### Title
Institutions and the Economic Efficiency of Managed Aquifer Recharge as a Mitigation Strategy Against Drought Impacts on Irrigated Agriculture in California




### Journal
Water Resources Research, 58(6)

### ISSN
0043-1397



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### Publication Date
2022-06-01

### DOI
10.1029/2021wr031261

Peer reviewed

Water Resources Research

RESEARCH ARTICLE
10.1029/2021WR031261

**Key Points:**

- Managed aquifer recharge (MAR) is economically efficient for the region, and its value depends on institutional design and aquifer hydrogeological properties
- Institutional arrangements matter, affecting optimal water management strategies and their economic impact across the region
- Cooperation among stakeholders is fundamental for the economic efficiency of MAR
- Property rights to groundwater introduced through recent legislation in California incentivizes MAR related arrangements

**Supporting Information:**

Supporting Information may be found in the online version of this article.

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**Citation:**

Reznik, A., Dinar, A., Bresney, S., Forni, L., Joyce, B., Wallander, S., et al. (2022). Institutions and the economic efficiency of managed aquifer recharge as a mitigation strategy against drought impacts on irrigated agriculture in California. *Water Resources Research*, *58*, e2021WR031261. https://doi.org/10.1029/2021WR031261

Received 21 SEP 2021
Accepted 16 MAY 2022

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# Institutions and the Economic Efficiency of Managed Aquifer Recharge as a Mitigation Strategy Against Drought Impacts on Irrigated Agriculture in California

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**Abstract** Managed aquifer recharge (or intentional recharge) is a purposeful human intervention designed to supplement natural enrichment processes of groundwater aquifers by various methods. It holds the potential to mitigate the impact of climate uncertainty on irrigated agriculture by restoring storage levels in depleted aquifers, the economic value of which increases during droughts. We use a high-resolution dynamic regional hydroeconomic framework that endogenizes farming decisions in response to water quantity-quality changes, as well as complex hydrogeological principles to analyze several institutional designs and climate scenarios applied to the Kings Groundwater Basin in California. Our analysis demonstrates that intentional recharge is of high benefit to the region, potentially increasing average groundwater levels in the region by 20% over a 20 year horizon. Additionally, we show how this practice could become the subject of second-best arrangements among water users in the region in view of property rights to groundwater derived from recent legislation in California, thus increasing its materialization potential. However, we also find that the quantity recharged is sensitive to climate conditions and hydrological properties.

## 1. Introduction

Managed aquifer recharge (MAR) is a set of practices and methods used for the intentional recharge of water of various types and qualities (surface water, recycled wastewater, and even groundwater from different locations) into a given aquifer. While replenishing depleted groundwater stocks is the main objective of MAR, the numerous direct and indirect benefits associated with it have been identified in the literature (Maliva, 2014; Perrone & Rohde, 2016; Vanderzalm et al., 2015, list several examples). Dillon et al. (2019) report that since the 1960s, global implementation of MAR has accelerated at a rate of 5% per year, but is not keeping pace with the increase in groundwater extraction. Specifically, authors estimate the annual volume of recharged quantities at less than 2.5% of groundwater extractions in countries practicing MAR. The occurrence of droughts intensifies with climate change and concurrently increases groundwater reliance of irrigated agriculture while exacerbating groundwater depletion, elevating the importance of MAR as a mitigation strategy (Scanlon et al., 2012). The objective of this paper is to examine the feasibility and economic efficiency of MAR, specifically as a strategy to mitigate drought effects in irrigated agriculture. A second objective is to understand the role of policies and institutional designs in determining the efficiency of this strategy. For that purpose, we develop a hydroeconomic regional dynamic optimization model to analyze several institutional design and climate scenarios, focusing on the Kings Groundwater Basin in California's Central Valley.

California is characterized by growing urban populations in proximity to productive agricultural regions, widespread underground aquifer systems, many stochastic flash floods and prolonged drought periods. Groundwater is an important resource, supplying nearly 40% of water consumed in the state in an average year, with some municipal and agricultural communities relying on that resource exclusively for water supply (Hanak et al., 2021). The reliance on groundwater increased significantly during the recent consecutive droughts, resulting in severe groundwater depletion, including in the Central Valley aquifer system, threatening the sustainability of groundwater resources in these severely over-drafted basins. These conditions called for the introduction of a new regulation on groundwater use—the Sustainable Groundwater Management Act (SGMA), with the objective to recover sustainable groundwater levels in the next 20 years and avoiding undesirable results (e.g., chronic lowering of groundwater levels, groundwater storage loss, deteriorating groundwater quality, land subsidence, sea

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water intrusion, and surface water depletion). Concerns have been raised that groundwater management practices derived from SGMA could impose significant losses in terms of agricultural revenues. Concurrently, researchers say that MAR could play an important role in mitigating some of these tradeoffs (Hanak et al., 2019). According to Scanlon et al. (2012), the spatiotemporal variability in natural conditions and the resulting heterogenous groundwater depletion in the Central Valley basins are contributing factors to the conclusion that MAR could be a promising strategy to mitigate the impacts of future droughts on the Central Valley's water balance.

As an approximation of MAR potential, Perrone and Rohde (2016) report on 106 planned projects of different water agencies across California that were approved for state support through designated funding schemes (Propositions 13, 50, 84, 1E) in the last two decades. The authors qualify that water availability could be a limiting factor in the materialization of this potential. Hanak et al. (2018) point also to regulation and institutional arrangements as important factors that would determine the implementation rate of these projects. This last observation is also prevalent in earlier studies covering California's experience with MAR (Thomas, 2001). Related to this discussion, a growing body of research focuses on the role of MAR within California's evolving water economy. For example, Scanlon et al. (2016) argue for complementarity of surface water and groundwater storage practices in order to achieve sustainable management of water resources. Authors exploit spatial regional differences in the connectivity of groundwater and surface water sources to demonstrate the effectiveness of MAR in preventing historical groundwater depletion. Alam et al. (2020) suggest that water transfers from north to south of the Central Valley are imperative for the success of MAR projects in restoring groundwater levels of severely depleted basins, highlighting the limited potential of the MAR strategy applied alone in achieving groundwater sustainability. Dahlke et al. (2018) provide a review of MAR strategies implemented in California. Authors point to regulatory constraints and MAR water source quality as potential barriers for future expansions of MAR practices in California, and offer excess irrigation of agricultural fields using flood flows as a promising remedy, however, requiring regulatory adaptation. Ulibarri et al. (2021) also identify similar barriers jeopardizing the full implementation of planned MAR projects by water agencies in the Central Valley aimed at meeting SGMA objectives. In their concluding remarks, the authors argue that a portfolio approach combining MAR with demand management tools (e.g., groundwater extraction limitations) could be a promising strategy, meeting SGMA objectives. For these considerations, California is an excellent experimental field for the analysis herein, which endogenizes regional heterogeneity in spatiotemporal natural conditions and agricultural productivity, and explores the role of different groundwater management policies and changes in institutional design on MAR efficiency.

The analysis performed in this paper is concerned with the conjunctive management of groundwater, surface water, and wastewater over time. In that respect, it is directly connected to the knowledge accumulated in the subfield of economics studying the issues of groundwater resources management (Koundouri, 2004). Studies exploring conjunctive use, starting from the seminal work of Burt (1964), have mostly focused on the fact that groundwater and surface water are substitutes for consumption purposes, ignoring the nuances associated with the hydrologic connectivity between them. Within that context, intentional recharge (or MAR) as a potential optimal strategy is mostly absent in such frameworks. One exception to that statement is the work of Knapp and Olson (1995), who compared common pool behavior to socially optimal management of a groundwater stock conjunctively with stochastic surface flows, allowing for artificial (or intentional) recharge (i.e., MAR). In their empirical analysis of Kern County in California, the authors find intentional recharge unwarranted. However, they point to several factors contributing to that result, such as the level of natural recharge, variability of surface flows, and the costs of energy in the studied region, all empirical in nature.

As a direct extension, hydroeconomic models have become the main tool used in empirical investigations of the general findings suggested in the literature of groundwater management models (Booker et al., 2012; Harou et al., 2009). The CALVIN model (California Value Integrated Network; Draper et al., 2003) is probably the one most associated with such investigations in California. For example, Harou and Lund (2008) used this model to evaluate the performance of different strategies to end groundwater overdraft in the Tulare Basin in the Central Valley. Another application of that framework (Dogan et al., 2019) explores the potential water supply effects of ending long-term groundwater overdraft in California's Central Valley for several general water policies with historical, and warmer/drier climates. Several recent hydroeconomic model applications (Tran et al., 2019; Tran, Kovacs, & Wallander, 2020; Tran, Kovacs & West, 2020) investigated the role of MAR and the different factors determining its effectiveness, focusing on Eastern Arkansas and the overdrafted Mississippi Valley Alluvial Aquifer. We complement these contributions by explicitly accounting for water quality differentiation of water sources and its impact on crop yield—established within the economic literature on irrigation water salinity (Connor et al., 2012; Feinerman & Yaron, 1983; Knapp, 1992; Letey & Dinar, 1986; Schwabe et al., 2006, to name a few examples). A nationwide hydroeconomic framework of competing demand sectors, incorporating salinity response functions in agriculture was recently developed by Slater et al. (2020). We supplement that work through explicit representation of hydrogeological principles, within a framework termed by MacEwan et al. (2017) as embedded hydrologic response function model integration, also accounting for MAR costs and benefits, as will be detailed below.

Considering the role attributed to future regulation and institutional design in the potential success of recharge projects in California (Hanak et al., 2018; Thomas, 2001), we examine in this paper the impact of institutional arrangements on MAR efficiency. According to North (1990) and as further developed by Saleth and Dinar (2004), water institutions include Water Law, Water Policy, and Water Administration components. The incorporation of institutional changes within water modeling frameworks such as the approach utilized herein has been reviewed by Booker et al. (2012), who stated that:

> "Water allocation institutions have emerged over a long and contentious history as a complex set of local rules, regulations, and rights. These can be modeled by sets of constraints and allocation priorities in the spatially explicit models that we discussed. One of the major policy advances in water resource allocation in recent years is the gradual replacement of fixed allocation rules by market-based institutions. The testing of the economic impact of such institutional changes has been a natural extension for hydroeconomic models with their detailed specifications and physical constraints on the ability to move water between different locations. The simplest and easiest way to specify hydroeconomic models is in the perfect market equilibrium situation without additional property rights constraints. The ability to represent alternative levels of market innovation in water resource allocation comes naturally to such models."

Thus, relying on common definitions and practices, as part of our analysis we aim to investigate the role of institutional changes on the economic efficiency of MAR. This would be, to the best of our knowledge, a unique contribution.

## 2. Materials and Methods

In order to assess the applicability of MAR within a regional setting and its sensitivity to policies and institutions under different climate conditions, we construct an economic optimization model (EOM) that accounts for agronomic, hydrologic, and hydrogeologic principles. Specifically, the model incorporates response functions to capture crop-yield sensitivity to water application levels and their quality, allowing agricultural adaptation through redistribution of land among a given set of crops. In terms of groundwater dynamics, the model tracks deep percolation resulting from precipitation, agricultural irrigation, and infiltration basins supplied by different water sources (e.g., surface water diversions, treated wastewater, or groundwater), as well as groundwater extractions, and lateral flows between adjacent subbasins. Surface water and groundwater supply, wastewater treatment and reuse, water conveyance and intentional recharge infrastructures' capacities, costs, and limitations are all included in the model.

The region studied in our analysis, as many others worldwide, is part of a larger and interdependent water and agricultural systems, hence subject to exogenous hydrological and regulatory constraints. In order to guarantee that the regional optimization framework decisions adhere to these larger-scale exogenous constraints, as well as to capture some of the hydrological and regulatory complexities, on which we reported earlier, affecting MAR implementation potential, we linked the EOM to perform iterations with a hydrologic and water management model (termed CVPAM), building on the foundations of the work done by Forni et al. (2016). The hydrologic model is developed using the Water Evaluation and Planning software and simulates water resources systems, including rainfall-runoff hydrology, water resources infrastructure, agricultural, urban, and environmental demanded quantities, and applies complex operating rules and constraints to the water allocation problem (for further detail on the CVPAM model, see Supporting Information S1).

### 2.1. The Economic Optimization Framework

Instead of developing the modeling capacity required for the regional analysis from scratch, we adopt the integrated model presented in Slater et al. (2020) and adapt it to our needs. Our analytical framework differs from the one shown in Slater et al. (2020) by explicitly representing several hydrogeological principles relevant for the analysis in this paper. These adaptations include: (a) the inclusion of intentional recharge, using designated infrastructure (e.g., infiltration basins); (b) deep percolation (that originates from intentional recharge, irrigation or treated wastewater discharge), which is included in our groundwater stock equation of motion; and (c) accounting for lateral flows between subbasins.

The dynamic optimization problem for a given agricultural region composed of several decision-makers $u \in (1, \ldots, U)$, controlling an area $L_u$, which is subdivided to the subdistrict level $d \in (1, \ldots, D)$, is depicted in Equation 1.

$$
\begin{gathered}
\max_{x_{udjt}, w_{udjt}, q^{\varphi}_{udt}, R^{M}_{udt}} \sum_{t} \rho_t \cdot \sum_{u,d,j} \pi\left(x_{udjt}, w_{udjt}, \boldsymbol{q}^{\varphi}_{udt}, R^{M}_{udt}\right) \\
\text{s.t.} \\
x_{udjt}, w_{udjt}, \boldsymbol{q}^{\varphi}_{udt}, R^{M}_{udt} \in \Omega : \Omega \equiv \left\{\mathbf{H}\left(x_{udjt}, w_{udjt}, \boldsymbol{q}^{\varphi}_{udt}, R^{M}_{udt}, \boldsymbol{z}\right)\right\} \\
G_{udt}(t=0) = G_{ud0}
\end{gathered}
\tag{1}
$$

Problem (Equation 1) is concerned with maximizing the present-value net gains from agricultural production, given a set of constraints $\boldsymbol{\Omega}$. This is achieved by finding at each time step $t$ of the planning horizon $t \in (1, \ldots, T)$, optimal decisions at the subdistrict level with respect to land allocation among $j \in (1, \ldots, J)$ crops ($x_{udjt}$), water application level per unit of land for each crop ($w_{udjt}$), the use of water of different types ($\boldsymbol{q}^{\varphi}_{udt}$), and intentional recharge ($R^{M}_{udt}$) through designated infrastructure. Water types $\varphi$, include groundwater ($g$), surface water ($s$), and treated wastewater ($r$). Where $\rho$ is the discount factor, and $\pi(\cdot)$ represents net benefits from crop production. The set $\boldsymbol{\Omega}$ of constraints $\mathbf{H}$, guarantees that the optimal levels and paths of decisions and states of the system comply with all hydrological, engineering, and feasibility conditions in the region. The vector $\boldsymbol{z}$ represents the different parameters of the system (e.g., pumping capacities, rainfall, available cultivable land, and others, as is described below), and $G_{ud0}$ is the boundary condition for groundwater head at the subdistrict level at the onset of the planning horizon.

We ignore benefits from water consumption in the urban sector. We, therefore, assume domestic water demand to be perfectly inelastic, and denote the quantity consumed in that sector as $Q_{udt}$, which we assume increases according to population and income growth trends. We define net gains from agricultural production $\pi(\cdot)$ in Equation 2:

$$
\pi\left(x_{udjt}, w_{udjt}, \boldsymbol{q}^{\varphi}_{udt}, R^{M}_{udt}\right) = x_{udjt} \cdot \left[p^{y}_{jt} \cdot y\left(w_{udjt}, \boldsymbol{q}^{\varphi}_{udt}\right) - \gamma_j - \delta_{1j} - \delta_{2j} \cdot x_{udjt}\right] - C\left(\boldsymbol{q}^{\varphi}_{udt}, R^{M}_{udt}\right) \tag{2}
$$

In Equation 2, revenues from crop sales are defined as the periodic market price of each crop $p^{y}_{jt}$ multiplied by the per-acre yield, which is the function $y\left(w_{udjt},\ \boldsymbol{q}^{\varphi}_{udt}\right)$. These are then deducted by the crop-specific variable costs of production (excluding water costs), $\gamma_j$, and the economic costs quadratic function $x_{udjt} \cdot (\delta_{1j} + \delta_{2j} \cdot x_{udjt})$, representing optimality considerations of farmers, which is manifested by observed land allocation to crops (Howitt, 1995). Finally, we account for the costs of water supply and intentional recharge in each subdistrict, as represented by the function $C\left(\boldsymbol{q}^{\varphi}_{udt}, R^{M}_{udt}\right)$ in Equation 2.

We follow Kan and Rapaport-Rom (2012) and Kan et al. (2002), and define per-acre yield as a linear function of evapotranspiration, which in turn is a nonlinear function of applied water quantity, $w_{udjt}$, precipitation, $\tilde{w}$, and salinity level, $\psi\left(\boldsymbol{q}^{\varphi}_{udt}\right)$, as depicted in Equation 3:

$$
y\left(w_{udjt}, \boldsymbol{q}^{\varphi}_{udt}\right) = \theta_{1udj} + \theta_{2udj} \cdot \frac{\overline{e}_{udj}}{1 + \alpha_{1udj}\left[\alpha_{2udj} \cdot \psi\left(\boldsymbol{q}^{\varphi}_{udt}\right) + \alpha_{3udj} \cdot \left(w_{udjt} + \tilde{w}\right)^{\alpha_{4udj}}\right]^{\alpha_{5udj}}} \tag{3}
$$

Salinity level itself is a function of all blended water sources at the subdistrict level. In Equation 3, $\overline{e}_{udj}$ is the potential evapotranspiration level, $\theta_{1udj}$ and $\theta_{2udj}$, and $\alpha_{1udj}$ through $\alpha_{5udj}$ are crop- and subdistrict-specific parameters. The costs of water supply, wastewater treatment, and intentional recharge $C\left(\boldsymbol{q}^{\varphi}_{udt}, R^{M}_{udt}\right)$ will be explicitly formulated and described in detail in the calibration section that will follow.

Groundwater dynamics is included in the set $\boldsymbol{\Omega}$ of constraints $\mathbf{H}$. We define groundwater changes at the temporal and spatial dimensions according to Equation 4:

$$G_{udt} - G_{ud(t-1)} = \frac{R^{M}_{ud(t-\tau)} + \sum_j s_j \cdot x_{udj(t-\tau)} \cdot (w_{ud(t-\tau)} + \tilde{w} - et(\cdot)) + \left(\kappa \cdot Q_{ud(t-\tau)} - q^{r}_{ud(t-\tau)}\right) + \bar{l}_{udt} - q^{g}_{udt}}{\nu \cdot L_{ud}} \quad (4)$$

where $G_{udt}$ is groundwater head in each subdistrict at each time step. Vertical and horizontal travel time to groundwater table are explicitly represented by the time lag $\tau$. Therefore, change in groundwater level between time periods in the model is increasing with intentional recharge from infiltration basins ($R^{M}_{ud(t-\tau)}$), which is the diversion of water away from production for the sole purpose of recharging groundwater and is capped in the model by the existing recharge capacity $\overline{R}^{M}_{ud}$. Deep percolations resulting from agricultural irrigation are also increasing groundwater level ($\sum_j s_j \cdot x_{udj(t-\tau)} \cdot (w_{ud(t-\tau)} + \tilde{w} - et(\cdot))$, and $et(\cdot)$, defined in Equation 3, is the evapotranspiration function), where $s_j$ indicates the share of land in each subdistrict suitable for groundwater recharge on agricultural land, acknowledging spatial differences in hydrogeological soil characteristics. The third form of recharge is deep percolations of treated wastewater that are not reused in agricultural irrigation, and due to lack of other safe disposal alternatives are left to percolate to the ground ($\kappa \cdot Q_{ud(t-\tau)} - q^{r}_{ud(t-\tau)}$, where $\kappa$ is a fixed share of sewage out of the quantity consumed in the domestic sector). Groundwater head decreases with pumping ($q^{g}_{udt}$). The net of all flows in and out of the basin are divided by the term of subdistrict land area ($L_{ud}$) multiplied by the basin specific yield ($\nu$).

Lateral flows are also affecting groundwater head in the model. The net sum of lateral flows to, and out of the subdistrict are denoted in Equation 4 as $\bar{l}_{udt}$, where lateral flows are defined as in Equation 5:

$$l_{udd^{-1}t} = f_{dd^{-1}} \cdot b_{dd^{-1}} \cdot \left(G_{ud^{-1}(t-\tau)} - G_{ud(t-\tau)}\right) \quad (5)$$

Thus, lateral flows at each period are determined based on groundwater head difference between adjacent subbasins (subdistricts) prevalent on the time lag $\tau$, multiplied by the border length between subdistricts ($b_{dd^{-1}}$) and a factor $f_{dd^{-1}}$, that its calibration is achieved through the iteration process with the CVPAM model (for further detail on this iteration process see Supporting Information S1). Thus, Equation 4 becomes an embedded hydrologic response function as defined in MacEwan et al. (2017).

Listed in Equation 4, the different sources for groundwater recharge in our model (i.e., recharge through designated infrastructure, percolation of treated wastewater that are not used for other consumptive purposes, and deep percolation from excess irrigation of crops) all fall in the broad definition of MAR. Intentional recharge through designated infrastructure (e.g., infiltration basins) differs from the other two because it requires the diversion of water away from production. It, therefore, bears an opportunity cost, which its value is obviously negatively correlated with water availability. The other two forms of recharge, assuming that reliance on groundwater for consumption purposes is significant, are analogous to making a withdrawal from a checking account and depositing in a savings account, accruing instantaneous benefits in the interim. Knapp and Olson (1995) already demonstrated that higher cost of groundwater pumping has an ambiguous effect on the value of MAR for a single basin under homogeneous conditions. Our approach, which captures spatial heterogeneity in hydrological conditions and internalizes groundwater flow externalities accounts for an additional contributing factor for MAR. That is, for every subdistrict $d$, groundwater reliance and higher cost of pumping in a neighboring subdistrict would increase the value of MAR regardless of the cost of groundwater pumping for subdistrict $d$. Furthermore, under our approach, deficit irrigation of certain crops and MAR in a given subdistrict would be simultaneously warranted if the value of production in a neighboring subdistrict, which relies on groundwater, surpasses the cost of groundwater extraction, the cost of conveyance to MAR basins and the forgone benefits of crop production in the subdistrict in which recharge occurs.

Equation 6 requires that groundwater head will not fall below some minimal threshold $\underline{G}_{ud}$.

$$G_{udt} \geq \underline{G}_{ud} \quad (6)$$

Equation 7 specifies the limitation on surface water deliveries to each subdistrict:

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$$q^s_{udt} \leq S_t - \sum_{u^{-1}} \sum_{d^{-1}} q^s_{u^{-1}d^{-1}t} \tag{7}$$

Such that, quantity delivered to a specific subdistrict cannot exceed the periodic availability of each source $S_t$ net of upstream diversions to all other connected subdistricts. Equations 8 and 9 are common input use constraints in the agricultural production process. Equation 8 caps land use by the total land area in each subdistrict, whereas Equation 9 limits the use of water in each subdistrict according to water quantities delivered to that subdistrict from all sources:

$$\sum_j x_{udjt} \leq L_{ud} \tag{8}$$

$$\sum_j x_{udjt} \cdot w_{udjt} \leq q^g_{udt} + q^s_{udt} + q^r_{udt} - R^M_{udt} - Q_{udt} \tag{9}$$

Further specifications of constraints and their parametrization are described in the data collection and calibration section that follows.

## 3. The Kings Groundwater Basin, Data Collection, and Calibration Procedures

The Kings Groundwater Basin is located in the southern part of the San Joaquin Valley groundwater basin in the Central Valley of California. The San Joaquin River signals the northern border of the basin and the alluvium-granitic rock interface of the Sierra Nevada foothills constitutes its eastern border. The west and south boundaries of the basin are formed by the borders of several agricultural irrigation districts as well as the south fork of the Kings River (see Figure 1, and Figures S2–S4 in Supporting Information S1). Agricultural production, which accounts for about 75% of the basin's 1-million-acre area, comprises mainly vineyards, nut and deciduous trees,

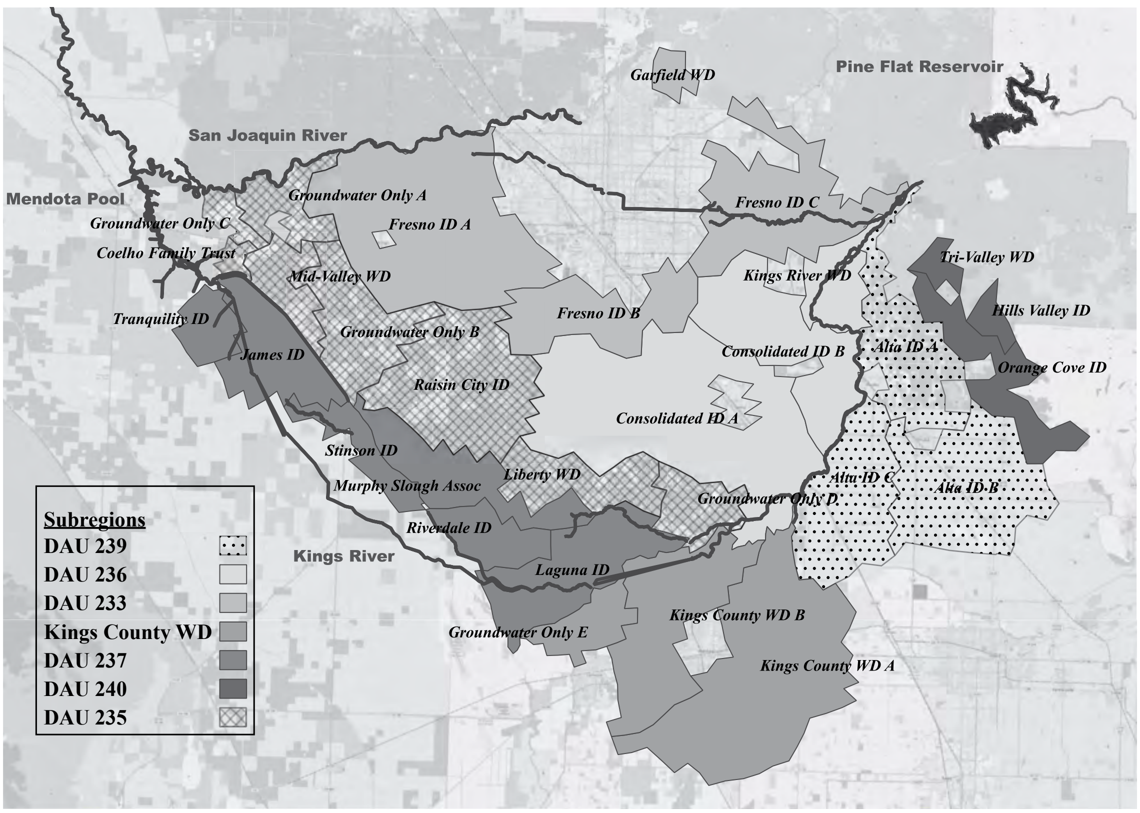


**Figure 1.** Subdistrict and Decision Analysis Units (DAUs) division in the model for the Kings Groundwater Basin.

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as well as field crops. Urban population in the basin, which grows steadily is nearly 800 thousand, of which 75% reside in the cities of Fresno and Clovis—located in the northern part of the basin.

The main source of surface water supply for agricultural irrigation in the region is the Kings River, which flows from Pine Flat Reservoir southwest to the bottom of the basin, and continues northwest meeting the San Joaquin River at Mendota Pool (Figure 1). Roughly 30 water agencies in the region rely on Kings River flows, as well as on surface water diversions from the San Joaquin River and the Central Valley Project through the Friant-Kern Canal, although to a smaller extent (for further detail see Table S1 in Supporting Information S1). Reliance on groundwater in the region is substantial, primarily in the urban sector, and increases in times of droughts when reductions in surface flows occur due to lower snowpack in the Sierra Nevada. Historically, groundwater overdraft in the region is estimated at 100–150 thousand-acre-feet (TAF) annually. This results in severe depletion, designating it as one of 21 groundwater basins in critical overdraft, and one of 94 groundwater basins subject to required actions under the SGMA legislation in California. A third source of water for agricultural irrigation and other purposes in the region that grows in importance due to the steady increase in population and the lack of disposal alternatives is treated wastewater. The comprehensiveness of the Kings Groundwater Basin in terms of water sources, crop selection, and predicted changes in groundwater management due to SGMA implementation, makes it an ideal case study for the analysis herein, and the results relevant to other regions in the Central Valley and across the world.

### 3.1. Data Collection and Calibration Procedures

The calibration process of the EOM included several steps and was validated in each step using the calibration tests suggested by Howitt et al. (2012). First, we divided the region of interest to 30 subdistricts (Figure 1), based on differences in hydrological and climatic conditions, as well as crop patterns and irrigation district affiliation. Subdistricts are then grouped to subregions according to the delineation to Decision Analysis Units (DAUs) to match the spatial resolution in the CVPAM model. We model 20 different land categories in the study area, including land fallowing. Data on land allocation for the different crops were collected from the California Department of Water Resources (CADWR/DWR) Land Use Viewer (n.d.) for the year 2014. That data is presented in Figure 2, according to DAUs delineation and by DWR land categories (detailed description of categories, crops included in each one, and the equivalent land category according to DWR definitions is presented in Appendix Table A1). Almonds and grapes are the largest crops in terms of land cultivated, covering almost 50% of the area. Other significant crops grown are corn, alfalfa, and cotton from the field crops category, and citrus, peaches, nectarines, and plums from the fruit category, covering cumulatively 23%, and 17% of land area in the region, respectively. Fallowed land accounts for 5% of farmland in the region, and all other crops grown cover less than 8% of the area. Spatial variation in crop specialization across the region is noticeable in Figure 2, which displays that field crops are predominantly grown in the southwest periphery of the region.

We import the parameters for the evapotranspiration functions defined in Equation 3 from previous work (Kan & Rapaport-Rom, 2012) conducted in Israel (i.e., $\overline{e}_{udj}$ and $\alpha_{1udj}$ through $\alpha_{5udj}$ as reported in Appendix Tables A2 and A3 through A5) based on the assumption that under similar growing conditions (soil type and climate) the agronomic growth process for each crop remains the same. We performed a detailed comparative analysis of soil structure and climate between regions in Israel and in our Kings Groundwater Basin study area, and validated that growing conditions in both regions are indeed similar. Soil taxonomy comparison is performed according to the Great Group classification (Hirmas, 2019), and based on data collected from the Natural Resources Conservation Service of the United States Department of Agriculture (NRCS-USDA, n.d.) for the Kings Groundwater Basin. In addition, we used soils map of Israel (Dan et al., 1975), translated by the International Arid Land Consortium (IALC, n.d.) to match the Great Group classification. We used data from meteorological stations in both Israel and the Kings Basin in California to compare climate conditions (for further detail on these comparisons, see Subsection 3.2.2.1 in Reznik et al., 2020).

The calibration process of the production function parameters ($\theta_{1udj}$ and $\theta_{2udj}$) follows the procedure prescribed in Kan and Rapaport-Rom (2012). First, to extract $\theta_{2udj}$ from Equation 3, we equate, for each crop, the value of marginal product (VMP) of water (which is the derivative of $y\left(w_{udjt}, \boldsymbol{q}^{\varphi}_{udt}\right)$ with respect to $w_{udjt}$ multiplied by crop price $p^{y}_{jt}$), with the observed price of water. The derivative of $y\left(w_{udjt}, \boldsymbol{q}^{\varphi}_{udt}\right)$ with respect to $w_{udjt}$ itself is termed the Marginal Product (or the Marginal Productivity of water). Once calibrated, we use $\theta_{2udj}$ and observed yield,

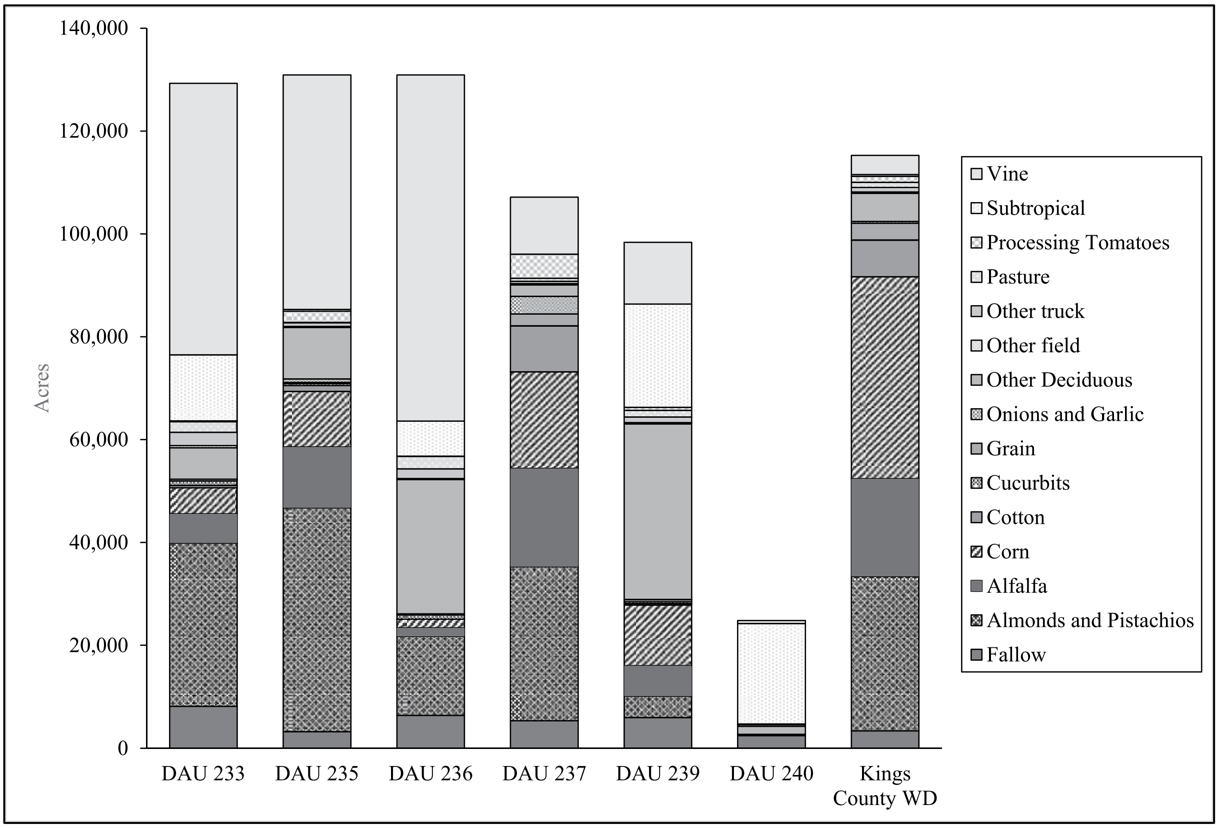


**Figure 2.** Land allocation in the Kings Groundwater Basin by Decision Analysis Units (DAUs) (source: CADWR Land Use Viewer, statewide crop mapping, 2014).

and applied water quantity and quality per acre to find $\theta_{1udj}$. For that purpose, data for yield, output prices and cost of water were collected from the University of California Cooperative Extension cost and return studies (UCCE, n.d.) and from California County Agricultural Commissioner's reports published by the USDA for Fresno, Kings, and Tulare counties. Water quantity and quality data were retrieved from the Kings River Watershed Coalition Authority (KRWCA) Groundwater Assessment Report (2014), and the State Water Resources Control Board (SWRCB) Groundwater Ambient Monitoring and Assessment Plan (n.d.).

Calibration of $\delta_{1j}$ and $\delta_{2j}$ parameters of the quadratic cost function in Equation 2 is performed using the two-stage PMP calibration procedure developed by Howitt (1995). Land use data required for this procedure was collected from the CADWR Land Use Viewer (n.d.) and displayed in Figure 2. Values for the parameter $\gamma_j$ from Equation 2, representing per unit of land variable costs of production (excluding water costs) for each crop, were collected from the UCCE (n.d.) cost and return studies (detailed data used in the calibration process, at the subdistrict, and crop levels is displayed in Appendix Tables A6 and A7). We acknowledge the uniqueness of perennial crops production (Franklin et al., 2017) and therefore impose further structure on land allocation decisions. We do that by forcing lower yield in early years of perennials, if expansion of land devoted to these crops occurs within the planning horizon of the model. This yield constraint is set to 50% of the yield of a mature plantation (orchard). Number of years until maturity vary by crop, and is calibrated based on data from the UCCE cost and return studies.

We distinguish in the model between the costs of groundwater pumping, surface water deliveries, wastewater treatment, and intentional recharge through infiltration basins. We attribute the cost of conveyance to all water sources available for each subdistrict. Conveyance costs are calculated based on average distance of conveyance within the subdistrict from each source. We use fine resolution well-level data (DWR's Water Data Library, n.d.) to determine lift, and use estimates on energy costs from the literature (MacEwan et al., 2017) to compute pumping costs per acre-foot (AF) at approximately 27.5 cents. Cost of wastewater treatment is assumed monotonically increasing with quantity treated at a decreasing rate. We use the parameters estimated by Fraas and

Munely (1984), adjusted for inflation, for that cost relationship. Intentional recharge through infiltration basins can accommodate all available water sources to the subdistrict and does not carry additional costs other than conveyance from these sources.

Annual rainfall and river flow volumes in the region are assumed constant at their long-term annual averages throughout the planning horizon. These are equal to 8.2 (Inches), 1,737, 475, and 1,076 TAF per year for the Kings River, San Joaquin River, and the Friant-Kern canal, respectively. Diversions from each of the surface water supply sources to the region are capped by their maximum historical record. These maximal quantities are 2,314, 92, and 233 (TAF) for the Kings River, San Joaquin River, and the Friant-Kern canal, respectively (WRIME, 2006). In order to assign values for $s_j$ from Equation 4, we use the Soil Agricultural Groundwater Banking Index (SAGBI), which is a suitability index for groundwater recharge on agricultural lands (O'Geen et al., 2015). The shares $s_j$, are calculated for each subdistrict as the portion of land rated Moderately Good, Good, or Excellent according to the SAGBI Index (values at the subdistrict level are presented in Appendix Table A7). The share of sewage generated in urban centers out of the quantity consumed by city inhabitants is calibrated at 60% (City of Fresno, 2013). Initial groundwater head at the subdistrict level $G_{ud0}$ is calibrated based on fine resolution well-level data (DWR's Water Data Library, n.d.) and equals 170 feet (above sea level) on average in the region (for detailed description see Figures S12–S18 in Supporting Information S1). A value of 0.113 is used for specific yield (CADWR, 2006), and is assumed fixed for the entire basin. The time lag $\tau$, representing travel time to groundwater table in Equation 4 is assumed homogeneous throughout the basin and is set equal to 1. Thus, for the calculation of groundwater dynamics in the first time period according to Equation 4 we rely on the initial conditions of the hydrologic model as specified in Supporting Information S1. Initial values for recharge through infiltration basins in the region and their capacities are taken from the groundwater sustainability plans (GSPs) of groundwater sustainability agencies (GSAs) in the region (CADWR SGMA, n.d.). Motivated by the objectives presented in SGMA, we set the planning horizon of the model to 20 years—starting at the baseline year of 2014.

## 4. Institutional Design Scenarios

We develop three alternative scenarios of water management institutions, in order to evaluate the feasibility of MAR and its long-term efficiency, as well as to understand its sensitivity to institutional arrangements. The first, which sets the benchmark for the other two, is the social planner solution (*Social*), corresponding to the solution that maximizes the present value of net gains of the entire region, ignoring income distribution implications among the subdistricts, which is also the equivalent of a free-market competitive equilibrium institution. The second, coined *Sustainable*, is constructed in the spirit of SGMA. In this scenario, added to the set of constraints $\mathbf{\Omega}$ of the *Social* scenario, we require that for each subdistrict, groundwater head at the end of the 20 yr planning horizon will be greater or equal to its initial level at the onset of the planning horizon. We adopt this criterion from the GSPs of the two largest (in terms of land area) GSAs in the region—Central Kings GSA and North Kings GSA. This scenario also resembles the "no overdraft" scenario in Harou and Lund (2008), who examined potential strategies to end groundwater overdraft in the Tulare Groundwater Basin in California. The third scenario, coined *Credit*, uses the principles of "capacity sharing" of an aquifer system (Dudley & Musgrave, 1988). According to this institution, the annual groundwater amount that can be extracted from the aquifer is limited by a credit account for each DAU and is based on the storage capacity of the aquifer. An initial endowment of annual credit is assigned to each DAU at the onset of the planning horizon based on land area and built capacity of infiltration basins for MAR. Accumulated credit increases with MAR (through infiltration basins) and decreases with groundwater pumping throughout the planning horizon. Such limitation on groundwater extraction is unique to the *Credit* scenario, and we implicitly assume that all other institutional design components (e.g., laws, public administration, and monitoring and accounting of credit accumulation) operate efficiently. Differently from the other two institutions, the *Credit* scenario limits cooperation by constraining groundwater use in correlation to subregion boundaries, and therefore implicitly assigns groundwater property rights according to subregion borders—discriminating DAUs with limited sources other than groundwater. Hence, comparing this scenario to the *Social* and *Sustainable* scenarios, which assume full cooperation in the region, offers the opportunity to examine the effect of an allocation mechanism of property rights to groundwater stock versus a perfect market design, and follows the research efforts we cited earlier, which examined the economic impacts of institutional changes in water modeling frameworks (see Section 4.2.3 in Booker et al., 2012, p. 191 and references therein).

## 5. Results

According to the results of the *Social* scenario, treated wastewater discharge in DAU 233 is the primary source of groundwater recharge in the region. The second important source for groundwater recharge in the region is excess irrigation of field crops in Kings County WD subregion. This finding strengthens previous predictions by Dahlke et al. (2018) regarding the potential of this strategy, which they coin as Ag-MAR. The choice of field crops land for recharge in the model is explained by the lower marginal productivity of water for that group of crops (and therefore higher percolation rates) at high water application levels compared to tree crops (Figure A1), and their larger land shares compared to vegetables (Figure 2). Compared to observed levels, regional land share of field crops increases at the expense of fallowed land. Land allocated to permanent crops in the region under the *Social* scenario is similar to its observed level. The exception is DAU 235, in which field crops and land fallowing area replaces that of fruit crops and nuts. Groundwater extraction in the region under this scenario is significantly lower than observed levels. It is mostly concentrated in DAU 235—a subregion with very little access to surface water sources and is increasing throughout the planning horizon at a rate of about 1% annually. Total agricultural water use in the region remains constant throughout the planning horizon at about 1.25 million acre-feet (MAF), which is equivalent to 60% of actual use in an average year, and deficit irrigation is preferred for most fruit and nut crops in the region. Consequently, the annual volume of irrigation water recharged into the groundwater basin slightly decreases from about 74 TAF to about 66 TAF over the planning horizon. At the same time, recharge of treated wastewater increases from about 83 TAF to about 125 TAF, such that total volume recharged into the basin increases over time. Spatial and temporal groundwater dynamic outcomes of these land allocation and irrigation strategies are depicted in Figure 3 at the subregion level.

As illustrated in Figure 3, the development of groundwater head differences over the planning horizon creates a cone of depression toward DAU 235. Consequently, deep percolation of treated wastewater that are not being reused for beneficial purposes in DAU 233, as well as deep percolation resulting from excess irrigation of field crops in DAU 237 and Kings County WD, all flow underground to enable groundwater extraction to support agricultural production in DAU 235. This concurrently minimizes overdraft in that subregion over the planning horizon. On average, groundwater head in the region remains almost unchanged.

The value of water in production (VMP) is in the range of \$36 and \$244 per AF. VMP is lowest in the northeastern part of the region where surface water is most abundant and increases as surface water availability decreases and reliance on groundwater becomes more important. VMP of water is negatively correlated with water application levels, hence increases with time in DAU 233, DAU 235, and Kings County WD—where water application levels per acre slightly decrease over the planning horizon. Water value in DAU 235, although affected by groundwater

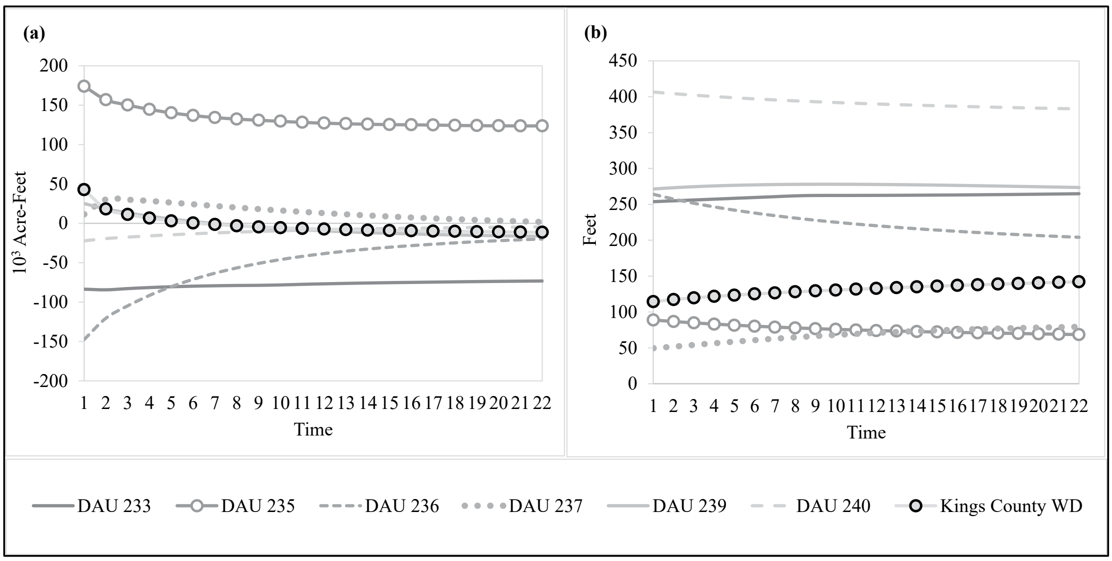


**Figure 3.** Temporal groundwater dynamics at the subregion level: (a) net lateral groundwater flows; (b) groundwater head.

**Table 1**
*Regional Land Allocation to Crops as Percentage of Observed Levels*

| | Fallow | Almonds and pistachios | Field crops | Fruit | Vegetables | Vine |
|---|---|---|---|---|---|---|
| *Sustainable* | 88 | 96 | 113 | 95 | 103 | 96 |
| *Credit* | 304 | 80 | 100 | 93 | 116 | 82 |

*Note.* For example, according the optimal plan under the *Sustainable* scenario, fallowed land should reduce by 12% in the region compared to total observed acreage. Similarly, land devoted to nut trees and grapes need to shrink by 4%, and fruit crops area should decrease by 5%, compared to observed land allocation according to the *Sustainable* scenario results. Field crops and vegetables area should increase by 13%, and 3%, respectively, according to the *Sustainable* scenario optimal plan compared to observed land allocation.

scarcity, is lower than for the other DAUs mentioned, due to the considerably higher salinity level in groundwater. Overall, the optimal plan suggested by the model predicts an annual regional profit of about $2.2 billion, distributed roughly according to DAU size.

We report the results from the two other institutional design scenarios in Table 1 and Figure 4. Land allocation under the *Sustainable* scenario is similar to that of the *Social* scenario (Table 1). However, water management strategies of the *Sustainable* scenario differ substantially than those of the *Social* scenario. Total water use in agriculture is higher at the onset of the planning horizon by 40% compared to the *Social* scenario, and exhibits a decreasing trend over time. It increases again under the *Sustainable* scenario towards the end of the planning horizon, peaking at 1.85 MAF then dropping to the level of agricultural water use under the *Social* scenario as the planning horizon ends (Figure 4a). Unlike in the *Social* scenario, excess irrigation of agricultural crops is the main source of groundwater recharge under the *Sustainable* scenario. Kings County WD subregion leads in terms of quantities recharged using this approach under the *Sustainable* scenario followed by DAU 236. Similar to the *Social* scenario, treated wastewater discharge is another source for groundwater recharge that grows in importance over time. Differently from the *Social* scenario, recharge of groundwater using infiltration basins is found optimal under the *Sustainable* scenario, averaging at 60 TAF annually. In total, recharged quantities under the *Sustainable* scenario are significantly higher compared to the *Social* scenario (Figure 4b). Groundwater extraction paths of the *Social* and the *Sustainable* scenarios are almost identical. Therefore, the average groundwater head in the region increases over time in the *Sustainable* scenario by 20%. Differences in income distribution among subregions between the *Social* and the *Sustainable* scenarios are negligible.

The land allocation results of the *Credit* scenario presented in Table 1 suggest a dramatic increase in land fallowing, mostly at the expense of tree and fruit crops. These changes are mainly concentrated in DAU 235. For the rest of the region, land allocation differences compared to the *Social* and *Sustainable* scenarios are far less significant. Results of the *Credit* scenario also suggest lower use of water in agriculture, compared to the *Social*

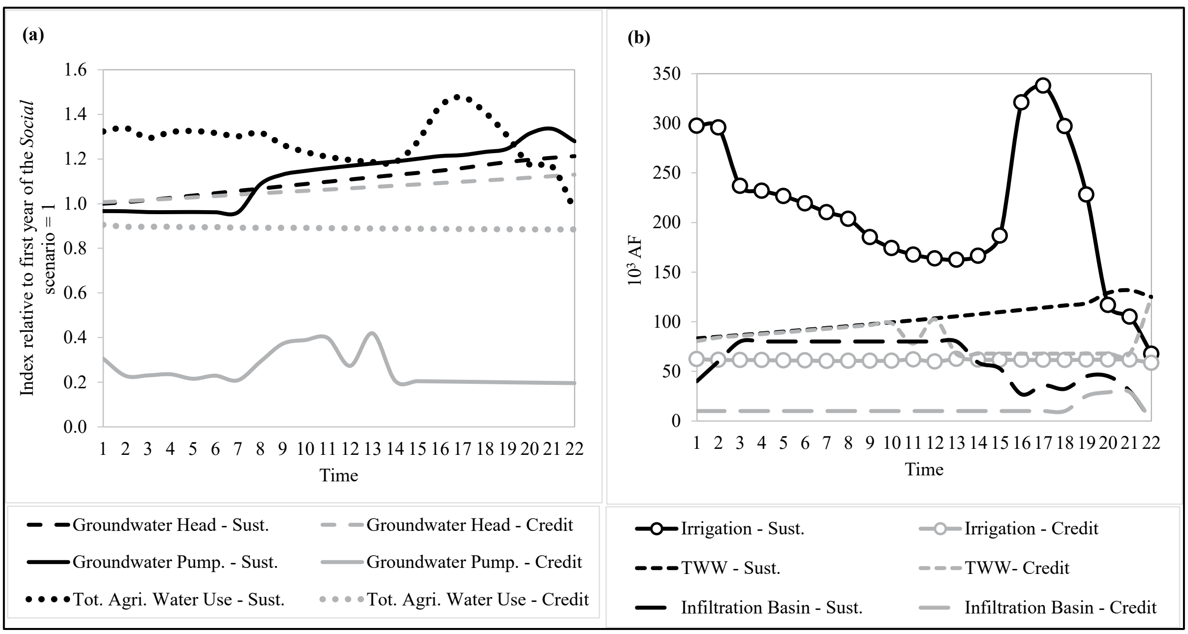


**Figure 4.** Water use and groundwater dynamics over time, under the *Sustainable* and *Credit* scenarios: (a) indices of total water use in agriculture, groundwater pumping, and groundwater head (first-year value of the corresponding index under the *Social* scenario, 1); (b) recharged quantities by source (TWW, treated wastewater).

**Table 2**
*Reductions in Economic Welfare Compared to the Social Scenario (1,000 USD)*

| | Constant-Climate | Hist1 | Hist2 |
|---|---|---|---|
| Time lag ($\tau = 1$) | | | |
| *Sustainable* | 96,441 | 119,344 | 105,197 |
| *Credit* | 1,639,823 | 1,605,203 | 1,615,791 |
| Time lag ($\tau = 5$) | | | |
| *Sustainable* | 70,366 | 77,443 | 74,821 |
| *Credit* | 1,715,766 | 1,697,051 | 1,707,836 |
| Time lag ($\tau = 10$) | | | |
| *Sustainable* | 123,326 | 132,882 | 115,563 |
| *Credit* | 2,041,966 | 1,927,023 | 2,044,836 |

scenario (Figure 4a). Groundwater pumping is profoundly lower (Figure 4a), and reused quantities of treated wastewater are higher under this scenario, which is implied by the decreased quantities of groundwater recharge from this source (Figure 4b), compared to the results of the *Social* scenario. Due to lower water use in agriculture, recharged quantities are lower for this scenario by 10% compared to the *Social* scenario (Figure 4b). However, as already mentioned, pumping is also considerably less. Therefore, average groundwater levels increase over time in the *Credit* scenario, whereas they remain constant in the *Social* scenario (Figure 4a). Differently from the *Social* scenario and similarly to the *Sustainable* scenario, MAR through infiltration basins is found to be optimal under the *Credit* scenario. This is because some subdistricts in DAU 235 and Kings County WD rely solely on groundwater, which forces that type of recharge as a means to accumulate credit and to enable groundwater extraction throughout the planning horizon under this institutional arrangement.

Comparing agricultural profits between the *Social* and the *Credit* scenarios reveals that the economic loss associated with the latter institutional arrangement is concentrated in DAU 235 and amounts to roughly \$1.75 billion (Table 2). It is implied that under the allocation of groundwater property rights as prescribed by the *Credit* institution, a market mechanism enabling exchange of income with groundwater extraction credit is economically warranted for an income transfer lower than the calculated economic loss of \$1.75 billion from DAU 235 to one or more of all other subregions. According to the results of both the *Credit* and the *Sustainable* scenarios groundwater head in DAU 235 increases by about 40% over the entire planning horizon. Thus, in order to maintain the same time-trend in groundwater head under higher groundwater extractions in DAU 235 and in view of the *Credit* institution, groundwater recharge in adjacent subregions to DAU 235 (i.e., DAU 233, DAU 236, and Kings County WD) should also be higher and similar to their level according to the results of the *Sustainable* scenario. Therefore, it is further implied that cooperation in the form of income transfers from DAU 235, specifically to neighboring subregions, aimed to incentivize higher recharge in these subregions, would also be economically warranted if the sum of all transfers from DAU 235 for both credit rights and recharge purposes do not exceed \$1.75 billion.

### 5.1. Sensitivity Analysis of Climate Conditions and Travel Time to Groundwater Table

We examine the sensitivity of our results with respect to two limiting assumptions in the model. First, we examine the impact of the *Constant-Climate* assumption using two alternative climate scenarios and study their impact on the outcomes of the different institutional design scenarios. The first, termed *Hist1*, assumes that regional climate conditions are similar to those in the period 1975–1996. The second, termed *Hist2*, refers to the climate conditions in the region during the period 1983–2004. The time-series values for regional rainfall and surface water availability that are used under these two climate scenarios are presented in Figure 5.

Comparing results between climate simulations and under the different institutions, we find that treated wastewater and groundwater storage are used as sources for stabilizing supply and consumption (see Figures B2–B4). This is when significant reductions in surface water supply occur under the *Hist1* and *Hist2* climate simulations. This finding strengthens previous contributions to the literature concerning benefits of conjunctive use (Tsur & Graham-Tomassi, 1991), and of treated wastewater reuse in agriculture (Feinerman & Tsur, 2014). As a consequence, the quantity recharged from both excess irrigation of surface water and from treated wastewater discharge are lower under the *Hist1* and *Hist2* climate simulations, compared to the *Constant-Climate* simulation for all institutional design scenarios under the different assumptions regarding travel time to groundwater. This, in turn, implies that intensification of dry-year sequences can impair the effectiveness of MAR strategy.

A second simplifying assumption in our modeling framework is that travel time to groundwater table of water applied to the surface and deep percolate through the porous is fixed and homogeneous throughout the region and equals 1 yr. We use two alternative values for the travel time ($\tau = 5$) and ($\tau = 10$) in order to examine the impact of this assumption. Table 2 presents total regional economic welfare differences, compared to the *Social* scenario in annual terms, across institutions, climate simulations, and under different assumptions regarding travel time to

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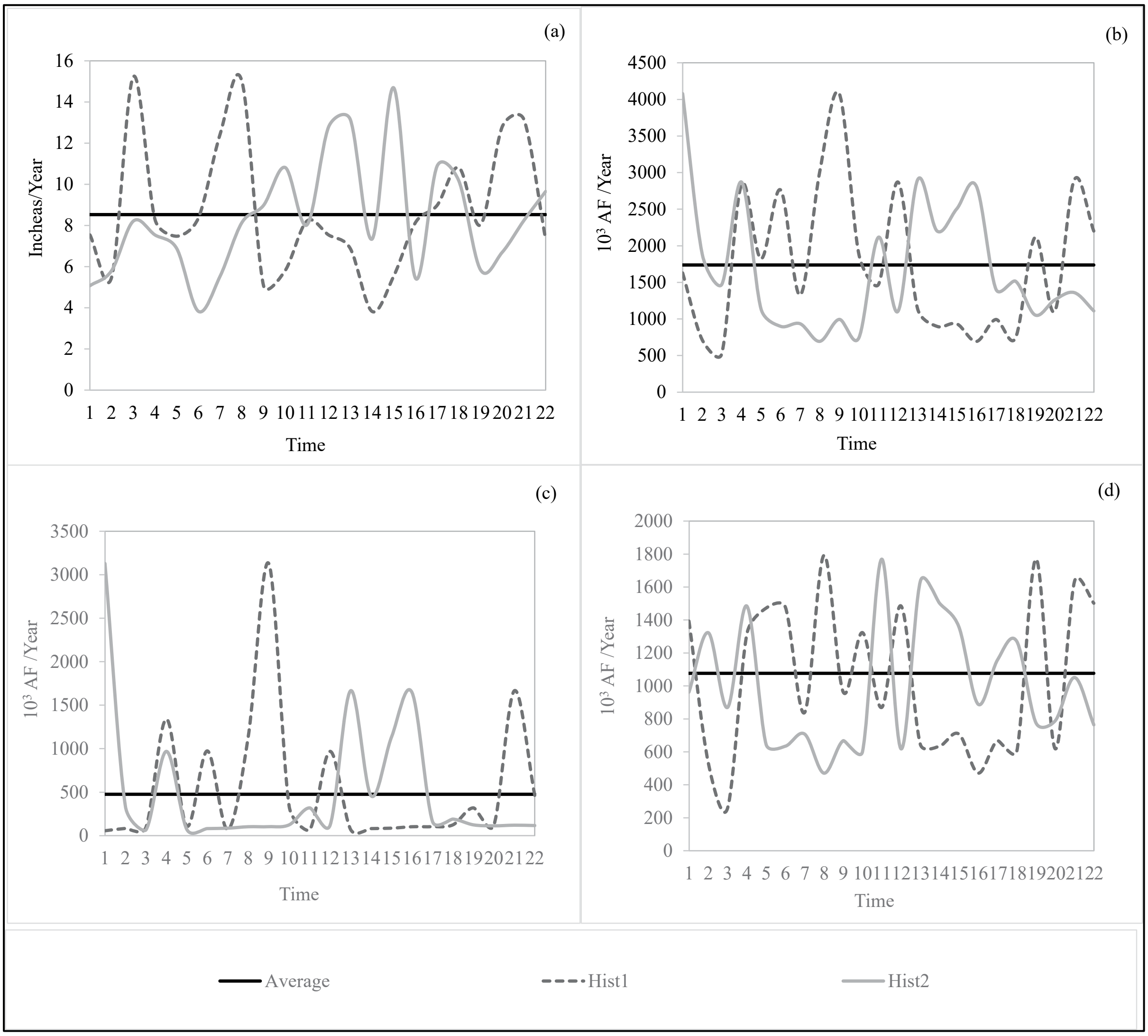


**Figure 5.** Simulated annual surface water availability and rainfall in the region: (a) regional rainfall; (b) Kings River flow; (c) San Joaquin River flow; (d) Friant-Kern Canal flow.

groundwater table. According to the results presented in Table 2, the institutional arrangement under the *Credit* scenario inflicts significant welfare reductions on the region relative to the benchmark, and specifically on DAU 235. As we already presented, economic loss is driven by changes in cropping and water use decisions. The economic cost of the *Sustainable* scenario is in the range of \$70–\$130 million USD annually, which is relatively mild. By comparison, revenues from agricultural commodities in the Kings Groundwater Basin are estimated at \$6–\$8 billion USD annually. Under both institutions, the economic welfare difference compared to the *Social* scenario increase the longer it takes to water to reach the groundwater table, although for the *Sustainable* scenario the impact is nonmonotonic.

We do not find any qualitative differences between the results reported for the scenarios of different assumed travel time to groundwater (Figures B1–B4). However, quantitively some differences are worth mentioning. We find that longer travel time promotes intuitive temporal shifts in water management decisions primarily under the *Sustainable* and *Credit* scenarios. For example, groundwater extraction across all different institutions is concentrated in earlier time periods under the ($\tau$ = 5) and ($\tau$ = 10) assumptions compared to the ($\tau$ = 1) assumption. Quantitative differences in results associated with the travel time assumption are summarized in Figure 6, which presents the total recharged quantities in the region over the entire planning horizon by the source of recharge, as

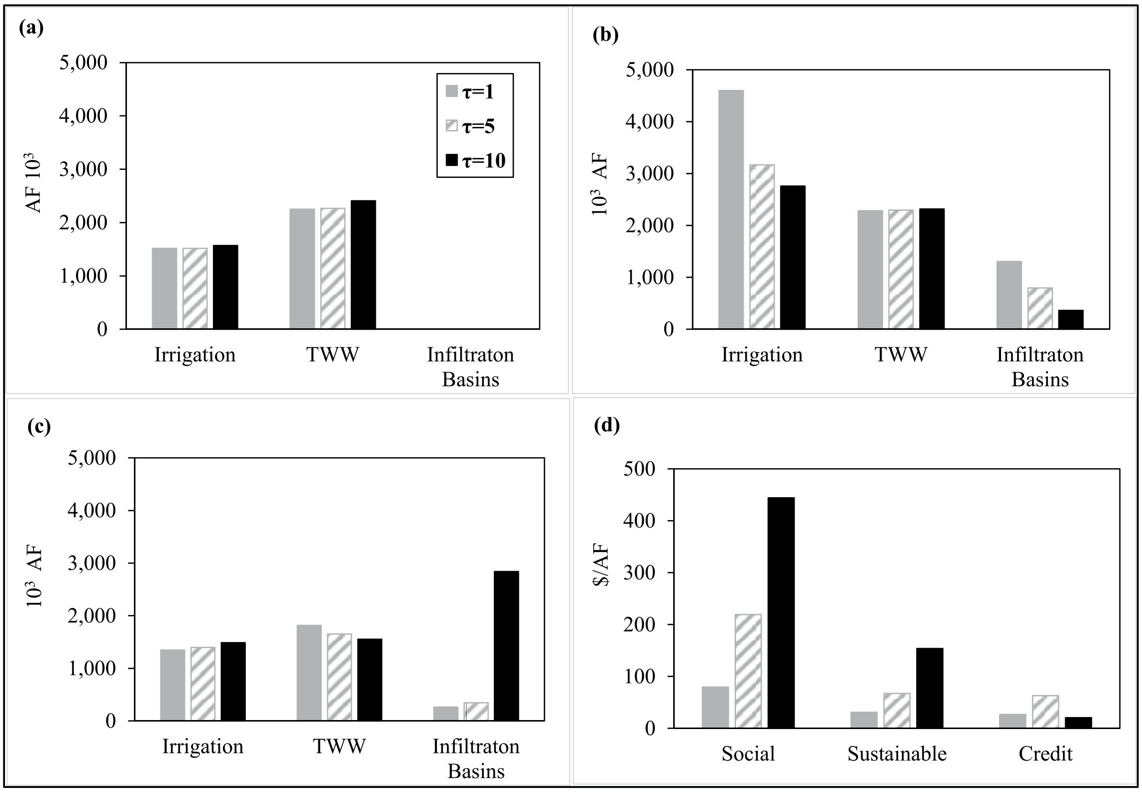


**Figure 6.** Groundwater recharge according to sources and the value of recharge across institutions under the *Constant-Climate* conditions for different travel time assumptions: (a) total recharged quantities over the planning horizon under the *Social* scenario; (b) total recharged quantities over the planning horizon under the *Sustainable* scenario; (c) total recharged quantities over the planning horizon under the *Credit* scenario; (d) value of groundwater recharge across institutional design scenarios.

well as the calculated value for the region of an AF of MAR. To compute the latter, for each institutional design scenario under each climate simulation and for every value of assumed travel time of groundwater, we calculate the economic welfare difference to an equivalent scenario (coined *No Recharge*). In the *No Recharge* alternative cropping and water use decisions are set to their optimal outcomes; however, groundwater recharge through excess irrigation and infiltration basins is not applicable. That economic welfare difference is then divided by the quantity of water recharged in excess of treated wastewater discharge to groundwater aquifers. The underlying assumption is that in the absence of safe discharge alternatives for treated wastewater, under the *No Recharge* alternative that wastewater quantity percolates to the ground, which is also consistent of actual existing conditions in the Kings Groundwater Basin region. The value of recharge then emerges as the pumping costs saved due to higher groundwater levels and additional benefits generated from reallocation of cheaper groundwater among crops.

It can be seen in Figure 6 that travel time assumption affects the optimal water management in the region inconsistently across institutional design scenarios. Under the *Social* scenario recharged quantities from both excess irrigation and treated wastewater discharge sources increase the longer the travel time to groundwater is assumed (Figure 6a). For the *Sustainable* scenario, the impact on recharge from excess irrigation and infiltration basins is the opposite. However, similar to the *Social* scenario, recharged quantities of treated wastewater discharge under the *Sustainable* scenario also increase the longer it is assumed that water travels to the groundwater table (Figure 6b). Under the *Credit* scenario, recharged volumes from excess irrigation and infiltration basins increase with longer assumed travel time and recharge of treated wastewater decreases (Figure 6c). Consequently, the value of recharge ranges between $14 and $444 per AF across institutions, climate conditions and travel time assumed. It increases with assumed travel time to the groundwater table for the *Social* and *Sustainable* scenarios, and presents nonmonotonic behavior under the *Credit* scenario. Compared to the VMP, the value of an AF recharged surpasses 50% of the average VMP in two-thirds of all scenarios examined, and is higher than the average VMP for a third of all scenarios examined.

## 6. Conclusions and Policy Implications

Economic research of groundwater management promotes MAR under spatial homogeneity assumptions and unique conditions in terms of natural recharge, variability of surface flows, and groundwater depletion (Knapp & Olson, 1995). However, as argued by Scanlon et al. (2012) and supported by the results of our analysis, heterogeneity in spatiotemporal natural conditions and groundwater depletion support MAR implementation as an optimal strategy for a wider set of realities. In California, such heterogeneity is associated in part to the distortive historical allocation of surface water rights (Burness & Quirk, 1979), which renders many agricultural lands exclusively reliant on groundwater use. As demonstrated by the results of our analysis, this heterogeneity in view of groundwater property rights allocation implied from SGMA, holds the potential for inducing detrimental economic consequences if stringent institutions concerning groundwater use are adopted (e.g., the *Credit* institution), as well as creating opportunities for intra-basin arrangements that promote MAR implementation and the reallocation of water resources in the studied region. The latter could potentially be achieved through higher flexibility in regulatory structure and the introduction of market mechanisms—a conclusion that we share with other studies exploring the future of California's water supply (Hanak et al., 2018).

Our results show that total recharged quantities in the region over the entire planning horizon across institutions, climate simulations, and assumed travel time to groundwater are substantial. They range between 3.13 MAF and 8.19 MAF, an equivalent of about 1.5 times to little over 4 times the total observed annual water use in the region. In most cases, the calculated value of a unit of water recharged is substantial compared to the direct VMP. The results of our first-best social planner scenario indicate that recharge of treated wastewater using existing capacity and of surface water through excess irrigation of field crops is beneficial for the region, maintaining high-value agricultural production and keeping groundwater level in the basin unchanged over time. This strategy is complemented by deficit irrigation of most permanent crops, reduced groundwater extractions, and increased quality of applied water on crops with respect to observed levels in the region. The same recharge strategy is found optimal and even supplemented by intentional recharge through designated infrastructure under the *Sustainable* scenario—designed according to the principles of the SGMA legislation, suggesting that this institution incentivizes MAR. This latter conclusion is congruent with previous findings by Haruo and Lund (2008) regarding the outcomes of preventing groundwater overdraft.

Under the *Credit* scenario, significant land fallowing is warranted, replacing permanent crops and inflicting detrimental economic consequences compared to the other institutional design scenarios, manifested mainly in groundwater-reliant areas. We find that this economic welfare loss is about $1.75 billion USD annually and is concentrated in DAU 235. It is implied that under the allocation of rights to groundwater storage as prescribed in the *Credit* scenario and in view of subbasin boundaries derived from SGMA, introducing market mechanisms to facilitate an exchange of groundwater extraction credits and income would be economically warranted for the region resulting in a second-best solution. It is further implied that such an institution incentivizes MAR implementation through intra-basin arrangements, supporting high-value agricultural production in groundwater-reliant areas while meeting SGMA objectives. Interestingly, while reviewing GSPs in the region we find evidence for such an arrangement between the McMullin Area GSA (which almost completely overlaps DAU 235), and the North Kings GSA (which perfectly overlaps DAU 233), in which the McMullin Area GSA will fund a project designed to construct infiltration basins on the border between the two GSAs, recharging excess surface water from the North Kings GSA.

Our analysis, as well as the SGMA framework, assumes a high level of cooperation and coordination between stakeholders in the region. As highlighted by the results of our analysis, regional heterogeneity in terms of access to water resources, available water quantities, and agricultural-growing conditions imply, for example, that some subregions specializing in field crop agriculture will act as a buffer for the entire region—decreasing their surface water diversions and increasing land fallowing when water supply fluctuates. Another example is intentional recharge through excess irrigation, recommended according to the model results in subregions that do not rely on groundwater, for the sole purpose of affecting groundwater flow direction. These behaviors are a direct outcome of the assumption that subregions fully cooperate and ignore income distribution, and are highly unlikely to sustain under less-lenient institutions or more extreme changes in climate and other exogenous conditions. According to Hanak et al. (2019), such cooperation is essential for the sustainability of water supply in this region. Our analysis supports this argument by demonstrating the importance of regional cooperation, and its ability to mitigate the asymmetric economic consequences associated with different institutional arrangements, which result from regional heterogeneity in terms of exogenous conditions. It is also demonstrated that market mechanisms introduced in view of property rights allocation to groundwater storage could potentially induce win-win arrangements reducing the need for coordination and relying on stakeholders' individual incentives to achieve a second-best solution for the region. Thus, exploring a wider set of institutional arrangements under different coalitional structures, different strategic behavior assumptions, and under equilibrium solutions is a promising endeavor for effective policy recommendation purposes, and where we aim our future research. Future work will explore more closely the different aspects of regional cooperation over groundwater management as a possible direction to sustain such resources under uncertain future supplies.

It should be noted that while the results of our analysis warrant recharge of significant quantities through excess irrigation of crop land, this result is partially related to the assumption that no yield loss is associated with this strategy (within the range of excessive amounts of applied water). This assumption is based on recent evidence from California (Dahlke et al., 2018) and obviously if relaxed would render this strategy of recharge less attractive. As a final comment, our analysis somewhat simplifies hydrological and hydrogeological complexities in the region, and only partially captures climate uncertainty. By performing a sensitivity analysis, we find that our hydrological simplifying assumptions have no qualitative impact on our results and therefore have very little significance regarding our conclusions and for policy recommendation purposes. Nevertheless, it is demonstrated that assumed values representing hydrological complexities could induce significant quantitative differences in reported outcomes. Thus, by incorporating higher resolution hydrogeological characteristics one could elicit more accurate results supporting better recommendations for local practices and improve performance of individual project implementation. Additionally, our sensitivity analysis of climate scenarios suggests that variation in surface water availability reduces the quantities allocated for MAR, which corroborates previous hypotheses regarding the opportunity costs associated with the different allocations of water sources in the region. Unfortunately, the full impact of climate uncertainty can only be captured through utilization of stochastic optimization methods. We leave these endeavors for future research.

## Appendix A: Calibration Data

Table A1 lists all 20 different land use categories represented in the EOM, their equivalent land categories according to DWR definitions, and the group of specific crops aggregated in each category.

**Table A1**
*Land Use Categories Representation in the EOM and DWR Definitions*

| DWR/CVPAM land category | Land category EOM | DWR: statewide crop mapping 2014 |
|---|---|---|
| Fallow | Fallowed | Idle |
| Almonds and pistachios | Almonds | Almonds, pistachios, and walnuts |
| Alfalfa | Alfalfa | Alfalfa and alfalfa mixtures |
| Corn | Corn | Corn, sorghum, and sudan |
| Cotton | Cotton | Cotton |
| Cucurbits | Melon | Melons, squash, and cucumbers |
| Dry beans | Garbanzo beans | Beans (dry) |
| Tomatoes for market | Tomatoes | |
| Grain | Wheat | Wheat |
| MultiCrop | Other field crops | |
| Onions and garlic | Onions | Onions and garlic |
| Other deciduous | Peaches and nectarines | Peaches/nectarines |
| | Plums | Plums, prunes, and apricots |
| | Cherries | Cherries |
| | Pomegranates | Pomegranates |
| | Apples | Apples, pears, young perennials, and miscellaneous deciduous |

**Table A1**
*Continued*

| DWR/CVPAM land category | Land category EOM | DWR: statewide crop mapping 2014 |
|---|---|---|
| Other field | Sorghum | Miscellaneous field crops and miscellaneous grain and hay |
| Other truck | Broccoli | Peppers, carrots, strawberries, bush berries, lettuce/leafy greens, miscellaneous truck crops, cole crops, flowers, nurseries and Christmas tree farms |
| Pasture | Pasture | Mixed pasture and miscellaneous grasses |
| Potatoes | Potatoes | Potatoes and sweet potatoes |
| Processing tomatoes | Tomatoes | Tomatoes |
| Rice | Rice | |
| Safflower | Safflower | Safflower |
| Sugar beets | Sugar beets | |
| Subtropical | Oranges | Citrus |
| | Olives | Olives, kiwis, and miscellaneous subtropical fruits |
| Vine | Grapes | Grapes |

*Note.* DWR crop categories are widely defined over the entire Central Valley. Hence, some categories are irrelevant for the case of the Kings Groundwater Basin and therefore are not included in our model. These are: dry beans, tomatoes for market, multi crop, potatoes, rice, safflower and sugar beets.

Table A2 presents the assignment of functions imported from four regions in Israel to each of the subdistricts in the Kings Groundwater Basin, based on the comparison of soil and climate described in the text. The values for parameters of each of the evapotranspiration functions imported (Function A through Function D) is detailed in Tables A3 through A5.

**Table A2**
*Subdistrict Division According to Imported Evapotranspiration Functions From Regions in Israel*

| Function A | Function B | Function C | Function D |
|---|---|---|---|
| Alta ID B | Alta ID A | Fresno ID C | Kings County WD A |
| Alta ID C | Consolidated ID B | Garfield WD | |
| Consolidated ID A | Kings River WD | Kings County WD B | |
| James ID | Fresno ID A | Laguna ID | |
| Tranquility ID | Fresno ID B | Hills Valley ID | |
| | Groundwater Only E | Tri-Valley WD | |
| | Murphy Slough Assoc. | Mid-Valley WD | |
| | Riverdale ID | | |
| | Stinson ID | | |
| | Orange Cove ID | | |
| | Coelho Family Trust | | |
| | Groundwater Only A | | |
| | Groundwater Only B | | |
| | Groundwater Only C | | |
| | Groundwater Only D | | |
| | Liberty WD | | |
| | Raisin City ID | | |

*Note.* Groundwater Only A through E are names of subdistricts in which farmers are not affiliated with any water agency and rely only on local groundwater extractions for irrigation purposes.

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**Table A3**
*Values of Evapotranspiration Functions for Fruit Crops*

| | $\bar{e}_{udj}$ | $\alpha_{1udj}$ | $\alpha_{2udj}$ | $\alpha_{3udj}$ | $\alpha_{4udj}$ | $\alpha_{5udj}$ |
|---|---|---|---|---|---|---|
| Almonds | | | | | | |
| Function A | 5.204 | 0.122 | 1.000 | 8.434 | −1.192 | 1.972 |
| Function B | 5.404 | 0.126 | 1.000 | 8.694 | −1.175 | 1.932 |
| Function C | 5.146 | 0.117 | 1.000 | 8.681 | −1.177 | 1.924 |
| Function D | 4.802 | 0.105 | 1.000 | 8.663 | −1.180 | 1.914 |
| Peaches and nectarines | | | | | | |
| Function A | 5.234 | 0.115 | 1.000 | 8.157 | −1.163 | 2.032 |
| Function B | 5.495 | 0.125 | 1.000 | 8.670 | −1.175 | 1.971 |
| Function C | 5.357 | 0.122 | 1.000 | 8.672 | −1.173 | 1.951 |
| Function D | 5.174 | 0.117 | 1.000 | 8.676 | −1.171 | 1.925 |
| Plums | | | | | | |
| Function A | 5.241 | 0.102 | 1.000 | 8.561 | −1.164 | 2.044 |
| Function B | 5.519 | 0.115 | 1.000 | 9.214 | −1.190 | 1.973 |
| Function C | 5.433 | 0.115 | 1.000 | 9.249 | −1.192 | 1.943 |
| Function D | 5.318 | 0.116 | 1.000 | 9.294 | −1.194 | 1.904 |
| Cherries/pomegranates/apples | | | | | | |
| Function A | 5.247 | 0.286 | 1.000 | 4.719 | −1.089 | 2.262 |
| Function B | 5.545 | 0.309 | 1.000 | 4.913 | −1.097 | 2.195 |
| Function C | 5.535 | 0.302 | 1.000 | 4.947 | −1.087 | 2.180 |
| Function D | 5.522 | 0.292 | 1.000 | 4.992 | −1.074 | 2.160 |
| Oranges | | | | | | |
| Function A | 5.245 | 0.081 | 1.000 | 9.953 | −1.191 | 2.033 |
| Function B | 5.535 | 0.092 | 1.000 | 10.481 | −1.204 | 1.963 |
| Function C | 5.491 | 0.095 | 1.000 | 10.562 | −1.209 | 1.926 |
| Function D | 5.432 | 0.099 | 1.000 | 10.671 | −1.215 | 1.877 |
| Olives | | | | | | |
| Function A | 5.247 | 0.007 | 1.000 | 29.274 | −1.180 | 2.152 |
| Function B | 5.545 | 0.009 | 1.000 | 31.393 | −1.194 | 2.072 |
| Function C | 5.535 | 0.011 | 1.000 | 32.331 | −1.217 | 2.006 |
| Function D | 5.522 | 0.013 | 1.000 | 33.583 | −1.248 | 1.917 |
| Grapes | | | | | | |
| Function A | 5.067 | 0.061 | 1.000 | 14.924 | −1.297 | 1.794 |
| Function B | 5.083 | 0.062 | 1.000 | 15.821 | −1.319 | 1.749 |
| Function C | 4.646 | 0.054 | 1.000 | 15.925 | −1.345 | 1.749 |
| Function D | 4.063 | 0.044 | 1.000 | 16.062 | −1.380 | 1.749 |

We report in Tables A6 and A7 the data used for the calibration of the different production and cost functions' parameters, distinguishing between data at the crop level (fixed for all subdistricts) and data at the subdistrict level (fixed for all crops), respectively.

Figure A1 demonstrates how the values presented in Tables A3 through A5 translate water application levels to changes in evapotranspiration levels (and effectively also in yield) for the different crops bundled according to five main categories. According to Figure A1, field crops and vegetables differ from tree crops. For the former

**Table A4**
*Values of Evapotranspiration Functions for Field Crops*

| | $\bar{e}_{udj}$ | $\alpha_{1udj}$ | $\alpha_{2udj}$ | $\alpha_{3udj}$ | $\alpha_{4udj}$ | $\alpha_{5udj}$ |
|---|---|---|---|---|---|---|
| Alfalfa | | | | | | |
| Function A | 1.915 | 0.024 | 1.000 | 5.066 | −1.209 | 2.157 |
| Function B | 2.139 | 0.030 | 1.000 | 5.657 | −1.237 | 2.044 |
| Function C | 2.132 | 0.037 | 1.000 | 5.708 | −1.278 | 1.949 |
| Function D | 2.124 | 0.046 | 1.000 | 5.777 | −1.332 | 1.821 |
| Corn | | | | | | |
| Function A | 1.821 | 0.051 | 1.000 | 4.253 | −1.233 | 1.836 |
| Function B | 1.823 | 0.042 | 1.000 | 4.530 | −1.141 | 1.896 |
| Function C | 1.823 | 0.043 | 1.000 | 4.503 | −1.166 | 1.883 |
| Function D | 1.824 | 0.045 | 1.000 | 4.467 | −1.199 | 1.865 |
| Cotton | | | | | | |
| Function A | 2.775 | 0.001 | 1.000 | 28.131 | −1.074 | 2.199 |
| Function B | 2.785 | 2.4E-10 | 1.000 | 81.257 | −0.366 | 5.245 |
| Function C | 2.788 | 2.7E-04 | 1.000 | 66.343 | −0.569 | 4.347 |
| Function D | 2.792 | 0.001 | 1.000 | 46.459 | −0.839 | 3.149 |
| Wheat | | | | | | |
| Function A | 1.485 | 0.003 | 1.000 | 8.242 | −1.215 | 2.344 |
| Function B | 1.733 | 0.004 | 1.000 | 9.660 | −1.174 | 2.298 |
| Function C | 1.735 | 0.005 | 1.000 | 9.918 | −1.182 | 2.212 |
| Function D | 1.737 | 0.006 | 1.000 | 10.262 | −1.194 | 2.097 |
| Onions | | | | | | |
| Function A | 5.249 | 0.073 | 1.000 | 8.388 | −1.123 | 2.287 |
| Function B | 5.559 | 0.076 | 1.000 | 8.741 | −1.111 | 2.258 |
| Function C | 5.635 | 0.080 | 1.000 | 8.932 | −1.119 | 2.225 |
| Function D | 5.736 | 0.085 | 1.000 | 9.186 | −1.130 | 2.181 |
| Sorghum | | | | | | |
| Function A | 1.485 | 0.001 | 1.000 | 14.219 | −1.132 | 2.508 |
| Function B | 1.733 | 0.001 | 1.000 | 15.390 | −1.103 | 2.499 |
| Function C | 1.665 | 0.001 | 0.820 | 44.930 | −0.898 | 2.685 |
| Function D | 1.576 | 2.3E−04 | 0.580 | 84.318 | −0.625 | 2.934 |
| Pasture | | | | | | |
| Function A | 1.915 | 0.024 | 1.000 | 5.066 | −1.209 | 2.157 |
| Function B | 2.139 | 0.030 | 1.000 | 5.657 | −1.237 | 2.044 |
| Function C | 2.132 | 0.037 | 1.000 | 5.708 | −1.278 | 1.949 |
| Function D | 2.124 | 0.046 | 1.000 | 5.777 | −1.332 | 1.821 |

crop bundles, maximal changes in evapotranspiration level occur in low levels of water application and thereafter rapidly diminish. Whereas for the latter group of crops, changes in evapotranspiration level are more moderate, which also implies a higher marginal productivity of water at higher application levels.

**Table A5**
*Values of Evapotranspiration Functions for Vegetable Crops*

| | $\bar{e}_{udj}$ | $\alpha_{1udj}$ | $\alpha_{2udj}$ | $\alpha_{3udj}$ | $\alpha_{4udj}$ | $\alpha_{5udj}$ |
|---|---|---|---|---|---|---|
| Melons | | | | | | |
| Function A | 1.433 | 0.054 | 1.000 | 3.178 | −1.278 | 1.769 |
| Function B | 1.454 | 0.051 | 1.000 | 3.338 | −1.253 | 1.781 |
| Function C | 1.457 | 0.052 | 1.000 | 3.324 | −1.265 | 1.774 |
| Function D | 1.460 | 0.053 | 1.000 | 3.306 | −1.281 | 1.765 |
| Broccoli | | | | | | |
| Function A | 0.979 | 0.005 | 1.000 | 4.158 | −0.971 | 2.526 |
| Function B | 1.041 | 0.007 | 1.000 | 3.467 | −1.127 | 2.450 |
| Function C | 1.051 | 0.007 | 1.000 | 3.513 | −1.132 | 2.432 |
| Function D | 1.063 | 0.007 | 1.000 | 3.575 | −1.137 | 2.409 |
| Tomatoes | | | | | | |
| Function A | 5.249 | 0.005 | 1.000 | 11.965 | −0.719 | 2.765 |
| Function B | 5.559 | 0.001 | 1.000 | 13.128 | −0.501 | 3.338 |
| Function C | 5.635 | 0.007 | 1.000 | 14.276 | −0.691 | 3.012 |
| Function D | 5.736 | 0.015 | 1.000 | 15.806 | −0.945 | 2.578 |

**Table A6**
*Crop Yields, Prices, and Variable Costs of Production for the Baseline Year (2014)*

| Crop | Yield (tons/acre-yr) | Crop price ($/ton) | Other variable costs ($/acre) |
|---|---|---|---|
| Almonds | 1 | 7,331 | 2,095 |
| Alfalfa | 8 | 237 | 607 |
| Corn | 25 | 64 | 787 |
| Cotton | 2 | 1,364 | 969 |
| Melons | 19 | 330 | 1,308 |
| Wheat | 3 | 256 | 581 |
| Onions | 29 | 321 | 7,634 |
| Peaches and nectarines | 10 | 1,327 | 5,035 |
| Plums | 9 | 1,251 | 9,877 |
| Cherries | 4 | 5,063 | 13,866 |
| Pomegranates | 5 | 1,576 | 5,552 |
| Apples | 17 | 1,150 | 14,764 |
| Sorghum | 16 | 50 | 379 |
| Broccoli | 7 | 1,000 | 5,414 |
| Pasture | 5 | 192 | 239 |
| Tomatoes | 53 | 83 | 2,601 |
| Oranges | 15 | 516 | 5,119 |
| Olives | 4 | 1,074 | 3,328 |
| Grapes | 13 | 1,496 | 16,725 |

**Table A7**
*Subdistrict Share of Suitable Agricultural Land for Groundwater Recharge and Average Water Price and Salinity for the Baseline Year (2014)*

| Subdistrict name | Av. price ($/AF) | Av. salinity of applied water (dS/m) | Share of suitable land for Ag. recharge (SAGBI index) |
|---|---|---|---|
| Alta ID A | 46.87 | 0.26 | 39.73 |
| Alta ID B | 46.36 | 0.38 | 16.22 |
| Alta ID C | 47.62 | 0.28 | 36.47 |
| Consolidated ID A | 95.74 | 0.19 | 75.60 |
| Consolidated ID B | 74.05 | 0.35 | 71.24 |
| Kings River WD | 81.20 | 0.13 | 24.06 |
| Fresno ID A | 104.97 | 0.25 | 27.11 |
| Fresno ID B | 94.85 | 0.20 | 59.14 |
| Fresno ID C | 80.93 | 0.30 | 26.02 |
| Garfield WD | 96.54 | 0.11 | 46.95 |
| Groundwater Only E | 100.77 | 1.49 | 72.81 |
| Kings County WD A | 86.80 | 0.42 | 65.76 |
| Kings County WD B | 110.26 | 0.51 | 98.24 |
| James ID | 70.79 | 0.30 | 0.84 |
| Laguna ID | 66.64 | 0.62 | 17.72 |
| Murphy Slough Assoc | 92.18 | 0.33 | 9.04 |
| Riverdale ID | 78.84 | 1.23 | 0.19 |
| Stinson ID | 78.54 | 0.93 | 0.56 |
| Tranquility ID | 67.96 | 0.31 | 0.50 |
| Hills Valley ID | 21.25 | 0.68 | 24.58 |
| Orange Cove ID | 20.73 | 0.61 | 15.22 |
| Tri-Valley WD | 15.47 | 0.66 | 7.42 |
| Coelho Family Trust | 20.18 | 0.67 | 0.92 |
| Groundwater Only A | 27.55 | 0.49 | 25.33 |
| Groundwater Only B | 54.43 | 0.73 | 9.28 |
| Groundwater Only C | 26.01 | 0.67 | 8.10 |
| Groundwater Only D | 28.39 | 0.81 | 40.57 |
| Liberty WD | 44.97 | 0.81 | 47.55 |
| Mid-Valley WD | 34.03 | 0.73 | 13.47 |
| Raisin City ID | 60.35 | 1.38 | 33.78 |

*Note.* The shares of suitable land for agricultural recharge are calculated as the share of land in each subdistrict rated Moderately Good, Good, or Excellent according to the SAGBI Index https://casoilresource.lawr.ucdavis.edu/sagbi/.

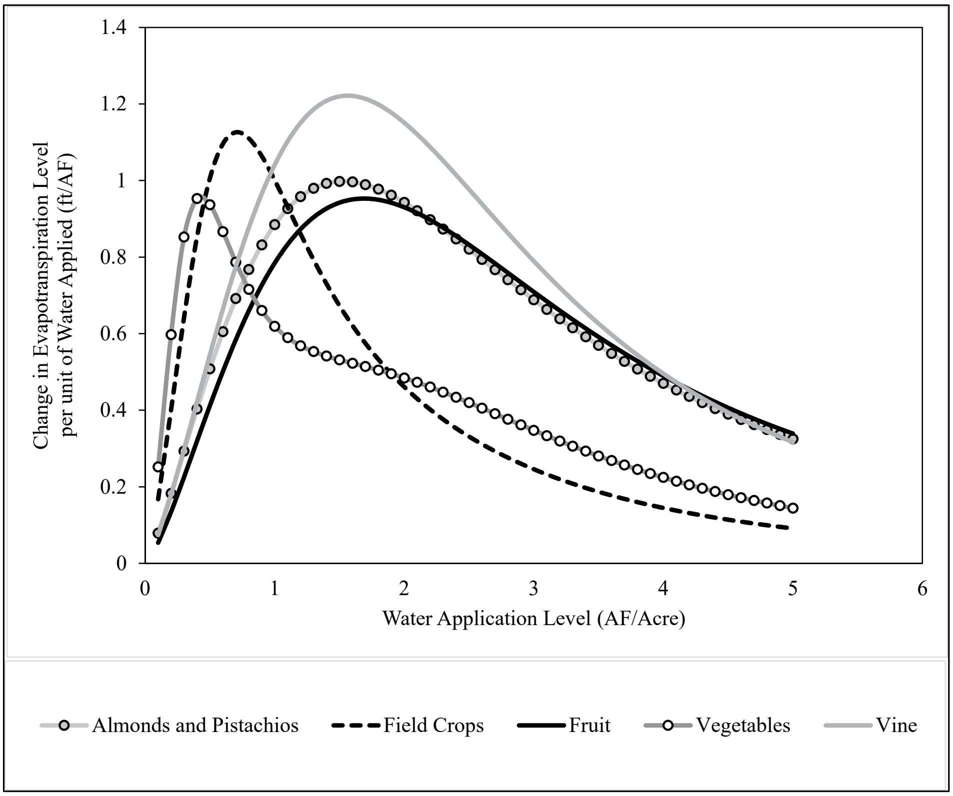


**Figure A1.** Changes in evapotranspiration levels per unit of water applied across crop categories.

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## Appendix B: Sensitivity Analysis Results

Figure B1 presents the impact of changing the assumption of travel time to groundwater on water management decisions in the model as well as the outcomes of groundwater dynamics under the *Social* scenario. Similar to Figure 4, Figures B2–B4 present water use decisions and outcomes of groundwater dynamics in the region for the *Sustainable* and *Credit* scenarios by climate scenario and separated to travel time length assumptions.

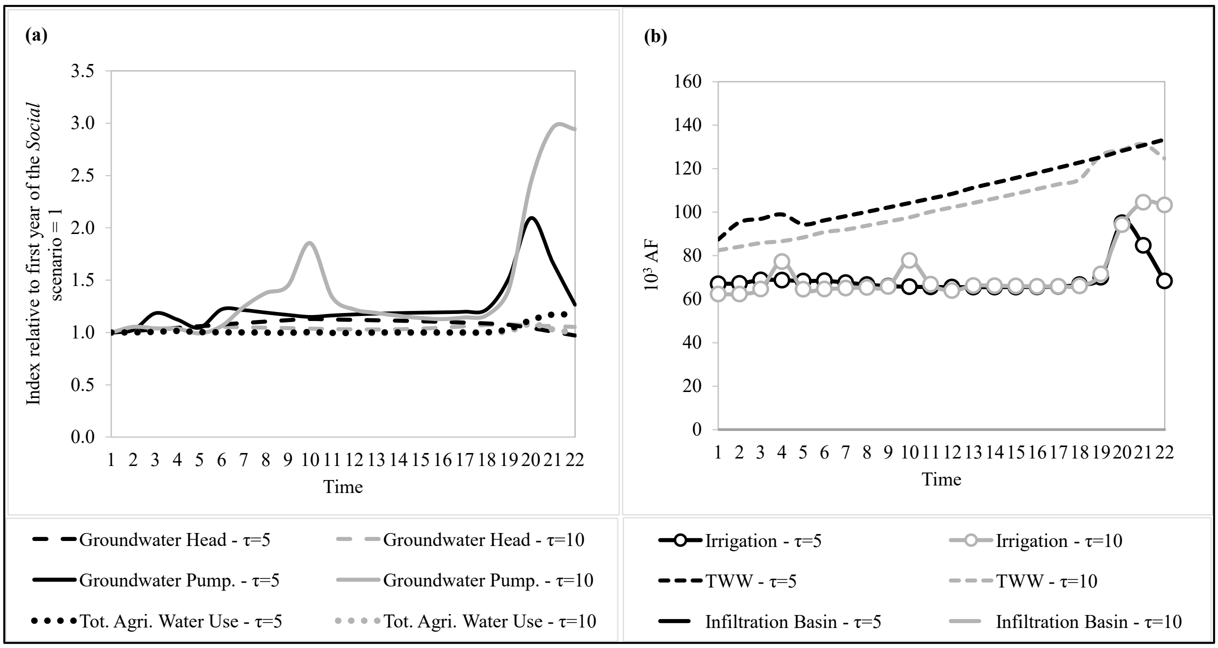


**Figure B1.** Water use and groundwater dynamics over time, under the *Social* scenario and the assumption that $\tau = 5$ or $\tau = 10$: (a) indices of total water use in agriculture, groundwater pumping, and groundwater head (first-year value of the corresponding index under the *Social* scenario = 1); (b) recharged quantities by source (TWW = treated wastewater).

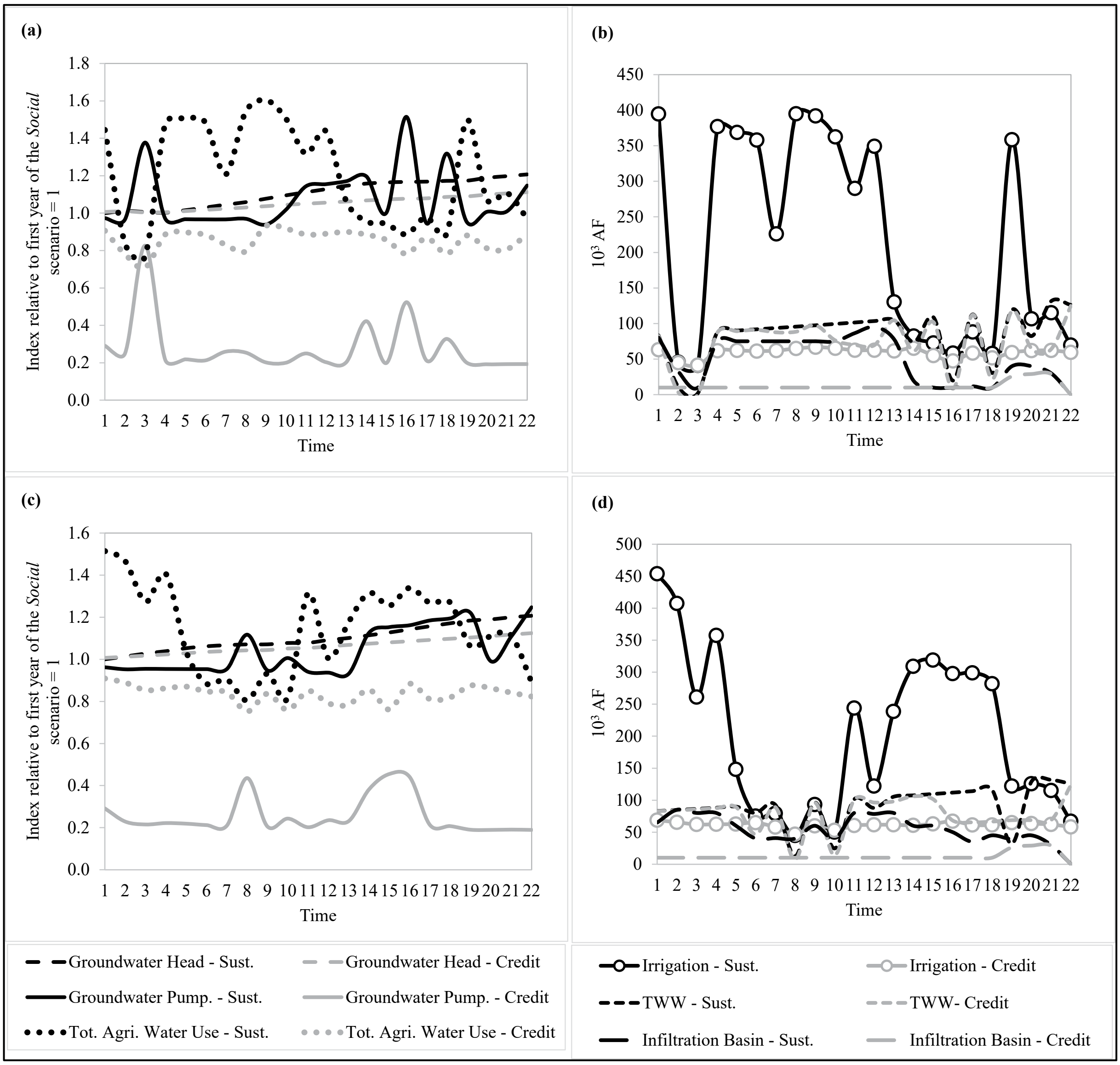


**Figure B2.** Water use and groundwater dynamics over time, under the *Sustainable* and *Credit* scenarios for the *Hist1* (panels (a and b)) and *Hist2* (panels (c and d)) climate scenarios under the assumption that $\tau = 1$: (a, c) indices of total water use in agriculture, groundwater pumping, and groundwater head (first-year value of the corresponding index under the *Social* scenario = 1); (b, d) recharged quantities by source (TWW, treated wastewater).

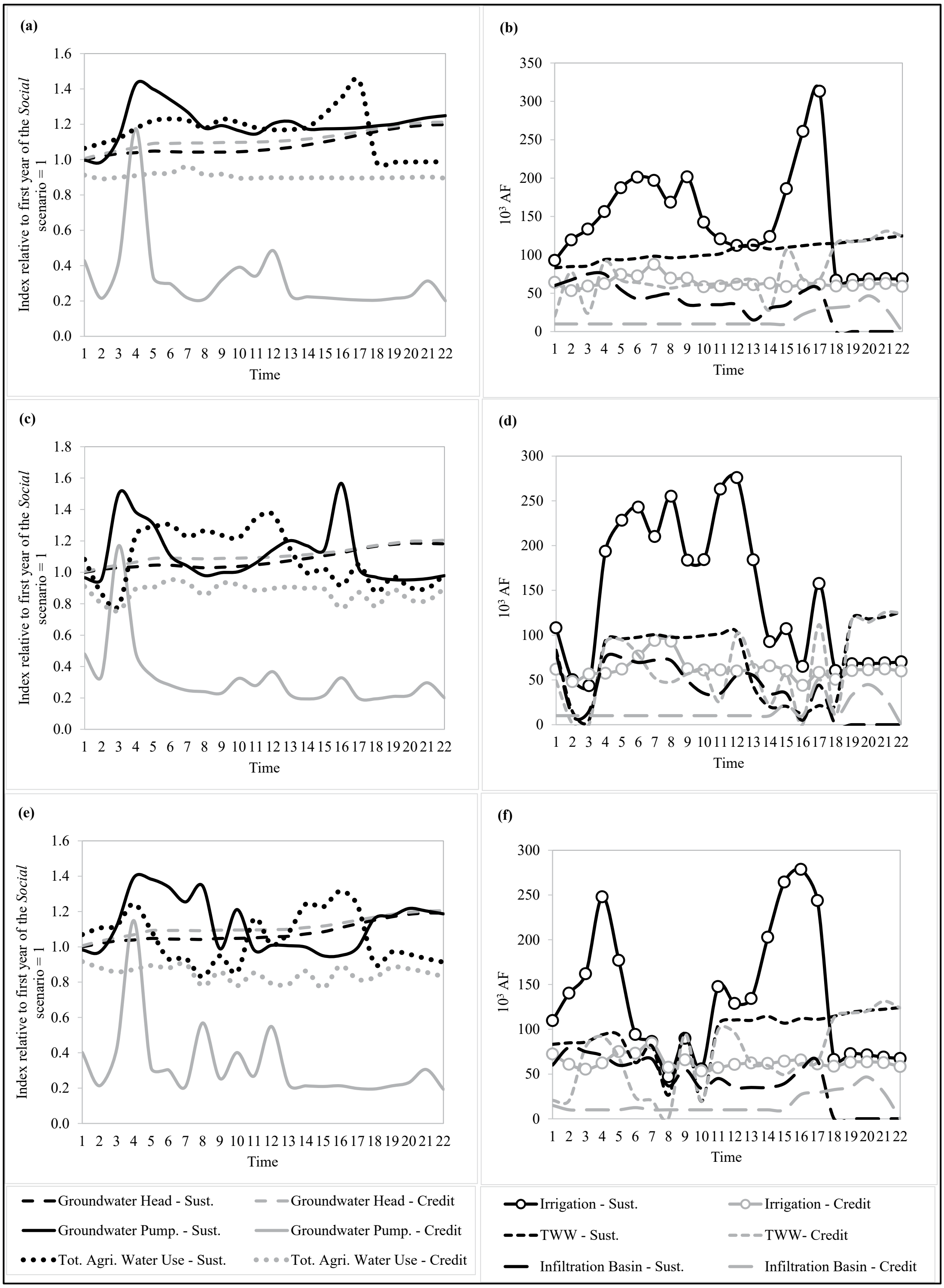


**Figure B3.** Water use and groundwater dynamics over time, under the *Sustainable* and *Credit* scenarios for the *Constant-Climate* (panels (a and b)), *Hist1* (panels (c and d)), and *Hist2* (panels (e and f)) climate scenarios under the assumption that $\tau = 5$: (a, c, e) indices of total water use in agriculture, groundwater pumping, and groundwater head (first-year value of the corresponding index under the *Social* scenario = 1); (b, d, f) Recharged quantities by source (TWW, treated wastewater).

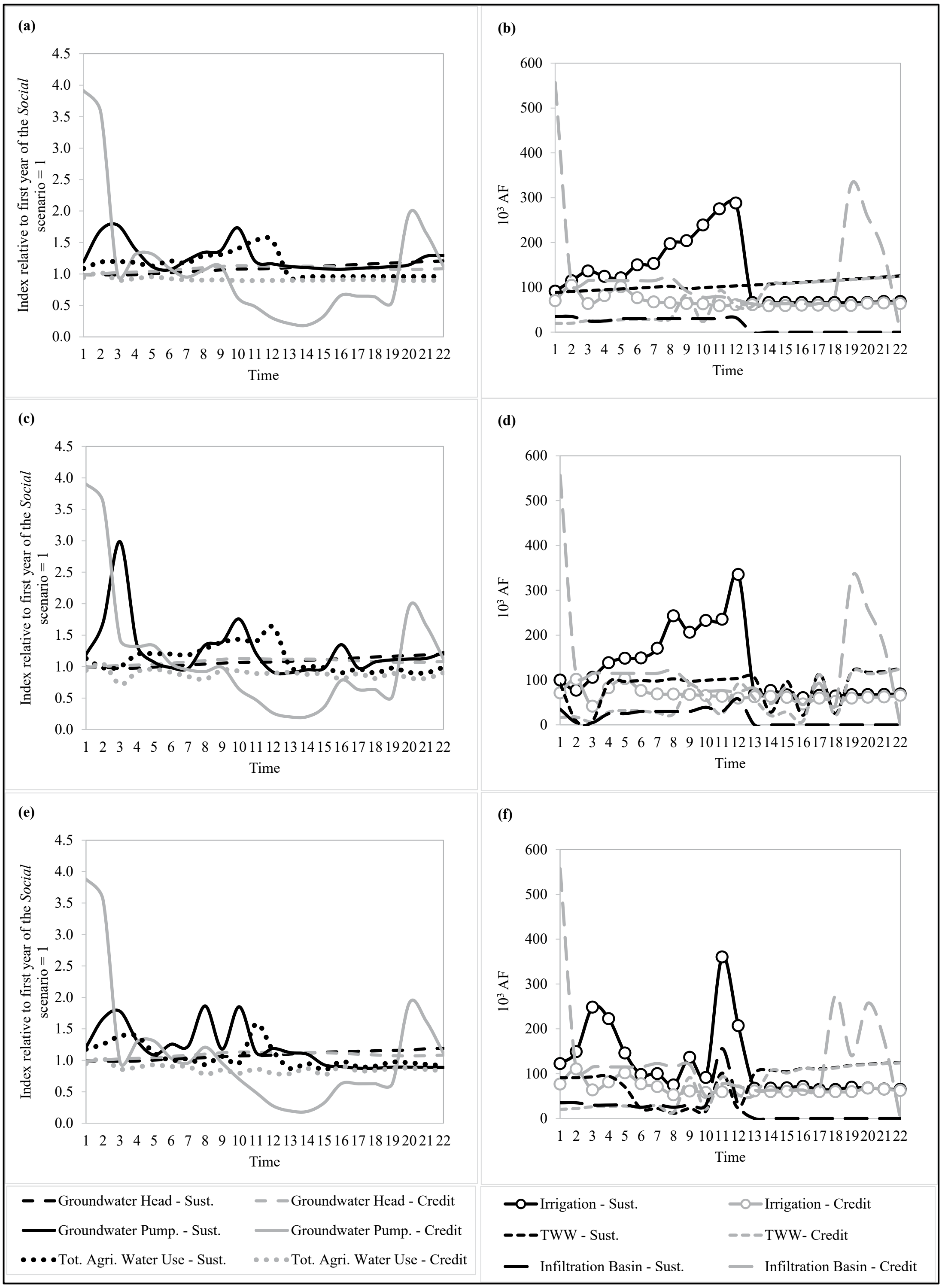


**Figure B4.** Water use and groundwater dynamics over time, under the *Sustainable* and *Credit* scenarios for the *Constant-Climate* (panels (a and b)), *Hist1* (panels (c and d)), and *Hist2* (panels (e and f)) climate scenarios under the assumption that $\tau = 10$: (a, c, e) indices of total water use in agriculture, groundwater pumping, and groundwater head (first-year value of the corresponding index under the *Social* scenario = 1); (b, d, f) recharged quantities by source (TWW, treated wastewater).

## Data Availability Statement

The empirical hydroeconomic model developed in this study (termed EOM) and the associated data set are available at Zenodo (https://doi.org/10.5281/zenodo.5518856).

**Acknowledgments**

We thank the Editor and three anonymous reviewers for their helpful comments. The research leading to this paper was funded by a cooperative agreement (58-6000-7-0093) between the USDA-ERS and the University of California, Riverside. The authors acknowledge the research and administrative leadership by the USDA-ERS Principal Investigator Steven Wallender. The authors would also like to extend our gratitude to Daniel Hirmas from the University of California, Riverside, for educating us on comparison methods of soil taxonomies. Ami Reznik would like to express his gratitude for the financial support of the Vaadia-BARD Postdoctoral Fellowship (No. FI-563-2017). Ariel Dinar would like to acknowledge support from the W4190 Multistate NIFA-USDA-funded project, "Management and Policy Challenges in a Water-Scarce World." and from the Giannini Foundation of Agricultural Economics Mini-grant Program.

## References

Alam, S., Gebremichael, M., Li, R., Dozier, J., & Lettenmaier, D. P. (2020). Can managed aquifer recharge mitigate the groundwater overdraft in California's Central Valley? *Water Resources Research*, *56*(8), e2020WR027244.

Booker, J. F., Howitt, R. E., Michelsen, A. M., & Young, R. A. (2012). Economics and the modeling of water resources and policies. *Natural Resource Modeling*, *25*(1), 168–218.

Burness, H. S., & Quirk, J. P. (1979). Appropriative water rights and the efficient allocation of resources. *The American Economic Review*, *69*(1), 25–37.

Burt, O. (1964). The economics of conjunctive use of ground and surface water. *Hilgardia*, *36*(2), 31–111.

California Department of Water Resources [CADWR]. (2006). *California's Groundwater Bulletin 118—Tulare Lake Hydrologic Region, San Joaquin Valley Groundwater Basin, Kings Subbasin*. Retrieved 11 March 2019 from https://water.ca.gov/-/media/DWR-Website/Web-Pages/Programs/Groundwater-Management/Bulletin-118/Files/2003-Basin-Descriptions/5_022_08_KingsSubbasin.pdf

California Department of Water Resources [CADWR]. (n.d.). *Statewide Crop Mapping 2014, Land Use Viewer*. Retrieved from 11 March 2019 https://gis.water.ca.gov/app/CADWRLandUseViewer/

California Department of Water Resources [CADWR]. (n.d.). *Sustainable Groundwater Management Act (SGMA) Portal*. Retrieved from 22 November 2021 https://sgma.water.ca.gov/portal/#intro

California Department of Water Resources [CADWR]. (n.d.). *Water Data Library*. Retrieved from https://wdl.water.ca.gov/Map.aspx. Accessed on March 11, 2019.

City of Fresno. (2013). *USBR Water Management Plan*. Retrieved 11 March 2019 from https://www.usbr.gov/mp/watershare/docs/2013/city-fresno.pdf

Connor, J. D., Schwabe, K., King, D., & Knapp, K. C. (2012). Irrigated agriculture and climate change: The influence of water supply variability and salinity on adaptation. *Ecological Economics*, *77*, 149–157.

Dahlke, H. E., Brown, A. G., Orloff, S., Putnam, D., & O'Geen, T. (2018). Managed winter flooding of alfalfa recharges groundwater with minimal crop damage. *California Agriculture*, *72*(1), 65–75.

Dahlke, H. E., LaHue, G. T., Mautner, M. R., Murphy, N. P., Patterson, N. K., Waterhouse, H., et al. (2018). Managed aquifer recharge as a tool to enhance sustainable groundwater management in California: Examples from field and modeling studies. In *Advances in chemical pollution, environmental management and protection* (Vol. 3, pp. 215–275). Elsevier.

Dan, Y., Yaalon, D., Koymdjisky, H., & Raz, Z. (1975). *The soils of Israel. Bulletin 168 of the Volcanic Institute of Agricultural Research, Beit Dagan, Israel 28p, in Hebrew, with 1:500,000 soil map of Israel in Hebrew*.

Dillon, P., Stuyfzand, T., Grischek, M., Lluria, R. D. G., Pyne, R. C., Jain, J., et al. (2019). Sixty years of global progress in managed aquifer recharge. *Hydrogeology Journal*, *27*, 1–30.

Dogan, M. S., Buck, I., Medellin-Azuara, J., & Lund, J. R. (2019). Statewide effects of ending long-term groundwater overdraft in California. *Journal of Water Resources Planning and Management*, *145*(9), 04019035.

Draper, A. J., Jenkins, M. W., Kirby, K. W., Lund, J. R., & Howitt, R. E. (2003). Economic-engineering optimization for California water management. *Journal of Water Resources Planning and Management*, *129*(3), 155–164.

Dudley, N. J., & Musgrave, W. F. (1988). Capacity sharing of water reservoirs. *Water Resources Research*, *24*(5), 649–658.

Feinerman, E., & Tsur, Y. (2014). Perennial crops under stochastic water supply. *Agricultural Economics*, *45*(6), 757–766.

Feinerman, E., & Yaron, D. (1983). Economics of irrigation water mixing within a farm framework. *Water Resources Research*, *19*(2), 337–345.

Forni, L. G., Medellín-Azuara, J., Tansey, M., Young, C., Purkey, D., & Howitt, R. (2016). Integrating complex economic and hydrologic planning models: An application for drought under climate change analysis. *Water Resources and Economics*, *16*, 15–27.

Fraas, A. G., & Munley, V. G. (1984). Municipal wastewater treatment cost. *Journal of Environmental Economics and Management*, *11*(l), 28–33.

Franklin, B., Knapp, K. C., & Schwabe, K. A. (2017). A dynamic regional model of irrigated perennial crop production. *Water Economics and Policy*, *3*(01), 1650036.

Hanak, E., Escriva-Bou, A., Gray, B., Green, D., Harter, T., Jezdimirovic, J., et al. (2019). *Water and the future of the San Joaquin Valley*. Public Policy Institute of California.

Hanak, E., Escriva-Bou, A., Mount, J., Pottinger, L., Ayres, A., Bardeen, S., et al. (2021). *Priorities for California's water: Responding to the changing climate*. Public Policy Institute of California.

Hanak, E., Jezdimirovic, J., Green, S., & Escriva-Bou, A. (2018). *Replenishing groundwater in the San Joaquin Valley*. Public Policy Institute of California.

Harou, J. J., & Lund, J. R. (2008). Ending groundwater overdraft in hydrologic-economic systems. *Hydrogeology Journal*, *16*(6), 1039–1055.

Harou, J. J., Pulido-Velazquez, M., Rosenberg, D. E., Medellín-Azuara, J., Lund, J. R., & Howitt, R. E. (2009). Hydroeconomic models: Concepts, design, applications, and future prospects. *Journal of Hydrology*, *375*(3–4), 627–643.

Hirmas, D. (2019). *Personal communication regarding rules for soil classification* (Vol. 14). Department of Environmental Sciences, University of California.

Howitt, R. E. (1995). Positive mathematical programming. *American Journal of Agricultural Economics*, *77*(2), 329–342.

Howitt, R. E., Medellín-Azuara, J., MacEwan, D., & Lund, J. R. (2012). Calibrating disaggregate economic models of agricultural production and water management. *Environmental Modeling & Software*, *38*, 244–258.

International Arid Land Consortium [IALC]. (n.d.). *Correlation List for the Soils of Israel*. Retrieved 21 March 2019 from https://cals.arizona.edu/OALS/soils/israel/correlation.html

Kan, I., & Rapaport-Rom, M. (2012). Regional blending of fresh and saline irrigation water: Is it efficient? *Water Resources Research*, *48*, W07517. https://doi.org/10.1029/2011WR011285

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

Kings River Watershed Coalition Authority [KRWCA]. (2014). *Groundwater Assessment Report*. Retrieved 21 March 2019 from https://www.waterboards.ca.gov/centralvalley/water_issues/irrigated_lands/water_quality/coalitions_submittals/kings_river/ground_water/2014_1120_kings_gar.pdf
Knapp, K. C. (1992). Irrigation management and investment under saline, limited drainage conditions: 1. Model formulation. Water Resources Research, 28(12), 3085–3090.
Knapp, K. C., & Olson, L. J. (1995). The economics of conjunctive groundwater management with stochastic surface supplies. *Journal of Environmental Economics and Management*, *28*(3), 340–356.
Koundouri, P. (2004). Current issues in the economics of groundwater resource management. *Journal of Economic Surveys*, *18*(5), 703–740.
Letey, J., & Dinar, A. (1986). Simulated crop water production functions for several crops when irrigated with saline waters. *Hilgardia*, *54*, 1–32.
MacEwan, D., Cayar, M., Taghavi, A., Mitchell, D., Hatchett, S., & Howitt, R. (2017). Hydroeconomic modeling of sustainable groundwater management. *Water Resources Research*, *53*(3), 2384–2403.
Maliva, R. (2014). Economics of managed aquifer recharge. *Water*, *6*, 1257–1279.
Natural Resources Conservation Service, United States Department of Agriculture [NRCS-USDA]. (n.d.). *Web Soil Survey*. Retrieved 21 March 2019 from http://websoilsurvey.sc.egov.usda.gov/
North, D. (1990). *Institutions, institutional change and economic performance (political economy of institutions and decisions)*. Cambridge University Press. http://doi.org/10.1017/CBO9780511808678
O'Geen, A., Saal, M., Dahlke, H., Doll, D., Elkins, R., Fulton, A., et al. (2015). Soil suitability index identifies potential areas for groundwater banking on agricultural lands. *California Agriculture*, *69*(2), 75–84. https://doi.org/10.3733/ca.v069n02p75
Perrone, D., & Rohde, M. M. (2016). Benefits and economic costs of managed aquifer recharge in California. *San Francisco Estuary and Watershed Science*, *14*(2), 1–13.
Reznik, A., Dinar, A., Bresney, S., Forni, L., Joyce, B., Wallander, S., et al. (2020). *Managed aquifer recharge as a strategy to mitigate drought impacts in irrigated agriculture: Role of institutions and policies with application to California. University of California, Riverside, School of Public Policy, Working Paper Series WP#20-04: Riverside, CA, USA*.
Saleth, R. M., & Dinar, A. (2004). *The institutional economics of water: A cross-country analysis of institutions and performance*. World Bank Publications and Cheltenham, UK: Edward Elgar Publishers.
Scanlon, B. R., Faunt, C. C., Longuevergne, L., Reedy, R. C., Alley, W. M., McGuire, V. L., & McMahon, P. B. (2012). Groundwater depletion and sustainability of irrigation in the U.S. high plains and Central Valley. *Proceedings of the National Academy of Sciences*, *109*(24), 9320–9325.
Scanlon, B. R., Reedy, R. C., Faunt, C. C., Pool, D., & Uhlman, K. (2016). Enhancing drought resilience with conjunctive use and managed aquifer recharge in California and Arizona. *Environmental Research Letters*, *11*(3), 035013.
Schwabe, K. A., Kan, I., & Knapp, K. C. (2006). Drainwater management for salinity mitigation in irrigated agriculture. *American Journal of Agricultural Economics*, *88*(1), 133–149.
Slater, Y., Finkelshtain, I., Reznik, A., & Kan, I. (2020). Large-scale desalination and the external impact on irrigation water salinity: Economic analysis for the case of Israel. *Water Resources Research*, *56*(9), e2019WR025657. https://doi.org/10.1029/2019WR025657
State Water Resources Control Board [SWRCB]. (n.d.). *Groundwater Ambient Monitoring and Assessment Program (GAMA)*. Retrieved 18 February 2018 from https://www.waterboards.ca.gov/water_issues/programs/gama/
Thomas, G. A. (2001). *Designing successful groundwater banking programs in the Central Valley: Lessons from experience*. Natural Heritage Institute.
Tran, D. Q., Kovacs, K., & Wallander, S. (2020). Water conservation with managed aquifer recharge under increased drought risk. *Environmental Management*, *66*(4), 664–682. https://doi.org/10.1007/s00267-020-01329-x
Tran, D. Q., Kovacs, K. F., & Wallander, S. (2019). Long-run optimization of landscape-level irrigation through managed aquifer recharge or expanded surface reservoirs. *Journal of Hydrology*, *579*, 124220.
Tran, D. Q., Kovacs, K. F., & West, G. H. (2020). Spatial economic predictions of managed aquifer recharge for an agricultural landscape. *Agricultural Water Management*, *241*, 106337.
Tsur, Y., & Graham-Tomasi, T. (1991). The buffer value of groundwater with stochastic surface water supplies. *Journal of Environmental Economics and Management*, *21*(3), 201–224.
Ulibarri, N., Escobedo Garcia, N., Nelson, R. L., Cravens, A. E., & McCarty, R. J. (2021). Assessing the feasibility of managed aquifer recharge in California. *Water Resources Research*, *57*(3), e2020WR029292. https://doi.org/10.1029/2020WR029292
University of California Cooperative Extension [UCCE]. (n.d.). *Cost and return studies*. Retrieved 21 March 2019 from https://coststudies.ucdavis.edu/en/
Vanderzalm, J. L., Dillon, P. J., Tapsuwan, S., Pickering, P., Arold, N., Bekele, E. B., et al. (2015). *Economics and experiences of managed aquifer recharge (MAR) with recycled water in Australia*. Australian Water Recycling Centre of Excellence.
Water Resources and Information Management Inc. [WRIME]. (2006). *Analysis of Water Supplies in Kings Basin—Tenical Memorandum Phase 1, Task 4*. Retrieved 18 February 2018 from http://www.kingsbasinauthority.org/_documents/reports_papers/Analysis_of_Water_Supplies_in_Kings_Basin_(Phase_1_Task4)_May_2006.pdf

## References From the Supporting Information

California Department of Water Resources [CADWR]. (2016). *CA Bulletin 118 Groundwater Basins*. Retrieved from https://sgma.water.ca.gov/webgis/?appid=SGMADataViewer#boundaries
Joyce, B. A., Mehta, V. K., Purkey, D. R., Dale, L. L., & Hanemann, M. (2011). Modifying agricultural water management to adapt to climate change in California's Central Valley. *Climatic Change*, *109*(S1), 299–316.
Sato, T., Qadir, M., Yamamotoe, S., Endoe, T., & Zahoor, A. (2013). Global, regional, and country level need for data on wastewater generation, treatment, and use. *Agricultural Water Management*, *130*, 1–13.
Yates, D., Sieber, J., Purkey, D., & Huber-Lee, A. (2005). WEAP21—A demand-, priority-, and preference-driven water planning model. *Water International*, *30*(4), 487–500.