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

By

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

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

3
4 **Abstract**

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

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

22 **JEL Classification:** Q15, Q25, Q28

23 **1. Introduction**

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

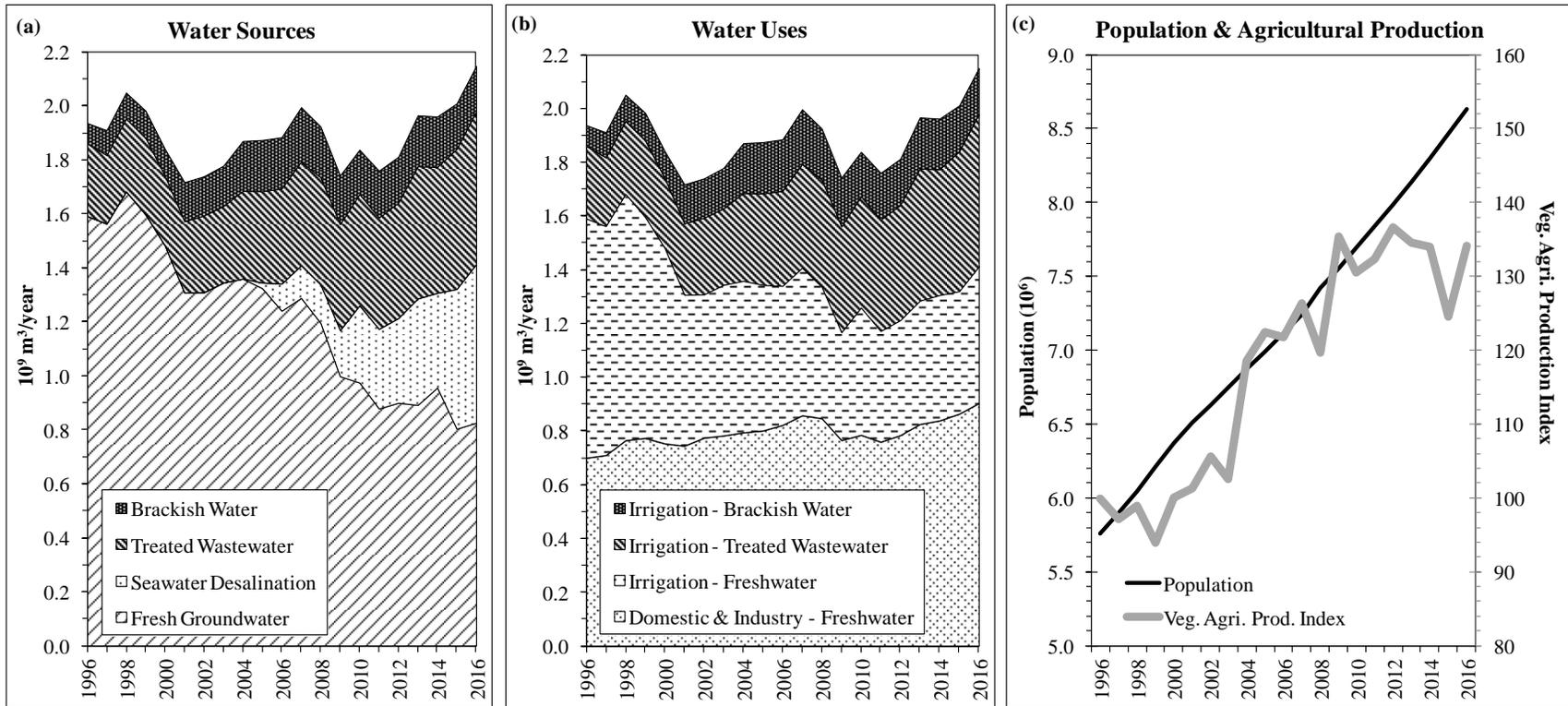
38 Given the long construction period of water infrastructures, designers of water-
39 supply systems should foresee scarcities and decide ahead of time which, when, where,
40 and to what extent infrastructural elements should be installed or expanded. While “no
41 model solves all problems” (Darper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

46 and Housh et al. (2011) refer to desalination as a means of resolving water-quantity
47 scarcities, but overlook the benefits of reducing irrigation-water salinity; Howitt et al.
48 (2010) and Welle et al. (2017) employed the van Genuchten and Hoffman (1984)
49 production function to internalize salinity's agronomic impacts into a static analysis of
50 irrigation-water management in California. However, to fully account for irrigation-water
51 salinity, one should allow the model to track the dynamics of water salinity throughout
52 the water-delivery system during the planning horizon, to affect salinity by allowing the
53 establishment of desalination capacities at various nodes in the distribution network, and
54 to comprise the salinity damage caused to agriculture and the environment, as well as the
55 adaptation measures employed by relevant stakeholders. Moreover, the model should
56 enable simulating exogenous changes, such as population growth, and agricultural and
57 environmental policies. The above attributes are incorporated herein in the analysis for
58 the case of Israel.

59 Israel today could manifest the present and future of developed economies that are
60 encountering increasing water scarcity, such as California, Australia and Spain. Israel's
61 water economy faces three basic difficulties: first, while precipitation is relatively
62 abundant in the north, the population is concentrated in the center, and most of the
63 agricultural lands are located in the south; second, whereas rainfall events occur only in
64 winter, summer is the main irrigation season; third, natural freshwater is scarce,
65 successive drought years are common, and the water scarcity is steadily increasing due to
66 population growth (Kislev, 2011). Israel has confronted these challenges by establishing a
67 complex water-distribution network to connect the various water sources and users,
68 which spans most of the country's regions, in particular based on a national water carrier

69 that essentially turns almost the entire state into a single basin in terms of water
70 management. That is, the network interlinks scarcities across points in time and space,
71 such that water use at a particular place and time may have opportunity costs with respect
72 to usage for various purposes at alternative locales and times (Fisher et al., 2005). In
73 addition, according to the Israeli Water Law (IMNI 1959), the government centrally
74 controls water consumption by administrative prices and quotas.

75 Severe water scarcity in Israel motivated the exploitation of brackish water and
76 TWW for irrigation, and in recent years, extensive seawater desalination has also been
77 used to supply freshwater to both the urban and agricultural sectors (Burn et al., 2015;
78 Martinez-Alvarez et al., 2016). Figure 1 presents the evolution of water sources and uses
79 (IWA, 2016), population growth and vegetative agricultural production (ICBS, 2017) in
80 Israel during the period 1996–2016. Fresh groundwater extractions declined dramatically
81 from 1,590 million m³/year in 1996 to only 830 million m³/year in 2016, and since 2005,
82 have been partly substituted by seawater desalination, which increased gradually to 590
83 million m³/year in 2016 (Figure 1a). At the same time, the population increased by 50%
84 (Figure 1c), generating larger amounts of sewage, which was reclaimed (Figure 1a) and
85 reused for irrigation (Figure 1b). Concomitantly, domestic and industrial freshwater
86 consumption increased by 30%, whereas freshwater for irrigation was cut by more than
87 40%. Nevertheless, with the growing agricultural use of TWW and brackish water
88 (Figure 1b), the total irrigation water did not vary much along that period. The population
89 growth also increased the demand for food and, while the index of local vegetative
90 agricultural production increased by 35% from 1996 until 2010, it stabilized thereafter.
91



92

93 **Figure 1. Trends of (a) water sources, (b) water uses and (c) population growth and vegetative-agriculture production in Israel**

94 **during 1996–2016**

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

109 By simulating a hypothetical scenario of zero salinity in Israel’s water sources, we
110 evaluate the statewide welfare loss caused by the presence of salts at about \$1,340 million
111 a year, which amounts to nearly \$4,468 per hectare of arable land. This estimate, which
112 accounts for the indirect salinity effects on domestic water supply and agricultural output
113 prices, is an order of magnitude larger than the direct agricultural salinity damage of
114 \$440/hectare reported by Qadir et al. (2014). We further use the model to assess a case in
115 which the designers of water infrastructures overlook the agronomic salinity effects; this
116 scenario entails a welfare loss of more than \$408 million a year compared to the optimal
117 solution, which is about \$1,360 per hectare of cultivable land. Moreover, ignoring salinity

118 leads to higher water prices and lower water supply, which in turn increase water
119 suppliers' surpluses at the expense of urban and agricultural water users, as well as of
120 consumers of agricultural products.

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

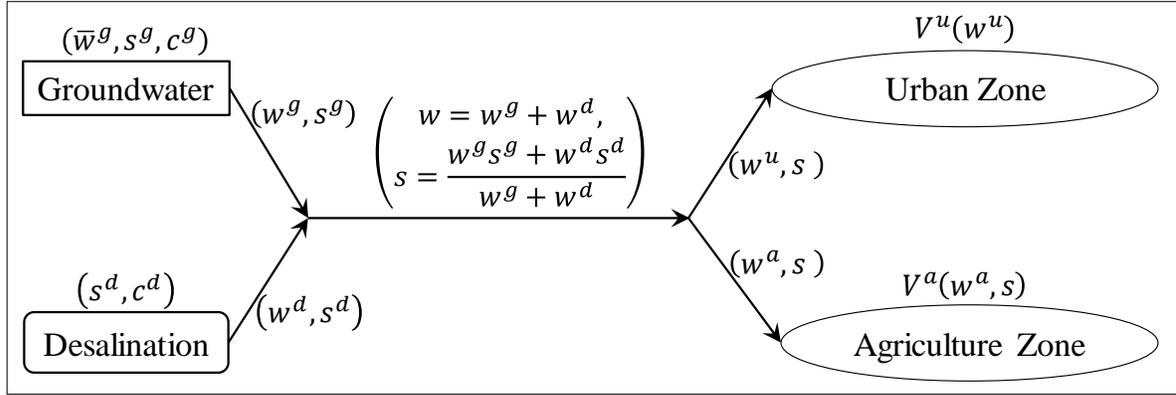
128 **2. Hydro-Economic Model**

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

137 2.1 Theoretical Analysis

138 Consider a static hydro-economic model, encompassing two water sources with different
139 salinities—groundwater and desalinated seawater—which are mixed and then delivered

140 to two water consumers—urban and agricultural, with the latter's water demand being
 141 sensitive to salinity (Figure 2).



142

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

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

149 The total water supply, $w \equiv w^d + w^g$, is a mixture of the two sources, with average
 150 salinity $s = \frac{w^d s^d + w^g s^g}{w}$. The blended water is split such that the urban and agricultural
 151 sectors obtain amounts w^u and w^a , respectively, where $w^u + w^a = w$. Let the function
 152 $V^u(w^u)$ and $V^a(w^a, s)$ be, respectively, the value of water consumed by agents in the
 153 urban and agricultural zones gross of water-purchasing expenses, where water quantity is
 154 beneficial to both sectors (i.e., $V_w^u(w^u) > 0$ and $V_w^a(w^a, s) > 0$), and water salinity is
 155 harmful only to agriculture ($V_s^u(w^u) = 0$ and $V_s^a(w^a, s) < 0$).

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

$$160 \quad \max_{w^d, w^g, w^u, w^a, s^d, s^g} V^u(w^u) + V^a(w^a, s) - c^d w^d - c^g w^g$$

$$s.t. \quad w = w^u + w^a = w^d + w^g, \quad w^g \leq \bar{w}^g, \quad s^d \geq \underline{s}^d, \quad s^g \geq \underline{s}^g, \quad s = \frac{w^d s^d + w^g s^g}{w}$$

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

$$165 \quad V_w^u(w^u) = V_w^a(w^a, s) \quad (1)$$

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

$$171 \quad p^{OS} = c^d - V_s^a(w^a, s) \frac{w^g(s^d - s^g)}{(w^d + w^g)^2} \quad (2)$$

$$172 \quad p^{OS} = c^g + V_s^a(w^a, s) \frac{w^d(s^d - s^g)}{(w^d + w^g)^2} + \lambda^g \quad (3)$$

173 where λ^g is the shadow price of the groundwater constraint $w^g \leq \bar{w}^g$.

174 Equations 2 and 3 imply $c^d > p^{OS} > c^g$. The term $V_s^a(w^a, s) \frac{w^g(s^d - s^g)}{(w^d + w^g)^2}$ in Eq. 2

175 reflects the marginal benefits accrued to the agricultural sector by salinity reduction

176 through desalination, which warrants the supply of a desalinated-water quantity beyond
 177 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
 178 represents the marginal damage caused by groundwater due to increased salinity, which
 179 renders the optimal groundwater supply lower than that equating p^{OS} to c^g , even under
 180 an ineffective groundwater constraint (i.e., $\lambda^g = 0$).

181 From Eqs. 2 and 3, efficiency requires

$$182 \quad \frac{V_s^a(w^a, s)}{w} (s^d - s^g) = c^d - (c^g + \lambda^g) \quad (4)$$

183 Thus, the per-water-unit agricultural benefits associated with reducing salinity from
 184 s^g to s^d should be equal to the difference between the per-water-unit supply (plus
 185 scarcity) costs of the two sources. Equation 4 also ensures the cost-recovery condition
 186 $p^{OS}w \geq c^d w^d + c^g w^g$, which, based on Eq. 3, requires

$$187 \quad \frac{V_s^a(w^a, s)}{w} (s^d - s^g) \geq c^d - \left(c^g + \lambda^g + \lambda^g \frac{w^g}{w^d} \right)$$

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

189 The shadow values of the desalination and groundwater salinity constraints $s^d \geq \underline{s}^d$
 190 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
 191 that the larger the share of a water source in the total water supply, the larger the negative
 192 agricultural impact associated with a marginal increase in the salinity of that source.

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

197 $s^g = 0$). In this case, the optimal administrative water price, denoted p^{ZS} , is set such that
 198 $p^{ZS} = c^g$ (i.e., $p^{OS} > p^{ZS}$), the water consumptions of the two sectors satisfy $c^g =$
 199 $V_w^u(w^u) = V_w^a(w^a, 0)$, and water-supply costs are exactly recovered. The salinity shadow
 200 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
 201 those under the OS scenario due to both the different water volumes and the marginal
 202 effect of salinity on agricultural production at zero salinity, $V_s^a(w^a, 0)$ (our empirical
 203 crop-production functions imply $V_{ss}^a(\cdot) < 0$; i.e., the lower s , the more negative
 204 $V_s^a(w^a, s)$, and therefore the larger the marginal damage of salinity). To evaluate the
 205 overall damage caused by salinity, the economy's net benefits under the ZS and OS
 206 scenarios should be compared.

207 The purpose of the third scenario, termed *fixed salinity* (FS), is to compute the
 208 deadweight loss associated with overlooking the salinity-reduction effect of desalination.
 209 Suppose that the groundwater source is exhausted (i.e., $w^g = \bar{w}^g$), and additional
 210 desalination water is supplied ($w^d > 0$), where the water master refers to desalination as
 211 an instrument for resolving water scarcity, but ignores its impact on salinity; that is, she
 212 or he erroneously assumes that the salinity of the supplied blended water is fixed at s^g .
 213 Consequently, when determining the administrative water price, p^{FS} , according to Eqs. 2
 214 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;
 215 hence, $p^{FS} = c^d$ (i.e., $p^{FS} > p^{OS}$), and Eq. 3 implies $\lambda^g = c^d - c^g$. Nevertheless, the
 216 actual salinity of the mixed water, s , will be lower than s^g due to the desalinated water.
 217 To compute the actual levels of w^d and w^a , Eqs. 1 and 2 and the water-balance
 218 constraint $w^u = w^d + \bar{w}^g - w^a$ are used to obtain the conditions

219
$$c^d = V_w^u(w^d + \bar{w}^g - w^a) = V_w^a(w^a, s) + V_s^a(w^a, s) \frac{\bar{w}^g(s^d - s^g)}{(w^d + \bar{w}^g)^2}$$

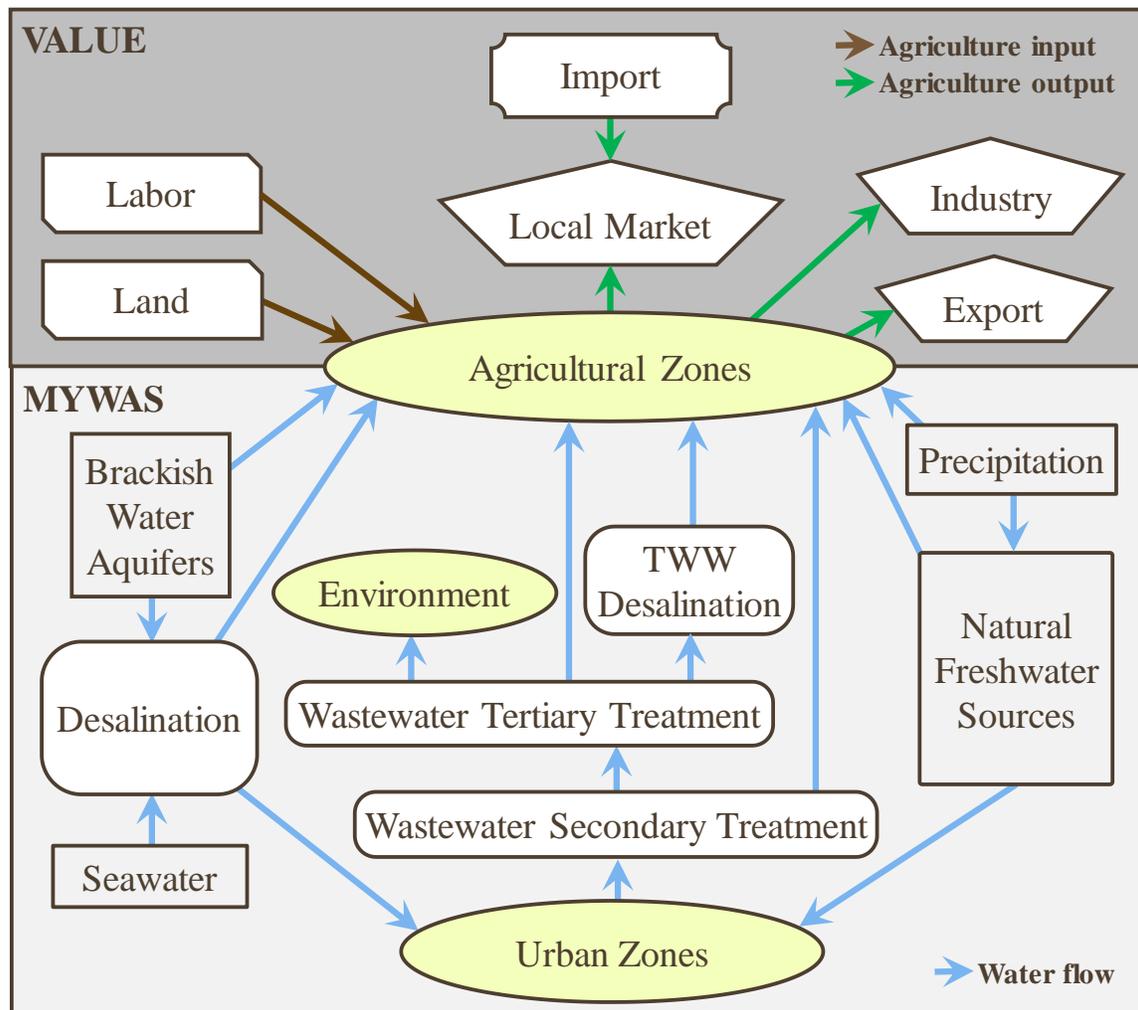
220 The difference between the economy's net benefits under FS and OS represents the
 221 welfare implications of ignoring the desalination's impact on irrigation-water salinity.
 222 Since the water price in the FS case exceeds that of the OS scenario, the consumer
 223 surpluses are lower than under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

225 2.2 Empirical Application

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

235 From MYWAS, we adopt the physical and economic setups of Israel's water
 236 economy and add, in specific nodes of the water network, optional infrastructures (e.g., a
 237 desalination facility), which may be constructed at an optimal timing and scope to be
 238 determined endogenously by the model. The investment criterion is maximization of the
 239 present value of the economic surplus created by the facility minus the investment cost.
 240 Precipitation-enriched freshwater sources include 16 natural water bodies (15 aquifers
 241 and the Lake of Galilee) and 3 groups of regional artificial reservoirs. There are 6

242 seawater-desalination plants (5 already exist in the center of Israel, and an additional
 243 optional one is assumed in the northwest region of Acre), and 4 brackish water-
 244 desalination plants, as well as 18 wastewater-treatment plants (WWTPs) applying
 245 secondary treatment, and 1 applying tertiary treatment to the sewage of the Tel Aviv
 246 metropolis. Each WWTP contains the option to introduce TWW-desalination activity
 247 (which necessitates tertiary pretreatment), where desalinated TWW can be delivered only
 248 to agricultural zones.



249

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

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

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

274 0.7 dS/m—the salinity added through domestic uses (IWA, 2012). Pursuant to
 275 regulations, the salinity of urban inflows is constrained to not exceeding 1.4 dS/m. The
 276 irrigation-water salinity in an agricultural region is the average of the salinities of the
 277 inflows to the region, weighted by their respective volumes.

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

$$290 \quad y_{lt}^i = \frac{ymax_l^i}{1 + \alpha_1 (\alpha_2 s_{lt}^i + \alpha_3 w_{lt}^i)^{\alpha_4}}^{\alpha_5} \quad (5)$$

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

296 To produce yield functions for the crops in the agricultural regions, we employ a
297 four-stage meta-analysis procedure. First, we generate a dataset of plant-level relative
298 yields (i.e., relative to $y_{max}_i^l$) under different combinations of water and salinity levels;
299 to this end, we use the crop model developed by Shani et al. (2007), adopting crop
300 salinity-tolerance parameters from Tanji (2002), and region-specific soil and climate
301 parameters from Kan and Rapaport-Rom (2012); second, to account for intra-field spatial
302 irrigation heterogeneity, we calculate field-level relative-yield levels by applying the log-
303 normal distribution function (Knapp 1992), calibrated for a Christiansen uniformity
304 coefficient of 85; third, we apply a non-linear regression to the generated dataset to
305 estimate the parameters α_1 through α_5 ; finally, we calibrate $y_{max}_i^l$ using the observed
306 yield, regional-water salinity and water application associated with each crop in each
307 agricultural region in the base year. The resultant functions exhibit the expected
308 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
309 the salinity of the water applied to a crop, the lower the salinity's marginal damage; this
310 implies that the lower the salinity in the water sources accessible by a water economy, the
311 larger (in absolute terms) the shadow value of the salinity constraint associated with those
312 sources.

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

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

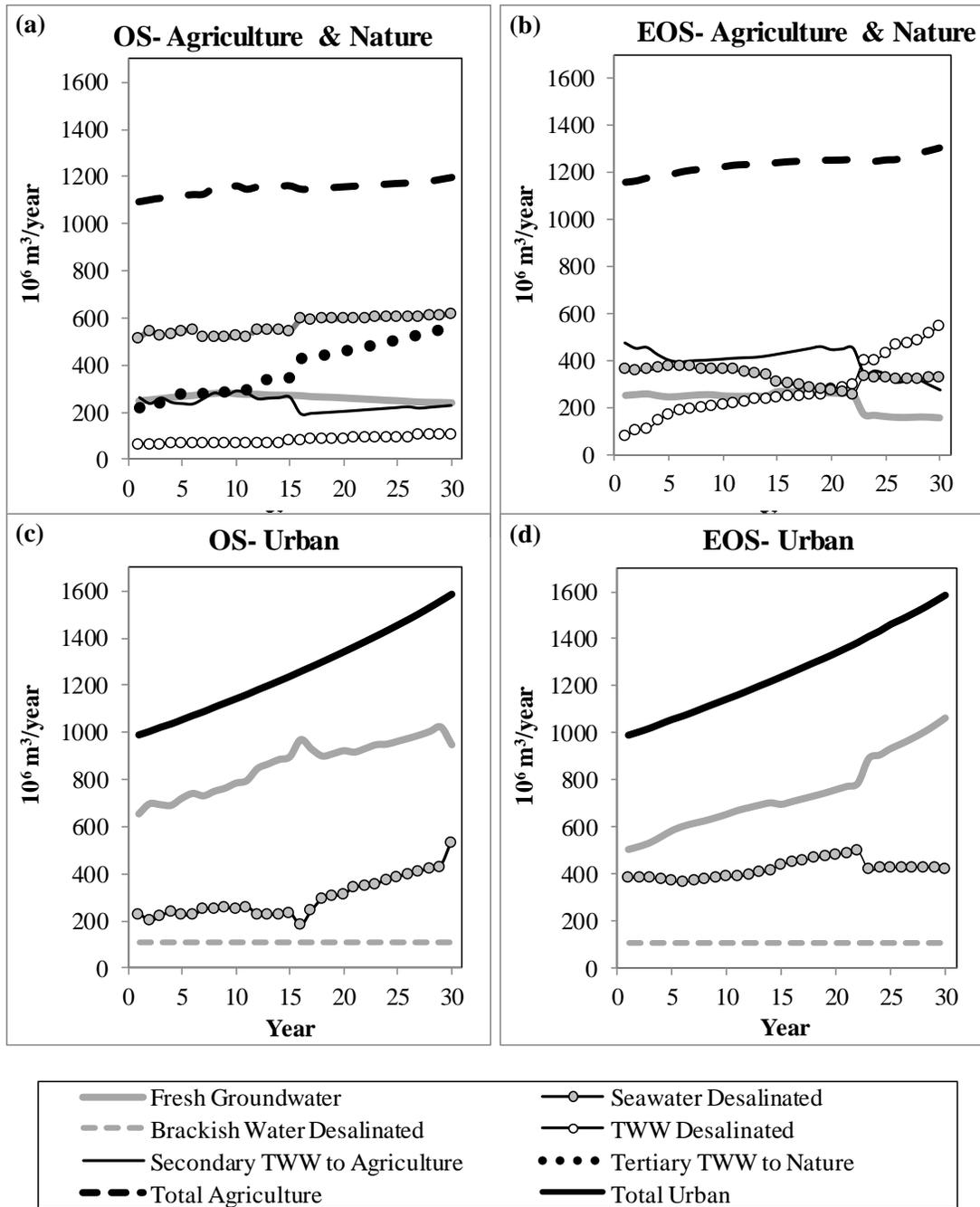
320 Our VALUE version extends that of Kan and Rapaport-Rom (2012) by introducing
321 the markets for vegetative agricultural products. Israel has a negligible impact on world
322 food prices, and employs import tariffs which mostly protect local fresh fruit and
323 vegetable products (Finkelshtain and Kachel 2009, Finkelshtain, Kachel and Rubin
324 2011). Thus, the local prices of agricultural outputs allocated to the processing food
325 industry and export markets equal the (exogenous) world prices, whereas the prices of
326 fresh vegetative products are determined in equilibrium subject to import tariffs. For the
327 import price of each crop's output, we select the country from which the import cost
328 (price plus transportation cost) is the lowest, and add the import tariff. We use constant-
329 elasticity demand functions for fresh agricultural produce, with elasticity parameters
330 adopted from Fuchs (2014). In addition, we account for Israel's immigration policy by
331 calibrating the model with regional constraints on foreign labor, upon which the Israeli
332 agriculture is heavily dependent (Kemp, 2010). These extensions enable us to study the
333 impact of changes in import tariffs and foreign-labor constraints on the water and
334 agricultural sectors.

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

341 (2007). Based on ICBS (2014) predictions, we assume an average annual population
342 growth rate of 1.8%, which shifts the demands for urban water and agricultural products
343 accordingly along the planning horizon. The average precipitation and annual enrichment
344 of natural freshwater sources (taken from Weinberger et al. (2012)) are assumed for the
345 entire simulated period. Initial monetary values are in 2015 US dollars, and scenario
346 results are reported for an average year throughout the 30-year planning period,
347 discounted to the 15th year (2030).

348 **3. Economic Implications of Salinity**

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



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361

362 **Figure 4. Trajectories of economy-wide annual water supplies to agriculture,**

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nature and the urban sector under the OS and EOS scenarios

364

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

	OS	EOS	ZS	FS
All Water Consumers				
Water Supply (10⁶ m³/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 (10⁶ m³/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 (10⁶ m³/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 (10⁶ day/year)				
Foreign	9.5	9.5	9.5	9.4
Local	5.1	5.1	5.1	5.0

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

368 **Table 2. Per-Annun Economy-Wide Average Shadow Values of Water-Quantity**
369 **and Water-Infrastructure-Capacity Constraints at Supply Points; Water-**
370 **Consumption Prices at the Consumption Points (\$/m³); Shadow Values of**
371 **Land and Foreign Labor^a**

	OS	EOS	ZS	FS
		Water Quantity		
Water Supply Constraint (\$/m³)				
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 (\$/m³)				
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 (\$/m³)				
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

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

373 **Table 3. Per-Annun Economy-Wide Average Salinity Levels and Salinity Shadow**
 374 **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 (\$10⁶ (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

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

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

377

Table 4. Per-Annun Economy-Wide Average Welfare Elements (\$10⁶/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

378

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

379

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

380

model.

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c. Since the computation of the areas beneath the calibrated constant-elasticity demand functions of urban

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water and agricultural products involves extrapolations, we normalize the surpluses of urban-water and

383

agricultural-product consumers to 0 under the OS scenario, and report their changes under each scenario

384

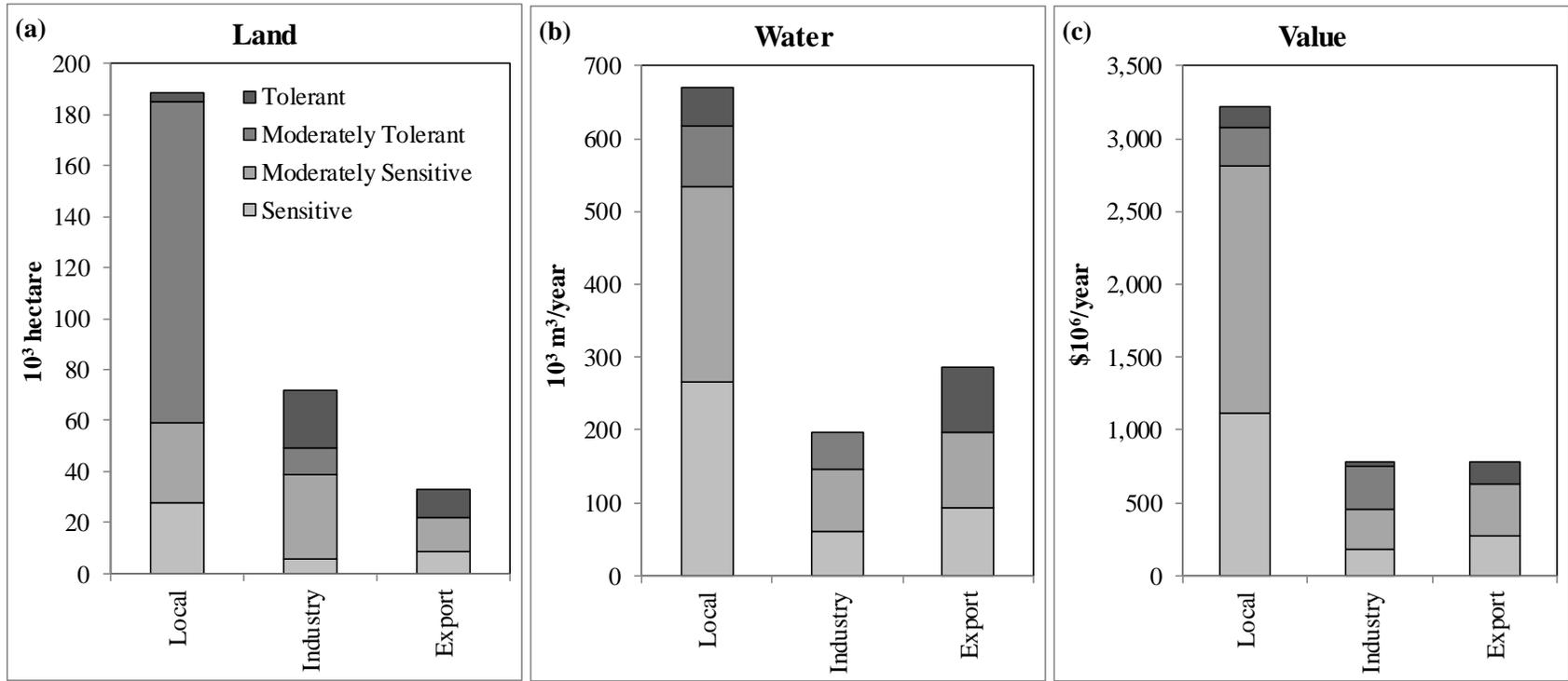
in comparison to the OS.

385

386 **Table 5. Laspeyres Quantity and Price Indices, and Value Index, of Agricultural**
 387 **Production by Salinity-Tolerance Crop Bundles, under the ZS and FS**
 388 **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

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



391

392 **Figure 5. Base-year (2015) economy-wide allocation of (a) land use, (b) water consumption and (c) production value across**

393 **four salinity-tolerance crop bundles, separated into production for local fresh produce, industry and export**

394 **agricultural markets**

395 3.1 Optimal Solution

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

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

418 of TWW, about half of which is disposed of by discharge to nature after tertiary treatment
419 (Figure 4a), and the rest delivered to irrigation.

420 If TWW discharge to nature is forbidden (scenario EOS), the average TWW
421 desalination increases by 195 million m³/year compared to the OS scenario, replacing
422 seawater desalination, which decreases by 119 million m³/year (Table 1). Returning to
423 OS, the total desalinated seawater remains at around 750 million m³/year during half of
424 the planning period, and then increases steadily, mostly through larger deliveries to urban
425 users. The total agricultural consumption of desalinated water gradually increases
426 throughout the period, from 581 to 724 million m³/year.

427 Desalinating water for irrigation was found economically viable by Hadas et al.
428 (2016) in a small-scale analysis for specific crops in the Arava region in southern Israel.
429 Our analysis finds this strategy to be warranted at the economy-wide level. We attribute
430 the motivation for the huge desalinated-water delivery to irrigation under the OS and
431 EOS scenarios to the agricultural protection policy in Israel, which for decades has led to
432 specific cropping specialization in the vegetative farming sector. Israel is a net importer
433 of grains and other low-value water-intensive agricultural products, which are consumed
434 mostly by the industrial sector free of import taxes (ICBS, 2017). On the other hand, as
435 already noted, a tariff policy effectively protects fresh produce of fruit and vegetables
436 from competing imports (OECD, 2018); hence, the prices of fresh products are generally
437 determined in equilibrium in the local markets, and the demand for those products
438 gradually increases via population growth. Apparently, these fresh-product crops require
439 large volumes of irrigation water, and are also relatively sensitive to salinity. To
440 elucidate, Figure 5 presents the agricultural land allocation, water use and production

441 value of crops in Israel in the base year, separated into production for the local fresh-
442 produce market, processing industry and export. In addition, each value is separated into
443 four crop bundles that classify the crops according to their salinity tolerance (Maas and
444 Hofmann 1977). Consider the salinity-sensitive and moderately sensitive crop bundles
445 that are produced for the local market: while only 20% of the total arable land in the
446 country is allocated to these two bundles (Figure 5a), their water consumption constitutes
447 46% of the total irrigation-water use (Figure 5b), and their production value amounts to
448 59% (Figure 5c). These patterns incentivize extensive allocation of desalinated water to
449 the agricultural sector to reduce salinity, increase the yields and per-hectare profits of
450 these crops, and in turn, the land allocated to them, and their total production value under
451 the optimal solution.

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

457 The computed shadow values and prices are instructive with respect to the relation
458 between the alternative water uses and the water salinity. Consider the negative shadow
459 value of the constraint requiring treatment of urban sewage at the WWTPs ($-\$0.09/\text{m}^3$
460 under OS, Table 2), implying that farmers should be compensated to voluntarily consume
461 non-desalinated TWW. This is because the salinity of TWW under OS (1.46 dS/m, Table
462 3) is higher than that of the irrigation water (0.6 dS/m) and therefore, farming use of
463 TWW, which saves the tertiary-treatment cost needed to allow TWW discharge to nature,

464 will reduce agricultural production. On the other hand, the salinity of desalinated TWW is
465 lower than that of the irrigation water, implying a positive shadow value ($\$0.38/\text{m}^3$). This
466 also explains the shadow value of the capacity constraint (Table 2) associated with TWW
467 desalination ($\$0.27/\text{m}^3$) being larger than that of the WWTPs ($\$0.05/\text{m}^3$).

468 Under the EOS scenario, TWW must be disposed of through irrigation; the statewide
469 average shadow value of this constraint is $\$0.10/\text{m}^3$ (Table 2). The social welfare is \$110
470 million a year lower than that under the OS scenario (Table 4), implying that a ban on
471 TWW discharge to nature is justified only if the associated (unknown) environmental
472 damage is larger than this welfare difference. The shadow value of the constraint
473 requiring sewage to be treated in WWTPs becomes $\$-0.40/\text{m}^3$, representing the
474 compensation to farmers for consuming TWW instead of desalinated TWW. In addition,
475 while the salinity of the irrigation water changes only slightly under the EOS vs. OS
476 scenario (Table 3), its total consumption rises (Table 1), which in turn reduces the
477 irrigation-water price (i.e., its value of marginal production) from $\$0.41/\text{m}^3$ under OS to
478 $\$0.22/\text{m}^3$ under EOS. Consequently, farming profits increase at the expense of those of
479 the water suppliers (Table 4). Note that as irrigation-water supply increases under EOS,
480 there is a rise in the shadow values of the constraints on agricultural land and foreign
481 labor as supplemental irrigation-water production factors (Table 2).

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

487 delivered to agriculture in the OS scenario, and therefore the salinity shadow values of
488 desalinated seawater and TWW become closer under EOS.

489 3.2 Zero Salinity

490 For the ZS scenario in our dynamic framework, envisage a case in which all of Israel's
491 water sources become pure freshwater (including the sea) exactly at the onset of the
492 planning period. Technically, given the infrastructures and crop portfolios in the
493 calibration year, MYWAS–VALUE searches for the optimal trajectories of water and
494 agricultural managements, where per-hectare yields increase to their maximum levels
495 under zero salinity (i.e., y_{max}), and the extraction cost of purely freshwater from all
496 primary sources (i.e., the sea and sources of natural fresh and brackish waters) is similar
497 to that of fresh groundwater (except that the water levels of the sea and brackish-water
498 aquifers are fixed, whereas that of fresh groundwater varies with the stock (Reznik et al.,
499 2017)). By comparing ZS to OS, we elicit the total welfare loss incurred by the presence
500 of salts in Israel's water sources, while accounting for the agronomic impacts and the
501 effects imposed on urban water users and agricultural product consumers.

502 The overall average annual welfare under the ZS scenario is \$1,340 million higher
503 than that of the OS case (Table 4). Without salinity, farmers' annual profits would be
504 \$733 million higher, consumers of agricultural products would gain an additional surplus
505 of \$281 million a year, the per-annum surplus of urban water users would increase by
506 \$907 million, and the yearly profit of water suppliers would decrease by \$581 million. To
507 put these findings in context, Qadir et al. (2014) estimated an average cost of \$440 per
508 hectare for salt-affected lands, amounting globally to \$27 billion a year. As already noted,
509 our analysis embeds direct and indirect salinity effects, and therefore obtains a much

510 larger damage of \$4,468/hectare to the economy as a whole, \$2,444/hectare of which is
511 farmers' losses.

512 In terms of water management, TWW desalination unsurprisingly vanishes, the
513 overall water use in the economy increases slightly, less TWW is discharged to nature,
514 and irrigation water use increases correspondingly (Table 1). Notice that the shadow
515 value of irrigation-water salinity becomes more negative (from \$-77 million per dS/m
516 annually under OS to \$-87 million per dS/m annually under ZS, Table 3), which is
517 explained by the combination of lower salinity and the production-function property
518 $\frac{\partial^2 y}{\partial s^2} > 0$. In addition, the irrigation-water price decreases (from \$0.41 to \$0.16/m³, Table
519 2) as theoretically predicted earlier, and the shadow values of agricultural land and
520 foreign labor increase accordingly.

521 The overall agricultural production value under ZS increases by 10% relative to OS
522 (Table 4). To examine this change, we present Laspeyres quantity and price indices of the
523 agricultural produce in Table 5 (using OS as the reference: OS = 100), separating the
524 production into the four salinity-tolerance crop bundles (recall Figure 5). We get a 13%
525 increase in the overall agricultural outputs (obtained through both per-hectare yield
526 changes and reallocation of regional arable lands across crops) and a 3% reduction in
527 average prices. Most of the effect of ZS is attributed to the increase in the outputs of crop
528 bundles that are sensitive and moderately sensitive to salinity, which together comprise
529 82% of the total production value for all three markets in the base year (Figure 5).

530 3.3 Fixed Salinity

531 We conduct the FS scenario in two stages: first, we run the MYWAS–VALUE model
532 while fixing the salinity concentrations in the agricultural regions at their base-year

533 levels; the output of this simulation is an infrastructure-development plan that the planner
534 considers optimal given her or his presumption of fixed irrigation-water salinity.
535 However, as already noted in our theoretical analysis, salinity will change, and in turn
536 affect crop production, agricultural output prices, farming profitability, land use, and
537 surpluses of producers and consumers in the agricultural sector. Therefore, in the second
538 stage, we rerun the model while allowing salinity to change, but at the same time forcing
539 the development of infrastructures to follow the infrastructure plan resulting in the first-
540 stage simulation. Thus, the FS scenario represents the expected evolution of the water and
541 vegetative-agriculture economies under efficient prices, given a non-optimally designed
542 water-infrastructure blueprint.

543 Compared to OS, under FS, the designer of the water-infrastructure system reduces
544 the desalination of brackish water, TWW and seawater by 41%, 100% and 25%,
545 respectively (Table 1). Instead, the reliance on fresh groundwater and secondary TWW
546 increases, which in turn elevates the irrigation-water salinity from 0.6 to 0.82 dS/m
547 (Table 3). In addition, water use by the agricultural sector decreases from 1,149 to 998
548 million m³/year (Table 1), where the scarcity of seawater desalination and TWW
549 capacities grow (Table 2). In accordance to the theory, the irrigation-water price increases
550 from \$0.41 to \$0.72/m³ (Table 2). Concomitantly, arable lands and foreign labor become
551 less scarce, and the total value of agricultural production declines by \$389 million a year
552 (7% compared to OS) (Table 4); the latter results from an 8% reduction in production
553 quantity, in combination with a 2% increase in agricultural output prices (Table 5), where
554 the shortage in local produce is partly covered by a 9% increase in the value of imported
555 products (Table 4). These changes reduce the surplus of consumers of agricultural

556 products and the farming profits by \$241 and \$81 million/year, respectively. In addition,
557 water consumption by the urban sector decreases by 21 million m³/year (2%) (Table 1),
558 the associated urban-water price rises from \$1.54 to \$1.77/m³ (Table 2), and urban water
559 users face a surplus reduction of \$465 million a year (Table 4). In line with the theory, the
560 only gainers from these changes are the water suppliers, whose net income increases from
561 \$549 to \$929 million a year. The overall annual deadweight loss compared to the OS
562 scenario is \$408 million, which is about \$1,360 per hectare of arable land.

563 3.4 Sensitivity to Changes in Agricultural Policies

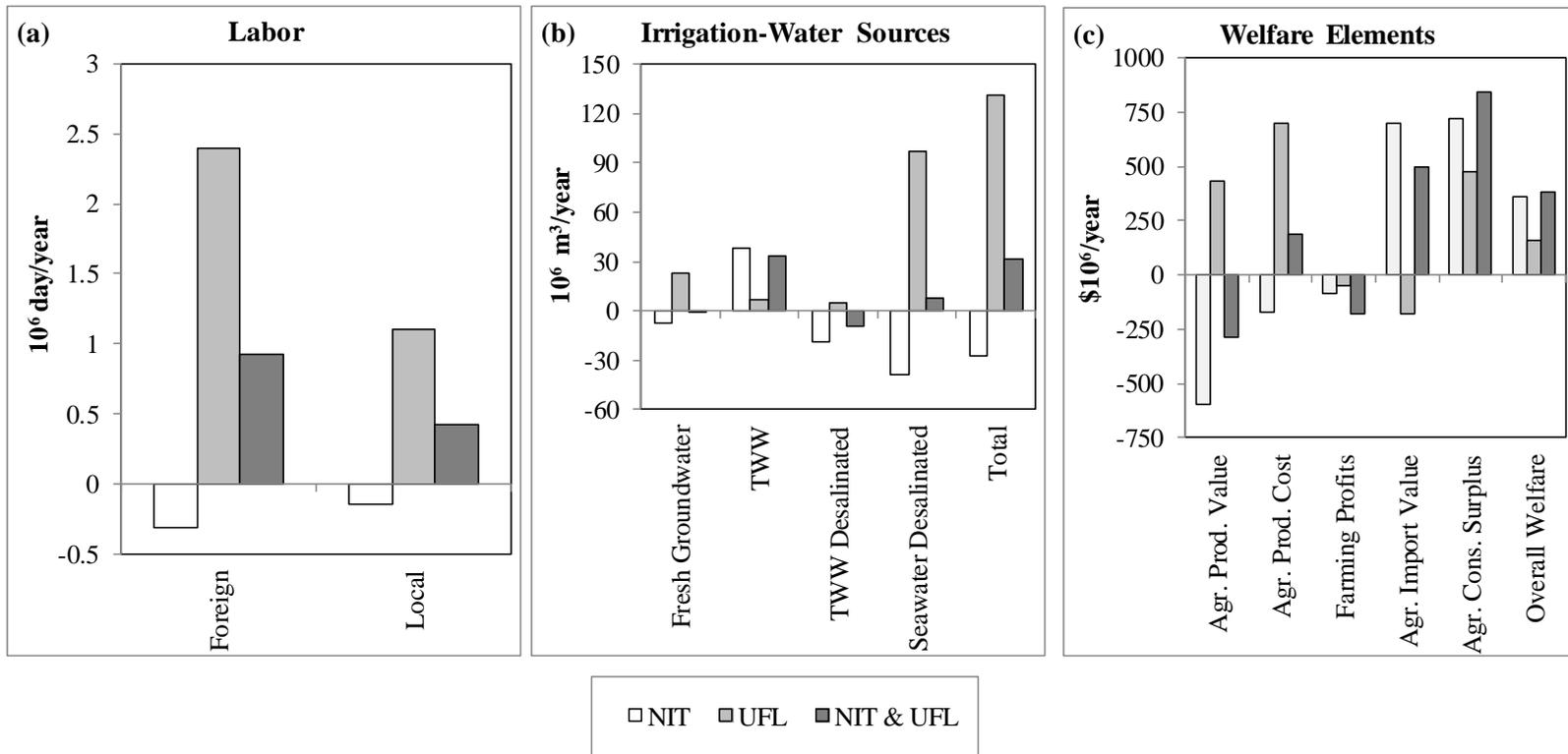
564 Israel's policies to protect its agricultural sector are continuously criticized economically
565 (OECD 2018), and are subject to public and political debate (Hendel et al., 2017).

566 Changes in Israeli agricultural policy can vary the irrigation-water demand, thereby
567 affecting the optimal infrastructural development of the water system. We study the
568 impact of three policy changes on the OS scenario; the first change assumes no import
569 tariffs on vegetative products (termed NIT). In the second, we remove labor quotas to
570 simulate an unlimited foreign labor (UFL) supply, where we assume no change in the
571 regulated minimum wage of the foreign workers. Moreover, based on Kimhi (2016), we
572 assume that foreign workers are not replacing local ones; hence, variations in foreign-
573 labor employment do not affect the local workers' wages. The third scenario combines
574 these two policy changes (termed NIT & UFL).

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

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

584 The UFL policy dramatically increases foreign (25%) and local (22%) labor, and also
585 enlarges water consumption (11%), particularly through an additional 97 million m³/year
586 of desalinated seawater. The vegetative-agriculture quantity index increases 15%, but the
587 price index declines 6% such that the production value rises only 8% (\$430 million/year).
588 However, since production costs increase to a larger extent, farmers lose \$53 million a
589 year from that policy, whereas consumers gain \$475 million a year, and the overall
590 welfare increases by \$156 million a year. If, in addition to UFL, the NIT policy is
591 applied, the welfare increase amounts to \$284 million a year. To summarize, while the
592 simulated labor reform has larger impacts on labor and water usage in the economy, the
593 abolishment of import tariffs induces a larger overall welfare increase, shifting surpluses
594 from farmers to agricultural-product consumers.



595

596

597 **Figure 6. Changes in (a) labor, (b) the use of irrigation-water sources and (c) welfare elements, in response to the abolishment**

598 **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 m³/year (Table 1), of which the agricultural sector consumes 262 million m³/year. In addition, based on land allocation in the base year, nearly 10⁹ m³/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 m³ of desalinated seawater and 0.03 m³ 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.

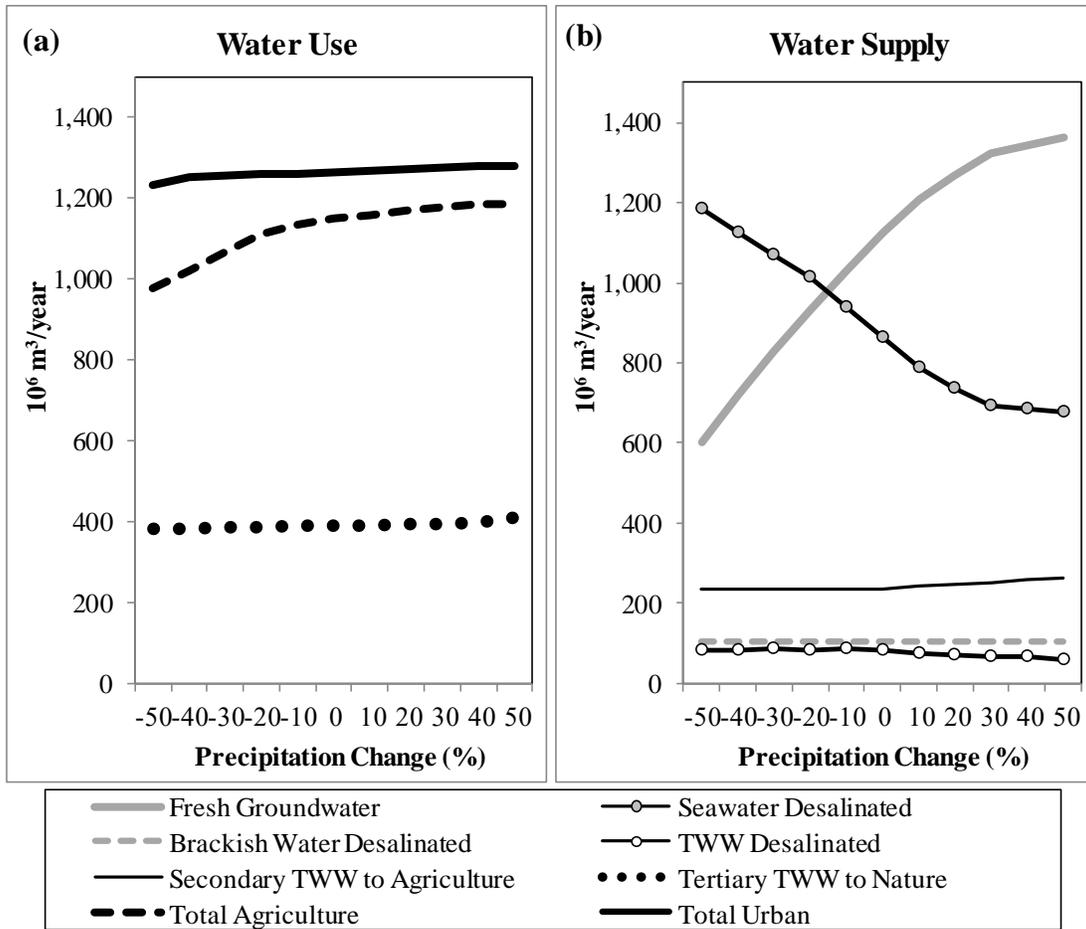


Figure 7. Effect of precipitation changes on economy-wide average annual (a) water use and (b) water supply under the optimal solution

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 $\mathbf{q}_{lt}^{ai} = (w_{lt}^{ai}, s_{lt}^{ai})$, 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, $\mathbf{q}_{lt}^{ai} = (\mathbf{q}_{lt}^{a1}, \dots, \mathbf{q}_{lt}^{al})$ 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 $\mathbf{q}_{lt}^{ao} = (\mathbf{q}_{lt}^{a1}, \dots, \mathbf{q}_{lt}^{ao})$. Note that locations of inflows and outflows of a particular activity may differ, as in the case of water-delivery activities. The vector $\mathbf{q}_{lt}^a = (\mathbf{q}_{lt}^{ai}, \mathbf{q}_{lt}^{ao})$ 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 Γ_{lt}^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 Γ_{lt}^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 $\mathbf{o} \rightarrow i = \{1 \rightarrow i, \dots, O \rightarrow i\}$ be the set of O outflows that feed inflow i . We denote by $\mathbf{b}_{lt}^{ao \rightarrow i} \equiv (w_{lt}^{ao \rightarrow i}, s_{lt}^{ao \rightarrow i}) = b_{lt}^{ao \rightarrow i}(w_{lt}^{a1 \rightarrow i}, \dots, w_{lt}^{aO \rightarrow i}, s_{lt}^{a1 \rightarrow i}, \dots, s_{lt}^{aO \rightarrow i}) = \mathbf{0}$ the set of balance constraints associated with inflow i , where $b_{lt}^{ao \rightarrow i}(\cdot)$ is a water quantity and salinity balance function;

accordingly, $\mathbf{b}_{lt}^{a0 \rightarrow i} = \mathbf{0}$ 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: $\mathbf{b}_{lt}^{ai \rightarrow o} \equiv (w_{lt}^{ai \rightarrow o}, s_{lt}^{ai \rightarrow o}) = b_{lt}^{ai \rightarrow o}(w_{lt}^{a1 \rightarrow o}, \dots, w_{lt}^{aI \rightarrow o}, s_{lt}^{a1 \rightarrow o}, \dots, s_{lt}^{aI \rightarrow o})$. For instance, a desalination plant produces freshwater and brine, where the function $b_{lt}^{ai \rightarrow o}(\cdot)$ specifies the dependence of one of the two outflows on the inflow elements, given the desalination technology. We denote by $\mathbf{b}_{lt}^{ai \rightarrow o} = \mathbf{0}$ 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 $\bar{\mathbf{q}}_{lt}^{aik} = (\bar{w}_{lt}^{aik}, \bar{s}_{lt}^{aik})$ and $\bar{\mathbf{q}}_{lt}^{aok} = (\bar{w}_{lt}^{aok}, \bar{s}_{lt}^{aok})$, 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 $\bar{\mathbf{q}}_{lt}^{ak}$ and $\underline{\mathbf{q}}_{lt}^{ak}$ activity a 's set of maximum and minimum constraints, respectively, and $\mathbf{q}_{lt}^{ak} = (\bar{\mathbf{q}}_{lt}^{ak}, \underline{\mathbf{q}}_{lt}^{ak})$.

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 $\Delta \mathbf{q}_{lt}^{ak}$ the changes in the set of constraints \mathbf{k} ; hence, given the initial level of constraints \mathbf{q}_{l0}^{ak} , there is $\mathbf{q}_{lt}^{ak} = \mathbf{q}_{l0}^{ak} + \sum_{t=1}^t \Delta \mathbf{q}_{lt-1}^{ak}$ 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 $\bar{\Delta} \mathbf{q}_{lt}^{ak}$ and $\underline{\Delta} \mathbf{q}_{lt}^{ak}$, respectively. The cost function $c_{lt}^a(\mathbf{q}_{lt}^{az})$ represents the capital and maintenance costs associated with \mathbf{q}_{lt}^{az} 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 $\Delta\mathbf{q}_{lt}^{az}$ and the non-water variables \mathbf{x}_{lt}^a associated with each activity, location and time. Given the linkage matrix Γ_{lt}^a and the initial level of constraints, \mathbf{q}_{l0}^{ak} , the decision variables should be set so as to maximize the net benefits of the water economy:

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

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

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

the set of exogenous constraints

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

the infrastructure capacity-expansion constraints

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

and non-negativity constraints associated with the respective variables.

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