economic models has been limited. For example, Housh et al. (2011), Lavee et al. (2011), Reznik et al. (2016), Saglam (2013) and Tanaka et al. (2006) refer to desalination as a means of resolving water-quantity scarcities, overlooking 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 impact into a static analysis of agricultural irrigation-water management in the Central Valley of California. However, that framework considered agriculture as a single water consumer, and therefore did not capture interlinks with domestic and industrial water users, for example, when agriculture consumes the TWW produced by urban regions. In addition, economies of scale associated with water conveyance warrant a mixture of water sources with different qualities for long-distance delivery. The economics of blending irrigation-water sources with diverse salinities has been studied at the field and regional levels (Dinar et al., 1986; Kan, 2008; Kan & Rapaport-Rom, 2012; Kan et al., 2002); however, to our knowledge, less attention has been given to the policy and welfare implications of salinity in spatially widespread water-distribution networks that supply mixed water to both agricultural and urban sectors.

The objective of this paper is to assess the welfare contribution of large-scale desalination through its impact on irrigation-water salinity. Specifically, we compare optimal water management at the regional scale under two scenarios: in the first, termed *Salinity Internalized* (SI), the salinity effects are considered in the design of water infrastructures, whereas the second, *Salinity Externalized* (SE), presumes that temporal and spatial changes in salinity are disregarded.
The following section presents a general dynamic hydro-economic model that optimizes water allocations and infrastructure development while accounting for water salinity and its agricultural damage. Based on an analytical solution, we show that the salinity reduction following desalination is an external effect of water consumption by all sectors, which can be internalized by subsidizing the supplied water. Otherwise, disregarding salinity leads to higher-than-optimal water prices and welfare loss, where water suppliers benefit at the expense of urban and agricultural water users.

In Section 3 we develop an empirical version of the model based on the Israeli water system. The case of Israel seems to correspond to the assertion that: “…for water-scarce countries that already implement all other measures for freshwater generation, desalination may serve as the only viable means to provide the water supply necessary to sustain agriculture, support population, and promote economic development” (Elimelech & Phillip, 2011, p. 717). Indeed, to cope with its growing water scarcity, Israel has established a complex water-supply system, incorporating extraction of freshwater, agricultural reuse of TWW, and extensive desalination of brackish water and seawater. The empirical model tracks the dynamics of water salinity throughout the water-delivery system, and controls the supplied water salinity by both the allocation and mixture of water from different sources and the establishment of plants for the desalination of seawater, brackish water and TWW. The model also comprises the farmers’ adaptation to irrigation-water salinity through land allocation across crops in relation to the crops’ salt tolerance.

Section 4 presents the empirical results. For the SI scenario, we obtain that the negative effect of irrigation-water salinity on agricultural production in Israel warrants
large-scale desalination of seawater and TWW for irrigation. We attribute this result to the large share of high-value, salinity-sensitive fruit and vegetables in Israel’s vegetative agriculture, where the Israeli farmers’ specialization in the production of these crops is motivated by supporting regulations. We then show that disregarding the salinity effects in the design of water infrastructures yields a considerable deadweight loss, evaluated at $1,200 per average hectare of cultivable land. This estimate, which accounts for the indirect salinity effects on domestic water supply and agricultural output prices, is nearly three times larger than the direct salinity damage to agriculture of $440 ha$^{-1}$ year$^{-1}$ reported by Qadir et al. (2014). Section 5 summarizes with a discussion of the model’s limitations and extensions.

2. General Framework

We first present a general dynamic hydro-economic model; then, to obtain theoretical insights, we solve the optimization problem of a simplified static version analytically.

2.1. Hydro-economic Model

Consider a water-distribution network that is centrally managed by a water master whose objective is to maximize welfare throughout a planning horizon of $T$ discrete periods by setting efficient prices and developing infrastructures. Let $\mathcal{K}$ be a set of nodes representing spatially linked water sources, treatment infrastructures, and water-use regions, and $q^k_t = (w^k_t, s^k_t)$ denote the pair of quantity ($w^k_t$) and salinity ($s^k_t$) of the water flowing through node $k \in \mathcal{K}$ during period $t$ ($t = 1, ..., T$). The elements in $q^k_t$ are
constrained by the vector of exogenous factors $\mathbf{b}^k_t$ (i.e., a volumetric balance constraint in a pipeline junction, the water salinity in groundwater sources, etc.), $\mathbf{q}^k_t \leq \mathbf{b}^k_t$, where $\mathbf{b}^k_t = f^k_t (\mathbf{b}^k_{t-1}, \mathbf{q}^k_{t-1})$ represents the intertemporal dependence between water flows and constraints (e.g., the extractable aquifer stock depends on the initial stock and extraction during the previous period). A related set of constraints, $\mathbf{b}^k_{t+1} \geq \mathbf{b}^k_t$, represents end conditions; for example, minimum extractable stocks in an aquifer.

The vector $\mathbf{q}^k_t$ is also constrained by the node’s capacity attributes $\mathbf{z}^k_t$ (e.g., the maximum annual volume and minimum salinity level of a desalination plant’s outflow), which are endogenous. We denote by $\Delta \mathbf{z}^k_t$ the change in the capacity features of node $k$ during period $t$, such that, given the initial capacity $\mathbf{z}^k_0$, $\mathbf{z}^k_t = \mathbf{z}^k_0 + \sum_{t=1}^{t-1} \Delta \mathbf{z}^k_t$. The total per-period costs associated with the capacities incorporated in $\mathbf{z}^k_t$ (i.e., maintenance and capital costs) are denoted $C^k_t (\mathbf{z}^k_t)$.

We denote by $V^k_t (\mathbf{q}^k_t, \mathbf{x}^k_t)$ the value associated with water flow through node $k \in \mathbb{K}$ during period $t$, where $\mathbf{x}^k_t$ is a vector of non-water input variables (e.g., land allocation to crops) which may be constrained from above by $\mathbf{x}^k_t$ (e.g., regional agricultural land). The value function $V^k_t (\cdot)$ may vary in time (e.g., due to population growth or climate change), and it is negative when node $k$ is a water-treatment infrastructure (e.g., the variable costs of a groundwater-pumping station, or a desalination or wastewater-treatment plant), and positive if the node is an urban or agricultural water-consumption region.

Let $\mathbf{Q}$, $\mathbf{X}$, $\mathbf{B}$ and $\mathbf{Z}$ be the vectors incorporating $\mathbf{q}^k_t$, $\mathbf{x}^k_t$, $\mathbf{b}^k_t$ and $\mathbf{z}^k_t$ for all $k \in \mathbb{K}$ at all the $T$ periods, respectively, and the vector $\Delta \mathbf{Z}$ represent all the capacity changes throughout the planning horizon. Given the initial levels of the exogenous and capacity constraints for all $k \in \mathbb{K}$, $\mathbf{B}_0$ and $\mathbf{Z}_0$, the planner’s objective is to solve the problem
\[
\max_{Q, \Delta Z, X} \mathbb{W} = \sum_{t=1}^{T} (1 + r)^{-t} \sum_{k \in \mathbb{K}} \left[ V_t^k (q_t^k, x_t^k) - C_t^k (z_t^k) \right]
\]

\text{s.t. } \begin{aligned}
& b_t^k = f_t^k (b_{t-1}^k, q_{t-1}^k) \quad \forall \ k \in \mathbb{K}, t = 1, \ldots, T, \\
& z_t^k = z_0^k + \sum_{i=t}^{t-1} \Delta z_i^k \quad \forall \ k \in \mathbb{K}, t = 1, \ldots, T, \\
& x_t^k \leq \bar{x}_t^k \quad \forall \ k \in \mathbb{K}, t = 1, \ldots, T, \\
& Q \leq \min (B, Z), B_{T+1} \geq B, Q, \Delta Z, X \geq 0
\end{aligned}

(1)

where \( \mathbb{W} \) denotes welfare and \( r \) is the discount rate. The solution defines the optimal infrastructure-development plan and allocation of water throughout the distribution network, induced by water quotas or prices. In the SI scenario, the water master accounts for the impact of salinity changes embedded in the value function \( V_t^k (q_t^k, x_t^k) \) for all \( k \in \mathbb{K} \) and \( t = 1, \ldots, T \), whereas the SE case implies \( s_t^k = s_0^k \) for all \( k \in \mathbb{K} \) and \( t = 1, \ldots, T \).

### 2.2. Theoretical Insights

To obtain an analytical solution, we consider a simplified static version of the optimization problem expressed in equation (1). Figure 1 depicts a water economy in which groundwater and desalinated seawater are two water sources with different salinities, which are mixed and then delivered to two water-consumption zones—urban and agricultural.

![Figure 1](image)

We denote by \( s^g \) and \( s^d \) the salinities of the groundwater and desalinated water, respectively, where \( s^g > s^d \). These salinity levels constitute minimum constraints for the salinities of the water supplied by the respective sources: \( s^g \geq s^g \) and \( s^d \geq s^d \). 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, \( w = w^d + w^g \), is a mixture of the two sources with a salinity that is the volumetric weighted average
\[ s = \frac{w^d s^d + w^g s^g}{w} \] The blended water is split such that the urban and agricultural sectors obtain amounts \( w^u \) and \( w^a \), respectively, and \( w^u + w^a = w \).

We assume constant per-water-unit marginal costs, where the marginal cost of desalination, \( c^d \), exceeds that of the groundwater, \( c^g \). Unlike groundwater, desalination is also associated with fixed cost \( C^d \), contingent on the supply of desalinated water. The functions \( V^u(w^u) \) and \( V^a(w^a, s) \) are, 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., \( V^u(w^u) > 0 \) and \( V^a(w^a, s) > 0 \)), whereas water salinity is harmful only to agriculture (\( V^u(w^u) = 0 \) and \( V^a(w^a, s) < 0 \)).

A benevolent water authority manages water centrally by setting the capacities of groundwater extraction and desalination, as well as the administrative water prices at the consumption zones. As already noted, under the SI scenario, the planner solves the problem in equation (1) which, with simplifying specifications, becomes

\[
\begin{align*}
\max_{w^d, w^g, w^u, w^a, s^d, s^g} \quad & W = V^u(w^u) + V^a(w^a, s) - c^d w^d - c^g w^g - C^d \\
\text{s.t.} \quad & w = w^u + w^a = w^d + w^g, \quad w^g \leq \bar{w}^g, \\
& s^d \geq \underline{s}^d, \quad s^g \geq \underline{s}^g, \quad s = \frac{w^d s^d + w^g s^g}{w} \\
& w^d, w^g, w^u, w^a, s^d, s^g \geq 0
\end{align*}
\]

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

\[ p^{SI} = V^u_{w^u}(w^u) = V^a_{w^a}(w^a, s) \]

implying that a single optimal administrative water price (denoted \( p^{SI} \) to indicate the SI scenario) should be set for both consumption sectors. Optimization with respect to \( w^g \) and \( w^d \) subject to the condition in equation (3) yields
\[ p^{SI} = c^d - V_s^a(w^a, s) \frac{w^g(s^d - s^g)}{(w_{SC})^2} \]  \hspace{1cm} (4)

\[ p^{SI} = c^g + V_s^a(w^a, s) \frac{w^d(s^d - s^g)}{(w^{SI})^2} + \lambda^g \]  \hspace{1cm} (5)

where \( w^{SI} \) denotes the total water consumption under price \( p^{SI} \), and \( \lambda^g \) is the shadow value of the groundwater constraint \( w^g \leq \bar{w^g} \). Taken together, equations (4) and (5) yield the efficiency condition underlying the price \( p^{SI} \)

\[ \frac{V_s^a(w^a, s)}{w^{SI}} (s^d - s^g) = c^d - (c^g + \lambda^g) \]  \hspace{1cm} (6)

which requires the (left-hand-side) per-water-unit agricultural benefits associated with reducing salinity from \( s^g \) to \( s^d \) to be equal to the (right-hand-side) difference between the per-water-unit supply (plus scarcity) costs of the two sources.

Equations (4) and (5) imply \( c^d \geq p^{SI} \geq c^g \). Equation (4) expresses the price of the supplied blended water in relation to the supply of desalinated water. The term

\[ V_s^a(w^a, s) \frac{w^g(s^d - s^g)}{(w^{SI})^2} \] in equation (4) reflects the marginal benefits accrued to the agricultural sector by the decline in salinity of the mixed water due to the last unit of desalinated water. However, from the perspective of any urban or agricultural water consumer, this is an external effect, as she or he overlooks the salinity reduction associated with the consumption of an additional unit of desalinated water. Therefore, efficiency requires internalizing this external effect as a subsidy for the desalinated water incorporated in the supplied blended water. The subsidy incentivizes the consumption of desalinated-water quantities beyond the level equating the water price to the marginal desalination cost \( c^d \). Clearly, supplying the water quantity demanded under price \( p^{SI} \) requires the establishment of a sufficiently large desalination capacity.
The optimal price \( p^{SI} \) is also obtained in equation (5) by inspecting the effect of the groundwater incorporated in the supplied blended water. The term \( V_s^a(w^a, s) \frac{w^d(s^g - s^d)}{\langle w^{SI} \rangle^2} \) in equation (5) represents the marginal damage caused to agriculture by an additional unit of groundwater due to the increased salinity of the mixed water. This is a negative external effect, which should be introduced as a tax to all water consumers in order to render the optimal groundwater supply lower than that equating the water price to \( c^g \) (even under an ineffective groundwater constraint; i.e., \( \lambda^g = 0 \)).

The optimal solution is illustrated schematically in Figure 2a. It is assumed that groundwater is supplied first at the marginal cost \( c^g \), and water deliveries beyond the groundwater constraint \( \bar{w}^g \) require desalination at the marginal cost \( c^d \). The brown dashed curve denoted \( V^u_w(w^u) \) is the inverse urban-water demand. The solid pink curve marked by \( V_w(w, s) \) represents the inverse aggregate demands of the urban and agricultural sectors. At \( \bar{w}^g \), the curve \( V_w(w, s) \) kinks upward due to the increase in agricultural produce resulting from the reduction in the blended water’s average salinity \( s \), which occurs under positive supplies of desalinated water. The solid green curve \( V_w(w, s^g) \) illustrates the inverse aggregate demand for the (hypothetical) case in which the salinity of the blended water is unaffected by desalination, and remains similar to that of the groundwater. The optimal conditions in equations (3), (4), and (5) imply that the optimal water supply, \( w^{SI} \), is set such that \( V_w(w^{SI}, s) = c^d \). Then, the water price can be computed based on the equality \( p^{SI} = V_w(w^{SI}, s^g) \).

Figure 2 about here
Are the variable water-supply costs covered under the SI optimal price $p^{SI}$? Based on equation (5), the cost-recovery condition with respect to the variable costs, $p^{SI}w^{SI} \geq c^d w^d + c^g w^g$, becomes
\[
\frac{V^a_s(w^a,s)}{w^{SI}}(s^d - s^g) \geq c^d - \left( c^g + \lambda^g + \lambda^g \frac{w^g}{w^d} \right)
\]
which, in view of equation (6), indicates that the surplus of water suppliers is positive provided that $\lambda^g \frac{w^g}{w^d} > 0$; that is, provided that $w^g = \bar{w}^g > 0$.

The economic implications of changes in the salinities of the water sources can be learned from the shadow values of the respective desalination and groundwater salinity constraints $s^{d} \geq s^{d}$ and $s^{g} \geq s^{g}$, which are defined by $\lambda^{sd} = V^a_s (w^a,s) \frac{w^d}{w^{SI}}$ and $\lambda^{sg} = V^a_s (w^a,s) \frac{w^g}{w^{SI}}$, respectively. Thus, 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.

Turning to the SE scenario, suppose that the groundwater source is exhausted (i.e., $w^g = \bar{w}^g$), and desalinated water is supplied ($w^d > 0$); alas, the water master refers to desalination as an instrument for resolving water-quantity scarcity while ignoring 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 (denoted $p^{SE}$ for the SE case) according to equations (4) and (5), the terms
\[
V^a_s (w^a,s) \left( \frac{w^g (s^d - s^g)}{(w^d + w^g)^2} \right) \quad \text{and} \quad V^a_s (w^a,s) \left( \frac{w^d (s^d - s^g)}{(w^d + w^g)^2} \right)
\]
respectively, are dropped; hence, $p^{SE} = c^d$, and equation (5) implies $\lambda^g = c^d - c^g$. Nevertheless, the actual salinity of the mixed
water, $s$, will be lower than $s^g$ when desalinated water is supplied, and the actual levels of $w^d$ and $w^a$ can be computed by the conditions

$$c^d = V_w^u(w^d + \bar{w}^g - w^a) = V_w^a(w^a, s) + V_s^a(w^a, s) \frac{\bar{w}^g(s^d - s^g)}{(w^d + \bar{w}^g)} \tag{8}$$

which follow from equations (3) and (4), together with the water-balance constraint $w^u = w^d + \bar{w}^g - w^a$.

Figure 2b illustrates the SE solution. The policy maker mistakenly refers to the (green) curve $V_w(w, s^g)$ as the inverse aggregate demand, and therefore sets $p^{SE} = V_w(w, s^g) = c^d$. Since $p^{SE} > p^{SI}$, the desalination capacity and the surpluses of the urban and agricultural consumers are lower than their SI counterparts, whereas the surplus of the water suppliers is positive, and equals $(c^d - c^g)\bar{w}^g$. The area marked $F$ indicates the deadweight loss associated with ignoring the desalination impact on salinity.

So far, we have assumed that desalination is warranted under both scenarios; that is, the welfare with desalination exceeds the welfare without it. It may well be that this condition holds under the SI scenario, but not under that of SE, as illustrated in Figure 2c. The total water-consumption value associated with desalination under the SI scenario (i.e., the area of the dashed-yellow-line trapezoid underneath the (pink) inverse aggregate demand curve $V_w(w, s)$ in the segment $w \in [\bar{w}^g, w^{SI}]$) exceeds the total desalination cost (the area of the pink rectangle, $c^d(w^{SI} - \bar{w}^g) + C^d$). In the SE case, however, the total consumption value erroneously perceived by the water master (the area of the dashed-brown-line trapezoid below the (green) inverse aggregate demand curve $V_w(w, s^g)$ in the segment $w \in [\bar{w}^g, w^{SE}]$) is smaller than the respective desalination costs (the area of the green rectangle, $c^d(w^{SE} - \bar{w}^g) + C^d$). Therefore, in the SE case, the water master would reject desalination altogether, and may set a market-clearing water price according
to \( p^{SE} = V_w(\bar{w}^g, s^g) \). The deadweight loss under SE increases relative to the case in which the desalination infrastructure is warranted or already in place, as assumed in Figure 2b.

To conclude, we show that overlooking the effect of desalination on the supplied-water salinity leads to higher water prices, lower desalination, and lower consumer surplus, whereas the water suppliers’ profits increase. Moreover, the establishment of desalination infrastructures may be wrongly perceived as economically unwarranted. Thus, in a dynamic framework, where water demands increase over time, the development rate of desalination capacities will be slower under SE than under SI.

3. **Empirical Model of Israel’s Water Economy**

Our objective is to evaluate empirically the economic implications of ignoring the salinity impact of large-scale desalination in the design of Israel’s water economy.

3.1. **Israel’s Water Economy**

Israel faces three water-management challenges: 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 water scarcity is steadily increasing due to rapid population growth (Kislev, 2011).
Israel has confronted these challenges by establishing a complex water-distribution network that 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. Severe water scarcity motivated the exploitation of brackish water and TWW for irrigation, and in recent years, extensive seawater desalination has been employed (Burn et al., 2015; Martínez-Alvarez et al., 2016) (see Appendix A). However, desalination capacities were primarily designed to meet the growing water demand for domestic uses, whereas the impact of desalination on irrigation-water salinity was not considered explicitly (see Israel Water Authority, 2012a).

3.2. Empirical Model

By virtue of the Israeli Water Law (Israel Ministry of National Infrastructures, 1959), water is a public property that is centrally managed by the government, which designs water-supply infrastructures and sets administrative water prices and quotas to control consumption; accordingly, the model’s objective is to characterize the optimal levels of these policy instruments from the point of view of a welfare-maximizing benevolent government.

Our model is based on a partial-equilibrium setup, integrating a water-distribution module and an agriculture module, as depicted schematically in Figure 3. Adopting a holistic approach (Brouwer & Hofkes, 2008), the variables of both modules are solved simultaneously. The water-distribution module determines the allocation of water from sources to regions of urban and agricultural consumers, sets the development trajectories of infrastructure capacities, and computes the salinities, the shadow values of constraints,
and (assuming price-based water allocations) the efficient water prices throughout the
water-distribution network during the simulated period. The agriculture module
characterizes the optimal allocation of agricultural land across crops in each of the
agricultural regions subject to constrained regional land, foreign-labor quotas, and the
quantity and salinity of the water allocated to that region by the water-distribution
module. The irrigation water’s salinity affects crop yields in relation to crops’ tolerance
to salinity. In addition, the agricultural module includes demand functions for vegetative
agricultural products, such that the prices of those products constitute equilibria in the
local markets for fresh fruit and vegetables—the latter protected by import tariffs.

Figure 3 about here

We calibrate the model to a base year (2015) and run it for a 30-year period. The
demands for urban water and agricultural products shift along the years in relation to a
projected population growth—this incentivizes the extension of water-supply
infrastructures, particularly the supply of desalinated water, which affects the irrigation-
water salinity and thereby yields optimal agricultural land allocation, equilibrium prices
and quantities in the different markets for vegetative agricultural products, and the
allocation of surpluses among the urban-water users, farmers, consumers of agricultural
products, and water suppliers.

The model is programmed in GAMS (General Algebraic Modeling System), with
links to input-data and output-data Excel files (see supplemental materials), and solves
simultaneously for all variables throughout the entire planning horizon. The GAMS code
describes the model components in detail; here, we outline the main empirical elements.
3.2.1. Water-Distribution Module

The water-distribution module extends the hydro-economic model MYWAS (Multi-Year Water Allocation System) (Fisher & Huber-Lee, 2011; Reznik et al., 2017), which is an Israeli multiyear version of the one-period WAS (Water Allocation System) model developed in the 1990s for the areas of Israel, Jordan and the Palestinian Authority (Fisher et al., 2005). Appendix B presents schemes of the water network embedded in MYWAS. There are 19 naturally enriched freshwater stocks (15 aquifers, the Sea of Galilee, and 3 regional arrays of artificial reservoirs), 5 seawater-desalination plants in the center of the country, 4 non-enriched brackish-water aquifers which are connected to desalination plants, 18 wastewater-treatment plants (WWTPs) that apply secondary treatment, and 1 which applies tertiary treatment to the sewage of the Tel Aviv metropolis. Water users include 21 urban regions and 18 agricultural zones. Urban zones obtain freshwater for domestic and industrial uses from natural freshwater sources, desalinated seawater and desalinated brackish water, and generate sewage outflows that undergo mandatory treatment at the WWTPs. The agricultural zones can consume irrigation water from all sources. The prevailing base-year water system includes 163 freshwater pipelines and 74 pipelines for wastewater and brackish water.

We introduce infrastructures that are nonexistent in the base year, the timing and extent of which can be set as determined endogenously; these include an optional desalination plant in the northwest region of Acre, 12 pipelines that connect seawater-desalination plants directly to agricultural areas located close to the Mediterranean shoreline, and deliveries of desalinated TWW to agricultural zones (which necessitates tertiary wastewater pretreatment). For the wastewater-desalination activities we introduce
an output/input rate parameter of 0.85, meaning that the brine needed to be disposed of constitutes 15% (Pérez-González et al., 2012). We also allow discharge of TWW from WWTPs to the environment (at zero cost or benefit), conditional on being reclaimed to the level of tertiary treatment to meet regulated quality standards. Together with these potential activities, the water network includes 118 nodes, connected by 289 pipelines.

The extractions from naturally enriched freshwater sources are limited by stock constraints. Let \((j = 1, ..., 19)\) denote a freshwater-source node, and \(w_t^j\) (m\(^3\) year\(^{-1}\)) be the extraction from source \(j\) during year \(t\). The amount \(w_t^j\) is limited by the extractable stock \(\bar{b}_t^j\) (m\(^3\) year\(^{-1}\)), which is a state variable defined by \(\bar{b}_t^j = \max(0, R_{t-1}^j - \bar{b}_{t-1}^j - e_{t-1}^j)\), where \(R_{t-1}^j\) (m\(^3\) year\(^{-1}\)) is the previous year’s recharge, and \(e_{t-1}^j = \max(0, \bar{b}_{t-1}^j + R_{t-1}^j - w_{t-1}^j - \bar{b}_t^j)\) (m\(^3\) year\(^{-1}\)) is the spillover from the source to the environment when the stock exceeds its maximum extractable content \(\bar{b}_t^j\) (m\(^3\) year\(^{-1}\)). In addition, we set a minimum extractable stock for the last period, \(\bar{b}_{T+1}^j\), such that \(\bar{b}_{T+1}^j \geq b_{T+1}^j\).

The water salinity is defined for each pipeline as the average salinities of the inflows into the pipe, weighted by the respective inflow volumes. The source salinities are 1 dS m\(^{-1}\) for freshwater from naturally enriched sources, 0.25 dS m\(^{-1}\) for desalinated water (Yermiyahu et al., 2007), and 4 dS m\(^{-1}\) for brackish water. The salinity of freshwater pumped from the Sea of Galilee decreases linearly with the lake’s water stock. Pursuant to regulations, the salinity of urban inflows is constrained to not exceed 1.4 dS m\(^{-1}\). The salinity of the sewage produced in any urban region is 0.7 dS m\(^{-1}\) higher than the salinity of the inflow into that region (Israel Water Authority, 2012b).

Variable and fixed water-supply costs of all infrastructural elements are taken from Reznik et al. (2017). Note that the desalination cost of TWW ($0.63 m^3 on average) is
higher than that of seawater ($0.56 \text{ m}^3$) due to the multiple TWW-desalination pretreatments and the need to dispose of the brine produced by plants located away from the seacoast. To define the benefits of urban-water users we denote by $i$ ($i = 1, \ldots, 21$) an urban-region node, and by $w_t^i$ ($\text{m}^3 \text{ year}^{-1}$), the respective water consumption. The total benefits associated with $w_t^i$ equal the integrated inverse constant-elasticity demand function $V_t^i(w_t^i) = (\eta + 1)^{-1} u_t^i(w_t^i)^{\eta+1} \text{ ($\text{year}^{-1}$)}$, where $\eta^{-1}$ is the demand elasticity and $u_t^i$ is computed for any year $t$ by $u_t^i = p_t^0(w_0^i(1 + \rho)^t)^{-\eta}$, wherein $\rho$ is the population-growth rate and $p_t^0$ ($\text{m}^3$) is the observed base year ($t = 0$) urban-water price. The parameter $\eta = -0.1$ is taken from Bar-Shira et al. (2007).

3.2.2. Agriculture Module

Our analysis incorporates the 55 main crops grown by the Israeli agricultural sector on about 300,000 ha, supplying vegetative products to the processing industry, export, and the local fresh-produce markets. Nearly 60% of the arable area is devoted to high-value fruit and vegetable crops (Israel Central Bureau of Statistics, 2019), requiring intensive irrigation and a large labor force; the latter is highly dependent on foreign workers (Kemp, 2010), who are allocated to farmers based on cropping-acreage declarations. Whereas most of the field-crop outputs (particularly grains) are exempted from import tariffs, the production of fruit and vegetables is protected by import tariff-rate quotas (Finkelshtain & Kachel, 2009; Finkelshtain et al., 2011); we assume that the local-market prices and quantities of fruit and vegetables constitute a competitive equilibrium.

Figure 4 about here
Apparently, the fresh-product crops grown in Israel are also relatively sensitive to salinity. Figure 4 presents the aggregate agricultural land, water use, and production value of the 55 crops in the base year, categorized into production to the local 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 & Hoffman, 1977). Consider the salinity-sensitive and moderately sensitive crop bundles delivered to the local market, where the demand is steadily growing due to population growth, and where prices may rise due to the presence of import tariffs: while only 20% of the total arable land in the country is allocated to these two bundles (Figure 4a), their water consumption constitutes 46% of the total irrigation-water use (Figure 4b), and their production value amounts to 59% (Figure 4c). These patterns can potentially incentivize large allocations of desalinated water to the agricultural sector to reduce irrigation-water salinity, increase the yields and per-hectare profits of these crops, and in turn, the land allocated to them and their total production value. To capture these effects, we base the agriculture module on the VALUE (Vegetative Agriculture Land-Use Economics) model.

VALUE is a positive mathematical programming (PMP) partial-equilibrium model of Israel’s agriculture, used to analyze rural landscape amenities (Kan et al., 2009), organic-waste policies (Kan et al., 2010; Raviv et al., 2017), climate change (Palatnik et al., 2011; Zelingher et al., 2019), and water management (Baum et al., 2016; Kan & Rapaport-Rom, 2012). The decision variables in VALUE are the cultivated lands allocated to the 55 crops in each of the 18 nodes specified in MYWAS as agricultural water-consumption regions. We denote by $x_{ot}^a$ (ha) the land assigned in year $t$ to crop $o$
for which the output $y_{ot}^a$ (ton ha$^{-1}$ year$^{-1}$) is given by the production function

$$y_{ot}^a (w_o^a, s_{ot}^a) = \frac{\bar{y}_o^a}{1 + \alpha_{o1}^a (\alpha_{o5}^a s_{ot}^a + \alpha_{o2}^a (w_o^a)^{\alpha_{o3}^a})^{\alpha_{o4}^a}} \quad (9)$$

where $w_o^a$ (m$^3$ ha$^{-1}$ year$^{-1}$) is the total water available to the crop, including irrigation water and rainfall during the crop’s growing season, both of which are assumed constant; $s_{ot}^a$ (dS m$^{-1}$) is the salinity of $w_o^a$, which is a weighted average of the salinities of the water supply set by MYWAS for irrigation in region $a$ and the rainfall salinity (assumed to be 0.1 (dS m$^{-1}$)); $\bar{y}_o^a$ (ton ha$^{-1}$ year$^{-1}$) is the maximum potential yield obtainable under zero salinity and no water deficit; $\alpha_{o1}^a$ through $\alpha_{o5}^a$ are crop- and region-specific parameters.

We develop the production functions by a four-stage meta-analysis procedure. First, we use the yield–water salinity model developed by Shani et al. (2007) to generate a dataset of plant-level relative yields (i.e., relative to $\bar{y}_o^a$) under different combinations of water and salinity levels, adopting crop salinity-tolerance parameters from Tanji and Kielen (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 nonlinear regression to the generated dataset to estimate the parameters $\alpha_{o1}^a$ through $\alpha_{o5}^a$; finally, we calibrate $\bar{y}_o^a$ using the observed base-year yield $y_{o0}^a$ and salinity $s_{o0}^a$:

$$\bar{y}_o^a = y_{o0}^a \left(1 + \alpha_{o1}^a (\alpha_{o5}^a s_{o0}^a + \alpha_{o2}^a (w_o^a)^{\alpha_{o3}^a})^{\alpha_{o4}^a}\right).$$

The resultant functions exhibit the
expected properties of $\frac{\partial y}{\partial w} > 0$ and $\frac{\partial y}{\partial s} < 0$. In addition, $\frac{\partial^2 y}{\partial s^2} > 0$, meaning that the higher the salinity of the water applied to a crop, the lower the salinity’s marginal damage.

The statewide production of each crop $o$ in year $t$, $Y_{ot} = \sum_{a=1}^{18} x_{ot}^a y_{ot}^a$ (ton year$^{-1}$), is allocated to the local food-processing industry and export markets based on the respective constant market shares $\mu_{ot}^{L}, \mu_{ot}^{ln}$ and $\mu_{ot}^{Ex}$, where $\mu_{ot}^{L} + \mu_{ot}^{ln} + \mu_{ot}^{Ex} = 1$. The prices of crop $o$’s outputs supplied to the industry and to export, denoted, respectively, $p_{ot}^{ln}$ and $p_{ot}^{Ex}$ ($\text{\$ ton}^{-1}$), equal their corresponding (exogenous) import prices, whereas the price of the local-market product, $p_{ot}^{l}$, is determined in equilibrium subject to the import-tariff-rate quotas. We use a constant-elasticity demand function to represent the benefits to local consumers of crop $o$’s products, $V_{ot}(Y_{ot}, Lm_{ot}) = A_{ot} \frac{(\mu_{ot}^{L} Y_{ot} + \mu_{ot}^{ln} Lm_{ot})^{\vartheta_o + 1}}{\vartheta_o + 1} (\text{\$ year}^{-1})$, in which $Lm_{ot}$ (ton year$^{-1}$) is import, $\vartheta_o^{-1}$ is the demand elasticity, and $A_{ot} = p_{ot}^{l}((\mu_{ot}^{L} Y_{ot} + Lm_{ot})(1 + \rho)^{t})^{-\vartheta_o} (\text{\$ year}^{-1})$ is an inverse-demand-function parameter, calibrated for the base year based on the observed local price $p_{ot}^{l}$ and output $Y_{ot}$. The total import is given by $Lm_{ot} = Lm_{ot}^{0} + Lm_{ot}^{1}$, with $Lm_{ot}^{0}$ and $Lm_{ot}^{1}$ denoting the amounts imported free of tariff and with the tariff, respectively, where $Lm_{ot}^{0}$ cannot exceed the free-of-tariff quota $\overline{Lm}_{ot}^{0}$. For the import price of each crop, we select the country from which the import cost (including transportation to Israel) is the lowest. The cropping acreage, per-hectare yields, imports, prices and tariffs are from the Israel Ministry of Agriculture and Rural Development, and the crop-specific output-demand elasticities ($\vartheta_o$) are adopted from Fuchs (2014).

Following the PMP approach, we assume a quadratic regional profit function, $\pi_{ot}^a = x_{ot}^a (p_{ot} y_{ot}^a - \gamma_{ot}^a - \phi_{ot}^a - \delta_{ot}^a x_{ot}^a) (\text{\$ year}^{-1})$, where $p_{ot} = \mu_{ot}^{L} p_{ot}^{L} + \mu_{ot}^{ln} p_{ot}^{ln} + \mu_{ot}^{Ex} p_{ot}^{Ex}$ ($\text{\$}
ton⁻¹) is the average price of crop \(o\), \(\gamma_o^a\) (\$/ha⁻¹ year⁻¹) is a vector of input costs, and \(\phi_o^a\) (\$/ha⁻¹ year⁻¹) and \(\delta_o^a\) (\$/ha² year⁻¹) are parameters representing unobservable costs, calibrated by the two-stage procedure proposed by Howitt (1995). The calibration accounts for the base-year constraints of regional agricultural land, foreign labor, and water: \(\sum_{o=1}^{55} x_{o0}^a \leq x^a\), \(\sum_{o=1}^{55} x_{o0}^a f l_o^a \leq f l^a\), and \(\sum_{o=1}^{55} x_{o0}^a w_o^a \leq w_o^a\), respectively, where \(x^a\) (ha) is region \(a\’s\) total arable land, \(f l_o^a\) (day ha⁻¹ year⁻¹) and \(f l^a\) (day year⁻¹) are, respectively, crop \(o\’s\) consumption and regional aggregate quota of foreign labor, and \(w_0^a\) is the base-year total irrigation water allocated to the region.

3.2.3. Integrated Model

The integrated MYWAS–VALUE model solves the problem expressed in equation (1), the elements of which incorporate the empirical specifications of benefits, costs, and constraints described above. We employ a social discount rate \((r)\) of 3.5%, as suggested by Nordhaus (2007), and adopt the Israel Central Bureau of Statistics (2014) prediction of 1.8% annual population growth rate \((\rho)\). As an end condition, we mandate retaining a minimum of 5% of the extractable stock from each naturally enriched freshwater source. The historical average annual precipitation and enrichment of natural freshwater sources (taken from Weinberger et al., 2012) are assumed for every year during the simulated period. Monetary values are in 2015 US dollars, discounted to the 15th year (2030).

Note that while the demands for urban water and agricultural products are calibrated for 2015 based on the assumption of market-clearing prices, the observed base-year water prices and the associated water distributions and infrastructures are not optimal! This is because of the Israeli Balanced Budget Water Economy legislation (Reznik et al., 2016);
therefore, as shown in Appendix C, the model does not reproduce the observed values of the decision variables in the base year. Thus, the base-year conditions do not serve as a reference in our analyses. Instead, we compare the optimal solutions under the SI and SE scenarios.

While the solution for the SI case is obtained by solving the problem in equation (1), the SE scenario requires a two-stage procedure: first, we run the model with the water salinity in each agricultural region fixed at its base-year level (i.e., in equation 9, $s_{ot}^a = s_{o0}^a$ for all $t = 1, \ldots, 30$); the output of this simulation is an infrastructure-development plan that the planner erroneously considers optimal given her or his presumption of nonresponsive irrigation-water salinity. However, as already noted in the 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 that resulted from the first stage. Thus, the SE scenario represents the expected evolution of the water and vegetative-agriculture sectors under efficient prices, given a non-optimal water-infrastructure blueprint.

4. Economic Implications of Externalizing Salinity

We first evaluate the differences between the SI and SE scenarios with respect to water supplies, salinities, prices, and surplus allocation across sectors. Then, we assess how these differences are affected by changes in precipitation and in the agricultural product
import tariffs and foreign-worker quotas. Appendix D provides detailed results tables; here, we present the main findings.

Figure 5 about here

4.1. Internalized versus Externalized Salinity

Figure 5 presents optimal trajectories of statewide water supply to the agricultural regions and discharge of tertiary-TWW to natural waterways under the two scenarios. The SI optimal solution suggests massive delivery of desalinated water to the agricultural zones, mostly from seawater-desalination plants, and to a lower extent by desalinating TWW (Figure 5a). It is found optimal to install 8 out of the 12 optional pipelines connecting seawater-desalination plants directly to agricultural regions, as well as the optional seawater-desalination plant in the north of the country. Irrigation with natural freshwater and secondary-TWW is stable throughout the period, seawater desalination for irrigation increases slightly with time, and most of the growing amount of sewage produced by the urban zones (not shown) is discharged to nature after tertiary wastewater treatment.

Compared to the SI scenario, the total irrigation water under the SE scenario is 45% lower (Figure 5b), particularly because of the lower capacity of the infrastructures supplying desalinated seawater to the agricultural zones, and because desalination of TWW becomes unwarranted due to the higher costs relative to that of seawater desalination. Concomitantly, a larger share of the TWW is used for irrigation, such that lower amounts are discharged to nature. Moreover, the water-supply trends differ from the SI solution: seawater desalination for agriculture declines over time, continuously
replaced by the growing amounts of TWW produced in the urban zones and by freshwater from natural sources.

Figure 6 presents per-annum statewide averages of water supply, use and salinity under the two scenarios. Clearly, the total water deliveries (Figure 6b) are lower under SE, and the average irrigation-water salinity (Figure 6c) is 0.2 (dS m⁻¹) higher due to the larger shares of natural freshwater and TWW in the supplied water (Figures 5a, 5b, and 6a).

Figure 6 about here

Figure 7 reports average shadow values of water availability at the sources (Figure 7a), infrastructure-capacity constraints (Figure 7b), water prices in the urban and agricultural zones (Figure 7c), and shadow values of the salinity of the water sources (Figure 7d). In the SE case, water is scarcer due to the lower water-infrastructure capacity; hence, the higher water prices compared to SI. Notice the switch in sign of the shadow value of the inflow–outflow balance constraint imposed on the WWTP (Figure 7a), meaning that instead of subsidizing TWW to encourage farmers to consume it under SI, farmers are willing to pay for TWW in the SE case. In addition, while irrigation salinity is higher under SE (Figure 6c), the shadow value of the salinity in the irrigation water is slightly lower (Figure 7d), which is attributed to the aforementioned positive sign of the second derivative of the production function with respect to salinity \( \frac{\partial^2 y}{\partial s^2} > 0 \).

Also notable is the relatively large negative shadow value of the salinity of the desalinated TWW under SI, which can be explained by the fact that this water source supplies low-salinity water only for agricultural irrigation, whereas the salinity of the TWW supplied to agricultural regions is high, and desalinated brackish water and
seawater are also supplied to the urban users who are assumed immune to salinity changes (up to the maximum urban-water salinity limit of 1.4 (dS m\(^{-1}\))). Thus, as TWW desalination vanishes under the SE scenario, the share of the other sources rises and therefore, their salinity shadow values become higher.

Figure 7 about here

As already noted, we assume that prices are set efficiently, and use the demand functions and shadow values to compute the water and agriculture-product prices, and the corresponding allocation of surpluses across sectors. Figure 8a presents average yearly surplus allocations under the SI and SE scenarios (consumer surpluses are normalized to 0 under the SI scenario). As shown analytically, all sectors except for water suppliers face a welfare loss under the SE vs. SI scenario, where the overall annual deadweight loss is $366 million. On average, this damage amounts to about $1,200 per hectare of arable land, caused by the salinity rise (of 0.2 (dS m\(^{-1}\)) on average, Figure 6c), of which farming losses are $1,087 ha\(^{-1}\) year\(^{-1}\) (29\% of farmers’ profit under the SI scenario).

Figure 8 about here

4.2. Sensitivity Analyses

We commence by analyzing a change in precipitation. On a per-annum average, Israel’s arable lands obtain 375 mm and 333 mm of water from naturally enriched freshwater sources and direct rainfall, respectively. However, climatological studies (Bentsen et al., 2013; Gent et al., 2011; Watanabe et al., 2010) predict a precipitation reduction of 10–30\% throughout Israel in the coming decades. In Figure 8b, we present the impact of a 20\% reduction in precipitation for every year of the 30-year period on the allocation of
surpluses under the SI and SE scenarios. All sectors except for the water suppliers lose from the change, farmers in particular. The overall deadweight loss is 37% and 38% under the SI and SE scenarios, respectively. Regarding water management (not shown), in both scenarios, increasing seawater desalination is the main means of offsetting the reduction in availability of natural freshwater sources, as well as cuts in agricultural irrigation.

As already noted, Israel employs import tariffs to support local agriculture and controls the allocation of foreign labor. In Figures 8c and 8d we present, respectively, the surplus change caused by the abolishment of import tariffs on vegetative products and dismissing the control of foreign labor by quotas (for the latter, we assume no change in the regulated minimum wage of the foreign workers or in the crop-specific per-hectare employment of foreign and local workers). Both policy changes effectively transfer surpluses from farmers to consumers of agricultural products in both the SI and SE cases, raising the overall welfare, with a larger effect under the SE scenario.

The agricultural intervention policies employed in Israel are continuously criticized economically (Organization for Economic Co-operation and Development [OECD], 2018), and are the subject of public and political debate (Hendel et al., 2017). The overall producer-support estimate for Israel is 17.7%, slightly below the OECD average of 19.2% (OECD, 2019). However, the share of domestic price support, border measures, and other market-distorting interventions is relatively high (91%); therefore, the effect of changes in such policies is expected to be larger than in other countries with similar semiarid conditions (e.g., Australia, Spain, and California).
5. Limitations and Extensions

This study shows that, under the specific circumstances of Israel, large-scale supply of desalinated water for irrigation is economically warranted, and overlooking the impact of desalination on irrigation-water salinity entails significant deadweight loss. Nevertheless, “no model solves all problems” (Draper et al., 2003, p. 156), and the economic valuations provided herein may vary with the inclusion of a range of exempted factors; a few examples follow.

Our ability to introduce salinity impacts on agricultural outputs into our hydro-economic model builds on decades of theoretical and experimental agronomic research. However, aside from irrigation, salinity has additional economic implications, such as health effects associated with drinking water (Dasgupta et al., 2016). Moreover, salinity is but one of a large set of water-quality measures that determine the economic value of water in various water uses. For example, the contents of Mg\(^{2+}\) and SO\(_4^{2-}\) in desalinated seawater are lower than recommended for drinking and irrigation water (Yermiyahu et al., 2007), and desalination removes nutrients that might otherwise save on fertilization expenses (Ben-Gal et al., 2009; Dawson & Hilton, 2011). On the other hand, TWW desalination also eliminates various contaminants (Gur-Reznik & Dosoretz, 2015) that damage irrigation systems (Tarchitzky et al., 2013) and cause environmental and health risks (Fatta-Kassinos et al., 2011; Kasprzyk-Hordern et al., 2009) which, in turn, may affect the demand for agricultural products (Messer et al., 2017). In addition, while our model allows for environmental discharge of tertiary-TWW and spillovers from
freshwater aquifers, we do not assign values to these flows because there are as yet no monetary valuations of the associated impacts on ecosystem services (Tsur, 2020).

Our modeling approach is suited to centrally managed water systems, as in Israel. We assume benevolent policy makers; however, centralization attracts political pressure to bend policies in favor of interest groups. Such distortions can be captured in a political-equilibrium analytical framework (Finkelshtain & Kislev, 1997). For the case of decentralized water systems, one should account for imperfections of water markets and water-right allocations, asymmetric information, and transition and transaction costs (Garrick et al., 2013; Richter et al., 2013).

Finally, our model does not explicitly incorporate uncertainty with respect to key factors, such as population growth, climate conditions, and environmental effects, some of which may be associated with deep uncertainty. Moreover, we maximize a single objective, whereas water systems may involve multiple objectives. The analysis provided here may be extended by employing many objective robust decision-making procedures (e.g., Kasprzyk et al., 2013; Moallemi et al., 2020; Trindade et al., 2017).

Acknowledgments

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empirical hydro-economic model developed in this study (entitled MYWAS-VALUE) is available at: 10.5281/zenodo.3702053.

References


Figure 1. A water economy with urban and agricultural zones in which water consumers obtain a mixture of groundwater and desalinated water.
**Salinity Internalized**

- $V^u_w(w^u)$ - Inverse urban demand
- $V_w(w, s)$ - Inverse aggregate demand under average salinity $s$
- $V_0(w, s^g)$ - Inverse aggregate demand under groundwater salinity $s^g$

**Graphical representation:**

- **A**
- **B**
- **C**

**Formulas:**

- $V_w(w, s) = V_0(w, s^g) + \lambda^g$

**Economic measures:**

- Consumer surplus = $A+B$
- Producer surplus = $C-B$
- Welfare = $A+C$

---

**Salinity Externalized**

- $p^SE = c^d$

**Graphical representation:**

- **D**
- **E**

**Economic measures:**

- Consumer surplus = $D$
- Producer surplus = $E$
- Welfare = $D+E$
- Deadweight loss due to disregarding salinity = $F$
Figure 2. Schematic illustration of water prices, quantities and surplus allocations under (a) the Salinity-Internalized scenario, (b) the Salinity-Externalized scenario, and (c) where water-use benefits from desalinated water exceed the desalination variable and fixed costs under the Salinity-Internalized scenario, but not under the Salinity-Externalized scenario.
Figure 3. Scheme of the integrated water and agriculture modules of the hydro-economic model.
Figure 4. Base-year statewide allocation of (a) land use, (b) water consumption, and (c) production value across four crop bundles with different salinity-tolerance levels, separated into production for the local fresh-produce, industry and export markets.
Figure 5. Optimal trajectories of statewide annual water supplies to agriculture and discharges of tertiary-TWW to nature under (a) the Salinity-Internalized and (b) the Salinity-Externalized scenarios.
**Figure 6.** Optimal average (a) annual water supplies, (b) annual water uses, and (c) supplied-water salinities under the Salinity-Internalized and Salinity-Externalized scenarios.
Figure 7. Average (a) water-source shadow values, (b) shadow values of infrastructures, (c) water prices at consumption points, and (c) shadow values of water-source salinity (per supplied cubic meter of water) under the Salinity-Internalized and Salinity-Externalized scenarios.
Figure 8. (a) Average annual surplus allocation under the Salinity-Internalized and Salinity-Externalized scenarios. Surplus change caused by (b) 20% reduction in precipitation, and (c) and (d) policy reforms in which agricultural import tariffs and foreign-labor quotas, respectively, are abolished.
Appendix A. Water and Agricultural Production in Israel

Figure A1 presents the evolution of water sources and uses (Israel Water Authority, 2016), population growth and vegetative agricultural production (Israel Central Bureau of Statistics, 2017) in Israel during the period 1996–2016. Fresh groundwater extractions declined dramatically, and were partly substituted by seawater desalination (Figure A1a). At the same time, the population increased by 50% (Figure A1c), generating larger amounts of sewage, which was reclaimed (Figure A1a) and reused by irrigation (Figure A1b). The population growth also increased the demand for food; however, while the index of local vegetative agricultural production increased until 2010, it stabilized thereafter.
Figure A1. Trends of (a) water sources, (b) water uses, and (c) population growth and vegetative agricultural production in Israel during 1996–2016.
Appendix B. Schemes of the MYWAS Model

**Model Topology**

<table>
<thead>
<tr>
<th>Agricultural Demand Nodes</th>
<th>Urban Demand Nodes</th>
<th>Waste Water Treatment Plants</th>
<th>National Carrier Junctions</th>
<th>Sea Water and Ground Water Desalination Plants</th>
<th>Brackish and Surface Water Sources</th>
<th>Natural Fresh Water Sources</th>
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<td>1006 North Mountain</td>
</tr>
<tr>
<td>3107 Judea and Samaria</td>
<td>3008 Judea and Samaria</td>
<td>1207 North Coast</td>
<td></td>
<td></td>
<td></td>
<td>1007 Carmel Coast</td>
</tr>
<tr>
<td>3108 Hadera</td>
<td>3009 Tel Aviv</td>
<td>1208 Central Coast</td>
<td></td>
<td></td>
<td></td>
<td>1008 Northern Coast</td>
</tr>
<tr>
<td>3109 Sharon</td>
<td>3010 Lod</td>
<td>1209 Yarkon</td>
<td>5003 Rosh Haain</td>
<td>1103 Ashkelon</td>
<td></td>
<td>1009 Central Coast</td>
</tr>
<tr>
<td>3110 Petah Tikva</td>
<td>3011 Jerusalem</td>
<td>1210 Nesher</td>
<td></td>
<td></td>
<td></td>
<td>1010 Central Mountain</td>
</tr>
<tr>
<td>3111 Tel Aviv</td>
<td>3012 Modeen</td>
<td>1211 Shafdan</td>
<td></td>
<td></td>
<td></td>
<td>1011 Southern Coast</td>
</tr>
<tr>
<td>3112 Ramle</td>
<td>3013 Abidim</td>
<td>1212 Southern Coast</td>
<td></td>
<td></td>
<td></td>
<td>1012 Negev Coastal GW</td>
</tr>
<tr>
<td>3113 Rehovot</td>
<td>3014 Granot</td>
<td>1213 Judea and Samaria</td>
<td></td>
<td></td>
<td></td>
<td>1013 Southern Mountain</td>
</tr>
<tr>
<td>3114 Jerusalem</td>
<td>3015 Western Negev</td>
<td>1214 Shfela</td>
<td></td>
<td></td>
<td></td>
<td>1014 Negev Aquifer</td>
</tr>
<tr>
<td>3115 Ashkelon</td>
<td>3016 Lachish</td>
<td>1215 Negev Coast</td>
<td></td>
<td></td>
<td></td>
<td>1015 Arava</td>
</tr>
<tr>
<td>3116 Negev</td>
<td>3017 Ramat Negev</td>
<td>1216 Negev</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>3117 Arava</td>
<td>3018 Dead Sea</td>
<td>1217 Dead Sea</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td></td>
<td>3019 Arava</td>
<td>1218 Arava</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td></td>
<td>3020 Elat</td>
<td>1219 Elat</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td></td>
<td>31131 Elat</td>
<td>1220 Elat</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
</tbody>
</table>

Natural Fresh Water Sources

- 1000 Sea of Galilee
- 1001 Golan
- 1002 Western Kinneret GW
- 1003 Western Galilee
- 1004 Lower Galilee
- 1005 Eastern Aquifers
- 1006 North Mountain
- 1007 Carmel Coast
- 1008 Northern Coast
- 1009 Central Coast
- 1010 Central Mountain
- 1011 Southern Coast
- 1012 Negev Coastal GW
- 1013 Southern Mountain
- 1014 Negev Aquifer
- 1015 Arava
Fresh and Brackish water for Agriculture

Western Galilee
- Western Kineret
- Sea of Galilee

Golan GW

Jerusalem
- Tzfat
- Kineret
- Telat

Lower Galilee

Jeezrael Valley
- Golan

Haifa
- Hadera
- Sharon
- Petah Tikva

Ramle
- Rehovot
- Tel Aviv

Ashkelon

Judea and Samaria

Negev

Jordan Valley

Western Aquifers

Zalmon

Menashe

Metzer

Rosh Haain

Haain

Central Coast
- Peltah Tikva
- Tel Aviv

Central Mountain

Eastern Aquifers

Eastern Local

Arava

Negev Coastal GW

Southern Coast
- Ashkelon

Southern Mountain

Northern Coast
- Hasifa
- Haifa

Northern Mountain
- Sharon

Carmel Coast
- Rosh Haain

Carmel Coast Brackish

Arava Brackish

Menashe

Holda

Zohar

Negev Brackish

Negev Aquifer

Arava Aquifer

Negev Aquifer

Arava
Appendix C. Calibration Data vs. Optimization Results

Figure C1 presents the statewide patterns of Israel’s water economy for the first year of the optimal solution under the Salinity-Internalized scenario in comparison to the observed base-year calibration data.

![Figure C1](image)

**Figure C1.** (a) Water supplies, (b) water uses, (c) water prices, and (d) salinities of the supplied water in the first year of the Salinity-Internalized scenario and the base-year calibration data.

As already shown by Reznik et al. (2016), the water supply costs in Israel are characterized by economies of scale. Since water prices are set subject to a regulatory requirement to recover variable and fixed costs, water prices in the urban sector are higher than their optimal counterparts (Figure C1c), and hence the larger water supply in the Salinity-Internalized case relative to that of the base year (Figure C1a). The larger share
of desalinated water in the total water supply causes the lower salinity under the Salinity-Internalized scenario (Figure B1d).
Appendix D. Results of the Salinity-Internalized and Salinity-Externalized Scenarios

Table D1

*Per-Annun Statewide Average Water Volumes*

<table>
<thead>
<tr>
<th></th>
<th>Salinity Internalized</th>
<th>Salinity Externalized</th>
</tr>
</thead>
<tbody>
<tr>
<td><strong>All Water Consumers</strong></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Water Supply (10^6 m^3 year^{-1})</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Natural Freshwater</td>
<td>1,124</td>
<td>1,153</td>
</tr>
<tr>
<td>Brackish Groundwater</td>
<td>0</td>
<td>0</td>
</tr>
<tr>
<td>Brackish Groundwater Desalinated</td>
<td>106</td>
<td>63</td>
</tr>
<tr>
<td>Treated Wastewater</td>
<td>626</td>
<td>714</td>
</tr>
<tr>
<td>Treated Wastewater Desalinated</td>
<td>81</td>
<td>0</td>
</tr>
<tr>
<td>Seawater Desalinated</td>
<td>865</td>
<td>652</td>
</tr>
<tr>
<td>All</td>
<td>2,802</td>
<td>2,582</td>
</tr>
<tr>
<td>Water Consumption (10^6 m^3 year^{-1})</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Urban</td>
<td>1,263</td>
<td>1,242</td>
</tr>
<tr>
<td>Agriculture</td>
<td>1,149</td>
<td>998</td>
</tr>
<tr>
<td>Nature</td>
<td>390</td>
<td>342</td>
</tr>
<tr>
<td>All</td>
<td>2,802</td>
<td>2,582</td>
</tr>
<tr>
<td><strong>Agricultural Sector</strong></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Water Supply (10^6 m^3 year^{-1})</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Natural Freshwater</td>
<td>262</td>
<td>270</td>
</tr>
<tr>
<td>Brackish Groundwater</td>
<td>0</td>
<td>0</td>
</tr>
<tr>
<td>Brackish Groundwater Desalinated</td>
<td>0</td>
<td>0</td>
</tr>
<tr>
<td>Treated Wastewater</td>
<td>236</td>
<td>373</td>
</tr>
<tr>
<td>Treated Wastewater Desalinated</td>
<td>81</td>
<td>0</td>
</tr>
<tr>
<td>Seawater Desalinated</td>
<td>569</td>
<td>355</td>
</tr>
<tr>
<td>All</td>
<td>1,149</td>
<td>998</td>
</tr>
<tr>
<td>Agricultural Labor (10^6 day year^{-1})</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Foreign</td>
<td>9.5</td>
<td>9.4</td>
</tr>
<tr>
<td>Local</td>
<td>5.1</td>
<td>5.0</td>
</tr>
</tbody>
</table>
Table D2

*Per-Annum Statewide Average Shadow Values of Water-Quantity and Water-Infrastructure-Capacity Constraints at Supply Points; Water Prices at the Consumption Points (\(\text{\$ m}^{-3}\)); Shadow Values of Land and Foreign Labor*\(^a\)

<table>
<thead>
<tr>
<th>Water Supply Constraint ((\text{$ m}^{-3}))</th>
<th>Salinity Internalized</th>
<th>Salinity Externalized</th>
</tr>
</thead>
<tbody>
<tr>
<td>Natural Freshwater</td>
<td>0.27</td>
<td>0.87</td>
</tr>
<tr>
<td>Brackish Groundwater</td>
<td>0.11</td>
<td>0.00</td>
</tr>
<tr>
<td>Brackish Groundwater Desalinated</td>
<td>0.10</td>
<td>0.75</td>
</tr>
<tr>
<td>Treated Wastewater</td>
<td>-0.09</td>
<td>0.14</td>
</tr>
<tr>
<td>Treated Wastewater Desalinated</td>
<td>0.38</td>
<td>0.82</td>
</tr>
<tr>
<td>Seawater Desalinated</td>
<td>0.00</td>
<td>0.00</td>
</tr>
<tr>
<td>Consumption Price ((\text{$ m}^{-3}))</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Urban</td>
<td>1.54</td>
<td>1.77</td>
</tr>
<tr>
<td>Agriculture</td>
<td>0.41</td>
<td>0.72</td>
</tr>
<tr>
<td>All</td>
<td>0.86</td>
<td>1.13</td>
</tr>
<tr>
<td>Capacity Constraint ((\text{$ m}^{-3}))</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Natural Freshwater</td>
<td>0.01</td>
<td>0.07</td>
</tr>
<tr>
<td>Brackish Groundwater</td>
<td>0.01</td>
<td>0.00</td>
</tr>
<tr>
<td>Brackish Groundwater Desalinated</td>
<td>0.47</td>
<td>0.25</td>
</tr>
<tr>
<td>Treated Wastewater</td>
<td>0.05</td>
<td>0.15</td>
</tr>
<tr>
<td>Treated Wastewater Desalinated</td>
<td>0.27</td>
<td>0.10</td>
</tr>
<tr>
<td>Seawater Desalinated</td>
<td>0.05</td>
<td>0.38</td>
</tr>
<tr>
<td>Water Conveyance</td>
<td>0.00</td>
<td>0.01</td>
</tr>
<tr>
<td>Non-Water Agricultural Production Factors</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Agricultural Land ((\text{$ ha}^{-1}))</td>
<td>5.12</td>
<td>2.49</td>
</tr>
<tr>
<td>Foreign Labor ((\text{$ day}^{-1}))</td>
<td>59.3</td>
<td>45.7</td>
</tr>
</tbody>
</table>

\(^a\) Shadow values and prices were discounted to the 15\(^{th}\) year, and then averaged.
Table D3

Per-Annum Statewide Average Salinity Levels and Salinity Shadow Values at the Water

Supply and Consumption Points

<table>
<thead>
<tr>
<th>Salinity (dS m(^{-1}))</th>
<th>Salinity Internalized</th>
<th>Salinity Externalized</th>
</tr>
</thead>
<tbody>
<tr>
<td><strong>Supply</strong>(^a)</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Natural Freshwater</td>
<td>1.05</td>
<td>1.06</td>
</tr>
<tr>
<td>Treated Wastewater</td>
<td>1.46</td>
<td>1.49</td>
</tr>
<tr>
<td>All</td>
<td>0.83</td>
<td>0.95</td>
</tr>
<tr>
<td><strong>Consumption</strong></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Urban</td>
<td>0.82</td>
<td>0.84</td>
</tr>
<tr>
<td>Agriculture</td>
<td>0.60</td>
<td>0.82</td>
</tr>
<tr>
<td>Nature</td>
<td>1.55</td>
<td>1.73</td>
</tr>
<tr>
<td>All</td>
<td>0.83</td>
<td>0.95</td>
</tr>
<tr>
<td><strong>Shadow Value ($10^6$ (dS/m)(^{-1}) year(^{-1}))(^b)</strong></td>
<td></td>
<td></td>
</tr>
<tr>
<td><strong>Supply</strong></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Natural Freshwater</td>
<td>-19.3</td>
<td>-31.3</td>
</tr>
<tr>
<td>Brackish Groundwater</td>
<td>0.0</td>
<td>0.0</td>
</tr>
<tr>
<td>Brackish Groundwater Desalinated</td>
<td>-6.6</td>
<td>-8.6</td>
</tr>
<tr>
<td>Treated Wastewater</td>
<td>-17.5</td>
<td>-43.1</td>
</tr>
<tr>
<td>Treated Wastewater Desalinated</td>
<td>-33.5</td>
<td>0.0</td>
</tr>
<tr>
<td>Seawater Desalinated</td>
<td>-95.5</td>
<td>-98.0</td>
</tr>
<tr>
<td><strong>Consumption</strong></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Agriculture</td>
<td>-77.4</td>
<td>-68.1</td>
</tr>
</tbody>
</table>

\(^a\) The salinity of brackish water and desalinated water is 4 and 0.25 dS m\(^{-1}\), respectively.

\(^b\) Shadow values were discounted to the 15\(^{th}\) year, and then averaged.
### Table D4

*Per-Annum Statewide Average Welfare Elements ($10^6$ year$^{-1}$)*

<table>
<thead>
<tr>
<th></th>
<th>Salinity Internalized</th>
<th>Salinity Externalized</th>
</tr>
</thead>
<tbody>
<tr>
<td><strong>Agricultural Products</strong></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Production Value</td>
<td>5,577</td>
<td>5,229</td>
</tr>
<tr>
<td>Import Value$^b$</td>
<td>662</td>
<td>724</td>
</tr>
<tr>
<td>Variable Costs</td>
<td>3,999</td>
<td>3,765</td>
</tr>
<tr>
<td>Capital Costs</td>
<td>336</td>
<td>322</td>
</tr>
<tr>
<td><strong>Water-Supply Costs</strong></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Variable Costs</td>
<td>1,432</td>
<td>1,291</td>
</tr>
<tr>
<td>Capital and Operation Costs</td>
<td>472</td>
<td>421</td>
</tr>
<tr>
<td>Total</td>
<td>1,903</td>
<td>1,712</td>
</tr>
<tr>
<td><strong>Water Purchase Expenses</strong></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Urban</td>
<td>1,855</td>
<td>2,100</td>
</tr>
<tr>
<td>Agriculture</td>
<td>442</td>
<td>655</td>
</tr>
<tr>
<td>Water to Nature</td>
<td>131</td>
<td>115</td>
</tr>
<tr>
<td>Total</td>
<td>2,297</td>
<td>2,755</td>
</tr>
<tr>
<td><strong>Surpluses</strong></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Urban Water Consumers$^c$</td>
<td>0</td>
<td>-465</td>
</tr>
<tr>
<td>Agricultural Product Consumers$^c$</td>
<td>0</td>
<td>-241</td>
</tr>
<tr>
<td>Farming Profits</td>
<td>1,136</td>
<td>810</td>
</tr>
<tr>
<td>Water Suppliers</td>
<td>263</td>
<td>929</td>
</tr>
<tr>
<td>Social Welfare</td>
<td>1399</td>
<td>1032</td>
</tr>
</tbody>
</table>

$^a$ 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 Salinity-Internalized scenario, and report their changes under the Salinity-Externalized scenario in comparison to the Salinity-Internalized one.
Table D5

Laspeyres Quantity and Price Indices, and Value Index, of Agricultural Production by Salinity-Tolerance Crop Bundles, under the Salinity Externalized Scenario Relative to the Salinity-Internalized Scenario (i.e., Salinity Internalized = 100)

<table>
<thead>
<tr>
<th>Crop Bundle Sensitivity to Salinity&lt;sup&gt;a&lt;/sup&gt;</th>
<th>Quantity</th>
<th>Price</th>
<th>Value</th>
</tr>
</thead>
<tbody>
<tr>
<td>Sensitive (36%)</td>
<td>92</td>
<td>104</td>
<td>95</td>
</tr>
<tr>
<td>Moderately Sensitive (47%)</td>
<td>94</td>
<td>102</td>
<td>97</td>
</tr>
<tr>
<td>Moderately Tolerant (14%)</td>
<td>95</td>
<td>100</td>
<td>95</td>
</tr>
<tr>
<td>Tolerant (3%)</td>
<td>64</td>
<td>102</td>
<td>66</td>
</tr>
<tr>
<td>All crops</td>
<td>92</td>
<td>102</td>
<td>94</td>
</tr>
</tbody>
</table>

<sup>a</sup> Values in parentheses indicate the share of the crop bundle in the production value under the Salinity-Internalized scenario.