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Sustainability of Regional Groundwater Quality in Response to Managed Aquifer Recharge

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Key Points:

- Transport of total dissolved solids (TDS) sourced from irrigation illuminates feasibility of mitigating regional groundwater quality deterioration via long-term managed aquifer recharge (MAR)
- MAR operations that exploit the geologic heterogeneity can reap significant, long-term benefits for regional groundwater quality
- This work provides a guidance and blueprint for sustainable, joint management of groundwater quantity and quality at a basin scale

Supporting Information:

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

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Abstract Growing demands on water supply worldwide have resulted in aquifer overdraft in many regions, especially in alluvial basins under intensive irrigation. This further leads to serious deterioration of groundwater quality. Managed aquifer recharge (MAR) has been shown to mitigate groundwater overdraft, but whether MAR can actually stabilize or reverse the ongoing declines in regional groundwater quality caused by non-point sources has not been demonstrated. This study was intended to address the question by investigating impacts of different MAR strategies on regional groundwater quality. A geostatistical model was first used to characterize a heterogeneous alluvial aquifer system in a portion of the Tulare Lake Basin. Three-dimensional numerical models were then employed to simulate groundwater flow and mass transport. Next, MAR strategies were applied in locations with different geological conditions or joint with different irrigation activities, and their performances were evaluated. Results demonstrate the potential of significant, long-term benefits for regional groundwater quality by applying strategic, high-intensity recharge operations on geologically favorable subregions. Siting MAR above the incised valley fill (IVF) deposit, a near-surface paleochannel containing unusually coarse, high-conductivity hydrofacies, leads to more extensive improvement in the groundwater quality in terms of salinity due to significant vertical flow and lateral outward flow from the IVF. Overall, decades would be required to alleviate groundwater quality concerns in the studied 189 km^2 region. The simulations indicate that the deep concentrations remain below the secondary maximum contaminant level as the solute mass migrates downward with the prominent contribution from the attenuation via dispersion and matrix diffusion.

1. Introduction

Population growth and the expansion of agriculture, coupled with climate uncertainties and unsustainable surface water storage, have accelerated groundwater extraction. In the U.S.A., 44% of the population relies on groundwater and approximately 60% of irrigation water comes from groundwater (Alley et al., 1999). In locations where the rate of groundwater pumping exceeds the rate of natural replenishment, groundwater level declines have been observed. Groundwater overdraft is observed worldwide in most alluvial basins that are intensely irrigated (Gleeson et al., 2012; Russo & Lall, 2017; Scanlon et al., 2012; Wada et al., 2014). Scanlon et al. (2012) studied several irrigated regions in the U.S.A., including the High Plains and California's Central Valley (CV). They estimate that at the current depletion rate, aquifer lifespans in these regions are as short as 140 years, and in some sub-regions, it will be impossible to support irrigation after several decades. Coupled with extreme climate conditions, such as the severe California drought (2012–2016) and prolonged future droughts under a warming climate, unsustainable surface water reliability and increased evapotranspiration will likely lead to more groundwater pumping and acceleration of aquifer depletion.

Groundwater pollution caused by non-point source contaminants from return flow, such as salts and nitrate, is a major concern in irrigated agricultural basins (e.g., Ayub et al., 2016; Fogg & LaBolle, 2006; Fogg et al., 1999; Han et al., 2016; Harter et al., 1998, 2002; He & Croley, 2008; Laitos & Ruckriegle, 2013; Pauloo, Fogg, et al., 2020; Scanlon et al., 2010; Weissmann et al., 2002; Xu, 2014). Contaminants leach through the root zone to the water table at greater concentration than in the irrigation water, as a large portion of pumped groundwater is evapo-concentrated due to crop evapotranspiration. Consequently, increased groundwater pumping may accelerate groundwater salinization in irrigated regions (Pauloo, Fogg, et al., 2020).

Managed aquifer recharge (MAR) is commonly used to replenish depleted aquifers. The success of this approach in mitigating overdraft, increasing groundwater levels and groundwater storage has been reported in many studies (e.g., Banerjee & Singh, 2011; Barnett et al., 2000; Kendy & Bredehoeft, 2006; Konikow & Kendy, 2005; Muniz & Ziegler, 1994; Stakelbeek et al., 1996). Castaldo et al. (2021) recently showed a correlation between clean sources of recharge and lower groundwater nitrate in California's CV. Sources for MAR include reclaimed water with proper treatment, desalinated seawater, river water, rainwater or imported groundwater (NGWA, 2017; Sun et al., 2020). The quality of water from these sources is usually high and is anticipated to improve groundwater quality after recharge. However, few studies investigate the impacts of MAR on regional scale groundwater quality, especially deep aquifers, and at the time scales of decades to centuries.

Fogg and LaBolle (2006) and Boyle et al. (2012) hypothesized that while the non-point source contamination from irrigation activities appears to be degrading broad regions of groundwater quality, a way to possibly reverse this trend is to add sources of recharge that are relatively fresh compared to irrigation water, while also reducing source concentrations as much as possible. Importantly, in many basins the process of irrigation leads to irrigation water becoming the dominant source of recharge, and work suggests that strategies focused solely on source control (i.e., reducing the contaminant concentrations or fertilizer inputs in the applied irrigation water leaching through the root zone), will not sufficiently mitigate the problem (Bastani & Harter, 2019; Boyle et al., 2012; Fogg et al., 1999; Harter et al., 2012; Levy et al., 2017). The above hypothesis, which to our knowledge has not been investigated, raises important questions concerning the feasibility of stabilizing groundwater quality in irrigated basins through MAR with water of better quality than the irrigation leachate. These questions include: Given plausible locations and amounts of recharge, what are the space and time scales over which groundwater quality is likely to change? Further, given that many of the irrigated lands are on top of sedimentary basins having horizontally stratified aquifer (sands and gravels) and aquitards (silts and clays) that form multi-story, semi-confined aquifer complexes, are the groundwater quality consequences of MAR likely to greatly vary spatially? In particular, are there locations or strategic geologic features at which the water quality benefits would be greatly improved, as was suggested by Weissmann et al. (2004) and Maples et al. (2019, 2020) who showed that relatively coarse-grained, incised-valley-fill (incised valley fill (IVF)) deposits (Meirovitz et al., 2017; Weissmann et al., 2004) produced by the most recent alpine glaciation event in the California CV sedimentary basin are capable of supporting much higher rates of recharge. Furthermore, Zhang et al. (2018) showed that the IVFs can also support much higher rates of pollutant transport. In turn, the IVF deposits may be key for effecting local and regional groundwater quality via MAR.

A corollary question to the issue of MAR influence on groundwater quality concerns the issue of contaminant remediation: Is it feasible to remediate the degraded groundwater quality at the regional scale? Unlike point-source groundwater contamination problems, which have received much attention from scientists and engineers attempting to cleanup or remediate contaminant plumes (e.g., NRC, 2013; Brusseau & Guo, 2014; Guo & Brusseau, 2017; Liu et al., 2020; O'Connor et al., 2018), non-point source groundwater contamination is typically regarded as virtually irreversible because of the immense scale of the myriad, coalesced plumes emanating from nearly the entire landscape (e.g., Harter et al., 2012; Kelsey et al., 2018; Lockhart et al., 2013). Nevertheless, given that MAR with relatively fresh water is increasingly necessary for addressing groundwater quantity sustainability, it is logical to investigate whether MAR can be implemented in ways that actually stabilize or reverse the ongoing declines in regional groundwater quality caused by non-point sources. Herein we examine not only the regional water quality impacts of MAR in an irrigated groundwater basin, but also the relative benefits of focusing MAR on the most permeable and interconnected portions of an alluvial multi-aquifer system.

In this work, we study MAR in a highly heterogeneous alluvial fan that includes the IVF deposit, investigating the time scales under which continuous MAR leads to water quality improvement. Changes in total dissolved solids (TDS), herein treated as a conservative solute, are used to represent the regional groundwater quality variation. This study is the first providing insight into the timescales under which MAR can change groundwater quality at regional scales and the potential benefits of siting MAR in geologically strategic locations. The results of this work provide guidance for the strategic application of MAR projects and have implications for joint groundwater quantity and quality management.

2. Methods

2.1. Study Area

The Kings River Alluvial Fan, located southeast of Fresno, California was selected as the study area because of intensive irrigation activity in the region (Figure 1) and detailed, 3D mapping and modeling of the system

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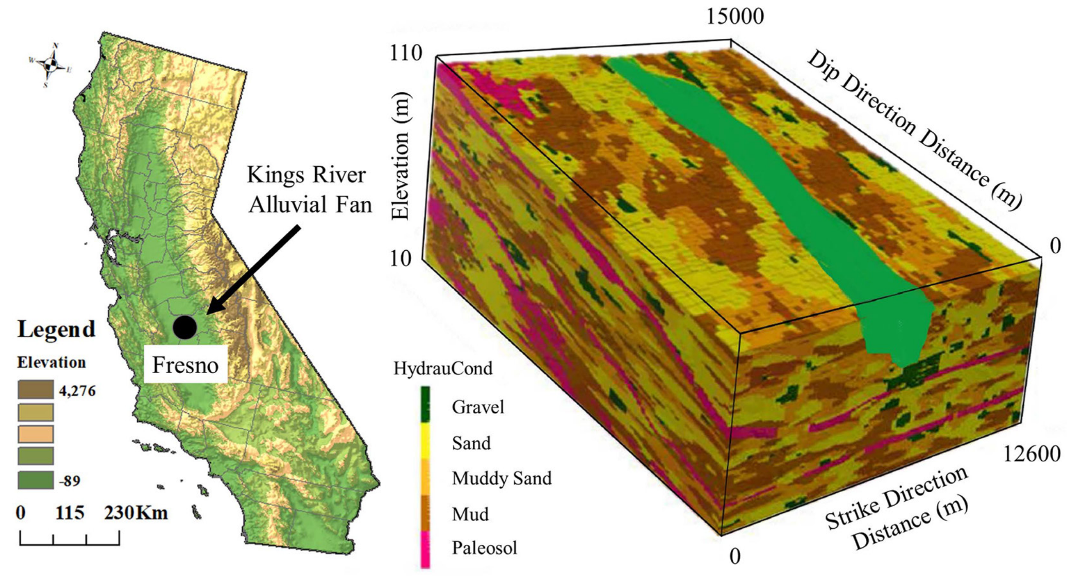

Figure 1. Location of Kings River Fan and the heterogeneity of the study site. The heterogeneous field was generated by the Markov chain transition probability model using five hydrofacies, gravel, sand, muddy sand, mud and paleosol. The incised valley fill deposit is a near-surface paleochannel containing unusually coarse, high-conductivity hydrofacies.

heterogeneity as well as flow and transport (Weissmann & Fogg, 1999; Weissmann et al., 1999, 2002, 2004; Zhang et al., 2018). The study area has dimensions of 12.6 km by 15 km. Groundwater quality has been degraded extensively by nitrate and TDS as reported by previous studies in the area (Hansen et al., 2018; Harter et al., 2012; Lindsey & Johnson, 2018). The investigated part of the aquifer system was deposited by a fluvially-dominated alluvial fan with a highly heterogeneous mix of unconsolidated sediments, and a prominent, shallow incised-valley-fill deposit IVF of high conductivity interconnected coarse sediment (Weissmann & Fogg, 1999; Weissmann et al., 1999). The IVF was produced by the most recent glacial cycle in the Sierra Nevada, the source area for the fan, and is particularly important because it creates relatively permeable pathways for vertical flow and recharge in an aquifer system that is otherwise dominated by fines. As is typical of CV sediments and many other sedimentary basins, the study area consists of over 60% fines. As a result, the IVF is likely to be critically important for effecting significant recharge from the surface. Five hydrofacies were recognized: gravel, sand, muddy sand, mud, and paleosol (Weissmann & Fogg, 1999; Weissmann et al., 1999) based on a high-quality soil survey by the U.S. Department of Agriculture (Huntington, 1971) and core data collected by the U.S. Geological Survey (USGS) (Burow et al., 1997; Harter et al., 1998). Moreover, geomorphic and stratigraphic studies identified five distinct depositional sequences bounded by laterally continuous paleosol hydrofacies with relatively low hydraulic conductivity (K) (Weissmann & Fogg, 1999; Weissmann et al., 1999, 2002). The regional groundwater flow within the fan is from west to southwest with a horizontal head gradient of about 0.002. Groundwater development in the region has significantly increased since the 1940s due to intensive irrigation and domestic use (Faunt, 2009).

2.2. Geostatistical Simulation of Heterogeneity

The FORTRAN program T-PROGS, which is based on transition probability–Markov chain random-field approach described by (Carle, 1997; Carle and Fogg, 1996, 1997), was used to generate the heterogeneous domain (Weissmann & Fogg, 1999; Weissmann et al., 1999; Zhang et al., 2018). Transition probabilities between identified hydrofacies are measured and fitted to a Markov chain model for each depositional direction (strike, dip and vertical), which incorporates geologic information, including proportions, mean length, and juxtaposition tendency for each hydrofacies. The 3D realizations of random fields were then generated by the Markov chain model using a sequential indicator simulation and annealing algorithm. The subsurface data include 11 cores, 132 drillers' logs, and soil survey data. The model was hard conditioned by the core and drillers' logs. Each stratigraphic sequence was modeled separately to deal with nonstationarity in a sequence stratigraphic framework

Table 1
Hydraulic Conductivity and Porosity Values Assigned to Five Hydrofacies

Hydrofacies	K (m/s)	Porosity
Gravel	1×10^{-2}	0.25
Sand	1×10^{-3}	0.3
Muddy Sand	1×10^{-5}	0.35
Mud	1×10^{-6}	0.4
Paleosol	1.3×10^{-7}	0.4

Note. "Mud" represents the fines and refers to silt and clay undifferentiated.

(Weissmann & Fogg, 1999; Zhang et al., 2018). The geostatistical conditional realizations were generated on a grid of 63 (strike) × 75 (dip) × 201 (vertical) nodes with 200.0 × 200.0 × 0.5 m spacing.

The fundamental hydraulic characteristics of the geostatistical model of heterogeneity, including K values and connectivity of the hydrofacies, were validated by Weissmann et al. (2002), who modeled groundwater ages in the system that agreed closely with measured chlorofluorocarbon apparent ages. Thus, as a sub-regional scale model of alluvial heterogeneity in a multi-aquifer system, this model is highly representative of the 3D flow and transport system. Because of the small differences observed among the resulting groundwater age distributions for 10 realizations in Weissmann et al. (2002) and the nonpoint source that can avoid the ergodicity issue used in this work (Guo, Fogg, & Henri, 2019, Guo, Fogg, Brusseau, et al., 2019), only one realization was adopted here that can fulfill the purpose to evaluate the MAR effects on groundwater quality. Because the 3D connectivity of the high-K facies varies little among the realizations, and because the important IVF deposit is conditioned with hard data into each realization, the regional transport behavior differs little among the realizations, making appropriate our use of just one realization. Consequently, rather than providing an ensemble of possible heterogeneous fields, the role of the stochastic model in this case is to provide a representative model of the heterogeneity and its regional effects on transport. Prior work on the heterogeneous model development (Weissmann & Fogg, 1999; Weissmann et al., 1999, 2004; Zhang et al., 2018) and its ability to reproduce measured groundwater mean ages in transport simulations (Weissmann et al., 2002), make this an extraordinarily reliable model of heterogeneity for exploring regional transport phenomena.

2.3. Groundwater Flow and Solute Transport Model

The flow model used in the study was MODFLOW, a 3D numerical (finite-difference) groundwater flow model (Harbaugh et al., 2000; McDonald & Harbaugh, 1988) using the same grid as the geostatistical model. An unconfined system was simulated. General head boundaries (GHB) were used along the front, back, left and right faces of the model, with a natural gradient (0.002) inducing lateral groundwater flow from northeast to southwest. Flux-weighted recharge was applied in each cell at the top layer, which was pre-determined based on the measured recharge amount in the region and scaled by the relative flux of each cell. The relative flux was calculated based on a vertical flow simulation, which applies the constant head boundaries at top and bottom layers and no flow for the rest faces. This accounts for non-uniform vertical flow caused by the heterogeneous distribution and connectivity of hydrofacies. The depths of supply wells for irrigation in this region are mostly deeper than 150 m (e.g., Faunt, 2009; Planert & Williams, 1995), which is deeper than the 100.5 m thickness of the simulated domain, therefore a GHB was used for the bottom boundary to represent water extraction in the deeper portion of the aquifer system. A vertical gradient of 0.02 was induced with recharge at the top boundary and GHB at the bottom (Pauloo et al., 2021). Spatially variable hydraulic conductivities (K) and porosities (Table 1) are assigned to individual cells for each hydrofacies that were determined by analyzing information generated from USGS core samples, slug tests and pumping tests for the site (Burow et al., 1997; Weissmann et al., 1999; Zhang et al., 2013).

The model represents the upper 100 m of the much thicker (~700 m) groundwater system because this is in effect a critical zone controlling recharge and groundwater quality in a system where the dominant groundwater flow direction has a strong downward component, driven by irrigation recharge at the surface and relatively high rates of pumping at depth (Pauloo, Fogg, et al., 2020, 2021). Moreover, the upper 100 m serves as an important groundwater source for numerous of smaller capacity wells including for domestic drinking water supplies (e.g., Pauloo, Escriva-Bou, et al., 2020), even though pumping from those smaller wells is not a significant portion of the overall groundwater budget. The entire system is leaky-confined with pervasive confining beds resulting in effective vertical hydraulic conductivities that are 100-to-10,000 times lower than effective horizontal hydraulic conductivities Pauloo, Escriva-Bou, et al. (2020). Accordingly, the deeper pumping creates a regionally augmented, downward hydraulic gradient that can be simulated with a general-head boundary condition at the bottom boundary of the model, as described above.

The 3D solute transport model MT3DMS (Zheng & Wang, 1999) was used to simulate solute transport via solution of the transient advection-dispersion equation (ADE). The third-order total-variation-diminishing scheme

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was used to minimize numerical dispersion and artificial oscillation. Longitudinal, transverse, and vertical dispersivities representing grid-scale dispersion were set to 20, 5, and 0.05 m, respectively. Experience has shown that in these aquifer-aquitard complexes having inherently high variance of K, most of the dispersion is imparted by the geostatistically modeled heterogeneity and that the local-scale dispersivities used in the ADE have a relatively minor effect (e.g., LaBolle & Fogg, 2001). The aqueous diffusion coefficient was 10^{-5} m^2/d according to the measurements from Hollins et al. (2004) for salts.

Because we are modeling transport of the TDS, we assumed reactions would have a minor impact on TDS concentration and hence were not modeled. We acknowledge that several of the main constituents of TDS are non-conservative, such as calcium, bicarbonate and sodium which vary as a function of the carbonate equilibria and cation exchange, among other reactions. While it would be valuable in future studies to add reactive transport to such simulations, in this study with the large TDS variations caused by 4- to 8-fold evapoconcentration, we restrict the analysis of non-reactive transport of TDS, similar to the typical scope of analysis used in seawater intrusion studies where the substantial contrasts of TDS are key. Our general approach is also supported by the fact that we observe significant freshening of the groundwater TDS locally beneath recharge areas (e.g., Boyle et al., 2012; Castaldo et al., 2021), indicating that conservative modeling of TDS is an appropriate and useful strategy at this stage of scientific inquiry on regional groundwater quality changes under different recharge regimes.

The simulated composite flux-averaged concentrations ($\overline{C}(t)$) for the entire domain at certain depths were used to evaluate the groundwater quality change with time (calculation is presented in Supporting Information S1). The proportions of the aquifer where the water quality is improved (concentration decrease) are calculated by dividing the number of cells with decreased concentration by the total number of cells, which can be used as an indicator of the pattern of water quality change. The State of California has an upper secondary maximum contaminant level (SMCL) of 1,000 mg/L for TDS and a recommended SMCL of 500 mg/L TDS (California State Water Resources Control Board, 2019). The EPA SMCL for TDS is 500 mg/L. The proportions of cells, for which the concentration stays below 500 mg/L were also calculated to evaluate the regional water quality variation, and the results presented in Supporting Information S1.

The flow and transport model neglects vadose zone processes for reasons given below. Although vadose zone transport can certainly be important, in this study where we simulate regional TDS spatiotemporal changes on century timescales and where the water table is not excessively deep (i.e., 9–24 m), we believe we are justified in neglecting the vadose zone effects. The vadose zone can serve both as a reservoir of high-TDS water that would be mobilized by recharge and as a significant delay on vertical transport to the water table. Regarding the mobilization of shallow contamination that may be stored in the vadose zone, we believe this potential to be small in general in the CV aquifer system because of the long history of vadose zone flushing caused by irrigation, which resulted in an approximately 7-times increase in recharge over pre-development conditions (Faunt, 2009). In other words, the subsurface hydrology of the CV aquifer system and its irrigation history indicate that although there are shallow sources of contaminated vadose water (e.g., Harter et al., 2012; Pauloo, Fogg, et al., 2020), those contaminants tend to already be in transit down to the water table, consistent with the ongoing deterioration of shallow groundwater quality (Hansen et al., 2018; Harter et al., 2012). Locally, however, it is true that MAR activities can initially exacerbate shallow groundwater quality.

Regarding the issue of the time delay of recharge, we assume this to be small relative to the century time scales of our simulations. A case in point is provided by the work of Domagalski et al. (2008) near the Merced River, California, located north of our study area in the same regional aquifer system. Depth to the water table in their study area was about 6.5 m, and modeled and measured nitrate concentrations in lysimeters, along with a bromide tracer test, indicated that up to 63% of the yearly applied nitrate was transported to the water table during a typical growing season, less than 1 year. In another study conducted by Maples et al. (2019) in northern CV in which they modeled both the entire vadose zone and the deep groundwater in 3D with a saturated-unsaturated model solving Richards equation, the model simulated rapid response of a 20- to 33-m deep water table to aquifer recharge in a range of 2–28 days. This prompt water table response to recharge was partly facilitated by an interconnected, daylighting IVF deposit originating from the same geologic processes that produced the IVF of this study.

Table 2
Managed Aquifer Recharge Scenarios

Component		IVF	Non-IVF	A	Non-A
	Irrigation	①	②	⑤	⑥
	MAR	③	④	⑦	⑧

	S1				S2				S3				S4			
	Apr- Oct		Nov - Mar		Apr - Oct		Nov - Mar		Apr - Oct		Nov - Mar		Apr - Oct		Nov - Mar	
Descriptio	Business as usual				irrigation entire domain		MAR entire domain		irrigation non-IVF		MAR IVF		irrigation entire domain		MAR IVF	
	①+②				①+②+③+④				②+③				①+②+③			
	IVF	Non-IVF	IVF	Non-IVF	IVF	Non-IVF	IVF	Non-IVF	IVF	Non-IVF	IVF	Non-IVF	IVF	Non-IVF	IVF	Non-IVF
Q	3R	3R	R	R	3R	3R	R+ HMF	R+ HMF	R	3R	R+ HMF	R	3R	3R	R+ HMF	R
C	C1	C1	C2	C2	C1	C1	C2	C2	C2	C1	C2	C2	C1	C1	C2	C2

	S5				S6				S7				S8			
	Apr- Oct		Nov - Mar		Apr - Oct		Nov - Mar		Apr- Oct		Nov - Mar		Apr- Oct		Nov - Mar	
Descriptio	irrigation non-IVF		MAR entire domain		irrigation non-A		MAR region A		irrigation non-IVF		MAR IVF		irrigation non-A		MAR region A	
	②+③+④				⑥+⑦				②+③				⑥+⑦			
	IVF	Non-IVF	IVF	Non-IVF	A	Non-A	A	Non-A	IVF	Non-IVF	IVF	Non-IVF	A	Non-A	A	Non-A
Q	R	3R	R+ HMF	R+ HMF	R	3R	R+ HMF	R	R	3R	15R	R	R	3R	2R	R
C	C2	C1	C2	C2	C2	C1	C2	C2	C2	C1	C2	C2	C2	C1	C2	C2

Note. C1, C2 are the concentrations for recharged water respectively during irrigation season (3,000 mg/L) and fallow season (400 mg/L), when MAR is implemented. *R* is the amount of natural recharge during fallow season (based on rainfall recharge in C2VSim [Brush et al., 2013]). 3R is three-times *R* and represents the estimated total recharge from irrigation and precipitation (Pauloo, Fogg, et al., 2020). Region A is the footprint of the IVF recharge area in a different location. High-magnitude streamflow represents additional recharge water potentially available if high-magnitude flows in streams of the basin are diverted for recharge (scaled from Kocis and Dahlke (2017).

2.4. MAR Scenarios

The scenarios are designed to answer two basic questions: What are the groundwater quality consequences of recharging with low-TDS water strategically on the relatively permeable IVF paleochannel; and can such a strategy accomplish regional reversal in the deteriorating groundwater quality better than simply increasing low-TDS recharge across the landscape? The imposed groundwater budget terms used to drive the flow model are based approximately on magnitudes used in California CV Groundwater-Surface Water Simulation Model (C2VSim), a preexisting regional groundwater flow model (Brush et al., 2013). We first established the flow model representing the water budget as estimated and modeled by Brush et al. (2013) (Table SM-1 in Supporting Information S1). To explore the MAR effects, the water components, such as recharge rate and spatial distribution, were modified in the flow models. In these cases, the underlying flow model is not intended to tightly represent the current or recent groundwater budget, but rather, other scenarios in which floodwaters or imported water would be used to implement MAR.

Table 2 lists each scenario and includes footnotes describing the rationale. A baseline scenario (S0) was simulated first, which represents a continuous non-point source contaminant loading from irrigation in the area from April to October (irrigation season). The initial concentration of TDS in the domain was set to 200 mg/L, based on estimated, pre-development background concentrations (Hansen et al., 2018). Continuous mass loading was imposed via irrigation for 50 years where recharge with contaminated water of 3,000 mg/L was applied during irrigation season. A concentration of 3,000 mg/L was used for irrigated water entering the domain via recharge across the top surface which represents the assumed current concentration of water infiltrating into groundwater (Pauloo, Fogg, et al., 2020). Evapotranspiration of the applied irrigation water results in evapo-concentration of the TDS. For example, the historical concentration (long term average from 1961 to 2001) of total applied water was around 400 mg/L and the estimated irrigation efficiency in the study area was 0.78–0.88 (Brush et al., 2013; Pauloo, Fogg, et al., 2020; Sandoval-Solis et al., 2013), TDS of the resulting recharge would be 1,900–3,300 mg/L (400/(1–0.78)–400/(1–0.88)). Thus, the assumed irrigation recharge TDS of 3,000 mg/L can be considered representative of the consequences of irrigating with moderately high TDS groundwater. During fallow season, November to March, the concentration for recharged water was 400 mg/L with recharge rates 1/3 of those during irrigation season, which estimates TDS for surface water by computing the sample TDS medians in Tulare Lake Basin (TLB) stream samples (USGS, 2016) from 1951 to 2019 combining with evaporation effects (Pauloo, Fogg, et al., 2020). The TDS of 200 mg/L flows into the domain from upgradient boundaries.

Eight scenarios were run (S1–S8) to evaluate the impacts of different aquifer management strategies on groundwater quality. The initial condition for these simulations was the TDS distribution after 50 years of mass loading in S0, and the simulation time for the scenarios was an additional 100 years with 200 stress periods to distinguish the irrigation and fallow season. Time steps were set small enough to avoid numerical instability.

In the first scenario (S1), it was assumed that no action was conducted and the irrigation in S0 continued for another 100 years. For the remaining seven scenarios with different strategies, MAR was applied during fallow season (November to March), which is reasonable in California's Mediterranean climate because most of the excess runoff available for recharge occurs during winter (e.g., Gailey et al., 2019; Kocis & Dahlke, 2017). For scenarios S2–S5, fallow recharge and irrigation were applied differently in the IVF or non-IVF regions as illustrated in Table 2. To explore the geology impact on the effectiveness and efficiency of MAR for groundwater restoration and regional scale groundwater quality mitigation, a scenario (S6) repeating S3 was simulated but with the footprint of the IVF recharge area moved to a different location (region A) that typifies the non-IVF portions of the system to ascertain more fully the relative benefits of recharging on the actual IVF paleochannel. The amount of fallow recharge was determined by scaling the high-magnitude streamflow (HMF) that can be used for groundwater banking in the CV calculated in Kocis and Dahlke (2017) to the study area, which was 53,900 m^3/d, increased 59% of recharge during irrigation season. However, there could be more available water for recharge besides HMF in the study region, such as water transfer from the north. Maples et al. (2020) found the recharge potential is highly dependent on subsurface geologic architecture that infiltration rates for interconnected coarse sediments can be nearly two order-of-magnitude higher than those for relatively low permeable zones. To investigate the ability of IVF to accommodate more recharge water that can improve water quality, two additional simulations were conducted. S7 and S8 were simulated repeating S3 and S6, respectively, but with ponded boundary conditions on the IVF or region A to simulate more aggressive MAR with imported water. The specified head boundaries assigned at the top of the IVF or region A with heads 0.5 m higher than the land surface, resulting in about 15 times the ambient recharge for S7 and 2 times for S8 during fallow season, which is approximately 9.3 times (S7) and 1.2 times (S8) of HMF used for S2–S6, respectively. The concentrations (evapo-concentration of the TDS) of 3,000 mg/L (C1) and 400 mg/L (C2) are used for irrigation and fallow seasons, respectively. A summary of scenarios is listed in Table 2.

2.4.1. Scenario Guide

- S2 and S3 show consequences of diverting the intrabasin HMFs for fallow recharge everywhere (S2) versus just on the IVF paleochannel and without irrigation of the IVF (S3).
- Because irrigation of the IVF is likely to continue unless it is set aside expressly for MAR, S4 differs from S3 only in that the IVF is irrigated during the growing season.
- S5 is the same as S3 but with the HMF water applied to the entire domain.
- S6 is a repeat of S3 but with the footprint of the IVF recharge area moved to a different location that typifies the non-IVF portions of the system to ascertain more fully the relative benefits of recharging on the actual IVF paleochannel.
- S7 repeats S3 but with ponded boundary conditions on the IVF to simulate more aggressive MAR with imported water, resulting in about 15 times the ambient recharge.
- S8 repeats S6 but with the ponded conditions applied to area A instead of to the actual IVF.

3. Results

3.1. Water Table Change After MAR Applications

The head differences for S1–S8 after 100 years of simulation compared to the initial heads are shown in Figure 2. Except S1, which has no MAR application, heads for S2–S8 all increased in varying degrees. It is obvious that continuous pumping and irrigation would lead to further decline of water table if no management strategies applied. Heads dropped over 1 m across the simulated domain after 100 years of irrigation and pumping in the business-as-usual case (S1). For S2 and S5, water recharge occurred across the entire domain, resulting in non-uniform increase of water table spatially. For S3, S4, and S6, water recharge only occurred in IVF or region A where the water table mounds and the water flows outward from IVF/A to the surrounding regions, resulting in an elevated water table outside. However, it is evident that more water accumulation above region A and less outward flow for S6 due to the greater occurrence of aquitard facies and poorer connectivity in A compared to

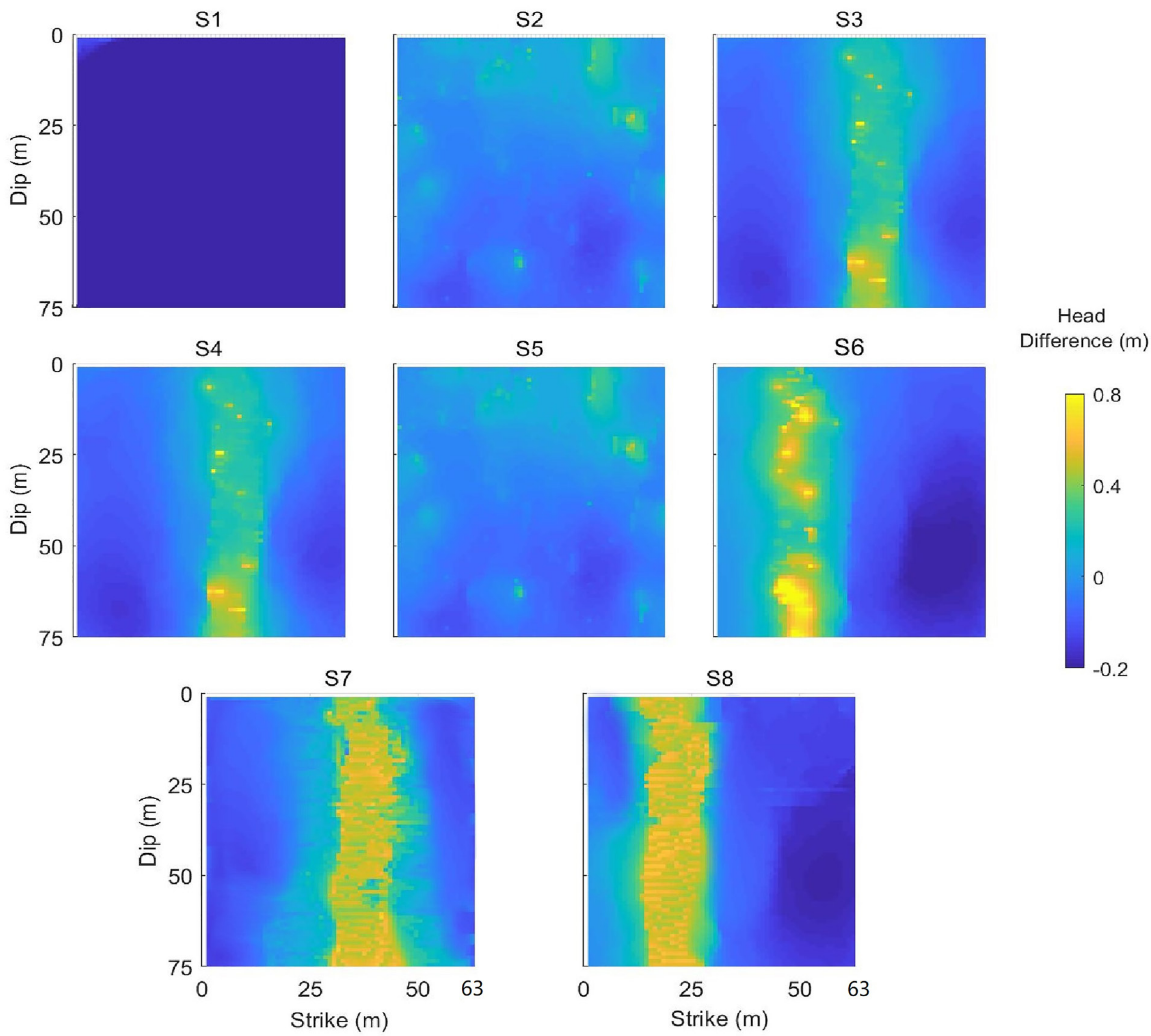

Figure 2. Head difference S1–S8 after 100 years of simulation compared to the initial head. It is 12.6 km in strike direction and 15 km in dip direction with dimension of 75 * 63 in the plain view. Less water table mounding and more outward flow are observed for S3, S4, and S7 that recharge was applied in incised valley fill compared to S6 and S8 that recharge was applied in a relatively less permeable region A. The total recharged water applied via Managed aquifer recharge in 100 years was 0. 97, 9.02, and 1.16 km^3 for S1–S6, S7, and S8, respectively.

IVF. Comparing the head differences for S7 and S8, less water table mounding and more outward flow are also observed for S7. This indicates that strategically sited MAR in areas of highly interconnected coarse hydrofacies can benefit not only these connected, coarse networks, but also the adjacent areas.

For S1, nearly 36% outflow from the model domain was across the bottom boundary, whereas for S2–S5 with extra recharge applied, more vertical flow was induced and outflow from the bottom boundary increased to 54%. For S6, the outflow from bottom is lower, around 46%. The water ponding condition in S7 and S8 resulting in higher recharge, which leads to higher vertical gradient, and more outflow from bottom, 72% and 51%, respectively.

3.2. Temporal Groundwater Quality Evolution

The Kings River Fan study site has undergone decades of intensive irrigation. Therefore, S1 represents the modeled change of water quality due to continuous irrigation and the time scale of water-quality degradation. The flux-averaged TDS concentrations with respect to time, at depths of 35, 70 and 100 m are plotted in Figure 3.

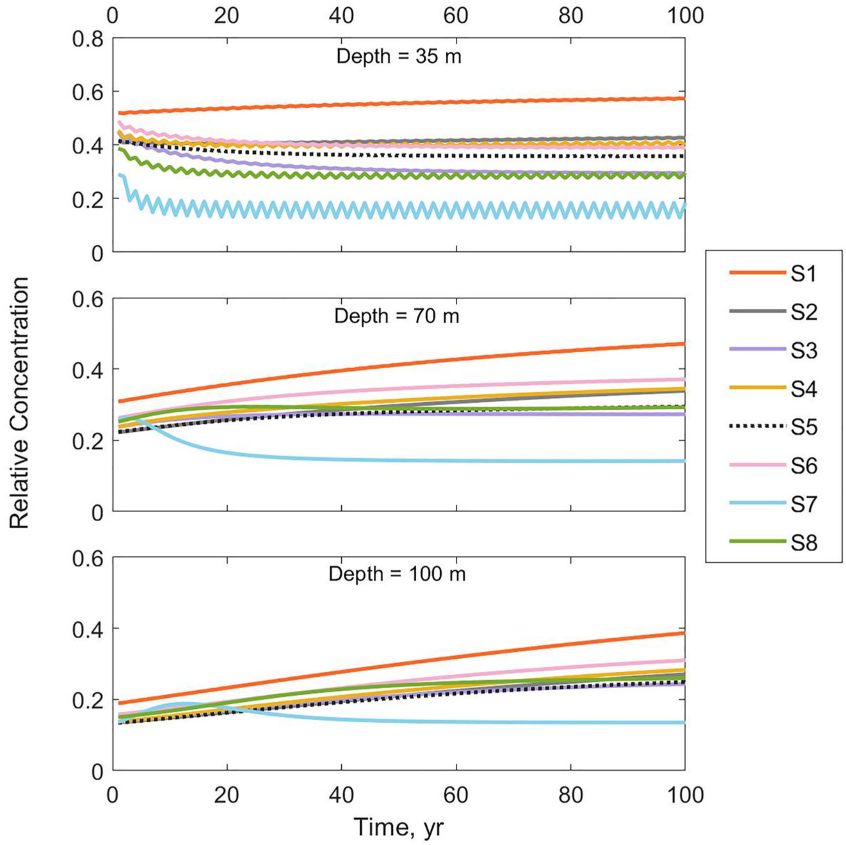

Figure 3. Total dissolved solids concentration change with time at depth of 35, 70, and 100 m for simulated scenarios (S1–S8). The decrease of concentrations (Depth = 35 m) or reduced rates of concentration increase (Depth = 70 and 100 m) with time are observed for every Managed aquifer recharge case (S2–S7) at three depths.

The decrease of concentrations or reduced rates of concentration increase with time are observed for every MAR case (S2–S7) at three depths. In the shallow zone, water quality improvement was observed soon after MAR was applied that TDS concentration for S2–S8 all decreased gradually till reached a steady state in contrast to the continuous TDS increase for S1. At depth of 70 m, TDS for S2–S6 were increased even with MAR applied, but less so than for S1. However, the trends leveled off in that concentrations stabilized for S3, S5 and S6 and approached stabilization for S2 and S4 after rising moderately. TDS for S7 dropped significantly until asymptotic condition after the temporary raise and TDS for S8 increased in first 20 years, then decreased slightly. In deeper zone at a depth of 100 m, TDS in all scenarios shows continuous increases but with reduced rates for S2–S6 and S8. The concentrations for S7 exhibited a rise in first 10 years and declined afterward.

The shallow zone is of course more vulnerable to water-quality degradation than the deeper zone. Due to aquifer heterogeneity, however, the contaminant travel times do not scale linearly. More mixing due to dispersion and matrix diffusion during deeper downward migration results in more resilience for deeper zone to nonpoint source contamination. The time continuous concentration increases for S2–S7 in the deeper zones indicate downward movement of TDS induced by fallow recharge via MAR. The TDS concentrations for S2–S8 were always lower than S1, indicating dilution effects of recharge. Oscillations are observed for curves in the shallow zone due to the seasonal recharge variation. These results show that recharge has more of a direct effect on the shallow system, and a delayed, and less pronounced effect on the deep system. For all scenarios, only S7 improved the groundwater quality to TDS below the EPA SMCL (equivalent to relative concentration of 0.17 in Figure 3), which provides a reference of the time scale and quantity scale of recharge to mitigate groundwater quality in an aquifer. Results presented below nevertheless show significant, regional improvement in shallow and deep groundwater quality for the cases involving strategically located recharge in the IVF.

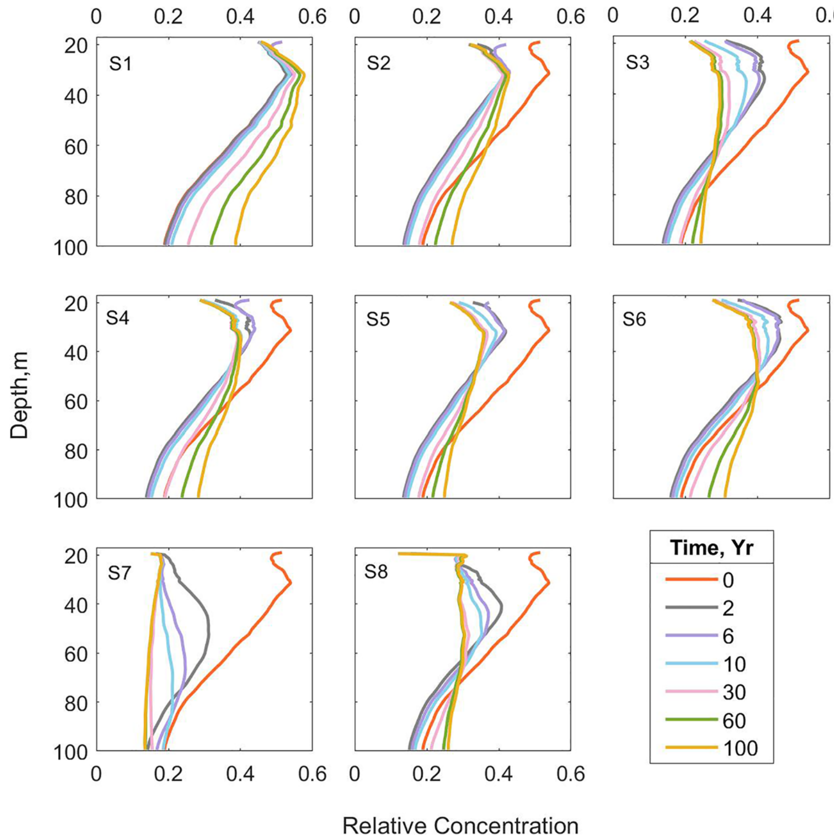

Figure 4. Vertical profiles for S1–S8 at different time periods. The resulting concentrations for S2–S8 with different Managed aquifer recharge (MAR) strategies applied are much lower compared with the business-as-usual S1, in which concentrations were increased continuously with time. The improvement of water quality for S2–S8 in shallow aquifer responded much faster after MAR initiated compared to the deeper aquifer. For S7, due to the strong capacity of incised valley fill to accommodate recharge water, the entire aquifer was diluted and the concentrations converged below secondary maximum contaminant level at later steady state.

Vertical average concentration profiles are plotted in Figure 4. For S1, under continuous irrigation, the concentration profile shifts rightward with time, and the groundwater concentration in the shallow zone reaches asymptotic conditions much faster. As the mass migrates downward with time, more mass accumulates per unit depth, and changes in the deeper zone are magnified. For S2–S8, as different strategies were applied, the resulting concentrations are much lower compared with the business-as-usual S1. The upper part of the concentration profiles in the shallow zone shift leftward and relatively quickly, whereas the lower part of the profiles needs much longer time to be impacted, simply because it takes time for recharged water to reach these depths. The downward movement of TDS and mixing of TDS and fresh water can be observed in the concentration profiles. For scenarios in which no irrigation occurred in IVF (S3 and S5), groundwater quality was improved faster, and the concentrations converged at lower values than other scenarios, indicating susceptibility of IVF to contamination. For S7 and S8, because of increased vertical flow due to greater recharge, the deeper zone was impacted much faster in that the concentrations approached asymptotic condition after approximately 30 years for S7. As a large amount of fresh water entered aquifer for S7, the entire aquifer was diluted and the concentrations converged below SMCL at later steady state. For all depths, the concentrations for S1 higher than other scenarios, and with time as the continued irrigation application, the differences between S1 and other scenarios become more significant. The concentration change in different hydrofacies at deeper zone were further evaluated to assess the impacts of MAR and induced contamination migration on groundwater quality in deeper aquifer. The flux-averaged concentration change for the lower 5 m at the bottom of the model for S7 was shown in Figure 5. The concentrations in all hydrofacies increased and reached the peaks in first 15–20 years followed by approximately 20 years of rapid drop. Afterward, the concentrations in coarse sediments (gravel and sand) exhibited continuous decrease with reduced rates, whereas the concentrations in fines (Muddy sand, Mud, Paleosol) increased gradually. The

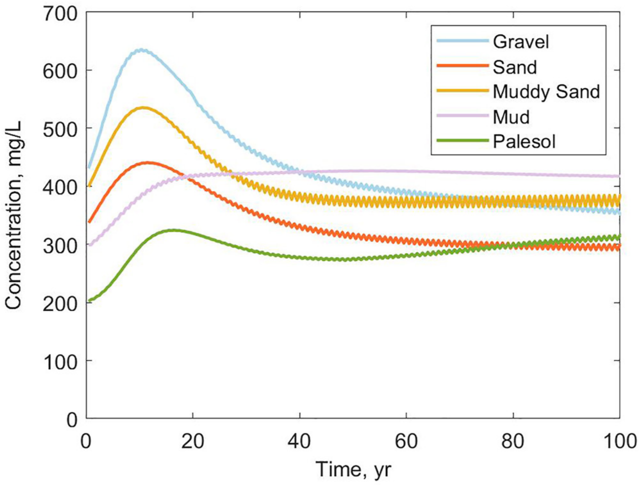

Figure 5. The concentration change in five hydrofacies (gravel, sand, muddy sand, mud and paleosol) for the lower 5 m at the bottom (depth = 96 m) of the model for S7. After Managed aquifer recharge (MAR) applied, the contaminated water was pushed downward and being diluted in the meantime, resulting in increase of concentrations. The concentrations would gradually decline after MAR water reach the deeper aquifer. After peaks, gradual decreases for concentrations in coarse facies (gravel and sand) following rapid drops are observed, whereas concentrations show increase in fines (mud, and paleosol). For muddy sand, the increase of concentrations after reaching the peak slows down approaching asymptotic condition.

concentration changes in fines, which stayed below SMCL in later period, demonstrates the effective mitigation of groundwater quality in the deeper part of system via attenuation process through dispersion and matrix diffusion (into the paleosols).

3.3. Spatial Distribution of Contaminants

The initial condition for eight scenarios was the concentration distribution after 50 years of continuous mass loading due to irrigation, forming a highly non-uniform, regional plume (Figure SM-1 in Supporting Information S1). The spatial distribution of TDS after applying different irrigation and MAR activities for 100 years are shown in Figure 6. The groundwater quality in S1 shows the trend of progressive deterioration, including migration of TDS downward to the underlying portion of the aquifer system. The non-uniform front illustrates the non-ideal transport processes due to distributions of connected channels and low-permeability aquitard facies. MAR in IVF could effectively improve the groundwater quality within IVF and nearby regions as shown by S3 and S5 because of its coarse-grained and connected features, and limiting irrigation activities in IVF can further mitigate the groundwater quality deterioration. Comparing S3 and S6, same application but in different regions, the impacts on groundwater quality also behaved differently. The effects of MAR in region A for S6 on groundwater quality were relatively minor and affected less area outside of A and in deeper zone, indicating less transport capability in region A and hence both less accommodation for recharge and less benefit to regional groundwater quality.

The maximal recharge cases in which the water ponding boundary condition was applied (S7, S8) demonstrated the potential for regional benefits to groundwater quality if the IVF is recharged intensively. The amount of recharge induced via water pond for S7 and S8, respectively, were approximately 15 times and 2 times of the ambient recharge, leading to more profound groundwater quality improvements, especially in the case of S7. In addition, with a large amount of fresh water entered subsurface for S7, which tended to flow both downward and outward, the contaminated water in the surrounding region was also diluted, resulting in a general improvement of groundwater quality and the TDS concentration falling below the SMCL extensively in the aquifer. Whereas, for S8, with the same set up of water ponding, the recharged water was much less compared to S7 due to the greater occurrence of aquitard facies and poorer connectivity in region A, resulting smaller positive effects on mitigation of groundwater quality. This demonstrates the potential magnitude of the benefits of siting MAR in areas with highly interconnected coarse sediments like the IVF.

Noticeably, the shallow zones far from the IVF is still very vulnerable to contamination as far enough away to not have dilution effects from the groundwater recharge, since irrigation outside of the IVF stays active, as shown by S3. Moreover, the hydraulic gradients were increased because of the elevated water table in the IVF, which results in outward flow toward regions where the concentration of TDS was originally low. This is illustrated by comparing the concentration distribution at the left boundary in Figure SM-3 in Supporting Information S1 for S1 and S7.

3.4. Overall Effects of MAR Application on Aquifer

The results show that MAR can improve and reverse negative trends in ongoing deterioration of groundwater quality underneath irrigated lands. Although the concentrations during the simulated period for S2–S6, and S8 did not reach the SMCL, the groundwater quality was improved significantly compared to S1, in which no strategies were applied, indicating the general benefit of MAR for water quality if the recharge water is of low to moderate TDS (Figure 7). This improvement was more significant in shallow zone for most scenarios due to the strong dilution, and the improvements slowed down going deeper before reaching steady state. Noticeably, the decrease of improvement vertically is not a linear relationship, reflecting the non-Fickian transport behavior caused by

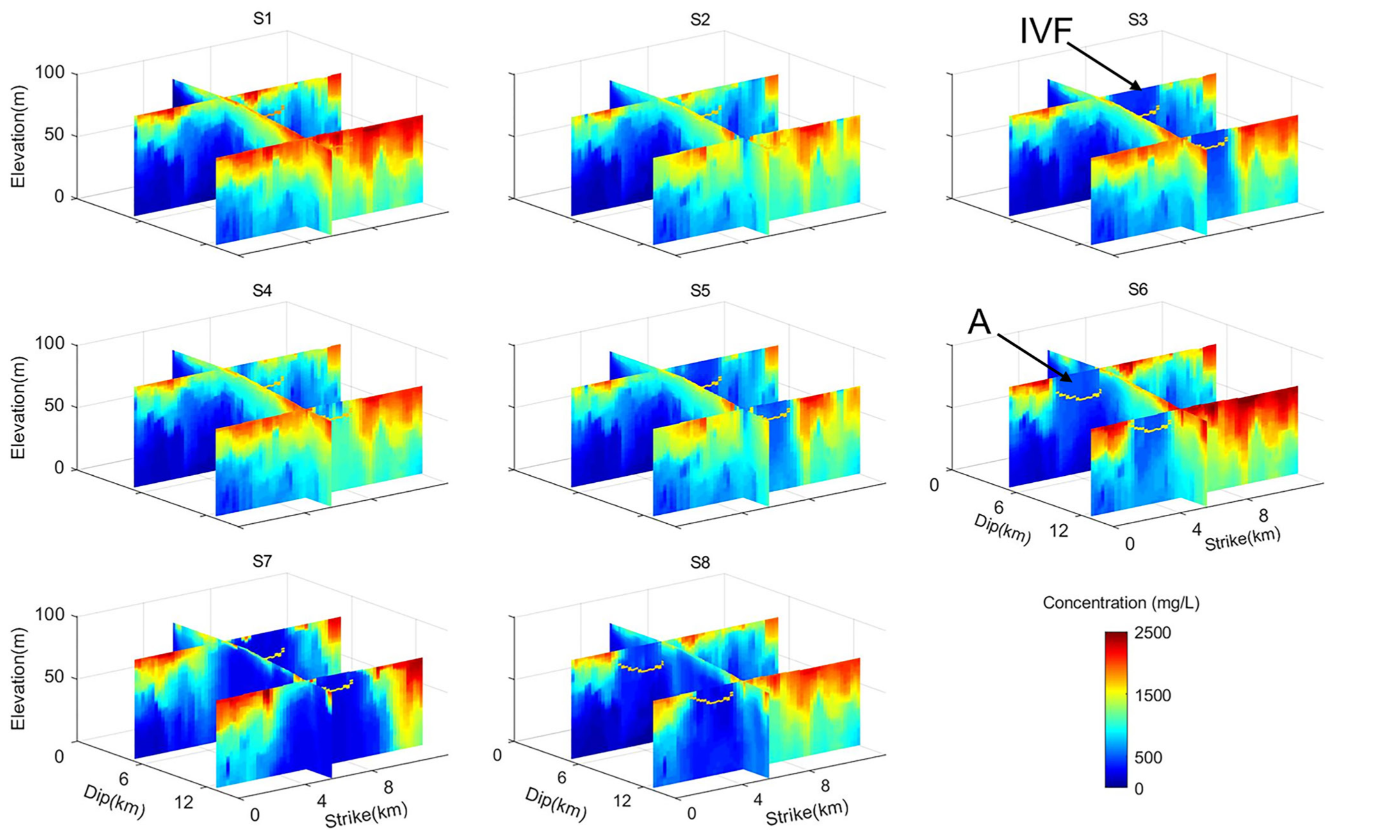

Figure 6. Total dissolved solids concentration distribution on three cross sections, 5 and 13 km in dip direction and 5.6 km in strike direction, for S1–S8 after 100 years. The lines in light yellow represent the bottom bounds of the incised valley fill (IVF) for S1–S5 and S7 or region A for S6 and S8. Recharge in IVF tends to induce more lateral flow into the surrounding open-fan deposits, resulting overall groundwater quality improvement.

aquifer heterogeneity. The larger differences between improvements in shallow and deep zone for S6 and S8 demonstrated relatively slow vertical migration as the less connected hydrofacies existed in region A (Figure 7a). In S7, because of the large gradient induced by water ponding and highly connected coarse hydrofacies in IVF region, the concentration in the entire system reached steady state in a shorter period, less than 50 years. Minor changes observed between improvements at year 50 and 100 indicated a balance between recharged water and the contaminated water due to mixing (Figure 7b). Whereas, for other scenarios, the groundwater quality in deep aquifer were improved gradually with time consistent to the TDS changes in Figure 3.

In terms of each hydrofacies, the mean concentrations in gravel and sand dropped significantly in first two decades of MAR application before reaching asymptotic conditions, whereas in low permeable hydrofacies, mud and paleosol, the concentrations experienced a rapid increase in earlier years followed by continuously gradual increase at a lower rate (Figure 8a). Because of the unchanged alternate irrigation and recharge, a stable condition would eventually be achieved that the mass residing in each hydrofacies equals the volume proportion of this hydrofacie in the system as shown in Figure 8b. This behavior, where the proportion of solute mass in the hydrofacies approaches the volume proportions of those hydrofacies, was also observed by LaBolle and Fogg (2001).

4. Discussion

4.1. Potential Benefits Siting MAR in Highly Connected Coarse Hydrofacies

Siting MAR in IVF generated more significant vertical migration within IVF and lateral spreading in surrounding regions, and therefore, more mixing between contaminated water and fresh water as shown in this study. Maples et al. (2019) studied the variability of MAR potential at different geologic sites, and found recharge rates, volumes, and the assimilation of water into an aquifer to be highly dependent on the subsurface geologic structure that the recharge rate for coarse materials could be 3.23 to 65.5 times of the recharge rates in other hydrofacies.

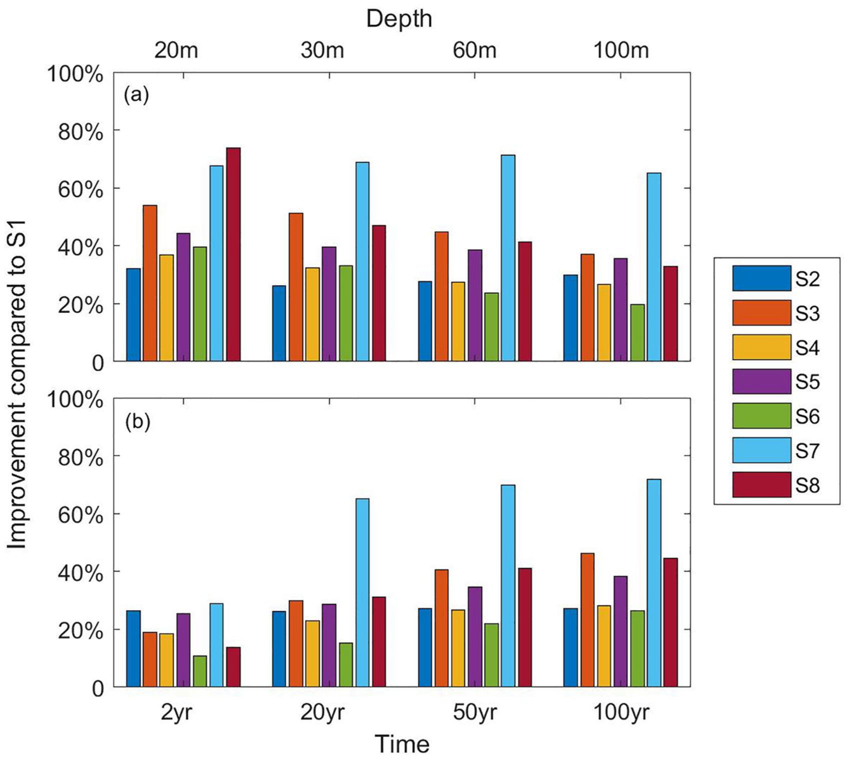

Figure 7. The improvement of groundwater quality for S2–S8 compared to S1. (a) Overall improvements in four depth (20, 30, 60, 100 m) after 50 years of Managed aquifer recharge operations; (b) improvements at depth of 80 m at different time.

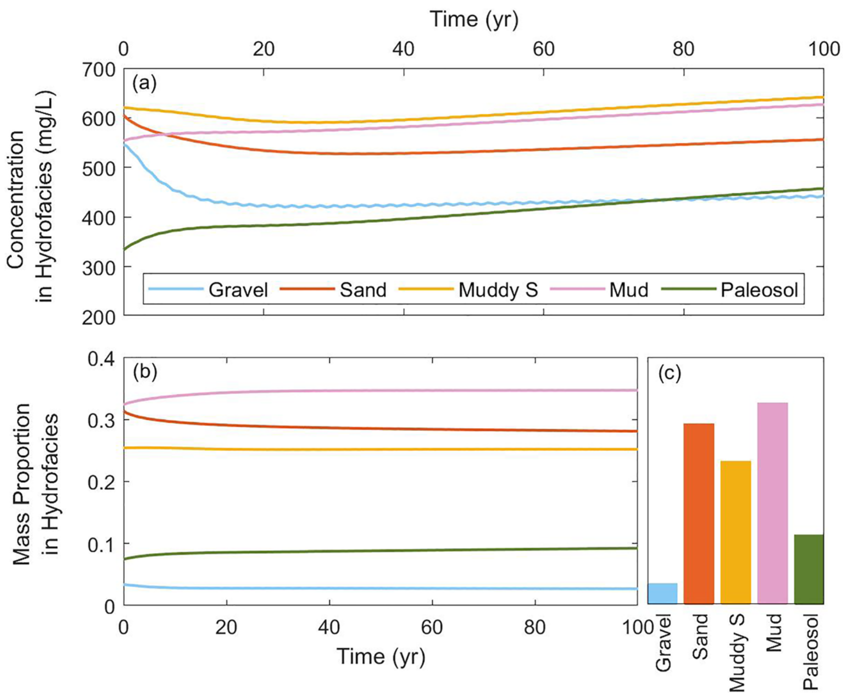

Figure 8. Total dissolved solids evolution in different hydrofacies for S7, (a) the mean concentration change with time in five hydrofacies; (b) the proportion of remaining mass in each hydrofacies; (c) the volume proportion of each hydrofacies. As expected, the mass proportions of solute in each hydrofacies approaches the proportions of those hydrofacies in the geostatistical model.

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In this study, the resulting recharge from ponded water above IVF was approximately 8 times of the recharge by applying ponded water outside of IVF (such as region A), which resulted in faster transport and stronger dilution of contaminated water. The relatively high flow rate in IVF leads to lateral flow into the surrounding open-fan deposits, resulting overall groundwater quality improvement. These results are consistent with the observation in Weissmann et al. (2004) that particles released in IVF would move to open fan deposits from IVF. Therefore, siting MAR in a region with highly connected coarse hydrofacies has evident benefits for mitigating groundwater contamination.

Importantly, whereas remediation of groundwater contamination resulting from nonpoint-source pollution is generally assumed to be impractical, these results suggest that strategic, high-intensity recharge operations on geologically favorable subregions can reap impressive, long-term benefits for regional groundwater quality. In other words, smart recharge operations that capitalize on the geologic heterogeneity offer a way of reversing the ongoing decline in regional groundwater quality under irrigated lands.

4.2. Effects on Deeper Portions of the Aquifer System

Recharge operations in systems already containing large amounts of degraded groundwater will push that groundwater into other portions of the system. As we demonstrate in this paper, this is likely to mobilize contaminants in both horizontal and vertical directions (Figure. SM-3 in Supporting Information S1). After implementing the recharge, the groundwater quality in shallow zone improved in a relatively short period. In contrast, the relatively contaminated water originally residing in the shallow zone will be pushed to deeper zones due to the strong downward flow (Figure 4). Clearly then, despite regional benefits to groundwater quality, MAR can temporarily worsen groundwater quality for certain wells, and these effects may play out over decades. MAR project will need to both monitor these effects and be prepared to compensate stakeholders whose well water quality is negatively impacted.

Moreover, after MAR was applied, the faster flush in coarse sediments can result in rapid concentration decrease in these hydrofacies, whereas in the finer-grained hydrofacies, mass accumulation was observed (Figure 8). Maples et al. (2019) in a similarly heterogeneous alluvial aquifer system also showed that low-K silt and clay facies accommodate the majority of recharge volume over long time periods. In alluvial aquifers where non-point source or point source contaminants have existed for a long time, much of the remaining contaminant mass may resides in low-permeability zones consisting of silt or clay. Re-saturating these low permeable zones with MAR can mobilize contaminants and facilitate mass transfer to highly permeable zones. Flow field changes induced by MAR can also mobilize previously contained contaminants, causing mass redistribution in the aquifer, and into water bearing facies that are tapped by wells.

On the optimistic side, even though downward migration of contaminants due to MAR was observed, the concentration at bottom (depth $\sim$ 100 m) for S7 still remained below the SMCL (Figures 3 and 4). As shown in Figure 5, the peak concentrations in five hydrofacies stay well under 1,000 mg/L, after which the concentrations in the aquifer facies (gravel and sand) decline gradually with time. These moderate concentrations at the bottom indicate that enough attenuation through dispersion and matrix diffusion (into the paleosols) occurred to mitigate effects of downward migration of contaminants on deeper parts of the system. Therefore, generally sustainable groundwater quality could be achieved due to the attenuation process via appropriate management of MAR. Clearly, much work remains to be done on management of regional groundwater quality in the presence of MAR and ongoing pumping from wells (e.g., Fogg & LaBolle, 2006).

Even in this study, a heterogeneous field was used to illustrate the impacts on groundwater, it is much more complex for a basin in real life with irrigation and pumping. As indicated from the results, heterogeneity always impacts the solute transport. Therefore, before water management strategies are applied, it would be prudent to evaluate and quantify the possible impacts on groundwater quality to avoid spreading contaminants into relatively uncontaminated portions of the aquifers. In this study, with the TDS as the main concern, a 3- to 10-times reduction in concentration that was observed in the results was enough to significantly mitigate deeper impacts. In contrast, for other contaminants, such as volatile organic compounds (VOCs), many orders of magnitude decrease in concentration would be required to achieve the remediation goal. Hence for VOCs, dispersion and matrix diffusion are much less consequential as attenuation mechanisms. In summary, the groundwater quantity needs to be managed in concert with the groundwater quality; hence the need for regional monitoring and groundwater quality management models.

4.3. Timescales to Mitigate Regional Groundwater Quality

In all simulated MAR scenarios, only S7 achieved aquifer concentration below SMCL. The amount of recharged water for S7 was 0.036 km^3/yr, nearly 15 times of the natural recharge. The concentration in gravel, from which a large fraction of extracted groundwater comes, dropped below SMCL and approached stable condition in approximately 20 years, which accommodated 0.73 km^3 recharged water. According to the Long-Term North to South Water Transfer Program in California (Bureau of Reclamation, 2011), the maximum conveyance of water transferring from water agencies in northern California to water agencies south of the Sacramento-San Joaquin Delta (Delta) and in the San Francisco Bay Area is 0.74 km^3/yr, over 20 times of the recharge amount for S7 each year in the study region that covers 189 km^2, indicating that the water supply implied by our modeling for MAR application, although sizable, is not inconceivable.

The amount of recharge needed to dilute groundwater below SMCL in 100 years was calculated roughly using mass balance by making assumptions that (a) concentration in groundwater was uniformly distributed and linearly changing with time; (b) both irrigation and MAR occur across entire domain, and the duration for each application is half year. The results showed extra recharge with at least 6.7 times of the natural recharge, which is approximately 2 times of the irrigated water, can reduce concentration below SMCL. This is a conservative estimation, which does not consider the effects of heterogeneity, such as preferential flow and slow mass transfer from low permeability zones. In cases of S1–S6 and S8, the extra recharged water was 0.5–2 times of the base-case recharge, which is lower than the 6.7 times estimated conservatively above for groundwater concentration reducing below SMCL. Therefore, for MAR scenarios S1–S6 and S8, the concentrations in later years approach steady state but,remain higher than SMCL.

Extreme climate conditions are expected in California's future. For example, intense winter precipitation in a short period of time followed by long-term periods of drought will not be uncommon (IPCC, 2014), and indeed have already been observed in recent decades (Swain et al., 2018). Given the space for water storage in the subsurface and the demand for water supply, storing excess rainwater underground instead of traditional reservoirs appears to be a viable management strategy. According to a recent study (DWR, 2013), statewide groundwater storage losses in California were between 0.6 and 1.85 km^3/yr (1,960–2003). Taking advantages of subsurface storage and available surface water, for which MAR could be an effective means, can help mitigate groundwater overdraft, and as shown in this paper, help alleviate groundwater quality problems within decades to centuries depending on aquifer of interest.

5. Conclusions

This work studied the potential for mitigating regional groundwater quality deterioration via MAR applications. An aquifer domain with a 3D heterogeneous hydraulic conductivity field generated by a geostatistical model was used to represent the study area. Different aquifer management scenarios (S2–S8) were simulated to explore the potential influence of MAR on groundwater. The water table and groundwater quality for these scenarios were compared with the results from the baseline scenario (S1) where irrigation water high in TDS was continuously applied. The results from our simulations indicate the effectiveness of MAR to mitigate overdraft, as well as improve groundwater quality. Groundwater quality in shallow aquifers responds rapidly to the management strategies explored in this study, whereas the deeper aquifer takes longer for groundwater quality to improve unless the recharge rate is increased significantly.

Siting MAR in regions with different geological formations shows great impacts on performance of MAR. MAR in highly connected coarse hydrofacies results in stronger flushing, more significant vertical migration and lateral spreading, and therefore, more mixing between contaminated water and fresh water. Ponding water above the relatively permeable and connected IVF induces 9 times more recharge compared to ponding water in other random region with footprint of the IVF recharge area, which further leads to faster and more extensive improvements on groundwater quality. During MAR, the concentrations in coarse sediments can decline rapidly whereas in fine materials, the concentrations show a gradual increase due to the preferred mass accumulation in low permeability zones. IVF and similar geologic deposits can act as conduits for MAR water to affect groundwater quality in surrounding regions. These results demonstrate the feasibility of strategic, high-intensity recharge operations on geologically favorable subregions to provide long-term benefits for regional groundwater quality under irrigated lands. Meanwhile, more careful management of irrigation activities in these geologically favorable regions can further alleviate aquifer contamination from the source, as these connected formations can also act as fast travel paths for contaminants.

Special attention must be paid applying MAR, as the recharge inevitably changes the flow field and can result in contaminants migrating to originally less contaminated regions. MAR may seem like a panacea to shortages in groundwater quantity, and if done with fresh water, it tends to improve groundwater quality, but in some cases it may mobilize unwanted contaminants, and thus the careful and strategic siting of MAR projects together with water quality monitoring are strongly advised. Importantly, our simulations show enough attenuation of TDS concentrations due to dispersion and matrix diffusion as the solute mass migrates downward, that the deep concentrations remain below the SMCL.

In systems like the TLB, where regional groundwater quality degradation is ongoing, the creation of improved groundwater quality in portions of the system will be beneficial for users able to access that resource. Moreover, regional groundwater quality management could involve preferentially pumping groundwater that has improved quality while reducing pumping of groundwater of poorer quality. MAR would in effect create “bubbles” of relatively good groundwater quality that needs to be both monitored and managed more carefully than is typical of historically ad hoc groundwater management approaches. Over time, this could have the benefit of enhanced mixing of higher and lower TDS groundwater, effecting gradual, basin-scale remediation. Overall, over the long term, MAR with appropriate recharge strategies could be an effective means for sustainable groundwater management.

Data Availability Statement

The geological borehole data is available at https://www.scidb.cn/en/s/amy2If and the flow and transport files are available at https://www.scidb.cn/s/NB3aIz.

Acknowledgments

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References

Alley, W. M., Reilly, T. E., & Franke, O. L. (1999). *Sustainability of ground-water resources* (Vol. 1186). US Department of the Interior, US Geological Survey.

Ayub, R., Obenour, D. R., Messier, K. P., Serre, M. L., & Mahinthakumar, K. (2016). Non-point source evaluation of groundwater contamination from agriculture under geologic and hydrologic uncertainty. In *World environmental and water resources congress 2016* (pp. 329–336).

Banerjee, P., & Singh, V. S. (2011). Optimization of pumping rate and recharge through numerical modeling with special reference to small coral Island aquifer. *Physics and Chemistry of the Earth, Parts A/B/C*, *36*(16), 1363–1372. https://doi.org/10.1016/j.pce.2011.04.014

Barnett, S. R., Howles, S. R., Martin, R. R., & Gerges, N. Z. (2000). Aquifer storage and recharge: Innovation in water resources management. *Australian Journal of Earth Sciences*, *47*(1), 13–19. https://doi.org/10.1046/j.1440-0952.2000.00760.x

Bastani, M., & Harter, T. (2019). Source area management practices as remediation tool to address groundwater nitrate pollution in drinking supply wells. *Journal of Contaminant Hydrology*, *226*, 103521. https://doi.org/10.1016/j.jconhyd.2019.103521

Boyle, D., King, A., Kourakos, G., Lockhart, K., Mayzelle, M., Fogg, G. E., & Harter, T. (2012). Groundwater nitrate occurrence. Technical report 4. In *Addressing nitrate in California's drinking water with a focus on Tulare Lake Basin and salinas Valley Groundwater. Report for the state water resources control board report to the legislature* (p. 277). Center for Watershed Sciences, University of California. (free public access).

Brush, C. F., Dogrul, E. C., & Kadir, T. N. (2013). *Development and calibration of the California Central Valley Groundwater-Surface water simulation model (C2VSim), version 3.02-CG*. Bay-Delta Office, California Department of Water Resources.

Brusseau, M. L., & Guo, Z. (2014). Assessing contaminant-removal conditions and plume persistence through analysis of data from long-term pump-and-treat operations. *Journal of Contaminant Hydrology*, *164*, 16–24. https://doi.org/10.1016/j.jconhyd.2014.05.004

Bureau of Reclamation. (2011). Long-term water transfers, environmental impact statement/environmental impact report, scoping report. Retrieved from https://www.usbr.gov/mp/cvp/ltwt/scoping_report/ltwt_scoping_report_full.pdf

Burow, K. R., Weissmann, G. S., Miller, R. D., & Placzek, G. (1997). Hydrogeologic facies characterization of an alluvial fan near Fresno, California, using geophysical techniques. *US Geological Survey*, *97*(46), 15.

California state water resources control board, A compilation of water quality goals. (2019). California state water resources control board, A compilation of water quality goals. Retrieved from https://www.waterboards.ca.gov/water%20issues/programs/water%20quality%20goals

Carle, S. F. (1997). Implementation schemes for avoiding artifact discontinuities in simulated annealing. *Mathematical Geology*, *29*(2), 231–244. https://doi.org/10.1007/bf02769630

Carle, S. F., & Fogg, G. E. (1996). Transition probability-based indicator geostatistics. *Mathematical Geology*, *28*(4), 453–476. https://doi.org/10.1007/bf02083656

Carle, S. F., & Fogg, G. E. (1997). Modeling spatial variability with one and multidimensional continuous-lag Markov chains. *Mathematical Geology*, *29*(7), 891–918. https://doi.org/10.1023/a:1022303706942

Castaldo, G., Visser, A., Fogg, G. E., & Harter, T. (2021). Effect of groundwater age and recharge source on nitrate concentrations in domestic wells in the san Joaquin valley. *Environmental Science & Technology*, *55*(4), 2265–2275. https://doi.org/10.1021/acs.est.0c03071

Domagalski, J. L., Ator, S., Coupe, R., McCarthy, K. A., Lamp, D. C., Sandstrom, M. W., & Baker, N. (2008). Comparative study of transport processes of nitrogen, phosphorous, and herbicides to streams in five agricultural basins, USA. *Journal of Environmental Quality*, *37*(3), 1158–1169. https://doi.org/10.2134/jeq2007.0408

DWR. (2013). *California natural resources agency and state of California 2013 California's groundwater update, a compilation of enhanced content for California*. Department of Water Resources.

Faunt, C. C. (2009). *Groundwater availability of the Central Valley aquifer, California* (Vol. 1766, p. 225). US Geological Survey Professional Paper.

Fogg, G. E., & LaBolle, E. M. (2006). Motivation of synthesis, with an example on groundwater quality sustainability: Motivation of synthesis. *Water Resources Research*, *42*(3), W03S05. https://doi.org/10.1029/2005WR004372

Fogg, G. E., LaBolle, E. M., & Weissmann, G. S. (1999). Groundwater vulnerability assessment: Hydrogeologic perspective and example from salinas valley, California. In L. Corwin, K. Loague, & R. Ellsworth (Eds.), *Geophysical monograph series* (Vol. 108, pp. 45–61). American Geophysical Union. https://doi.org/10.1029/GM108p0045

Gailey, R. M., Fogg, G. E., Lund, J. R., & Medellín-Azuara, J. (2019). Maximizing on-farm groundwater recharge with surface reservoir releases: A planning approach and case study in California, USA. *Hydrogeology Journal*, *27*(4), 1183–1206. https://doi.org/10.1007/s10040-019-01936-x

Gleeson, T., Wada, Y., Bierkens, M. F., & van Beek, L. P. (2012). Water balance of global aquifers revealed by groundwater footprint. *Nature*, *488*(7410), 197–200. https://doi.org/10.1038/nature11295

Guo, Z., & Brusseau, M. L. (2017). The impact of well-field configuration and permeability heterogeneity on contaminant mass removal and plume persistence. *Journal of Hazardous Materials*, *333*, 109–115. https://doi.org/10.1016/j.jhazmat.2017.03.012

Guo, Z., Fogg, G. E., Brusseau, M. L., LaBolle, E. M., & Lopez, J. (2019). Modeling groundwater contaminant transport in the presence of large heterogeneity: A case study comparing MT3D and RWhet. *Hydrogeology Journal*, *27*(4), 1363–1371. https://doi.org/10.1007/s10040-019-01938-9

Guo, Z., Fogg, G. E., & Henri, C. V. (2019). Upscaling of regional scale transport under transient conditions: Evaluation of the multirate mass transfer model. *Water Resources Research*, *55*(7), 5301–5320. https://doi.org/10.1029/2019WR024953

Han, D., Currell, M. L., & Cao, G. (2016). Deep challenges for China's war on water pollution. *Environmental Pollution*, *218*, 1222–1233. https://doi.org/10.1016/j.envpol.2016.08.078

Hansen, J. A., Jurgens, B. C., & Fram, M. S. (2018). Quantifying anthropogenic contributions to century-scale groundwater salinity changes, San Joaquin Valley, California, USA. *Science of the Total Environment*, *642*, 125–136. https://doi.org/10.1016/j.scitotenv.2018.05.333

Harbaugh, A. W., Banta, E. R., Hill, M. C., & McDonald, M. G. (2000). *MODFLOW-2000, the U.S. Geological Survey modular ground-water model—User guide to modularization concepts and the Ground-Water Flow Process*. USGS Open-File Report 00-92. : U.S. Geological Survey

Harter, T., Heeren, K., Weissmann, G., Horwath, W., & Hopmans, J. (1998). Non-point source contamination in a heterogeneous, moderately deep vadose zone: The Kearney research site. *IAHS Publication (International Association of Hydrological Sciences)*, *250*, 257–263.

Harter, T., Lund, J., Darber, J., Fogg, G., Howitt, R., Jessoe, K., et al. (2012). Addressing nitrate in California's drinking water. With a focus on Tulare Lake Basin and Salinas Valley Groundwater. In *Report for the state water resources control board report to the legislature*. UC Davis Center for Watershed Sciences.

Harter, T., Meyer, R., & Mathews, M. (2002). Nonpoint source pollution from animal farming in semi-arid regions: Spatio-temporal variability and groundwater monitoring strategies. In L. Ribeiro (Ed.), *Future groundwater resources at risk, proceedings of the 3rd international conference, june 2001* (pp. 363–372).

He, C., & Croley, T. E., II. (2008). Estimating nonpoint source pollution loadings in the great lakes watersheds. In W. Ji (Ed.), *Wetland and water resource modeling and assessment: A watershed perspective* (pp. 115–127). CRC Press.

Hollins, S. E., Ridd, P. V., & Read, W. W. (2004). Measurement of the diffusion coefficient for salt in salt flat and mangrove soils. *Wetlands Ecology and Management*, *8*(4), 257–262. https://doi.org/10.1023/a:1008470719913

Huntington, G. L. (1971). *Soil survey, eastern Fresno area*. US Department of Agriculture, Soil Conservation Service.

IPCC. (2014). In R. K. Pachauri & L. A. Meyer (Eds.), *Climate change 2014: Synthesis report. Contribution of working groups I, II and III to the fifth assessment report of the intergovernmental panel on climate change* (p. 151). IPCC.

Kelsey, R., Hart, A., Butterfield, H., & Vink, D. (2018). Groundwater sustainability in the San Joaquin valley: Multiple benefits if agricultural lands are retired and restored strategically. *California Agriculture*, *72*(3), 151–154. https://doi.org/10.3733/ca.2018a0029

Kendy, E., & Bredehoeft, J. D. (2006). Transient effects of groundwater pumping and surface-water-irrigation returns on streamflow. *Water Resources Research*, *42*(8), 1–11. https://doi.org/10.1029/2005WR004792

Kocis, T. N., & Dahlke, H. E. (2017). Availability of high-magnitude streamflow for groundwater banking in the Central Valley, California. *Environmental Research Letters*, *12*(8), 084009. https://doi.org/10.1088/1748-9326/aa7b1b

Konikow, L. F., & Kendy, E. (2005). Groundwater depletion: A global problem. *Hydrogeology Journal*, *13*(1), 317–320. https://doi.org/10.1007/s10040-004-0411-8

LaBolle, E. M., & Fogg, G. E. (2001). Role of molecular diffusion in contaminant migration and recovery in an alluvial aquifer system. *Transport in Porous Media*, *42*(1–2), 155–179. https://doi.org/10.1023/a:1006772716244

Laitos, J. G., & Ruckriegle, H. (2013). The clean water act and the challenge of agricultural pollution. *Vermont Law Review*, *37*, 1033–1070.

Levy, Y., Shapira, R. H., Chefetz, B., & Kurtzman, D. (2017). Modeling nitrate from land surface to wells' perforations under agricultural land: Success, failure, and future scenarios in a mediterranean case study. *Hydrology and Earth System Sciences*, *21*(7), 3811–3825. https://doi.org/10.5194/hess-21-3811-2017

Lindsey, B., & Johnson, T. (2018). *Data from decadal change in groundwater quality web site, 1988-2014, version 2.0*. U.S. geological survey. Retrieved from https://nawqatrends.wim.usgs.gov/Decadal/

Liu, J. W., Wei, K. H., Xu, S. W., Cui, J., Ma, J., Xiao, X. L., et al. (2020). *Surfactant-enhanced remediation of oil-contaminated soil and groundwater: A review*. Science of the Total Environment. 144142.

Lockhart, K. M., King, A. M., & Harter, T. (2013). Identifying sources of groundwater nitrate contamination in a large alluvial groundwater basin with highly diversified intensive agricultural production. *Journal of Contaminant Hydrology*, *151*, 140–154. https://doi.org/10.1016/j.jconhyd.2013.05.008

Maples, S. R., Fogg, G. E., & Maxwell, R. M. (2019). Modeling managed aquifer recharge processes in a highly heterogeneous, semi-confined aquifer system. *Hydrogeology Journal*, *27*(8), 2869–2888. https://doi.org/10.1007/s10040-019-02033-9

Maples, S. R., Foglia, L., Fogg, G. E., & Maxwell, R. M. (2020). Sensitivity of hydrologic and geologic parameters on recharge processes in a highly heterogeneous, semi-confined aquifer system. *Hydrology and Earth System Sciences*, *24*(5), 2437–2456. https://doi.org/10.5194/hess-24-2437-2020

McDonald, M. G., & Harbaugh, A. W. (1988). *A modular three-dimensional finite-difference ground-water flow model* (Vol. 6). US Geological Survey Reston.

Meirovitz, C., Fogg, G. E., Weissmann, G. S., Sager, J., Roll, L., & LaBolle, E. M. (2017). Non-stationary hydrostratigraphic model of cross-cutting alluvial fans. *International Journal of Hydrology*, *1*(1), 1–10. https://doi.org/10.15406/ijh.2017.01.00001

Muniz, A., & Ziegler, W. B. (1994). Aquifer storage and recharge in south-east Florida. In *Artificial recharge of ground water* (Vol. 311–17). ASCE.

NGWA (National Ground Water Association). (2017). *Managed aquifer recharge: A water supply management tool*. National Ground Water Association.

NRC (National Research Council). (2013). *Alternatives for managing the nation's complex contaminated groundwater sites*. National Academies Press

O'Connor, D., Hou, D., Ok, Y. S., Song, Y., Sarmah, A. K., Li, X., & Tack, F. M. (2018). Sustainable in situ remediation of recalcitrant organic pollutants in groundwater with controlled release materials: A review. *Journal of Controlled Release*, *283*, 200–213. https://doi.org/10.1016/j.jconrel.2018.06.007

Pauloo, R. A., Escriva-Bou, A., Dahlke, H., Fencl, A., Guillon, H., & Fogg, G. E. (2020). Domestic well vulnerability to drought duration and unsustainable groundwater management in California's Central Valley. *Environmental Research Letters*, *15*(4), 044010. https://doi.org/10.1088/1748-9326/ab6f10

Pauloo, R. A., Fogg, G. E., Guo, Z., & Harter, T. (2020). Anthropogenic Basin closure and groundwater salinization (ABCSAL). *Journal of Hydrology*, *593*, 125787. https://doi.org/10.1016/j.jhydrol.2020.125787

Pauloo, R. A., Fogg, G. E., Guo, Z., & Henri, C. V. (2021). Mean flow direction modulates non-fickian transport in a heterogeneous alluvial aquifer-aquitard system. *Water Resources Research*, *57*(3), e2020WR028655. https://doi.org/10.1029/2020WR028655

Planert, M., & Williams, J. S. (1995). *Ground water atlas of the United States: Segment 1*. US Geological Survey.

Russo, T. A., & Lall, U. (2017). Depletion and response of deep groundwater to climate-induced pumping variability. *Nature Geoscience*, *10*(2), 105–108. https://doi.org/10.1038/ngeo2883

Sandoval-Solis, S., Orang, M., Snyder, R. L., Orloff, S., Williams, K. E., & Rodríguez, J. M. (2013). Spatial and temporal analysis of application efficiencies in irrigation systems for the state of California.

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 US high plains and Central Valley. In *Proceedings of the National Academy of Sciences* (Vol. 109, pp. 9320–9325). National Academy of Sciences. https://doi.org/10.1073/pnas.1200311109

Scanlon, B. R., Gates, J. B., Reedy, R. C., Jackson, W. A., & Bordovsky, J. P. (2010). Effects of irrigated agroecosystems: 2. Quality of soil water and groundwater in the southern high plains, Texas. *Water Resources Research*, *46*(9), W09538. https://doi.org/10.1029/2009WR008428

Stakelbeek, A., Roosma, E., & Holzhaus, P. M. (1996). Deep well infiltration in the north-holland dune area. In *Artificial recharge of ground water* (Vol. 111–26). ASCE.

Sun, J., Donn, M. J., Gerber, P., Higginson, S., Siade, A. J., Schafer, D., et al. (2020). Assessing and managing large-scale geochemical impacts from groundwater replenishment with highly treated reclaimed wastewater. *Water Resources Research*, *56*(11), e2020WR028066. https://doi.org/10.1029/2020WR028066

Swain, D. L., Langenbrunner, B., Neelin, J. D., & Hall, A. (2018). Increasing precipitation volatility in twenty-first-century California. *Nature Climate Change*, *8*(5), 427–433. https://doi.org/10.1038/s41558-018-0140-y

USGS. (2016). National water information system data available on the world wide web (USGS water data for the nation). URL: Retrieved from http://waterdata.usgs.gov/nwis/

Wada, Y., Wisser, D., & Bierkens, M. F. (2014). Global modeling of withdrawal, allocation and consumptive use of surface water and groundwater resources. *Earth System Dynamics Discussion*, *5*(1), 15–40. https://doi.org/10.5194/esd-5-15-2014

Weissmann, G. S., Carle, S. F., & Fogg, G. E. (1999). Three-dimensional hydrofacies modeling based on soil surveys and transition probability geostatistics. *Water Resources Research*, *35*(6), 1761–1770. https://doi.org/10.1029/1999WR900048

Weissmann, G. S., & Fogg, G. E. (1999). Multi-scale alluvial fan heterogeneity modeled with transition probability geostatistics in a sequence stratigraphic framework. *Journal of Hydrology*, *226* (1), 48–65. https://doi.org/10.1016/S0022-1694(99)00160-2

Weissmann, G. S., Zhang, Y., Fogg, G. E., & Mount, J. F. (2004). Influence of incised-Valley-fill deposits on hydrogeology of a stream-dominated alluvial fan. https://doi.org/10.2110/pec.04.80

Weissmann, G. S., Zhang, Y., LaBolle, E. M., & Fogg, G. E. (2002). Dispersion of groundwater age in an alluvial aquifer system. *Water Resources Research*, *38*(10), 1–13. https://doi.org/10.1029/2001WR000907

Xu, Q. (2014). *The study of agricultural non-point source pollution control policy system*. Michigan Technological University.

Zhang, Y., Green, C. T., & Fogg, G. E. (2013). The impact of medium architecture of alluvial settings on non-Fickian transport. *Advances in Water Resources*, *54*, 78–99. https://doi.org/10.1016/j.advwatres.2013.01.004

Zhang, Y., Weissmann, G. S., Fogg, G. E., Lu, B., Sun, H., & Zheng, C. (2018). Assessment of groundwater susceptibility to non-point source contaminants using three-dimensional transient Indexes. *International Journal of Environmental Research and Public Health*, *15*(6), 1177. https://doi.org/10.3390/ijerph15061177

Zheng, C., & Wang, P. P. (1999). *MT3DMS: A modular three-dimensional multispecies transport model for simulation of advection, dispersion, and chemical reactions of contaminants in groundwater systems; documentation and user's guide*. Alabama Univ University.