Buying water for the environment: A hydro-economic analysis of Salton Sea inflows

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\section*{ABSTRACT}

As societies confront greater levels of water scarcity, conflict follows. Water transfers are increasing with irrigation water regularly considered in such discussions given it often comprises the bulk of water rights within a particular region, especially in water scarce environments. Here we analyze the impacts of three widely considered and implemented strategies to purchase water from irrigated agriculture—land fallowing, improvements in irrigation efficiency, and direct leasing. Our empirical application involves water transfers from Colorado River water rights holders to the Salton Sea, a critical ecological resource that has been in decline for many decades, with environmental damages estimated in the tens of billions of dollars. We develop a regional hydro-economic model that accounts for essential field-level agro-hydrologic processes related to crop production, irrigation, and salinity to evaluate the cost-effectiveness of these three programs. Results indicate that both fallow and direct water lease programs are capable of generating significant environmental water flows with relatively small decreases in agricultural production and no appreciable decrease in grower profits. Because these policies focus on a single input (applied water) rather than overall inflows to the Sea, the direct lease program—which is the most cost-effective approach for generating water conservation—may result in less inflows into the Sea than a land fallowing program.

\section*{1. Introduction}

As population, wealth, and climate change concerns continue to increase, significant attention and effort has been devoted to reducing water scarcity, particularly in the agricultural, environmental and urban sectors. Urban and agricultural users have traditionally reduced scarcity through water grabs, often from faraway places (Schwabe and Connor, 2012). This approach required the development of significant and expensive storage and conveyance systems that, in effect, changed the spatial and temporal distribution of water. With rising demand, and the lack of opportunities to find untapped (and unclaimed) inexpensive sources of water, efforts to reduce water scarcity increasingly have focused on a reallocation of water among sectors. Efforts to reallocate water are frequently, if not always, contentious, involving adversarial claims about water rights and the private and public benefits of different forms of water use and consumption (Hanak, et al., 2011). These efforts almost inevitably involve irrigated agriculture due to the significant amount of water resources it withdraws. For instance, nearly 69\% of freshwater withdrawals worldwide are withdrawn for irrigated agricultural production, with percentages in California and Australia’s Murray-Darling Basin exceeding 80\% (Schwabe et al., 2013).

From a policy perspective, appropriating water from irrigated agriculture for either urban or environmental purposes is challenging. Much of the water use by irrigated agriculture is based on senior water rights held under prior appropriation and beneficial use doctrines. Governments, in response, have considered, tried, and enacted a variety of approaches to appropriate water from irrigated agriculture. One approach is legislative, an example of which is found in California under the Central Valley Project Improvement Act (CVPIA) of 1992. The CVPIA is U.S. federal legislation that reallocated a portion of water away from irrigated agriculture for environmental concerns and an act that continues to face conflict and litigation today (Hanak, et al., 2011).

Another avenue is incentive-based approaches that encourage growers to irrigate more efficiently and allow them to maintain water rights. However, Scheierling et al. (2006) and Ward and Pulido-Velazquez (2008) found that subsidies for improved irrigation were
unlikely to reduce overall agricultural water consumption. A third approach is following programs where farmers are essentially paid for their water. These programs have been used in times of drought and as long-term solutions in arid regions (Radonic, 2014; Escriva-bou et al., 2016; Lofman et al., 2002). Lastly, water markets are being employed to reallocate water, often from irrigated agricultural uses to urban and environmental uses. When allocations are driven (at least in part) by economic markets and valuations, trade-offs between often higher value, low water volume urban uses and lower value, high water volume agricultural uses can be evaluated, and have been for many years with economic optimization models (Booker and Young, 1994; Colby-Saliba et al., 1987; Donohew, 2009; Howe et al., 1986; Brewer et al., 2006; Howitt, 1994). Australia provides a significant example of a water market in which trades are made for the environment, with most of the water being leased/purchased from irrigated agriculture (Wheeler et al. 2013; Connell, 2015). For example, Qureshi et al. (2010) found that a flexible water market was a much more cost-effective means of increasing environmental flows in the Murray Darling Basin than a less flexible irrigation subsidy scheme.

Given the continuing interest in and use of land fallingow and irrigation efficiency programs to generate water savings for short- or long-term reallocations, the intention of this paper is to compare these approaches with a water market, and thus highlight the perceived efficiencies of these schemes. Such a comparison has important policy implications along two fronts. First, further evidence of the cost-effectiveness of more flexible schemes such as water markets—which are advocated by economists—is important when one considers the challenges to appropriate scarce public dollars to shift water to the environment (or other public benefits). Second, a narrow focus on the costs to agriculture of reducing water use, or the required payments to incentivize such reductions, as the primary metric to evaluate alternative strategies may overlook potentially important externalities associated with such strategies, namely the impacts on return flows which can provide significant consumption, production, or preservation/recreational benefits to society. While our focus is comparing three different water leasing schemes for environmental flows into the Salton Sea in Southern California, the structure of our analysis and comparison of alternative water saving schemes can be applied to any region worldwide and for any beneficial use of water.

A second intention of this paper is to illustrate the importance of hydrological modeling and maintaining water balance in hydro-economic models evaluating the benefits of alternative schemes to make additional water available for sale or lease, especially when it comes to environmental water. Water market researchers and modelers are increasingly aware of the importance of maintaining water balance in their crop water production functions (Rosegrant et al., 2000; Zaman et al., 2009), yet less attention has been paid to agricultural return flows generated from drainage or run-off (Cai et al., 2003; Ward and Pulido-Velazquez, 2008; Qureshi et al., 2010). Indeed, all of the approaches mentioned above that are intended to save water require “designing institutional, technical, and accounting measures that accurately track and economically reward reduced water conservation” (Ward and Pulido-Velazquez, p. 18,215); without such attention, true savings may not be achieved due to either failure to acknowledge the loss in return flows or the additional water use that may occur via a “rebound effect” associated with behavioral adjustments that accompany changes in costs and/or efficiencies. Finally, we illustrate that water leasing schemes differ not only in terms of costs and return flows back to the environment, but also possibly in terms of the quality of the water returning to the environment, an issue that has not been illustrated in previous policy comparisons. In the example evaluated here, we find that the salinity of the return flows differ across schemes—an outcome that can be extremely important in regions worldwide, including significant parts of California and Australia that confront salinity issues (Schwabe et al., 2006).

To perform our analysis, we develop a regional hydro-economic model to assess leasing (i.e., year-to-year purchasing) of water for environmental flows, which could mitigate ongoing environmental damages while minimizing impacts to the local agricultural economy. Our programming model accounts for essential field-level agro-hydrologic processes, including nonlinear relationships and feedback mechanisms that exist among irrigation water volume, irrigation water quality, evaporation, soil salinity, drainage water volume, drainage water quality, and crop yield. The model and analysis are designed to provide a timely evaluation of one of the most pressing environmental and water related issues in California today, the Salton Sea.

Similar to other terminal lakes such as the Aral Sea Basin, the quantity and quality of flows into the Sea are threatened by upstream activity. Very expensive and protracted restoration plans have been developed which would reduce damages to the Sea under lower flow regimes (California Department of Water Resources, 2007; California Department of Water Resources and United States Bureau of Reclamation, 2015), yet little attention has been paid to the possibility of leasing/purchasing water from regional agricultural operators and the costs of alternative strategies to acquire that water. While legal and water rights issues—both at the state and federal levels—present significant barriers to such a scheme, there is precedence in the past for directing water conserved by agricultural operators with senior Colorado River water rights to the Salton Sea (QSA, 2003). Given the potential benefits of preserving the Salton Sea are enormous (Cohen, 2014), an analysis of water leasing schemes seems warranted regardless of current regulations and policies, particularly considering that these schemes could be implemented relatively quickly, as they would require no physical construction. While our analysis considers the costs of the different policy instruments to generate environmental water for the Sea with recognition of the consequent impact on quality and quality of return flows from drainage and run-off into the Sea, one could also view these strategies as generating beneficial use water for other sectors/purposes, including agricultural or urban uses.

2. Water for the environment: the Salton Sea

The Salton Sea, California’s largest lake by surface area (> 340 square miles), is a terminal lake located in southeastern California. Since its formation in the early 1900s, the Salton Sea has served as a rest stop for millions of migratory birds traveling along the Pacific Flyway, and year-round habitat for several endangered and sensitive species, including the desert pupfish, Yuma clapper rail, and burrowing owl (Audubon California, 2016; U.S. Fish and Wildlife Services, 2011). In the 1950s and 1960s, the Salton Sea was one of the most popular recreation sites in California, with more visitors annually than Yosemite National Park (Schwabe et al., 2008).

For much of the 20th century, the water level of the Sea was maintained by inflows consisting mostly of drainage (water traveling through the soil profile) and tail waters (water traveling across the soil surface) emanating from farmland around the Sea—waters laden with salts and fertilizers that have caused significant deterioration (Schwabe et al., 2008). As a terminal lake, its salinity increased rapidly during the last century due to evapoconcentration of salts. Few fish species can tolerate the present salinity levels of nearly 60 ppt, an outcome that also affects the bird species that rely on these fish. The primary source of drainage water is from the Imperial Irrigation District (IID), directly to the south. IID, which provides water for 440 to 540 thousand cultivated acres (King, 2007; Ross, 2010; Shields and Pacheco, 2016; Silva, 2013), has among the most senior rights and largest share of Colorado River water (~ 3.1 million acre-feet (MAF)). As most of the farmland uses furrow irrigation, there is significant drainage and tail water that ends up in the main drainage canals that empty into the Alamo and New Rivers, which empty into the Salton Sea (Fig. 1).

In addition to salinization, the Sea is shrinking due to decreased inflows, partly as a result of the 2003 Quantification Settlement Agreement (QSA). The QSA is a federal-state-local agreement that
allowed the transfer of a large amount of irrigation water from IID to urban users in Southern California (QSA, 2003). Because of the expected reduced inflows as a result of the QSA’s out-of-region transfers, the State of California mandated mitigation water transfers be made by IID directly to the Sea to keep increases in Salton Sea salinity at or below levels that would have occurred without the QSA urban water transfers. Nevertheless, flows into the Sea have decreased by approximately 10% since enactment of the QSA (Tetra Tech, 2016; USBR, 2000). In 2017, the mitigation transfers stopped and thus inflows will likely further decrease, resulting in an acceleration of the salinization and shrinking of the Sea if no other action is taken. By 2021, the Sea will become too saline for fish populations. Another 25 years will see the salinity even too high for brine shrimp to thrive, leading to algal and microbial populations rising unchecked. These ecological conditions will severely affect the ability of the Sea to sustain migratory birds (Cohen and Hyun, 2006).

The estimated public damages from allowing the Sea to continue to decline are significant. Expected damages from lower flows are estimated at between $11 and $70 billion over the next 30 years, and include health impacts from increased respiratory illnesses associated with higher concentrations of airborne particulates from exposed playa, decreased property values, decreased recreation, and the loss of nonuse values associated with wildlife habitat (Cohen, 2014). In response to these possible damages, the state proposed a 70-year plan that included making the Sea smaller and less saline, diverting salts into a brine sink, developing habitat, and employing dust management. Capital and operation and maintenance costs were estimated to be about $10.6 billion initially plus $169 million annually (California Department of Water Resources, 2007). With the long term plan currently lacking funding, the state more recently released a lower cost interim plan that identifies proposed actions to be taken over the next ten years (California Natural Resources Agency, 2017). While the ten year plan does not address the salinity of the Sea, nor does it alter the size of the Sea, it acknowledges that rising salinity will likely still render the Sea unusable to many species which, in turn, may have repercussions for the effectiveness of the habitat created (see Table A1 for a comparison of the plans).

In the next section we discuss the model we develop to evaluate possible policy strategies to generate additional beneficial use water to help address the region’s water scarcity, with particular attention towards maintaining environmental inflows to the Salton Sea. Similar to the efforts under the QSA to generate mitigation water as inflows to the Salton Sea, we evaluate an extension of those efforts under different policy strategies that incentivize water conservation by IID agricultural operators with recognition of the return flows to the Sea (which are tracked via imposing water balance on our model results). We emphasize that while our focus is on the cost-effectiveness of more flexible policies in generating environmental flows for the Salton Sea, environmental flows are just one of multiple beneficial uses that could be considered for such conservation efforts on the part of IID.

3. **Methods**

To evaluate three different possible programs to generate beneficial flows that may be used for the Salton Sea, we developed a constrained hydro-economic optimization model that simulates water flows, cropping patterns, crop yields, and profits in IID, and evaluates inflows for the Salton Sea.

3.1. **Salton Sea water inflows**

Annual inflow to the Salton Sea from IID can be expressed as:

\[
SI = DW + TW + EW
\]

where \(SI\) is inflows to the Sea from IID, \(DW\) is agricultural drainage (water that travels through the soil profile) flows, \(TW\) is tail water (agricultural surface runoff) flows, and \(EW\) represents direct transfers, or mitigation water, of Colorado River water to the Sea (environmental water). The Salton Sea water budget in Eq. (1) neglects rainfall (which is only a few inches per year) and assumes that the contribution of regional groundwater to agricultural drain flows is negligible.
3.3. Farm-level water budget and crop yield

Considering Eq. (1), it would seem that shifting water from \( IW \) to \( EW \) would directly increase Salton Sea inflow. However, it is not that straightforward. Reducing \( IW \) decreases drainage \( (DW) \) and tail water \( (TW) \), so increases in \( EW \) will be partially offset by decreases in \( DW \) and \( TW \). Additionally, the salinity of \( DW \) will likely increase.

To fully account for the effects of shifting water from \( IW \) to \( EW \), it is necessary to model the relationship between the depth of irrigation water applied at the farm-level, \( w \), and the drainage generated, \( D \). This relationship is nonlinear because of feedback mechanisms that exist between irrigation amount, irrigation water salinity, root zone soil salinity, and crop water use (or yield). Both drainage \( D \) and crop yield \( Y \) can be expressed as functions of \( w \) that depend on a number of crop and system parameters, including the potential crop transpiration rate, \( T_p \); the amount of direct evaporation, \( E \); the salt tolerance of the crop, \( E_{C_W} \); the salinity of the irrigation water, \( E_{C_W} \); and the fraction of applied water that runs off the field as tail water, \( \tau_f \):

\[
D(w) \equiv D(w; T_p, E, E_{C_W}, E_{C_W}, \tau_f)
\]

\[
Y(w) \equiv Y(w; T_p, E, E_{C_W}, E_{C_W}, \tau_f, \tau_t)
\]

As shown in Appendix A1, we modeled Eqs. (3) and (4) using the steady-state analytical approach of Skaggs et al. (2014). The model is an explicit analytical solution to a one-dimensional, steady-state, mass-conservative, physical-mathematical model of root zone water uptake and solute transport processes. The model accounts for the effects of irrigation and soil water salinity on crop growth. We expand upon the functions of Skaggs et al. (2014) by incorporating evaporation, as shown in Appendix A2. Crop-specific parameter values for \( T_p, E, Y_p, \) and \( E_{C_W} \) are provided in Table A2 for traditional irrigation methods, which in IID have been flood and furrow irrigation. Transpiration and evaporation calculations are described in Appendix A2. Tail water runoff in IID has been estimated to range from a few percent to 30 percent, with an IID average of about 17 percent (Bali et al., 2001). We assume \( \tau_f = 0.17 \) for traditional irrigation methods. A graphical representation of field level flows is shown in Fig. 3. Farms in the IID receive Colorado River water for irrigation via the All-American Canal, whereas surface runoff and tile drainage flows are conveyed to the Salton Sea via the New and Alamo Rivers. The irrigation water salinity \( E_{C_W} \) is approximately 1.2 dS/m in IID.

3.4. Optimization model and scenario modifications

Our optimization model follows the general farm-level framework employed in Levers and Schwabe (2017), but is applied at a regional level. The objective function maximizes regional farm profits, which depends on profit per acre for a given crop, \( c \), expressed as:

\[
k_c(p_c - H_c)Y_c - q_c - h_c - w_c p_w
\]

3.2. Imperial irrigation district (IID) water budget

The IID has rights to 3.1 MAF per year of Colorado River water, which is used for irrigation \( (IW) \), mitigation or other direct transfers to the Salton Sea \( (EW) \), transfers/sales to other Southern California urban water agencies \( (UW) \), unneeded or unused water \( (OW) \), and various miscellaneous and domestic uses in the IID service area \( (MW) \):

\[
3.1 \text{MAF} = IW + EW + UW + MW + OW \tag{2}
\]

By far the largest water budget component is \( IW \), averaging 2.523 MAF from 2013 to 2015. \( MW \) represents less than one percent of water allocation (Shields and Pacheco, 2016). \( UW \) and \( EW \) have both been increasing during the QSA period. From 2013 to 2015, average \( UW \) was 304 TAF, and the average \( EW \) was 105 TAF (Imperial Irrigation District, 2016a).

Fig. 2. Reported Imperial Irrigation District applied water, estimated IID sea inflows (adjusted for flows from Mexico), estimated drainage water, reported out-of-region transfers, and reported crop area. (California Natural Resources Agency, 2006; Imperial Irrigation District, 2016a,c; King, 2007; Ross, 2010; Shields and Pacheco, 2016; Silva, 2013; United States Geologic Survey, 2004-2015; United States Geologic Survey, 2004).

Fig. 3. Field level water flows.
where \( x_c \) is profit per acre; \( p_c \) is crop price; \( Y_c \) is actual crop yield (Eq. (4)); \( H_c \) is yield-related harvest costs; \( h_c \) is non-yield related harvest cost per acre; \( q_c \) is production costs; \( w_c \) is applied water depth; and \( p_c \) is the price of water paid by farmers. Costs and crop prices (high, middle, and low estimates) for specific crops are given in Table A2.

To validate the model, we first used it to evaluate a baseline scenario that is comparable to recent years (2013–2015) under the QSA. After illustrating that the model performs well in representing actual conditions and agricultural outcomes under the QSA, the model is used to evaluate three possible post-2017 water transfer programs: (1) a fallowing program, based on an IID program already in place; (2) an efficient irrigation program, also based on an existing program; and (3) a direct water leasing program. We evaluate these three hypothetical post-2017 water transfer programs in terms of costs, generation of beneficial use water, impacts on the Salton Sea in terms of return flows and beneficial use water that could be used as environmental water, and agricultural profits. We then provide estimates of the benefits from the literature to highlight whether and to what extent plans pass a benefit-cost test.

### 3.4.1. Baseline

For the 2013–2015 baseline scenario, the objective function for maximizing profits \( \Pi \) is:

\[
\text{max} \Pi = \sum_{x, w} p_c x_c
\]

subject to the following constraints:

\[
x_c^{\text{min}} \leq x_c \leq x_c^{\text{max}}
\]

(7)

\[
\sum_c x_c \leq 450,000 \text{ac}
\]

(8)

\[
x_{\text{carrot}} + x_{\text{broccoli}} + x_{\text{lettuce}} + x_{\text{onion}} \leq 112,500 \text{ac}
\]

(9)

\[
\sum_c x_c w_c \leq 2.523 \text{MAF}
\]

(10)

where \( x_c \) is the acreage for crop \( c \), and \( x \) and \( w \) are vectors of the decision variables. Eight crops—comprising about two-thirds of the cultivated area—were chosen to represent the most commonly grown crops in the valley, including four field crops—alfalfa, Bermuda grass, Sudan grass, and sugar beets—and four garden crops—broccoli, onions, carrots, and lettuce (Imperial Irrigation District, 2017). The bounds imposed by Eq. (7) ensure that the simulated baseline cropping pattern is reasonably consistent with recent actual acreages reported for IID (Table A2), which indicated the majority of the cultivated acreage consists of field crops. Eq. (8) caps total available farmland at 450 K acres, the approximate average of cropped acreage in 2013–2015 (Shields and Pacheco, 2016). Eq. (9) limits garden crop acreage to 25 percent of the available farmland, in accordance with recent cropping patterns (Imperial Irrigation District, 2017). Eq. (10) limits total irrigation to 2.523 MAF, which is the average water applied to crops in the baseline time period (Imperial Irrigation District, 2016a).

### 3.4.2. Post-2017 water leasing programs

We consider three hypothetical programs where beneficial use water is leased from growers that could be used for maintenance of the Salton Sea. The model allows us to analyze how different levels of flexibility offered to growers in terms of the strategies they can adopt to lease water impact grower profits and the amount of water growers are willing to lease. By evaluating grower willingness to lease for different offered prices, we can identify the opportunity costs to growers from engaging in particular types of water use reduction schemes to generate additional beneficial use water that can be used as inflows to the Sea under different programs.

While the focus of our analysis is on the opportunity costs of generating inflows into the Salton Sea, the policies evaluated focused on a single input—irrigation water—to achieve a particular level of inflows, even though inflows consist of both environmental flows (irrigation water purchases) and drainage and tailwater inflows. Since drainage and tailwater inflows are similar to nonpoint sources and thus difficult to assign any specific inflow quantity to any particular farm, it is reasonable to expect the policy to be implemented on the more easily measurable input—irrigation water. As such we are, in effect, comparing the cost-effectiveness of three second-best policies (Baumol and Oates, 1988; Larson et al., 1996). Finally, given the importance of crop prices to such opportunity costs, sensitivity analysis is performed over a range of price levels (high, medium, and low) for each crop, as shown in Table A2.

#### 3.4.2.1. Fallowing program

The land fallowing program we analyze is based on a current IID program (Imperial Irrigation District, 2016a). In exchange for fallowing land, a farmer receives a set price per acre foot of water that would have been used had the land remained in production. The fallowing program introduces a water lease payment to the farmer, as illustrated in Eq. (11):

\[
\text{max} \ Pi = \sum_{x, w} p_c x_c + (x_c^b - x_c) w_c^b p_f
\]

subject to the following constraints:

\[
0.8 x^b_c \leq x_c \leq x^b_c
\]

(12)

\[
w_c^b \leq w^b_c
\]

(13)

where \( w_c^b \) is the depth of water applied to crop \( c \) in the baseline scenario; \( p_f \) is the price per acre-foot paid for water leased (conserved); and \( x^b_c \) is the area of crop \( c \) in the baseline outcome. Eq. (12) limits the amount of land that a grower can fallow to 20 percent of their baseline acreage. The purpose is to restrict the fallowing program to active, productive land. This percentage is supported by California Department of Water Resources (CDWR), which recommends agencies not approve transfers that result in more than twenty percent of a crop being fallowed (CDWR and USBR, 2015). Eq. (13) constrains the amount of water that can be applied to non-fallowed land to baseline levels, thereby ensuring that water is actually conserved. To the extent that growers would shift water from their fallowed acreage over to non-fallowed acreage, our results underestimate the opportunity costs of such programs.

#### 3.4.2.2. Improved irrigation efficiency program

The second possible program we analyze, referred to as the irrigation efficiency program, is also based on an existing IID program (Imperial Irrigation District, 2016c). Under this program, farmers are paid for water that is conserved due to the adoption of more water-efficient irrigation practices and technologies. With traditional IID irrigation methods, opportunities for farm-level conservation could include reduced conveyance losses, reduced tail water fractions, and reduced evaporative water losses from improved irrigation systems or other methods. Given that tail water and drainage make up part of the Salton Sea inflows, real water savings from changing irrigation methods arise from reducing evaporative losses (assuming crop transpiration varies minimally across irrigation methods). Drip irrigation, particularly subsurface drip irrigation, has been touted as a means to minimize evaporative water losses due to reduced soil surface wetting (Ayars et al., 2015), although in some instances that effect has been found to be offset by need for more frequent irrigation (Burt et al., 2005). Nonetheless, subsurface drip has been found to reduce overall water use by up to 25 percent (Alam et al., 2009). Emerging technologies for reducing evaporation include sprayable, biodegradable polymer membranes that cover the soil surface (Adhikari et al., 2015; Johnston et al., 2016).

Recognizing that most irrigation in IID is flood or furrow, we
develop two estimates to represent the annual costs, \( I \), of improving irrigation efficiency—\$400/ac and \$800/ac. These values were chosen to show approximate yearly costs of improved irrigation. The University of California Cooperative Extension cost and return publications indicate approximately a \$400 difference in yearly cost between flood and drip irrigation for alfalfa (Mayberry and Meister, 2003a; Putnam, et al., 2014). A range of \$500 to \$1800 per year cost for drip irrigation on small vegetable farms is given by the University of Florida Extension (Simonne et al., 2012). Using data from Amosson et al., (2002) which estimates cost by applied water volume, the additional yearly cost of subsurface drip irrigation for alfalfa would be over \$600 (this does not include additional investment costs of \$1000 per acre). All publication costs were adjusted for inflation. We assume that implementation of this more efficient irrigation technology will reduce evaporative water losses by 50 percent and eliminate tail water runoff, yet drainage is still estimated by Eq. (3). Given these assumptions, our optimization framework is defined as follows:

\[
\max \pi \Pi = \sum \pi_i x_i + \pi_c x_c^c
\]  

subject to the following constraints:

\[
\pi_c^c = (p_c - H_i) \bar{Y}_c - (q_i + I) - h_i - w_c^c p_c + (w_c^c - w_c^e) p_c
\]

where \( w_c^c \) is the depth of water applied to crops with the improved irrigation system, \( x_c^c \) is the acreage with improved irrigation, and \( p_c \) is the price per acre-foot paid to lease water for other beneficial uses, including environmental transfers. Similar to the following program above, only 20 percent of the baseline crop area can be followed (Eq. (20)), and only 20 percent of the baseline water can be leased (Eq. (21)). Again, the constraints are intended to restrict the program to farmers who continue to be active, productive growers, and capture elements of current programs that are implemented for these reasons as well as to reduce potential third-party effects.

While each of these programs consists of leasing water, it is useful to recall that water rights in the western U.S. are generally tied to specific lands and beneficial uses. Whereas the first two programs actively conserve water through changes in crop management under constraints that eliminate any gaming by the growers (e.g., restricting water use on other crops), the flexibility associated with this last program may result in growers applying less water yet not suffering any crop yield or revenue losses if previously the growers were operating far to the right and on the relatively flat part of their crop-water production function. In principle, reducing water use with no adverse effect on yield may call into question the “beneficial use” of the water in the first place and thus create a potential challenge to the water rights themselves. Such a discussion, while important, goes beyond our focus.

4. Results

4.1. Baseline

The baseline scenario is intended to be comparable to IID water use and production during recent years (2013–2015) under the conditions of the QSA. We assumed \( EW \) was equal to the average mitigation water for those years, which was 105 TAF. The baseline results for crop acreages, applied water rates, relative yields, and regional profits using the midrange crop prices are shown in Tables 1 and 2.

The estimated water flows for the baseline scenario are in reasonable agreement with available data, and the optimized irrigation rates for different crops are compatible with reported irrigation rates. For example, the model estimated applied water of 7.8 feet for alfalfa is comparable to the value of 7 feet reported by Summers and Putnam (2008), and the estimate of 2.9 feet for onions is comparable to the range of 2.5 to 3 feet reported in Pelter and Sorensen (1999). The leaching fractions calculated for different crops varied greatly (between 9 and 47%), being much larger for garden crops than for field crops. However, the applied water for garden crops is significantly lower than that of field crops, so despite the wide-ranging leaching fractions, the amount of water leached was relatively uniform, ranging from 7 to 14 in. for field crops and 7 to 16 in. for garden crops. The calculated area-weighted average leaching fraction for IID was 20.8 percent, which is compatible with the historical IID leaching fraction of 19 percent reported by Gibson (2012). Total profits for the region, using midrange crop prices, were \$365 million. Results using low and high crop prices show no significant qualitative differences and are available

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**Table 1**

Baseline simulation results.

<table>
<thead>
<tr>
<th>Crop</th>
<th>Alfalfa</th>
<th>Bermuda</th>
<th>Sudan</th>
<th>Sugar Beets</th>
<th>Broccoli</th>
<th>Carrots</th>
<th>Onion</th>
<th>Lettuce</th>
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<tr>
<td>Area (10^3 ac)</td>
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<td>35</td>
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<td>20</td>
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<td>4.71</td>
<td>2.05</td>
<td>2.99</td>
<td>2.99</td>
</tr>
<tr>
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<td>0.994</td>
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<tr>
<td>Leaching Fraction</td>
<td>0.15</td>
<td>0.12</td>
<td>0.09</td>
<td>0.15</td>
<td>0.35</td>
<td>0.47</td>
<td>0.46</td>
<td>0.45</td>
</tr>
<tr>
<td>Profit ($/ac)</td>
<td>614</td>
<td>746</td>
<td>509</td>
<td>1218</td>
<td>1617</td>
<td>644</td>
<td>269</td>
<td>1417</td>
</tr>
</tbody>
</table>
from the authors upon request. We do not have data for actual profits, but the average revenue for field crops from 2013 to 2015 was $1559 per acre, which is similar to the average revenue generated in our model for field crops: $1521 per acre. For garden crops, the reported revenue was $6560 per acre, which compares with our generated value of $5810 per acre (Valenzuela, 2014, 2016). The modeled Sea inflow, equal to the sum of the mitigation water and calculated drainage and tail water flows, was 941 TAF, which compares reasonably well with the 965 TAF calculated from USGS data (See Section 3.2). The model predicts 81 TAF of the available 2.523 MAF was left unused. This is not necessarily unrealistic however, as the Imperial Irrigation District has "...consistently underused water between 2013 and 2017, with annual underruns varying between 797 acre-feet and 97,188 acre-feet" (James, 2018), with a mean of 56 TAF over the 2013–2015 period (Imperial Irrigation District, 2016a).

Besides simply not needing all the water, a number of possible other reasons may explain this. For example, the tail water fraction we use from Bali et al. (2001) could be too low; rerunning the baseline model with a slightly higher tail water fraction eliminates the unused water. It is also possible that actual irrigation depths in IID for some or all crops were greater than the optimal level determined by the model. The calculated salinity of the inflow, meanwhile, was 3.0 dS/m. While we do not have data for direct comparison, we note that the salinities of the main channels for conveying water to the sea, the Alamo River and the New River, were 3.1 dS/m and 4.1 dS/m, respectively, between 2004 and 2010 (CDWR, 2011).

The objective of the remainder of this paper is to evaluate the opportunity costs of generating beneficial use water that can be used as environmental flows into the Salton Sea under different water leasing programs in the absence of mandated mitigation and while accounting for the implications on return flows to the Sea. In the coming years, the terms of the QSA will allow IID to sell up to an additional 174 TAF of conserved water to San Diego County Water Authority (SDCWA) and to Coachella Valley Water District (Imperial Irrigation District and San Diego County Water Authority, 2003). Considering that the QSA mitigation transfer requirements will cease, the conservation required to generate a net increase of about 69 TAF of additional transfers is largely already in place. The profitability is great for these transfers—San Diego pays more than $600 per ac-ft (San Diego County Water Authority, 2016), a large sum relative to the $20 per ac-ft that farmers pay (Imperial Irrigation District, 2016b); consequently, any profit-maximizing analysis always chooses transfers (not shown). Hence, we assume that in coming years UW (urban water transfers) will be 174 TAF more than the average sold between 2013 and 2015, that the 105 TAF of mitigation water under the QSA will cease and that IID conservation programs currently in place remain, including fallowing. Thus, with these assumptions, plus the assumption that MW

### Table 2
Comparison of baseline simulation with reported estimates of actual values. Model values with an asterisk were model inputs; other model values were computed outputs.

<table>
<thead>
<tr>
<th>Model</th>
<th>Reported</th>
</tr>
</thead>
<tbody>
<tr>
<td>Regional leaching fraction</td>
<td>0.208</td>
</tr>
<tr>
<td>Profits and Revenue</td>
<td></td>
</tr>
<tr>
<td>Total profits (million dollars)</td>
<td>365</td>
</tr>
<tr>
<td>Field Crop Revenue ($/ac)</td>
<td>1521</td>
</tr>
<tr>
<td>Garden Crop Revenue ($/ac)</td>
<td>5810</td>
</tr>
<tr>
<td>Water Accounting (MAF)</td>
<td></td>
</tr>
<tr>
<td>Total water available</td>
<td>2.523* (IW)</td>
</tr>
<tr>
<td>Water applied to crops</td>
<td>2.442</td>
</tr>
<tr>
<td>Total Inflows to Salton Sea</td>
<td>0.941</td>
</tr>
<tr>
<td>Drainage</td>
<td>0.421</td>
</tr>
<tr>
<td>Tail water</td>
<td>0.415</td>
</tr>
<tr>
<td>Excess Water</td>
<td>0.081</td>
</tr>
<tr>
<td>Mitigation water</td>
<td>0.105*</td>
</tr>
</tbody>
</table>

### Table 3
Profit per acre foot of water ($ per acre-foot) for each crop under fallowing policy given low, middle, and high crop prices ($ per unit).

<table>
<thead>
<tr>
<th></th>
<th>Low Prices</th>
<th>Middle Prices</th>
<th>High Prices</th>
</tr>
</thead>
<tbody>
<tr>
<td>Sudan</td>
<td>44</td>
<td>76</td>
<td>99</td>
</tr>
<tr>
<td>Alfalfa</td>
<td>11</td>
<td>78</td>
<td>112</td>
</tr>
<tr>
<td>Onions</td>
<td>14</td>
<td>93</td>
<td>272</td>
</tr>
<tr>
<td>Bermuda</td>
<td>62</td>
<td>165</td>
<td>234</td>
</tr>
<tr>
<td>Sugar Beets</td>
<td>130</td>
<td>259</td>
<td>385</td>
</tr>
<tr>
<td>Carrots</td>
<td>42</td>
<td>308</td>
<td>545</td>
</tr>
<tr>
<td>Broccoli</td>
<td>204</td>
<td>789</td>
<td>952</td>
</tr>
<tr>
<td>Lettuce</td>
<td>36</td>
<td>1042</td>
<td>2534</td>
</tr>
</tbody>
</table>

(miscellaneous IID water) remains approximately 1% of the water allocations, we focus on the portion of the water budget that can be allocated to IW, EW, and OW as follows:

\[
2.454 \text{ MAF} = IW + EW + O
\]

(Rerunning the baseline model accounting for the 69 TAF lower water constraint has minimal to no effect on model outputs except for the unused water (OW) variable, which is lower by a negligible 11 TAF.

### 4.2. Fallowing program

Since crops have different potential profits, as well as different water requirements, the opportunity costs of fallowing a particular crop depends on the foregone profits per acre-foot of water conserved. Table 3 shows the average profit generated from an acre-foot of water by crop and crop price. This profit represents the value the water holds to the farmer from growing different crops and highlights how a farmer’s willingness to participate in a water leasing program is a function of both crop type and crop price. Knowledge of the extent to which particular low- (e.g., Sudan grass, alfalfa) and high-revenue (e.g., broccoli, lettuce) crops are being cultivated provides a rough estimate of the amount of land that might be enrolled in such a program at different water leasing prices.

Fig. 4 illustrates the relationship between the lease price, \( p_F \), and the regional water balance components (\( SI \), \( EW \), \( DW \), and \( TW \)), total cost, and salinity of the Salton Sea inflows assuming mid-level crop prices. Note that direct transfers, \( EW \), is the total water leased from the farmers; as such, the relationship between the \( EW \) quantity and the lease price is akin to a supply curve of water under this program. The “jumps” in the curves indicate the price points at which it becomes profitable to fallow additional land and thus make water available for lease. Recall that land fallowing for a given crop is capped at 20 percent.

Without any additional program in place following the end of the QSA’s requirement that mitigation water be sent to the Sea, Sea inflows are around 850 TAF, comprised entirely of water from drainage and tail water runoff (in Fig. 4(a), this is illustrated by the intersection of the SI curve with the y-axis). As the lease price for water via land fallowing increases, Sea inflows increase along with the total cost, while drainage, tail water flows, and salinity of the inflows decrease. If the state wanted to achieve historic Sea inflows of one MAF, for example, the total cost of the program would be approximately $28 million, which increases overall grower profits by about a third of a percent (value of the water to the growers, represented by lost agricultural sales, is almost $27 million as shown in Fig. 5(a)). This inflow is achieved with the per acre-foot lease price set at $79/ac-ft, compensating growers for the foregone agricultural profits from fallowing their land, and results in approximately 360 TAF of environmental water and 730 TAF of drainage and tail water inflows. Salinity of the inflows drops from the baseline inflow salinity of 3.5 dS/m to 3.2 dS/m. Of course, since this is a terminal lake, overall salinity will increase over time, but at a slower pace under the following program and at a level that will require less desalination if
and when such strategies are implemented as suggested in the 10-year plan (CDWR, 2017). As illustrated in Figs. 4 and 5, the low-cost strategy to generate inflows for the Salton Sea is the fallowing program, unless inflows exceeding the fallowing program’s upper limit are desired.

In Fig. 5(a), we show the relationship between total profits to growers from engaging in this program (Total Profit), profits from farming (Farming Profit), the profit from leasing the water, which is the same as the cost of purchasing the water (Water Profit, EW Cost) versus

---

**Fig. 4.** Regional water budget components and Salton Sea inflow (SI) salinity as a function of the price paid for direct transfers estimated for the fallowing (a), the reduced evaporation irrigation (b), and direct leasing (c) programs.
price per acre-foot of leased water. At $79/ac-ft, the revenue received from leasing the water is slightly greater than the lost agricultural profits from fallowing their land (i.e., the lease price is greater than or equal to the foregone profits for all acres enrolled). This result is obtained by fallowing approximately 20 percent of the Sudan grass and alfalfa acreage, which constitutes about 10 percent of the total farm area.

While not shown, for different crop price levels the opportunity cost of fallowing changes and thus so will the lease price to generate any particular level of environmental flows. For instance, under the high crop price scenario, a lease price of $113/ac-ft would be required to generate inflows of approximately one MAF (consisting of environmental water, drainage, and tail water). As a point of comparison, growers that participate in the existing fallowing program currently receive $175/ac-ft, a premium above and beyond the opportunity cost to the farmer thereby indicating a positive shadow value on the acreage constraint. Additional results from high and low price runs are available from the authors, but do not show any significant qualitative differences.

4.3. Improved irrigation efficiency program

Fig. 4(b) shows results for the improved irrigation efficiency program assuming the lower cost estimate for improving irrigation efficiency ($400 per acre annually). Similarly to Fig. 4(a), jumps in the curves occur at prices where it becomes profitable to invest in improved irrigation technology for a particular crop. Crop prices are less important here in determining the jump points because the decision to switch to efficient irrigation depends on whether the additional irrigation costs can be recouped with water conservation payments, a calculation that depends on the irrigation system costs as well as the amount of potential evaporation reductions. Thus as the lease price increases, irrigation efficiency improvements will occur first on land cultivated with crops that have higher total seasonal evaporative water losses (Table A2). In our case, irrigation improvements occur first on land cultivated with alfalfa, which would require a water lease price of about $183/ac-ft to trigger enrollment.

As shown in Fig. 4(b), at $183/ac-ft, 370 TAF of environmental water is generated. In comparison, the fallowing program generated almost the same amount of environmental water (∼360 TAF) at a much lower price of $79/ac-ft. Note also that while environmental flows may be higher with the irrigation program, overall Sea inflows are lower, and inflow salinity higher, relative to the fallowing scenario. The total expenditures at $183/ac-ft, shown in both Fig. 4(b) and Fig. 5(b), amount to approximately $68 million in environmental water costs, roughly 2.5 times the expenditures required under the fallowing program, with negligible effects on grower profits. Drainage and tail water inflows to the Sea are estimated about around 560 TAF, 70 TAF less than under the fallowing program.

It is important to highlight that the efficiency program costs are exorbitant because “saved water” (EW) is generated through reduced evaporation alone via improving irrigation system technologies. Farm profits are reduced significantly compared to fallowing, which increases the value of applied water, and therefore increases the cost of the program. If one assumes the higher irrigation cost estimate (I = $800/ac), the costs of this program approximately double. The takeaway here is that high payments are needed to induce changes in irrigation of lucrative garden crops, and that relatively little evaporative water is available for conservation from those crops. However, when different irrigation methods are used, agricultural yields are minimally affected.

4.4. Direct leasing program

Fig. 4(c) shows results for the direct leasing program using mid-range crop prices. Relative to the previous figures, these curves are smoother, a result of the more flexible program which allows growers a larger array of strategies to conserve water for leasing, including deficit irrigation. Notable is how for any positive lease price growers are incentivized to conserve some amount of water which is in stark contrast to the other two strategies. In effect, internal margin adjustments such
as lower applied water rates are a cost-effective means to save water rather than relying entirely on a more extensive-margin strategy such as land fallowing or changes in irrigation technology (Schwabe et al., 2006; Adamson et al., 2017). In terms of external-margin adjustments, at the lower lease prices, Sudan grass is fallowed first followed by alfalfa. Because of its relatively low water requirement and high profit margin per unit of water, lettuce is the last crop to be fallowed, and at a lease price of $1400/ac-ft or higher (not shown).

Under the direct lease program, we estimate that a lease price of $60/ac-ft would generate approximately 300 TAF of environmental water and approximately 600 TAF of drainage and tail water at a cost of around $18 million, significantly less on a per acre unit of environmental water relative to the other two scenarios. Relative to the other two scenarios, we see a significant reduction in drainage relative to tail water, a result of less applied water arising from deficit irrigation. Additionally, inflow salinity decreases by more relative to the irrigation scenario, yet achieves only a slight salinity reduction relative to the fallowing option at $60/ac-ft, largely a result of the fact that at that price there is no fallowing. To generate a historic inflow of around one MAF, the lease price would need to be $89/ac-ft, a value higher than the land fallowing option since the model is not being optimized to minimize the costs of generating inflows but rather to maximize grower profits given a particular lease price and the constraints identified above.

To illustrate the relative efficiency of the three water leasing strategies, Fig. 5 presents total costs of supply under each of the three programs, as well as farm profits and value of leased water (opportunity cost to farmers). However, if a remediation plan called for fresh water flows, not mixed inflow (which consist of “unpriced” drainage and tail water flows), the direct leasing program is the lowest cost as it can provide more environmental water at a lower cost. Recall that there is not a linear relationship between purchased water and inflows. This is not surprising—and is consistent with previous program evaluations (e.g., Fig. 1 in Qureshi et al., 2010)—given that the flexibility associated with this strategy allows growers to choose their least cost approach to generate water for leasing either through fallowing, irrigation improvements, deficit irrigation, or crop switching. Also notable is the extent to which the direct leasing program can appropriate significantly more water than either the fallowing program or the efficiency improvement program. With direct leasing, individual crop yield is minimally affected, never dropping below 97 percent. Again, results from high and low price runs do not show any significant qualitative differences.

5. Discussion and conclusions

The purpose of this paper was to compare three widely discussed and often used strategies to lease water in terms of the opportunity costs to generate water savings and the potential externalities associated with such savings. Our focus was on investigating the opportunity costs to agricultural operators in Southern California to generate water for leasing under a fallowing program, an irrigation efficiency program, and a flexible, direct lease program with the water going to the Salton Sea, a terminal lake which is likely to shrink in the coming years under a no-action policy, with severe ecological and human health repercussions. Such a reduction in both volume and shoreline has been estimated to result in damages of approximately $11 to $70 billion over the next thirty years under a no action policy (Cohen, 2014). Our model estimates that the opportunity costs of water associated with the three programs analyzed vary widely.

While the direct lease program is the least-cost method for purchasing environmental water given the wide array of low cost strategies growers can choose from to conserve water, it results in the greatest reduction in drainage flows and tail water runoff. Since both these and purchased environmental water contribute to the Salton Sea, consideration of the potential externalities of programs that incorporate such flexibility, e.g., water markets, is warranted. Because of the high water application rates, water use reductions may have limited impact on overall production, a positive benefit in response to concerns over third party effects (e.g., employment, income) that are often raised when such water transfers are discussed. Indeed, our model shows that the direct water leasing program results in reduced water application rates and some land fallowing, with total crop production reduced by only a few percent even at the lowest crop prices evaluated.

The fallowing program, given the relatively smaller impacts on return flows than the other two strategies, produces the greatest total inflows at the lowest costs. This is an important finding given that fallowing programs often get criticized as being an inefficient means of generating beneficial use water because of their limited options, especially when compared to water markets. Yet, when one expands the scope of analysis to consider return flows and the potential (external) benefits associated with such flows, the system efficiency of a fallowing program likely compares much more favorably with water markets. Of course, if individual contributions to the inflows were both measurable and assignable, the pricing of, or market for, inflows would achieve the efficient solution.

In terms of third-party effects, farm production is not substantially reduced given fallowing occurs on the low value field crops and, recalling Eq. (12), we limit fallowing to at most 20% of the acreage. Even when over a million acre-feet of inflows are generated, only 7% of production profit is lost (See Fig. 5). The crops that are fallowed are field crops such as Sudan grass and alfalfa, and not higher valued garden crops like broccoli and carrots. Given that much of the cultivated alfalfa and Sudan grass is exported, and that the fallowed field crops represent much less than one percent of the US production (e.g., 131 million tons of hay were produced in 2017), impacts on local feed markets or global crop prices are likely to be minimal (NASS, 2018; Blake, 2017; Guerrero, 2008; Pierson, 2014). Lastly, improving irrigation efficiencies was the most costly approach to producing inflows to the Salton Sea given the high costs of irrigation system upgrades and the fact that this method only generates water through reduced evaporation. The irrigation program is distinct from the other two in that the cost is driven by irrigation system costs and not crop prices. On the plus side, irrigation efficiency improvements did not change output and thus there were no third party effects associated with this strategy.

As discussed in the introduction, there are various entities vying for water. We have focused on the Salton Sea, as it represents an important environmental water use that is currently experiencing substantial water related damages. However, the Colorado River Delta in Mexico has long been deprived of its historical Colorado River flows, resulting in substantial ecological changes. One may argue the water could have higher value there than in the Salton Sea. Additionally, there are several other agricultural and urban regions who may be interested in purchasing water were it to be made available on an open market, particularly Mexican and Arizonan agricultural interests, and municipal interests in Southern California (e.g., San Diego and Los Angeles). These players would ultimately have to be considered in any potential water transfers.

Nonetheless, these hydro-economic analyses indicate that leasing programs are capable of generating significant environmental water flows for the Sea with relatively small decreases in agricultural production and limited to no decrease in overall grower profits. The estimated damages of doing nothing at the Salton Sea are in the hundreds of millions to billions of dollars per year (Cohen, 2014). The cost of the comprehensive restoration plan is in the low hundreds of millions per year. Our results indicate that a great deal of water could be purchased for the Sea at a fraction of the cost, potentially mitigating damages and helping save a great deal money.

In terms of sustainability, the programs described here can serve as an integral element to both a short- and long-term fix. In the short-term, they can provide the necessary water nearly immediately to (1) keep the system from crashing by reducing the rate of salinization and sustaining...
current bird and fish populations, and (ii) reduce the amount of exposed dust-emitting playa and the consequent air-quality related health impacts over the next 5 to 10 years as other short-term mitigation efforts are implemented. In the long-term, they can serve as a low-cost means to generate inflows to the Sea and free up scarce public dollars for other identified long-term mitigative efforts. Unlike the suggestions in the restoration plans, the water market solutions presented here already have the needed infrastructure in place and could theoretically be workable in short order.

Acknowledgements

The data used are listed in the references, tables, and appendices.

Appendix A1. Farm-level crop yield and drainage

We use a crop water production model derived in Skaggs et al. (2014) to estimate crop yield. Crop yield, \( Y \), as a function of the depth of applied water, \( w \), is calculated as:

\[
Y(w) = Y_p(w)Y_r
\]

where \( Y_p \) is potential crop yield and \( Y_r(w) \) is relative crop yield. The latter we calculated by modifying the functions in Skaggs et al. (2014) to include evaporation:

\[
Y_r = \left( \frac{N}{T_p} \right) \left( 1 - \frac{EC_0}{EC_D} \right)
\]

\( N = w(1 - t_f) - E \)

\( EC_0 = EC_{IW} \frac{N + E}{N} \)

where \( N \) is net infiltrated irrigation water, \( t_f \) is tail fraction, \( T_p \) is potential transpiration, \( E \) is actual soil evaporation, \( EC_{IW} \) is irrigation water salinity, \( EC_0 \) is effective irrigation water salinity, and \( EC_D \) is drainage water salinity. The ratio of the latter two terms is given by Skaggs et al. (2014)

\[
\frac{EC_D}{EC_0} = \left( \frac{R + \sqrt{\Delta}}{2\sqrt{-Q \cos(\varnothing/3)}} \right) \Delta \geq 0
\]

\[
\frac{EC_D}{EC_0} = \left( \frac{R - \sqrt{\Delta}}{2\sqrt{-Q \cos(\varnothing/3)}} \right) \Delta < 0
\]

where

\[
\Delta = Q^2 + R^2
\]

\[
Q = [2(1 - T_p/N)R - 1]/3
\]

\[
R = (EC_{IW}/EC_0)^3
\]

\[
\varnothing = \cos^{-1}\left( \frac{R}{\sqrt{-Q}} \right)
\]

where \( EC_{50} \) is the soil water salinity at which yield is reduced by 50 percent. Crop-specific values for the various model parameters are provided in Table A2.

Eq. (A1) is based on an assumption that crop yield is proportional to the actual transpiration rate, \( T \), and that relative yield and transpiration are equal,

\[
T/T_p = Y/Y_p
\]

From mass balance considerations, the drainage depth \( D \) is given by

Table A1

<table>
<thead>
<tr>
<th>Years</th>
<th>Sea Area (10^3 ac)</th>
<th>Sea Salinity ppt</th>
<th>Habitat Complex (ac)</th>
<th>Air Quality Manage. Area (ac)</th>
<th>Capital Costs (Million $)</th>
<th>Oper. &amp; Main. (Million $/yr)</th>
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<tbody>
<tr>
<td><strong>Preferred Alternative [2007]</strong></td>
<td></td>
<td></td>
<td></td>
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<tr>
<td>1 to 12</td>
<td>Construction</td>
<td>Construction</td>
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<td>–</td>
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<td>12 to 22</td>
<td>45</td>
<td>–35</td>
<td>71</td>
<td>23</td>
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<td>22 to 32</td>
<td>45</td>
<td>–35</td>
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<td>75</td>
<td>1,615</td>
<td>86</td>
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<tr>
<td>32 to 70</td>
<td>45</td>
<td>–35</td>
<td>108</td>
<td>75</td>
<td>1,401</td>
<td>173</td>
</tr>
<tr>
<td><strong>Management Plan [2017]</strong></td>
<td></td>
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<td></td>
</tr>
<tr>
<td>1 to 10</td>
<td>–220^1</td>
<td>&gt; &gt; 100^3</td>
<td>30^5</td>
<td>383</td>
<td>Unknown</td>
<td></td>
</tr>
</tbody>
</table>

1. Includes non-sea habitat, vegetated playa, and stabilized playa.
2. The 2017 plan does not include costs for monitoring and testing.
3. This is the current size of the Sea, as this plan does not reduce the size.
4. Current salinity is about 60 ppt. Projections for 2030 with no action are over 150 ppt. This plan does not address Sea salinity.
5. Plan refers to combined acreage as “Habitat and Dust Mitigation Area”.
\begin{equation}
D = N - T
\end{equation}

The leaching fraction is defined as

\begin{equation}
LF = \frac{D}{w(1 - f_t)}
\end{equation}

Appendix A2. Evapotranspiration

To estimate crop potential transpiration and evaporation, we used the Food and Agriculture Organization (FAO)'s procedure for splitting the crop coefficient, \( K_c \), into its components, \( K_{o_b} \), the transpiration coefficient, and \( K_r \), the soil evaporation coefficient, using crop growth periods for the California desert, and initial coefficients (Allen et al., 1998). As a crop grows throughout the season, it goes through different stages: the initial stage, the development stage, the middle stage, and the end stage, which have varying \( K_{o_b} \) values for the initial, middle, and end stages are published on FAO's website (http://www.fao.org/docrep/X0490E/x0490e0c.htm, along with the lengths of the stages for different crops in different areas. The initial stage \( K_{o_b} \) does not need to be modified, but the mid and end stages do, since the climate varies:

\begin{equation}
K_{o_b} = K_{o_b\text{(tab)}} + \left[ 0.04(u - 2) - 0.004(RH_{\text{min}} - 45) \right] \left( \frac{h}{3} \right)^{0.3}
\end{equation}

\[ K_r = \min[K_r, K_{r\text{max}} - K_{o_b}, E_{o_b}, K_{r\text{min}}] \]

where \( K_{o_b} \) is the calculated value for a given day, \( K_{o_b\text{(tab)}} \) is the estimated FAO value, \( u \) is wind speed, \( h \) is plant height, and \( RH_{\text{min}} \) is the minimum daily relative humidity. When the topsoil has just been wetted, \( K_r \) is at its maximum, which cannot exceed the maximum value, \( K_{r\text{max}} \). When the topsoil is dry, \( K_r \) is very small, approaching zero. \( K_r \) is a dimensionless evaporation coefficient dependent on the cumulative depth of water depleted from the topsoil, and \( E_{o_b} \) is the fraction of the soil that is wetted and exposed and is dependent on the irrigation method and the crops. For flood irrigation, we assume this is one hundred percent. After wetting, \( K_r \) is 1. As the soil dries, \( K_r \) decreases and becomes zero when all the water is gone. \( K_r \) estimation requires a daily water balance computation. Crop transpiration can then be calculated by multiplying the respective \( K_{o_b} \) values by the reference evapotranspiration for each day in the plant's growing season. Likewise, evaporation can be calculated by multiplying the respective \( K_r \) values by the reference evapotranspiration for each day in the plant's growing season. The summation of both over the growing season results in the estimated potential transpiration and evaporation. For crops with multiple cutting periods, transpiration and evaporation are estimated for each cutting period, then summed together. Alfalfa, Sudan grass, and Bermuda were assumed to have ten cuttings per year (Geisseler and Horwath, 2016; Miller, 2015).

Table A2

Crop Parameters¹ and Bounds.

<table>
<thead>
<tr>
<th>Field</th>
<th>Parameters</th>
<th>Alfalfa</th>
<th>Bermuda</th>
<th>Sudan</th>
<th>Sugar Beet</th>
<th>Broccoli</th>
<th>Carrots</th>
<th>Onion</th>
<th>Lettuce</th>
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</thead>
<tbody>
<tr>
<td></td>
<td>( p ) Mid</td>
<td>$/ac-ft</td>
<td>0.14</td>
<td>0.14</td>
<td>0.15</td>
<td>0.05</td>
<td>0.78</td>
<td>0.10</td>
<td>0.14</td>
</tr>
<tr>
<td></td>
<td>Low</td>
<td>$/ac-ft</td>
<td>0.25</td>
<td>0.25</td>
<td>0.22</td>
<td>0.08</td>
<td>0.98</td>
<td>0.13</td>
<td>0.17</td>
</tr>
<tr>
<td></td>
<td>High</td>
<td>$/ac-ft</td>
<td>0.21</td>
<td>0.21</td>
<td>0.19</td>
<td>0.06</td>
<td>0.94</td>
<td>0.12</td>
<td>0.15</td>
</tr>
<tr>
<td></td>
<td>( q )</td>
<td>$/ac</td>
<td>468</td>
<td>411</td>
<td>271</td>
<td>921</td>
<td>1604</td>
<td>1693</td>
<td>1644</td>
</tr>
<tr>
<td></td>
<td>( H )</td>
<td>$/kg</td>
<td>0.016</td>
<td>0.002</td>
<td>0.026</td>
<td>0.007</td>
<td>0.500</td>
<td>0.050</td>
<td>0.070</td>
</tr>
<tr>
<td></td>
<td>( Y_p ),( Y_p ) = ( 10^3 ) kg/ac</td>
<td>7</td>
<td>7</td>
<td>6</td>
<td>40</td>
<td>7.5</td>
<td>36</td>
<td>25</td>
<td>17</td>
</tr>
<tr>
<td></td>
<td>( T_p )</td>
<td>ft</td>
<td>4.53</td>
<td>2.73</td>
<td>4.48</td>
<td>2.74</td>
<td>0.76</td>
<td>0.69</td>
<td>1.09</td>
</tr>
<tr>
<td></td>
<td>( E )</td>
<td>ft</td>
<td>1.04</td>
<td>0.59</td>
<td>0.62</td>
<td>0.59</td>
<td>0.34</td>
<td>0.24</td>
<td>0.22</td>
</tr>
<tr>
<td></td>
<td>( E_{\text{Ca}} )</td>
<td>ds/m</td>
<td>17.7</td>
<td>29.4</td>
<td>28.9</td>
<td>30.9</td>
<td>17.7</td>
<td>9.1</td>
<td>8.7</td>
</tr>
<tr>
<td>Bounds</td>
<td>( x^{\text{min}} )</td>
<td>( 10^3 ) ac</td>
<td>170</td>
<td>60</td>
<td>45</td>
<td>30</td>
<td>15</td>
<td>20</td>
<td>15</td>
</tr>
<tr>
<td></td>
<td>( x^{\text{max}} )</td>
<td>( 10^3 ) ac</td>
<td>190</td>
<td>75</td>
<td>70</td>
<td>35</td>
<td>25</td>
<td>35</td>
<td>25</td>
</tr>
</tbody>
</table>

¹ \( p \) is crop price; \( q \) is production costs; \( H \) is yield-related harvest costs; \( h \) is non-yield related harvest cost; \( Y_p \) is potential crop yield; \( T_p \) is the potential crop transpiration rate; \( E \) is direct evaporation; \( E_{\text{Ca}} \) is a salinity tolerance parameter; \( x \) is crop acreage.

² Adapted from Imperial County Agricultural Commissioner Sealer of Weights and Measures Annual Crop Reports (Available at http://www.co.imperial.ca.us/ag/?page=iccr).


⁴ See Appendix A2.

⁵ Estimated as 100/B + 2A, where A and B are the Maas and Hoffman threshold and slope salt tolerance parameters.

References


