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Publication Date
2019-10-01

DOI
10.1016/j.jconhyd.2019.103521

Peer reviewed

Journal of Contaminant Hydrology 226 (2019) 103521

Contents lists available at ScienceDirect

Journal of Contaminant Hydrology

journal homepage: www.elsevier.com/locate/jconhyd

Source area management practices as remediation tool to address groundwater nitrate pollution in drinking supply wells

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ARTICLE INFO

Keywords:
Groundwater quality
Nitrate pollution
Agricultural management practices
Contributing recharge area
Crop change
Groundwater remediation

ABSTRACT

Nitrate in drinking water may cause serious health problems for consumers. Agricultural activities are known to be the main source of groundwater nitrate contaminating rural domestic and urban public water supply wells in farming regions. Management practices have been proposed to reduce the amount of nitrate in groundwater, including improved nutrient management practices and "pump and fertilize" with nitrate-affected irrigation wells. Here, we evaluate the feasibility and long-term impacts of agricultural managed aquifer recharge (Ag-MAR) in the source area of public water supply wells. A numerical model of nitrate fate and transport was developed for the Modesto basin, part of California's Central Valley aquifer system. The basin is representative of semi-arid agricultural regions around the world with a diversity of crop types, overlying an unconsolidated sedimentary aquifer system. A local public supply well in an economically disadvantaged community surrounded by farmland was the focus of this study. Model scenarios implemented include business as usual, alternative low-impact crops, and Ag-MAR in the source area of the public supply well. Alternative nutrient management and recharge practices act as remediation tools in the area between farmland and the public supply well. Improved agricultural source area management practices are shown to be an effective tool to maintain or even enhance groundwater quality in the targeted supply well while remediating ambient groundwater.

Best results are obtained when lowering nitrate load while also increasing recharge in the source area simultaneously. This scenario reduced nitrate in the supply well's drinking water by 80% relative to the business as usual scenario. It also remediated ambient groundwater used by domestic wells between the source area farmlands and the supply well and showed 60% more reduction of nitrate after 60 years of application. Increasing recharge led to shorter initial response time (five years) and showed the most sustainable impact. Our analysis further suggests that Ag-MAR in a highly discontinuous, wide-spread pattern leads to slow water quality response and may not yield sufficient water quality improvements.

1. Introduction

Groundwater is a critical resource for drinking water and irrigation in the Central Valley, California, a region dominated by intensive cultivation of mostly specialty crops (Burow et al., 2008). Nitrate is one of the most widespread groundwater contaminants threatening drinking water supplies here and in agricultural regions elsewhere (Nolan et al., 2002; Van Grinsven et al., 2015; Wang et al., 2016). Nitrate causes serious health problems if it exceeds 10 mg N/L in drinking water (Jordan and Weller, 1996). Nitrate transport through groundwater into connected surface water may also significantly affect surface water quality and ecosystem functioning (Sprague et al., 2011; Tesoriero et al., 2013).

Nitrate in groundwater is a global problem. The European Union (EU) has undertaken broad steps toward managerial regulations that prevent further nitrate pollution of their aquifers (Hansen et al., 2017). The issue is most threatening in some where drinking water suppliers entirely rely on groundwater resources, such as Denmark (Gejl et al., 2019). Studies conducted throughout the EU but also in China and elsewhere have repeatedly identified the historical nitrate loading from cropland as the main contributor to groundwater nitrate pollution in these areas (Gu et al., 2013; Wang et al., 2016; Zhang and Hiscock, 2016; Harter et al., 2017; Ransom et al., 2017).

In the subsurface, nitrate is the most readily transported form of reactive nitrogen (Jury and Nielson, 1989), lacking significant sorption or degradation potential, unless reducing geochemical conditions prevail. Sources include farmland, animal agriculture, septic systems, and naturally occurring nitrate (Liu et al., 2014). Irrigated croplands managed with synthetic nitrogen fertilizers or animal manure have been

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https://doi.org/10.1016/j.jconhyd.2019.103521
Received 10 December 2018; Received in revised form 24 June 2019; Accepted 3 July 2019
Available online 08 July 2019

shown to be the largest source of nitrate loading in agricultural areas (Burow et al., 2008; Harter et al., 2017).

The main driver of long-term average nitrate leaching from croplands is the balance between nitrogen inputs (atmospheric deposition, synthetic fertilizer, animal manure) and nitrogen harvested (Kirchmann et al., 2002). This excess nitrogen is lost in ionic or gaseous form through leaching, runoff volatilization, and denitrification (Liu et al., 2014). Once leached below the root zone, denitrification through microbial processes may provide for some or complete natural attenuation of nitrate depending on biogeochemical conditions (Ransom et al., 2017). These conditions also affect the vulnerability of drinking supply wells to nitrate contamination (Eberts et al., 2013).

Excess nitrogen is transported as nitrate to groundwater via soil percolation and recharge. In some instances, dry wells, abandoned wells, or improperly destroyed wells may also act as rapid conduits of nitrate contaminated surface runoff directly into groundwater (Gailey, 2017). In irrigated regions, minimizing recharge during the growing season and matching nitrogen applications to crop demand is the primary tool to nitrate source control (Rosenstock et al., 2014; Baram et al., 2016), and to protect vulnerable public supply wells from nitrate contamination (Eberts et al., 2013).

Land-use change, agricultural practice improvements, or technological advancements have been shown to help to reduce nitrate contributions to groundwater. These techniques (a) control water application rates to minimize nitrate leaching, and (b) reduce nitrogen quantity applied to the land (Dzurella et al., 2012; Lee et al., 2017). Somura et al. (2008) suggested reducing the concentration of nitrate leaching to groundwater by partly or entirely removing livestock area. Hiscock et al. (2007) and Zhang and Hiscock (2016) showed that land-use conversion to forest provides a considerable decrease in groundwater nitrate concentrations in the Sherwood Sandstone and Lincolnshire Limestone (aquifers in England). They used MODFLOW and MT3DMS as simulation tools to study different land-use scenarios that included partial or complete conversion between arable areas, woodlands (forest), and grasslands. They indicated that water quality change of public supply wells is on the order of years to centuries when land-use changes occur over days to decades. Moreover, Rudolph et al. (2015) conducted field experiments to study the performance of regional-scale nutrient management practices in the source area of supply wells. Neither of these studies considered changes to alternative crop types or agricultural managed aquifer recharge.

The groundwater age mixture in a public supply well is also an important factor on water quality dynamics in the well (Eberts et al., 2013). The travel-time distribution from source area to wells is partly due to the complexity of media. “Short-circuit pathways” in the flow system create a large fraction of young water in the well, and provide for faster response to land use change (McMahon et al., 2008). The vertical length of the well screen is typically an even more significant contributor than aquifer heterogeneity to wide distribution of travel times within the water exiting a well at any given time (Henri and Harter, 2019). Travel times of nitrate in the unsaturated zone range from less than one year to decades, depending mostly on depth to groundwater, and on recharge rates under agricultural activities. Below the water table, nitrate travel times to shallow domestic wells range from a few years to several decades, and from one to several decades, and even centuries, for deeper production wells (Jurgens et al., 2008; van der Schans et al., 2009).

It has recently also been suggested that farmland provides an untapped resource for recharge of seasonal flood and excess water when available during the non-growing season (Niswonger et al., 2017; Department of Water Resources, 2018). This promising technique is known as agricultural managed aquifer recharge (Ag-MAR) (Ghasemizade et al., 2019). Key considerations for implementation of Ag-MAR include strong coordination between growers and water managers, and selection of farms with suitable crop and soil characteristics. While primary consideration of Ag-MAR is for aquifer replenishment, we here consider its potential to significantly improve groundwater quality.

The implementation of water and nutrient management to improve groundwater nitrate conditions may incur significant cost on agriculture (Medellín-Azuara et al., 2012). Mayzelle et al. (2015) considered the economic feasibility and cost of the more drastic measure of converting irrigated agricultural land-uses surrounding economically disadvantaged communities (DAC) in the Tulare Lake Basin to reduce the risk of nitrate contamination in their public supply wells. The source area of all drinking water wells inside DCs was assumed to be a circular region called the “protective agricultural buffer zone”. These buffer zones were devoted to clean recharge basins and conversion of land-use to leguminous (N-fixing) crops such as alfalfa or low nitrogen use crops such as vineyards. Surprisingly, the work demonstrated economic gains over 20 years offset some or all the cost of land-use conversion for some scenarios. In addition, it minimizes the cost of drinking water treatment plants' operation and maintenance, which is important for communities facing financial constraint.

Source area agricultural practices play an important role in determining the water quality in targeted drinking water wells. These activities change (a) the volume of water that enters the groundwater system, and (b) the management of water and nutrients on crops to reduce nitrate mass transported to groundwater. We develop a high resolution, transient regional flow and transport model to more closely investigate the effects of agricultural practices on future nitrate concentration dynamics in public supply wells and on the quality of groundwater in nearby domestic wells. Specifically, we seek to quantify the effectiveness of (a) agricultural managed aquifer recharge (Ag-MAR) during the rainy winter season and (b) nitrate load reduction by either growing appropriate alternative crops or by significantly improving nutrient management practices.

2. Materials and methods

2.1. Site description

The study area of about 2700 km^2 is located in the northeastern San Joaquin Valley, bounded on the western edge by the San Joaquin River, on the northern edge near the Stanislaus River, in the south near the Merced River, and at the eastern edge by the bedrocks of the Sierra Nevada foothills (Fig. 1). The San Joaquin River drains the study area into the San Francisco Bay (Pacific Ocean). The land surface in the study area slopes westward from the Sierra Nevada foothills to the San Joaquin River; gradients range from < 1 m/km near the river to > 5 m/km in places near the foothills and adjacent to streams and rivers. The climate is semi-arid, characterized by hot summers and mild winters; the rainfall averaged 315 mm annually from 1931 to 1997 with most precipitation occurring from late fall through early spring. The headwaters of San Joaquin River and its tributaries, the Stanislaus, Tuolumne, and Merced Rivers originate from the Sierra Nevada Mountains to the north and east of the study area and provide an important source of irrigation water, managed through upstream reservoirs (Phillips et al., 2007a).

Agriculture is the major land use, covering about 65% of the study area. The primary crops are deciduous trees (almonds and walnuts, peaches) grapes, and forage crops (grain, corn, pasture, and alfalfa). In 2010, the cities of Modesto and Turlock, and a number of smaller communities occupy about 7% of the study area. The remaining land areas are predominantly dry land vegetation near the foothills and riparian wetlands along river corridors (Phillips et al., 2015).

The aquifer system consists of highly heterogeneous alluvial sediments. The aquifer is mainly unconfined within the study area, although discontinuous fine-grained units locally confine sand and gravel layers (Fig. 2). The Corcoran Clay, a semi-permeable barrier, is a key hydrogeological feature that limits vertical groundwater flow in the western part of the study area. The top of the Corcoran Clay zone is in the range of 25 to 80 m below land surface. The confining unit has a

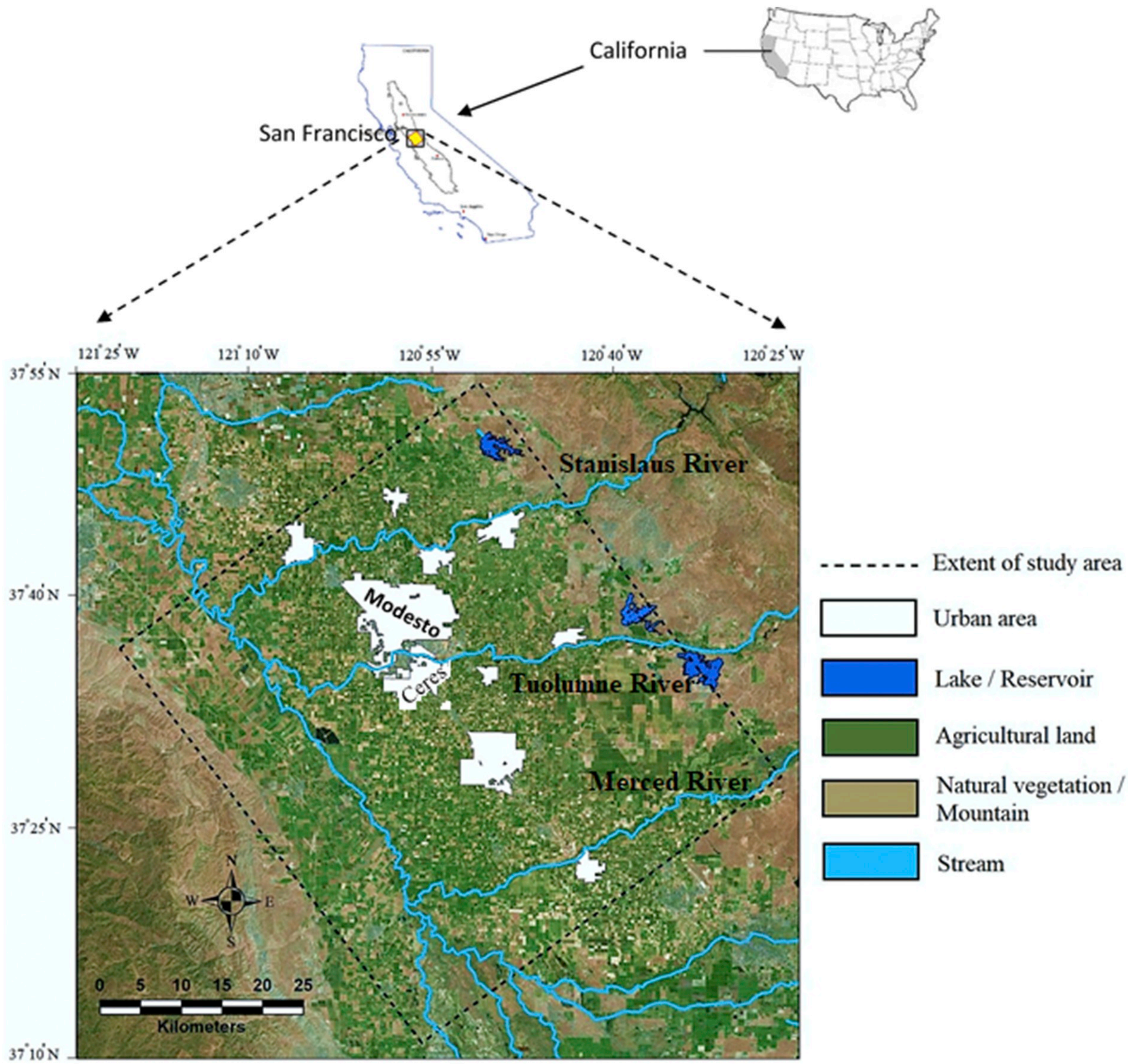


Fig. 1. Location of the study site near Modesto, San Joaquin Valley, California

maximum thickness of about 58 m. The confined aquifer below the Corcoran Clay is also an important regional water resource (Phillips et al., 2015).

Prior to groundwater development in the early 1900s, streams recharged groundwater predominantly near the mountain front to the northeast of the study area. Groundwater moved laterally across the basin, and discharged at the San Joaquin River. Subsequently, groundwater pumping and diversion of water from streams for irrigation added a vertical flow pattern to the groundwater system. Irrigation water provided significant additional groundwater recharge to the system while irrigation pumpage became the main groundwater discharge from the system (Burow et al., 2008). Nitrate is one of the main groundwater contaminants in the study area derived mainly from the application of agricultural nitrogen-based fertilizers (Harter et al., 2017). Elevated nitrate concentrations in the region are mostly associated with young groundwater in the shallow groundwater system (Bexfield and Jurgens, 2014).

2.2. Flow and transport model development

2.2.1. Groundwater flow simulation

A calibrated, transient, three-dimensional saturated flow model constructed in MODFLOW-OWHM (Phillips et al., 2015) was modified to develop simulation conditions for a prediction period from January 2005 to December 2064. The original model represents groundwater flow in the study region from January 1960 to December 2004 with 540 stress periods of one month each. The model uses the Farm Process (Schmid and Hanson, 2009), which allocates available surface water to water users and simulates processes associated with agricultural water management, recharge, groundwater pumping, surface-water interaction with groundwater, and crop parameters to set boundary conditions for solving the groundwater flow equation. The model domain covers a total area of 3353.76 km^2 using a grid discretized horizontally into 400 m × 400 m cells in 153 rows, 137 columns, and 16 layers. Layer thickness generally increases with depth, ranging from one to 28 m in the top layer. Total thickness of the model is about 220 to 430 m. The Corcoran Clay layer thickness, ranging from 3 to 43 m, has been assigned to western cells in layer eight of the model. General head-dependent boundaries are employed to simulate lateral flow across the model boundaries, except along the Sierra Nevada foothills (northeastern boundary) which is considered a no-flow boundary (Phillips et al., 2015).

For the transient flow and transport model in this study, we first needed to convert the MODFLOW-OWHM model into a MODFLOW-2005 framework to create a flow model platform that is compatible

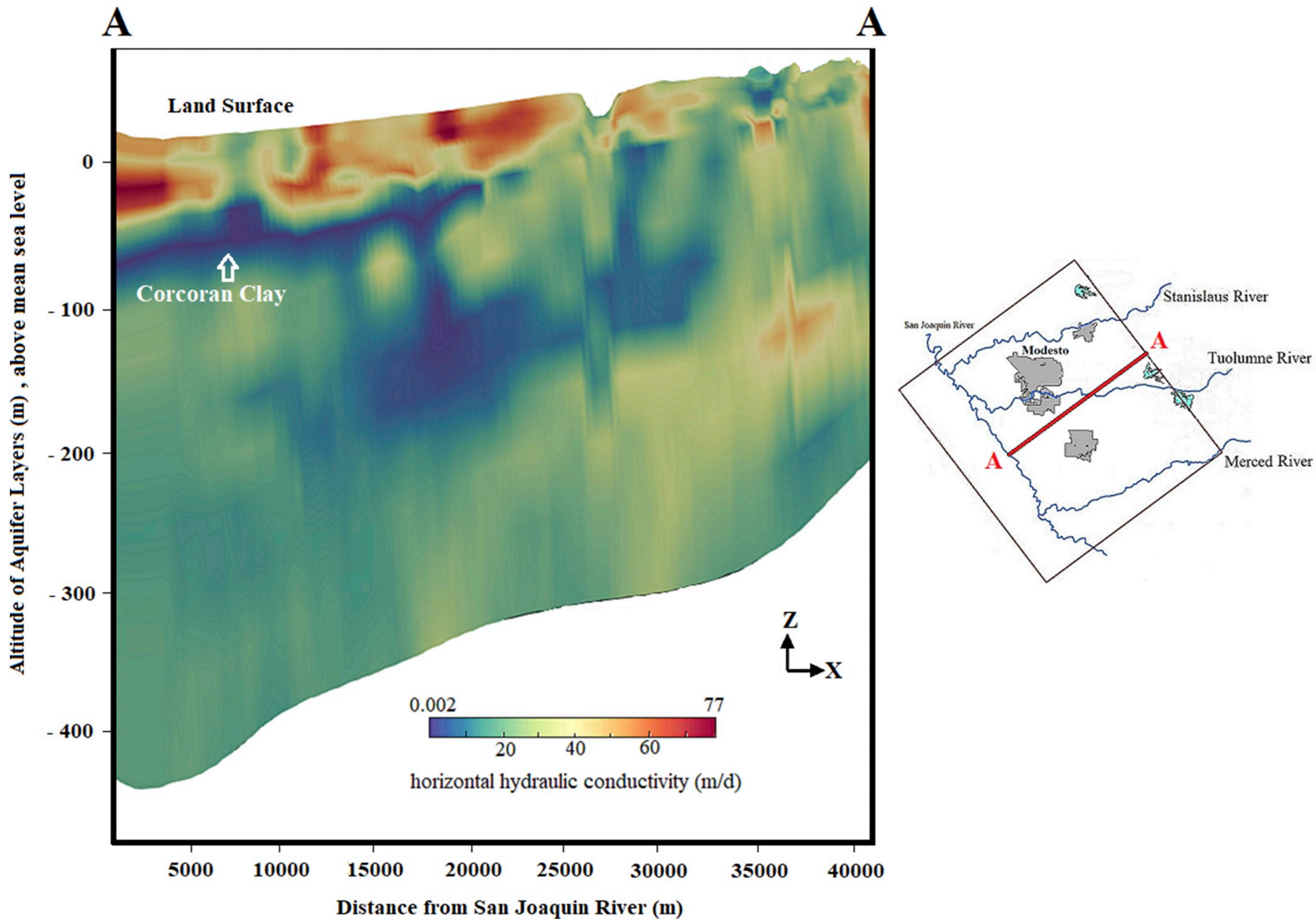


Fig. 2. Vertical distribution of aquifers in section AA of the study area.

with an MT3D (Zheng, 1990) based transport model. This was achieved by deriving OWHM-specific flux boundary conditions from output files of the original model, including recharge rates (deep percolation of irrigation and precipitation), evapotranspiration parameters, and private agricultural pumpage. Location and time-specific flux data were used to create suitable MODFLOW-2005 packages to effectively impose the same boundary conditions on the flow model.

In the second step, we modified boundary conditions to represent a cyclical steady-state condition for 60 years. The original model's year 2004 monthly recharge and pumping conditions were repeated for each of 60 years starting from January 2005, in 720 monthly stress periods (Fig. 3). Seasonal pumping and recharge cause significant periodic change in water levels. In addition, because the 2004 water budget was not entirely balanced, water levels showed a long-term decrease of about 3–4 cm/year over the 60-year period (Fig. 3).

2.2.2. Supply well selection

The regional groundwater flow model was used to delineate the capture zone and the contributing recharge area for a single public supply well located in one of the economically disadvantaged communities (DAC) in the study domain. Data related to the “non-farm” wells in the well package of the original MODFLOW-OWHM model were used to select an appropriate supply well for this study. These data include spatial distribution of wells, well depth, and pumping rates. Only shallow public supply wells were considered for the purpose of this study. The concentration of nitrate in groundwater samples from agricultural land-use in the area is reported to be much higher in shallow groundwater and less in the deeper aquifer (Burow et al., 2008; 2010). In addition, deep production wells are likely to receive water with a wide range of ages that increases the mixing of poor quality water with older, cleaner water (Burow et al., 2008). The selected well was among wells with longest possible screen interval above the confining layer, where agricultural managed aquifer recharge (Ag-MAR) practices may have most impact on groundwater quality. The selected well is located in the city of Ceres adjacent to agricultural lands. It has a maximum and minimum pumping rates of 4900 and 1400 m^3/d, over the scenario period (Fig. 3).

2.2.3. Recharge area delineation

Delineation of the production well's source area - the area of recharge - is a critical step in determining the spatial extent of the area within which to apply alternative agricultural practices that would reduce long-term nitrate levels. Here, we use the particle-tracking program MODPATH (Pollock, 2012) to simulate groundwater flow pathlines backwards from the public supply well to the recharge area and to estimate groundwater travel-times. The pathline particle-tracking technique uses advective transport but does not consider dispersion or diffusion.

Cell-by-cell groundwater flux and effective porosity (interconnected pores) are the main input information for the MODPATH program to track particles in time and space. The effective porosity used in the current contaminant transport and particle tracking simulations was distributed based on the percentage of coarse-grained texture in model cells (Table 1). The values for each category of porosity is derived from previous work in similar regions.

The well screen of the targeted public supply well is located along two vertically adjacent cells in the model, in layers five and six. The top

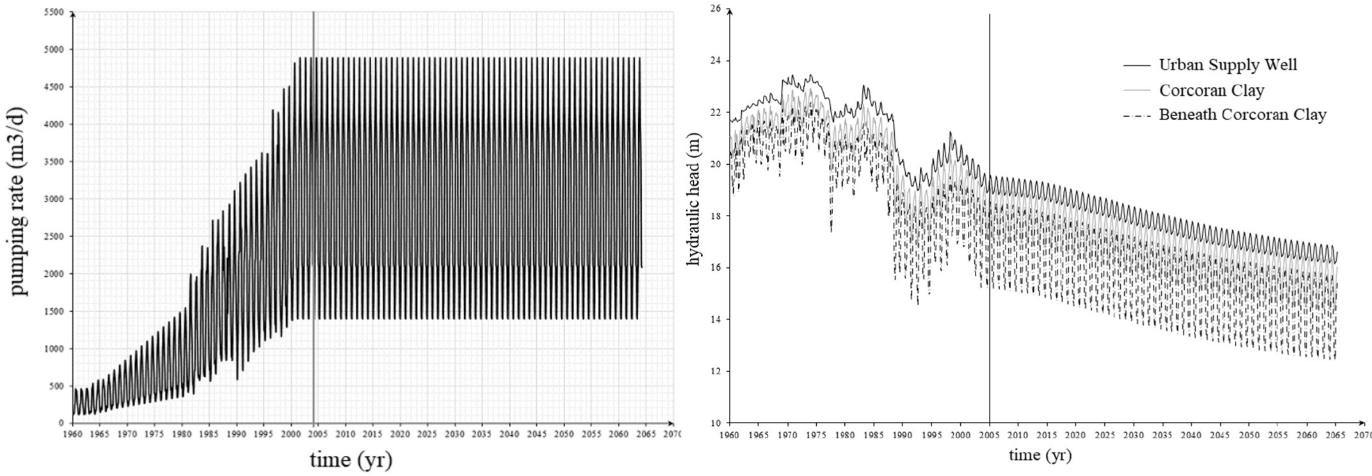


Fig. 3. Pumping rate and head changes of the urban supply well in the flow model (before 2005), and for scenario analysis purposes (after 2005)

Table 1
Effective porosity values (Phillips et al., 2007b; McMahon et al., 2008).

Material categories	Coarse texture proportion (%)	Effective porosity (%)
Gravel	> 75	25
Coarse sand	51–75	28
Fine sand	25–50	32
Silt and clay	< 25	35

of the screen is at a depth of 26.7 m below ground surface and the screen length is 13.9 m. Twenty particles were equally distributed at each side of the cells (200 particles in total) and released from the well screen every two stress periods (360 release times). This frequency of particle release takes the system's transient dynamics into account. Particles moved from the well screen to the water table, backwards over the 60-year simulation time, from where the public supply well's water originated. Twenty-five cells were identified as the source area for the drinking water well; land use in those cells includes urban and farm lands (Fig. 4).

2.2.4. *Nitrate transport model construction*

Three-dimensional transport of nitrate in saturated porous media was simulated using the advection-dispersion equation and solved numerically by use of MT3D program (Zheng and Bennett, 2002):

$$\frac{\partial(\theta C)}{\partial t} = \nabla.\,(\theta D\nabla C) - \nabla.\,(\theta v C) + q_s C_s$$

where θ is porosity [unitless], C is the concentration of nitrate in groundwater [M/L^3], v is the seepage velocity ($v = q/\theta$) [L/T], D is the hydrodynamic dispersion tensor [L^2/T], C_s is the concentration of the nitrate in source or sink [M/L^3], and q_s is the volumetric flow rate of sink or source per unit aquifer volume [T^{-1}]. The first term in the right side of the equation is diffusion-dispersion term, the second term is the advection term, and the last term is the source and sink term.

Three main assumptions were made to simulate nitrate transport in the study area's groundwater. First, the nitrate load from the bottom of the root zone was applied to the groundwater table in the same year. Impacts of the unsaturated zone on the nitrate level attenuation and travel time delays were not considered in the model. Second, the study aquifer is a largely oxic basin with slow denitrification rates (McMahon et al., 2008). Therefore, groundwater denitrification was neglected. Third, we assumed that nitrate is not adsorbed to solid phases.

Initial concentration of nitrate across the aquifer was estimated based on available, limited data from sampling wells distributed vertically and horizontally in the study area. The active transport model domain was the same as the active flow model domain. Prescribed nitrate flux was used to simulate boundary solute leached into the groundwater system using monthly stress periods (Table 2). Streams were presumed to have negligible nitrate relative to loading from the land surface in the model. Nitrogen fertilizers leached from agricultural lands were considered to be the main source of nitrate contamination. The nitrogen load was continuously applied on farms over a period of 12 months.

Nitrate concentration of the recharged water was calculated using a mass balance approach using the estimated amount of nitrogen fertilizer applied and the nitrogen available in harvested plants for each crop type (see Rosenstock et al., 2014, Harter et al., 2017, for details). Their work assumed that nitrogen mass inflows balances outflows (no change in root zone storage) over the long run (15 years and more). Hence, cropland nitrate leaching to groundwater was estimated by adding known applied fertilizer and other inputs to a field, and then subtracting the harvested nitrogen, atmospheric losses, and runoff. The annually leached mass of nitrate entered groundwater beneath farmlands (Table 3). The 2004 land-use distribution from the original MODFLOW-OWHM file was used for the prediction simulation period (i.e., 2005–2064). (See Table 3.)

Since there was no field measurement or test for transport parameters in the study area, the longitudinal dispersivity was estimated based on the work of Schulze-Makuch (2005). He presented comprehensive data sets at various scales and provided the following best fitted empirical power law relationship that describes longitudinal dispersivity data with respect to scale of measurement:

$$\alpha = c(L)^m$$

where $\propto$ is longitudinal dispersivity, c is characteristic parameter of the geological medium, L is the flow distance, and m is the scaling exponent. Selected values for the above parameters ($m = 0.81$, $c = 0.085$) are based on the table provided in Schulze-Makuch (2005) for an unconsolidated medium.

2.3. *Source area management scenarios*

A number of potential future scenarios are considered for the source area of the public supply wells: business as usual, a change in nutrient loading, a change to agricultural managed aquifer recharge (Ag-MAR) during winter months, and a combination of these alternative practices. Lower nitrate mass loading from the root zone may be achieved either by replacing high-intensity crops (e.g., almonds, corn) with low-intensity crops that require significantly less nitrogen input (e.g., alfalfa, vineyards) and therefore reduce the risk of nitrogen losses; or it may be achieved by significantly improving nitrogen management practices without changing the crop to achieve low nitrate mass losses. Winter

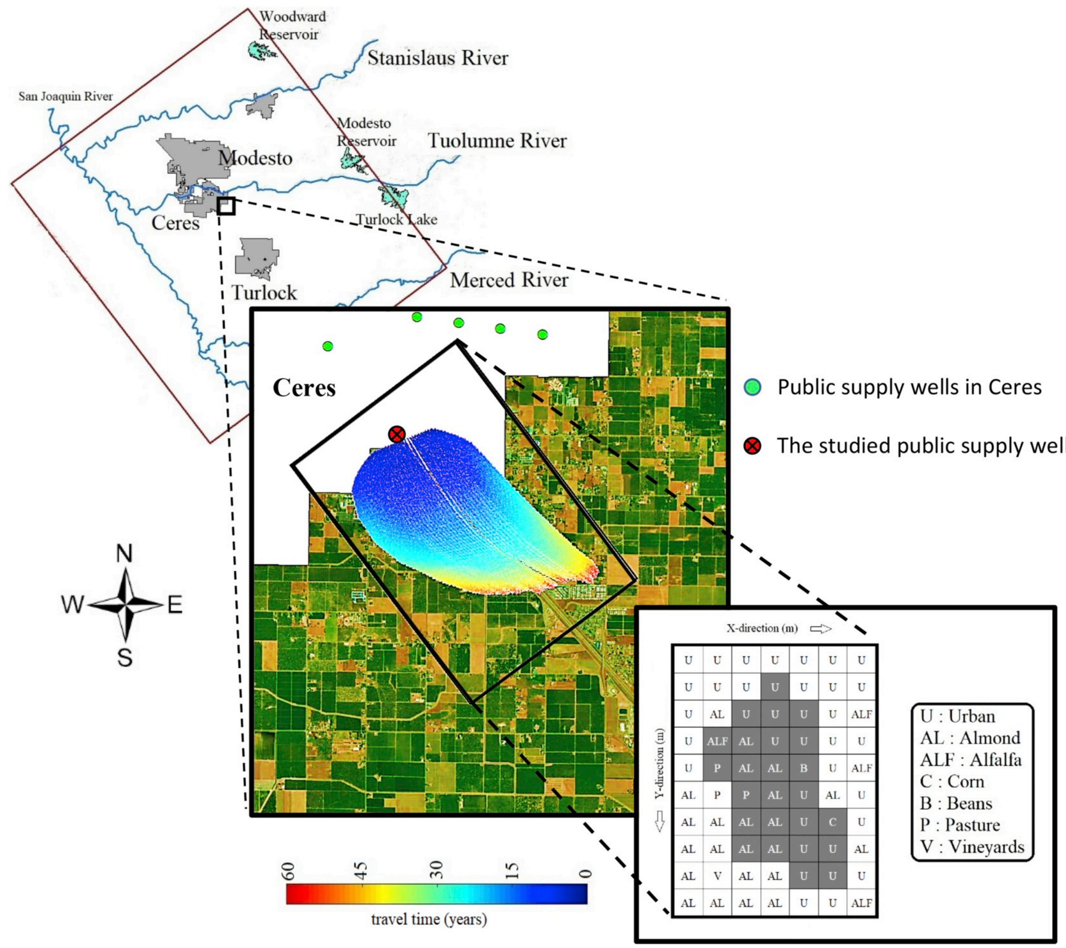


Fig. 4. Selected public supply well in the city of Ceres and its source area delineation.

Table 2
Basic parameters/inputs for numerical prediction of scenarios.

Parameter	Unit	Value
Input hydraulic parameters (Flow)		
Horizontal hydraulic conductivity range (K_H)	m d $^{-1}$	0.002–77.4
Vertical hydraulic conductivity range (K_V)	m d $^{-1}$	0.0018–19.38
Hydraulic conductivity for Corcoran Clay (K_H, K_V)	m d $^{-1}$	(0.002, 0.0018)
Specific storage range (S_s)	m^{-1}	3×10^{-6}–0.376
Input hydraulic parameters (Transport)		
Porosity range (n)	–	0.25–0.35
Longitudinal dispersivity (α_L)	m	11
Transverse horizontal dispersivity (α_{TH})	m	$0.1 \times \alpha_L$
Transverse vertical dispersivity (α_{TV})	m	$0.1 \times \alpha_{TH}$
Effective molecular diffusion (D*)	m^2 d $^{-1}$	ignored
Input reaction parameters (Reactive transport)		
Denitrification rate coefficient	–	ignored
Nitrate adsorption coefficient	–	ignored
Specified boundary conditions (Flow)		
Source packages	–	RCH, SFR, GHB
Sink packages	–	WEL, ET, GHB, SFR
Specified boundary conditions (Transport)		
Source packages	–	RCH, SFR, GHB
Sink packages	–	SFR, GHB

Table 3
Estimated nitrate mass load for simulation period of 1960–2004 (Modified from Harter et al., 2017).

Land use type	Nitrate load as N (kg/ha/yr)		
	Years 1960–1975	Years 1975–1990	Years 1990–2004
Fallow	0	0	0
Grain	73.3	48.8	89.3
Rice	7.9	20.3	39.1
Field crops	62.6	75.8	79.3
Cotton	27	54.1	115
Corn	60.7	63.6	153.2
Pasture	0	0	0
Alfalfa	30	30	30
Truck crops	82.7	133.4	185.7
Artichoke	50	50	50
Asparagus	109.8	144.7	143.6
Christmas tree	30	30	30
Strawberry	140.8	150.2	182.6
Almonds	29	48.9	136.9
Citrus tree	122.6	138.9	141
Vineyards	40.7	46.2	22.9
Natural vegetation	15	15	15
Urban	20	20	20

recharge in agricultural fields (Ag-MAR) provides additional recharge volume at a time when mobile nitrate availability in the vadose zone is presumed to be minimal. Ag-MAR is therefore assumed not to affect the amount of nitrate mass loss. Both management practices lead to reduction in concentration – one by reducing the nitrogen mass flux without similar reduction in recharge, the other by increasing the water flux without increasing nitrate mass flux (Fig. 5). One practice is focused on improving nutrient management, possibly by crop selection, the other focuses on increasing recharge in the source area, which may be achieved in a number of ways. Six scenarios were developed which are described in the following sub-sections.

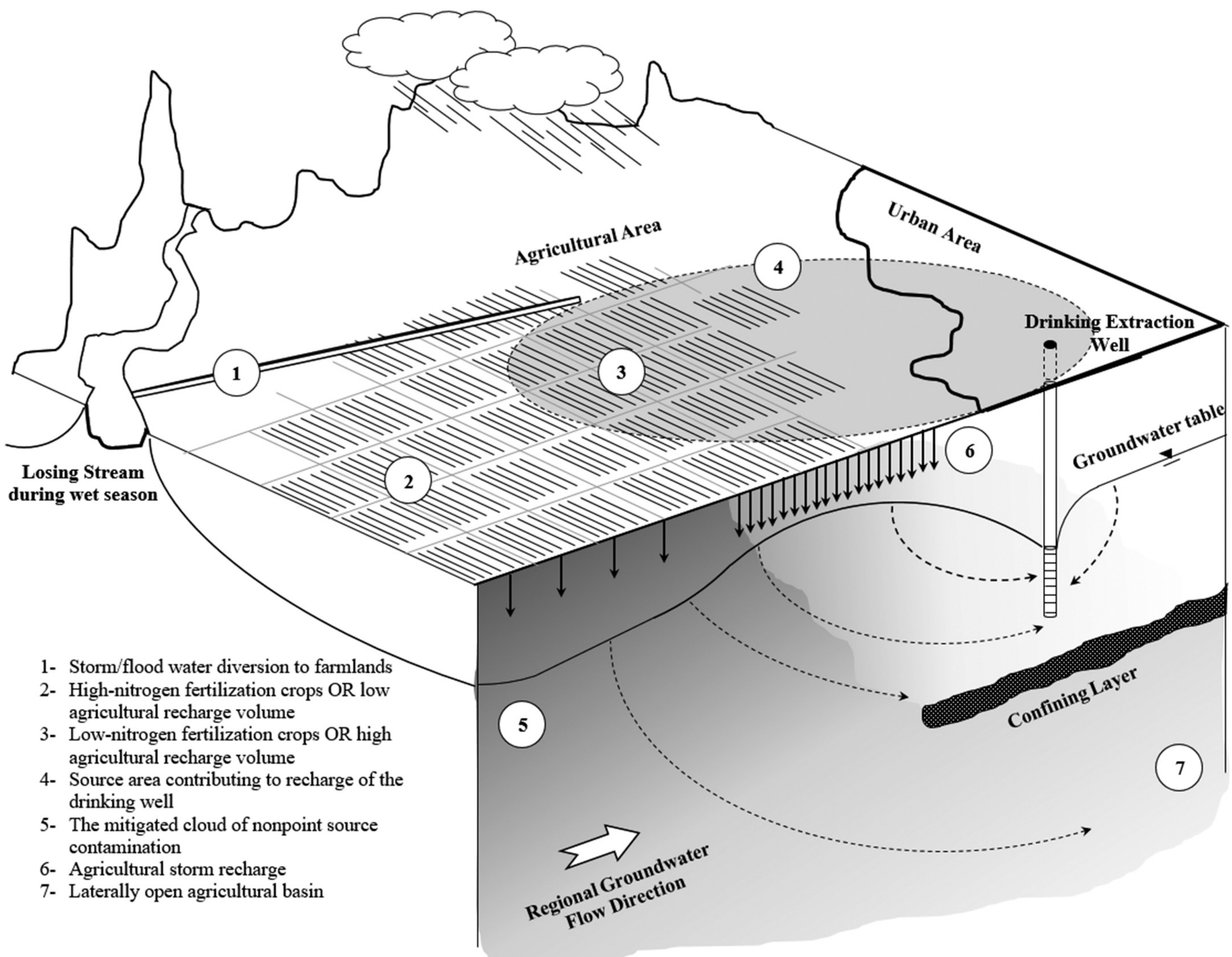


Fig. 5. Conceptual diagram of nitrogen load reduction and recharge volume increase at source area as two main agricultural management practices that potentially affect groundwater quality

2.3.1. Business as usual (BAU) scenario

The "business as usual" (BAU) scenario demonstrates the nitrate trends at the supply well in the absence of any changes to agricultural resources management during next 60 years. For this baseline scenario, we assume that the land use distribution during the last year of the Phillips et al. (2015) model continues for the remainder of the 60-year simulation period.

2.3.2. Low intensity crop (LIC) scenario

Harter et al. (2017) estimated the intensity of nitrate N mass leaching to groundwater [kg N ha^{-1} a^{-1}] from 58 crop categories in the Central Valley using a mass balance approach. Only alfalfa leaching was estimated from field experiments. Alfalfa is a nitrogen-fixing legume with little N fertilizer applied in practice. Non-crop sources of groundwater nitrate were assessed by reviewing permit records, literature sources, and by conducting surveys to estimate groundwater nitrate loading. Viers et al. (2012) introduced an operational benchmark of 35 kg N ha^{-1} a^{-1}, delineating low intensity from high intensity of nitrate leaching to groundwater. The benchmark was based on the maximum contamination level (MCL) of nitrate, 10 mg N/L, or 30 kg N ha^{-1} a^{-1} in 300 mm a-1 of recharge, which is a recharge rate typical for irrigated landscape, and assumed 10–20% denitrification of leached N. Crops with estimated leaching rates significantly above the benchmark include manured forage crops, vegetables, subtropical, tree-fruit, and nut crops. Crops with leaching rates at or below the benchmark include vineyards and alfalfa (Harter et al., 2017).

The LIC scenario reflects, for example, replacement of current high fertilizer intensity crop types (i.e., almonds in this study) with a low intensity crop (here: alfalfa) in the recharge source area of the public supply well, or a change in almond management practices that result in significantly reduced nitrate leaching rates, equivalent to those for alfalfa. The recharge rate for alfalfa lands was estimated from surrounding lands on which alfalfa was the main crop and had the same soil type. Concentration of nitrate in the recharged water was derived from the amount of nitrate load for alfalfa reported by Harter et al. (2017) (30 kg N ha^{-1} a^{-1}). Nitrate mass flux at the water table is assumed to be uniform throughout the year, without seasonal variations. This scenario may also represent other crop changes (e.g., to vineyards) that lead to this leaching rate.

2.3.3. Winter recharge (WR) scenario

Groundwater is one of the major sources for irrigation in semi-arid agricultural regions like the Central Valley of California. Optimum use of seasonal floodwater from excess rainfall to recharge aquifers is a promising approach to improve both quantity (Kocis and Dahlke, 2017) and quality of groundwater in these areas. Winter recharge in the agricultural landscape between November and April is achieved by directing stormwater runoff in streams onto farms via their irrigation system. We consider this approach here as a third scenario. Suitable crop types are here limited to those currently under consideration: almonds, alfalfa, and vineyards (Bachand et al., 2014). Winter recharge months include the period from January to June.

We consider two separate scenarios of winter recharge. The WR-Regular scenario assumes that recharge water is available each winter through water management arrangements that secure water for recharge from upstream reservoirs. A more constrained scenario, WR-Irregular, limits recharge to those winter months when additional floodwater is actually available, during wet years. Kocis and Dahlke (2017) estimated high-magnitude streamflow dynamics and availability for all major Central Valley streams. The public supply well studied is in proximity to the Tuolumne River (Fig. 4), which was used here to determine the frequency and amount of available floodwater.

In both scenarios, the annual winter recharge in alfalfa and almond was capped based on soil and agronomic considerations. Almond orchards are thought to accommodate up to 0.6 m/yr of winter recharge (Helen Dahlke, CWEMF presentation, 2018), while alfalfa is here assumed to tolerate winter recharge of up to 1.8 m/yr (Dahlke et al., 2018). Only source area locations overlying soils with "moderately good" to "excellent" rating in the California soil agricultural groundwater banking index (SAGBI) were considered here. The amount of recharge water diverted to the source area per monthly stress period was computed from the number of MODFLOW cells with landuse alfalfa or almond on suitable soils, the maximum allowable recharge rate on these crops, and (in WR-Regular) the length of the winter recharge period. In WR-Irregular, the diversion could not exceed the monthly available floodwater calculated for the Tuolumne River and was terminated once the annual cap for each crop was reached (Fig. 6).

2.3.4. Low intensity crop and winter recharge (LIC-WR) scenario

The fourth scenario combines the two previous scenarios (Table 4): Storm water is assumed to be available for recharge during the winter months and nitrate leaching in almond orchards of the source area is reduced to 30 kg N/ha/yr either by crop replacement or nutrient management changes. The total area of farmland receiving floodwater is the same as in the WR scenarios. The total volume of diverted water is calculated here assuming that LIC is achieved by changing landuse to alfalfa, allowing for 1.8 m/yr of recharge instead of 0.6 m/yr (Fig. 7). Both, regular recharge scenarios and irregular, hydrologically constrained recharge scenarios were simulated.

3. Results and discussion

3.1. Model validation

Concentration of nitrate in 170 production and domestic wells across the study region were obtained from various studies (https://www.waterboards.ca.gov/water_issues/programs/gama/online_tools.html) where screen depth was provided as part of the water quality data record. Data spanning the time period from 1960 to 2005 were assembled, although sampling frequency at any particular well and data density varies widely from decade to decade (Boyle et al., 2012; King et al., 2012). The measured water quality dataset was compared against the range of simulated nitrate-N concentrations at the same location, depth, and time as the measured wells, by decade, from 1960 to 2005 (Fig. 8).

Overall, measured data show a wider distribution than simulated data while simulated data capture long-term trends very well. The decadal medians of nitrate at measured locations on the date of sampling differ between measured and simulated datasets from < 0.5 (1990s) to < 2.0 mg N/L (1960s) indicating that simulated nitrate capture most of the measured variability but also the strong decadal concentrations trends: Measured and simulated nitrate-N concentration increase from < 5 mgN/L in the 1960s to over 7 mg N/L in the 1990s, but then show a decrease again in the early 2000s (Fig. 8). Both, measured and simulated data show increasing variability in nitrate-N concentrations among measured wells with time, especially during the last 15 years (1990–2005). The transport simulation captures regional changes in nitrate concentrations throughout time, but it does not capture the large variability among measured data.

For additional validation, a set of four sampling wells were selected from the data set, each with multiple nitrate sampling dates. All four wells are within or near study area cities (Fig. 9). Wells W1, W2, and W4 are in Modesto, Ceres, and Turlock, respectively and well W3 is near the public supply well of our case study (shown in Fig. 10). Each well had measured nitrate data available since at least 1995, sometimes earlier. Breakthrough curves for these specific wells demonstrate that the model is capable of capturing both the magnitude of nitrate concentration and long-term trends at a given location (Fig. 9), while it does not capture actual seasonal and annual fluctuations in nitrate concentration, some of which can be quite large.

The larger variability of the measured versus simulated data shown in Figs. 8 and 9 reflect multiple sources of variability in space and time that affect measured nitrate concentration but are not represented in the model: within-crop field-to-field and year-to-year variation in nitrate loading, farm-to-farm variability in nitrogen management that also affects groundwater nitrate loading (the model only represents long-term average differences in nitrate loading between crop types), aquifer heterogeneity at scales smaller than the model grid cell dimensions, and lack of accounting for localized denitrification processes, either during the recharge process or in groundwater. They may also represent sampling errors or unknown differences in the sampling protocol (e.g., sampling during pumping season versus sampling during non-pumping season).

The overall agreement between simulated and measured nitrate concentration during the validation period, even at individual well locations, and especially across the study region (Fig. 8) demonstrates that the selected modeling approach (nitrate loading as well as groundwater flow and transport) is sufficient to guide assessment and future decision-making and, more specifically, to develop a proof-of-concept for the scenarios simulated in this study. At the same time, the model validation serves to bracket the potential accuracy of the simulated outcomes and cautions for interpreting results as accurate in trend, relative difference, and magnitude, but not in absolute numbers.

3.2. Water quality improvements in the public supply well

The scenarios begin in the year 2005. In 2004, the last year under standard conditions across all scenarios, nitrate concentrations in the supply well were about 8 mg N/L and increasing. Under the BAU scenario, nitrate concentrations continue to increase, exceeding the MCL about one decade later. The rapid increase slows down after 2020, but nitrate concentrations continue to steadily increase throughout the simulation period. By 2065, rising nitrate-N is 20% above the MCL (Fig. 10).

3.2.1. Nitrate attenuation in alternative scenarios

All alternative source area management scenarios improved water quality in the public supply well relative to the BAU scenario. In fact, in all scenarios nitrate concentration improve relative to starting conditions in 2005, except in the WR-Irregular scenario. While improvements are notable relative to the BAU with WR-Irregular, the practice is insufficient to stop the rise in nitrate concentrations. It is merely slowed down. The continued rise in nitrate in concentration for WR-Irregular reflects continued high nitrate concentrations in recharge during years with no winter flood water applied in the source area. Relative to BAU, WR-irregular reduces nitrate concentration by < 10% during the 60-year scenario period. It is the least efficient technique in this study for nitrate attenuation at the public supply well. The results indicate that,

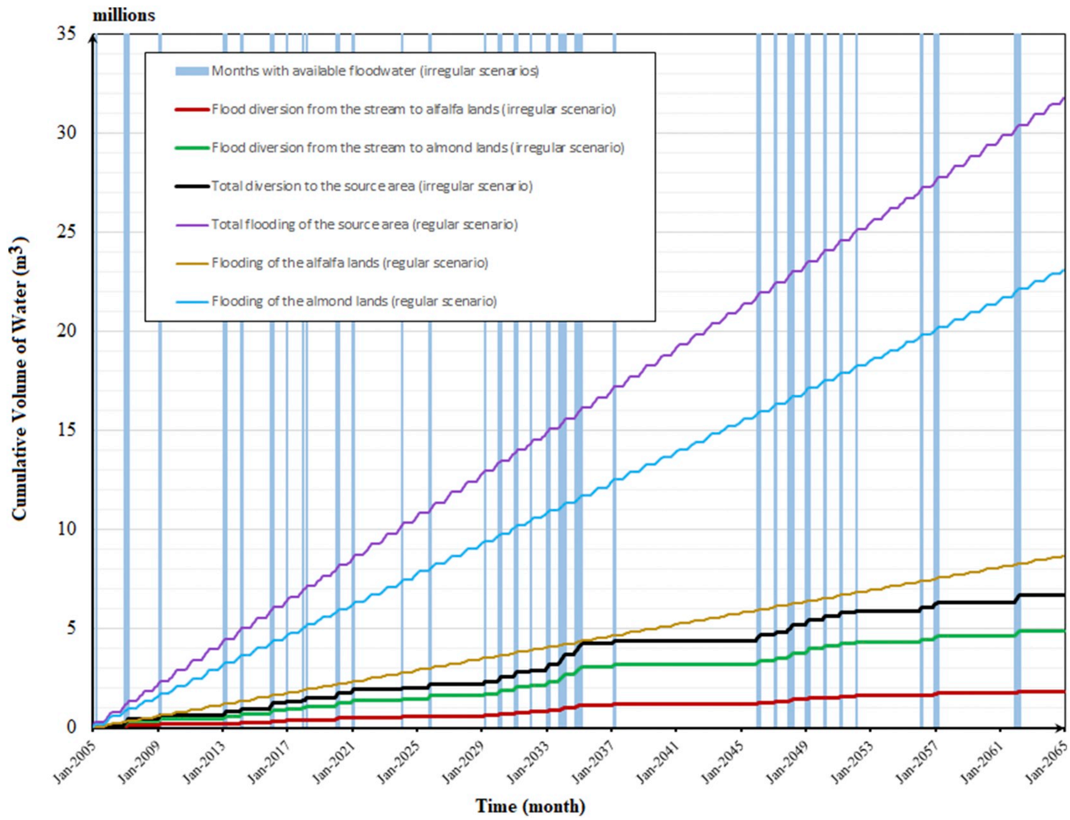


Fig. 6. The volume of extra recharge applied to the source area for WR scenario analysis

in this region, the application of winter recharge during years with sufficient flood water has benefits with respect to groundwater storage, but water quality benefits are relatively limited without additional measures, especially where nitrate concentrations are already near or above the regulatory level.

To achieve a more desirable long-term groundwater nitrate concentrations, without changes in nutrient management practices or crops, regular winter recharge is needed (scenario WR-Regular). This could be achieved for example, through water management arrangements with upstream reservoirs to deliver an additional 0.53 MCM (million cubic meters) per year (430 acre-feet/year) for winter recharge. The WR-Regular scenario reduces nitrate to stable levels near 9.5 mg N/L, below the MCL, but not lower than in 2005. The additional recharge is sufficient to dilute the nitrogen loading in the source area to levels not exceeding the MCL. The decrease in supply well concentration relative to BAU is three times larger than in the case of the WR-Irregular scenario.

The WR-Regular scenario also leads to a very small (< 5%) increase in nitrate concentration, relative to BAU, during the early years, immediately after the beginning of scenario, and before nitrate concentrations improve. The slightly higher nitrate concentrations in the WR-Regular scenario are due to the acceleration of flow between the recharge area, where water levels are higher, relative to BAU, and hydraulic gradients to the well are steeper, leading to faster travel times relative to BAU. The ameliorated higher concentrations, prior to arrival of recharge water at the well, therefore arrive earlier than in the BAU scenario, causing the relatively higher concentrations prior to arrival. In this case, the small increase would be of little practical consequence, however. We note that our WR scenarios did not consider the mobilization of existing legacy nitrate that may be stored in the unsaturated zone, by Ag-MAR.

The LIC scenario, under which nitrate leaching at the water table in the source area is limited to 30 kg N/ha/yr, e.g., by changing the crop type to alfalfa, begins to show a slow-down in the increase in nitrate concentrations within 5–7 years after initiation, then leads to a rapid decrease in nitrate concentrations after 2012 to levels that are about half of the nitrate concentration in 2005. Combining the LIC scenario with winter recharge leads to even more rapid response time and larger decrease in nitrate levels. The LICWR-Irregular scenario displays similar extent of nitrate removal as the LIC scenario. The highest nitrate attenuation occurs with implementation of the LICWR-Regular practice. This scenario removes > 80% of nitrate concentration from the drinking water at the public supply well after 60 years.

Table 4
Source area management scenarios.

Scenario	Current crop land use	Crops in source area changed to alfalfa	Current recharge rate	Recharge rate increased in source area	Regular winter recharge	Irregular winter recharge
BAU	✓	–	✓	–	–	–
LIC	–	✓	✓	–	–	–
WR-Regular	✓	–	–	✓	✓	–
WR-Irregular	✓	–	–	✓	–	✓
LICWR-Regular	–	✓	–	✓	✓	–
LICWR-Irregular	–	✓	–	✓	–	✓

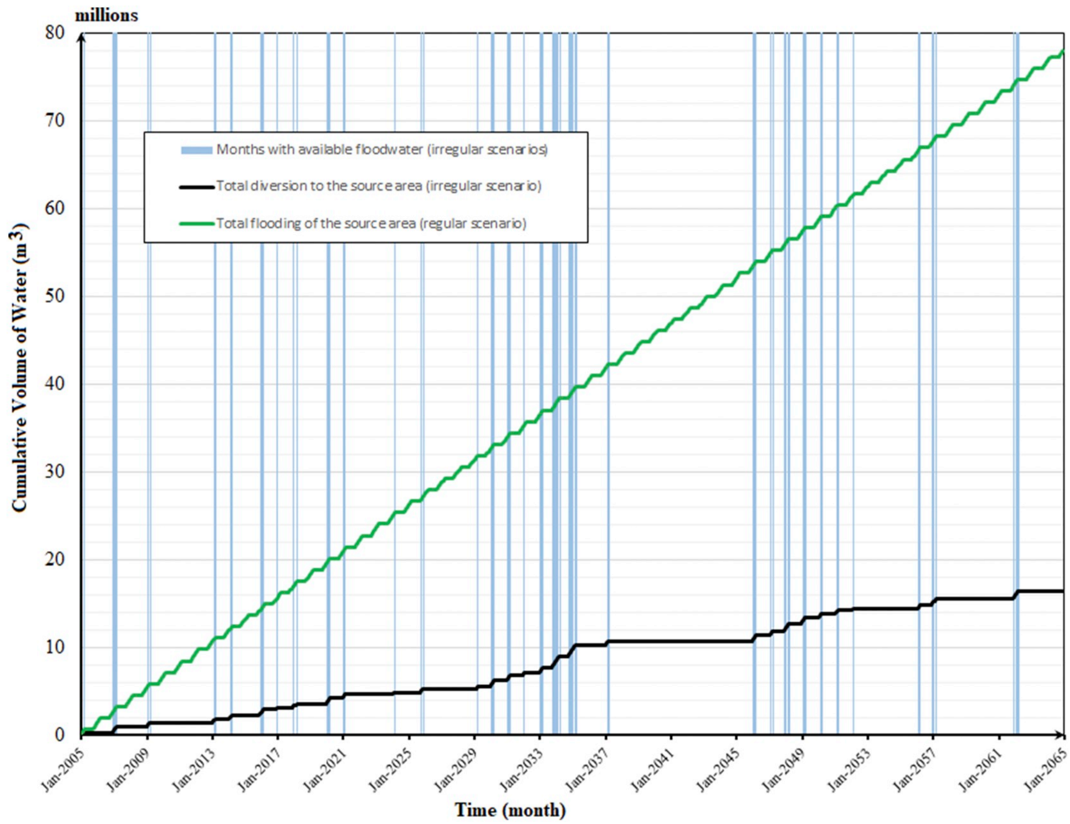


Fig. 7. The volume of extra recharge applied to the source area for the LIC-WR scenario analysis.

3.2.2. Response time for water quality improvements

Under all alternative scenarios, the rise in nitrate concentrations at the supply well continues for at least 5 years. This is expected given the travel time between the source area and the supply well screen location in the aquifer (Fig. 4). Under the most aggressive scenario, LICWR-Regular, concentrations begin to rapidly decrease after 2010. Winter recharge in average and wet years only, WR-Irregular, is found to be the least aggressive scenario, but even with this approach, improvements over BAU can be seen beginning in 2015, ten years after the scenario's initiation. This lag time is at about the time at which the supply well exceeds the nitrate MCL in both, BAU and the WR-Irregular scenario (Fig. 10). Other scenarios have an intermediate lag time between 5 and 10 years, before improvements begin.

Stable, steady nitrate concentrations in the public supply well are achieved only after a much longer period of time (Fig. 10), ranging from three decades in the LICWR-Regular scenario to nearly six decades in the LICWR-Irregular scenario. BAU and WR-Irregular do not reach steady conditions within the simulation period.

3.3. Water quality improvements in ambient groundwater

Rural households, outside community water supply boundaries, typically rely on shallow domestic wells with low extraction rate (< 0.002 MCM per year). The simulated scenarios offer significant insight on ambient groundwater quality relevant to these domestic wells. For the analysis, we consider the vertical nitrate profile throughout the capture zone of the public supply well. We average nitrate concentration across all depth-specific cells within the flow path zone emanating from the source area to the supply well. This averaging process includes only those cells that intersect the flow paths between source area and well (colored area in Fig. 11-top).

Flow paths to the public supply well are located above the Corcoran clay layer. For the BAU scenario, nitrate concentration above this layer is much higher than underneath the Corcoran clay. Highest BAU concentrations are found near the water table. Concentrations decrease with depth. Implementation of alternative future scenarios significantly changes ambient nitrate distribution above this confining layer relative to BAU.

WR-Irregular improves ambient groundwater in the shallow-most groundwater, within the upper 35 m. On the other hand, the WR-Regular scenario shows 20% more reduction of nitrate during the same period. The comparison of WR-Irregular and WR-Regular scenarios reveals the importance of recharge frequency to groundwater quality improvement that is relevant to shallow domestic wells. LIC doubles the percentage of nitrate improvements within ambient groundwater. Consistent with the breakthrough curves at the supply well, ambient groundwater quality improves most under LICWR scenarios. LICWR-Regular scenario improved ambient groundwater quality by 60% after 60 years of application.

Results confirm findings in similar groundwater modeling scenarios implemented for the Alta Irrigation District in the southern part of the Central Valley, California (Luhdorff and Scalmanini Consulting Engineers (LSCE) and Larry Walker and Associates (LWA), 2016). The study analyzed the application of selected practices including recharge, nutrient management, abandonment of agriculture, and local pump-and-fertilize scenarios across the entire region, not focused on the source area of wells. Surface restoration activities and nutrient management schemes were shown to improve the ambient groundwater quality in small geographic settings. On-farm recharge practices were shown to flush the root zone and move low-quality shallow water to deeper zones of the aquifer. Also, it was found that pump, treat, and reinject plans were not practicable for implementation across larger regions ($\gg$1 km^2) (Luhdorff and Scalmanini Consulting Engineers (LSCE) and Larry Walker and Associates (LWA), 2016, King et al., 2012).

3.4. Source area management and costs

Under the BAU scenario, the public supply well produces 69 MCM

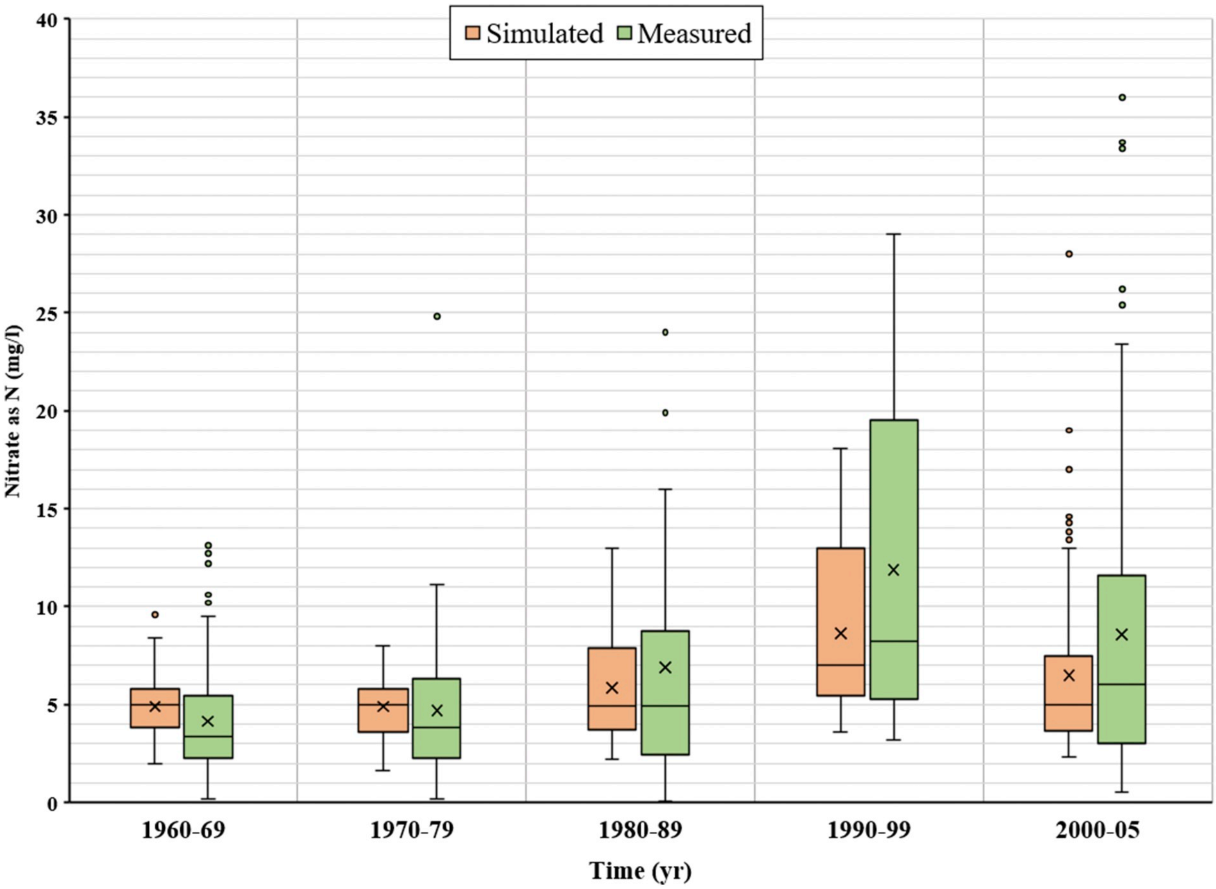


Fig. 8. Range of simulated nitrate concentration at the study region compare to the measured concentration, by decade. Boxplots show the median and interquartile range with whiskers representing data outside of the inter-quartile range and outliers shown as dots.

over the 60 scenario years (1.15 MCM/year) from a source area that extends over 400 ha (1000 acres, 25 model cells). That corresponds to an average BAU scenario source area recharge rate of 0.29 m/year. Nearly half of the source area (12 cells) is in urban land use (Fig. 4). Eight cells (128 ha, 320 acres) are almond orchards and one cell (16 ha, 40 acres) is an alfalfa field. The remainder of the area (4 cells) is in pasture, corn, and beans. Winter recharge water (WR scenarios) was diverted to almond and alfalfa land use (Table 5). In the LIC scenario, almond land use was replaced by alfalfa, improving nitrate mass loading to the water table. In the LIC-WR scenarios, the LIC land use change to alfalfa also allows for additional recharge, as alfalfa is more tolerant of winter irrigation.

The additional recharge to the system is lowest in the WR-Irregular scenario (9.5 MCM) but 8.2 times larger in the highest recharge scenario, LICWR-Regular (78 MCM). The difference in recharge between WR and LICWR scenarios is about 2.5-fold. Over the 720 month scenario period, 108 months were wet with floodwater available for the "Irregular" scenarios. The volume of water available for the Regular scenarios is about 3 times larger than under the corresponding irregular winter recharge scenario. The additional recharge water represents 14, 46, 33, and 113% of the public supply well pumping rate for the WR-Irregular, WR-Regular, LICWR-Irregular, and LICWR-Regular scenario, respectively.

The economic feasibility of the alternative scenarios is driven by the cost of additional winter recharge operations and the net cost of land use conversion to alfalfa or other low nitrogen intensity crops. Current irrigation water rates in the area vary from $0.0016/m^3 to $0.032/m^3 (http://www.mid.org/water/irrigation/allocation.html) compared to at least $0.50/m^3 for domestic water charged by community water suppliers (https://www.ci.ceres.ca.us/DocumentCenter/View/2192/Water-Rate-and-Connection-Fee-Update). For the scenario supplying the largest amount of water (LICWR-Regular), the cost of the additional recharge water (113% of the water pumped at the public supply well) would therefore be as much as 7% of the current cost to public water supply users (1.13 * $0.032/$0.50), assuming that the cost of winter recharge water is equal to the highest current irrigation water rate, including any operational and permitting expenditures.

Land use conversion from almond orchard to alfalfa requires upfront investments and, more importantly, will lead to decreased production income (Mayzelle et al., 2015). The initial cost to establish an alfalfa field is relatively small, at $3140/ha (Frate et al., 2008), while net total economic losses are on the order of $8000/ha/year due to the much lower net profit from alfalfa production when compared to almond production (Mayzelle et al., 2015). The cost of the conversion of 128 ha of almond orchard to alfalfa would be equivalent to $0.90/m^3 of water produced at the public supply well. Assuming that the additional cost would be born by the supply well utility, this would bring costs within the range of typical surface water-based drinking water supply in California (California Public Utilities Commission, 2016).

The cost analysis for the LIC and LIC-WR scenarios using alfalfa is likely a worst cost case. It does not account for flood risk reduction benefits, which may outweigh these costs; and it does not consider more valuable alternative low nitrogen intensity crops, including improved nitrogen management in the almond orchard or conversion to vineyard landuse (Mayzelle et al., 2015): Conversion of the almond orchard to wine grape vineyard, a crop that has similar low nitrate mass losses to groundwater as alfalfa, is suitable for winter recharge, and grown commonly in the area, and would likely erase production losses over the current almond production (California Department of Food and Agriculture, 2017). However, establishing the vineyard is expensive, on

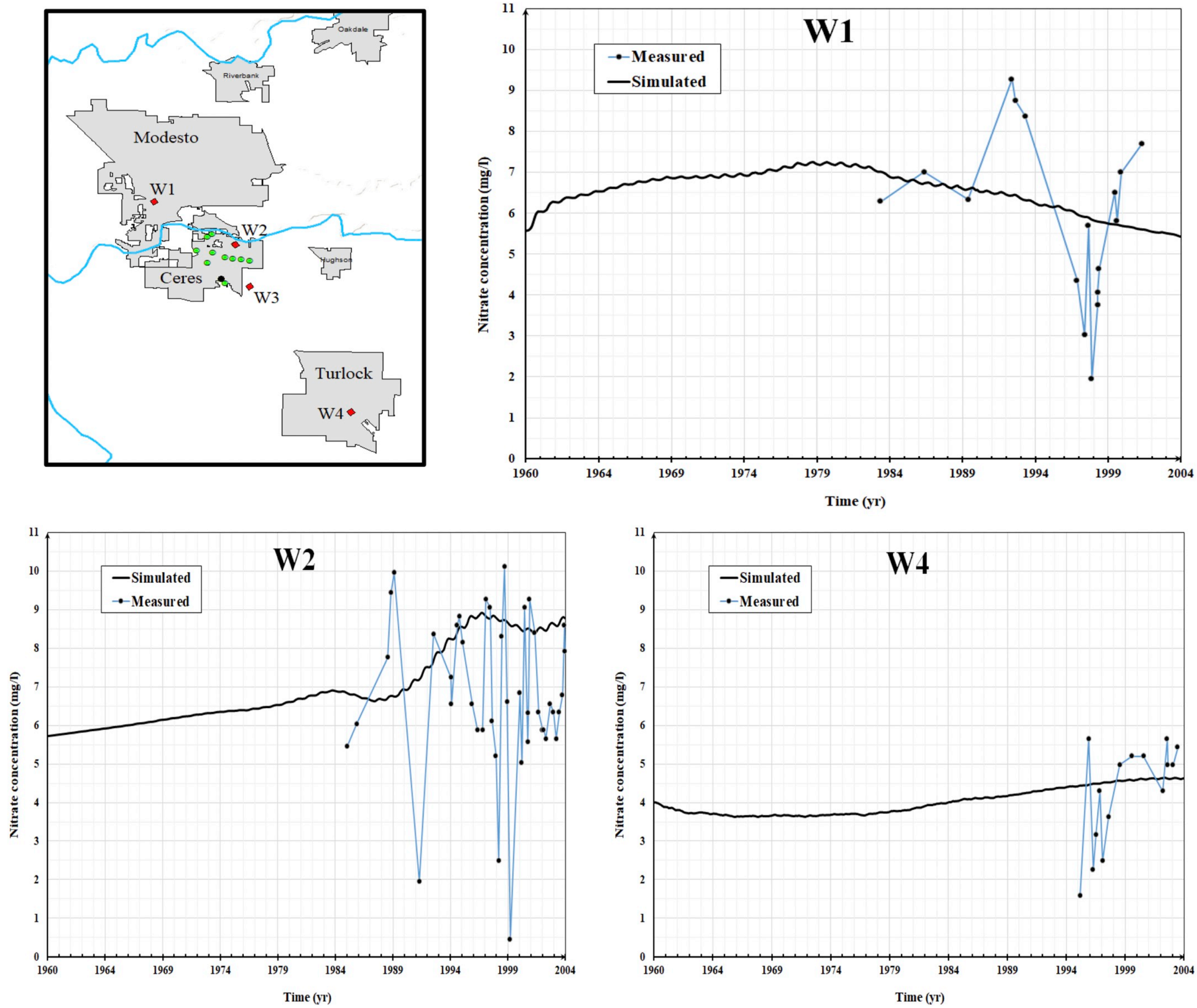


Fig. 9. Variation of simulated nitrate concentrations versus the available measured data at some pilot points derived from GAMA program.

the order of \$40,000/ha (Verdegaal et al., 2012). When spreading this cost over 20 years for the 128 ha of current almond orchard, this amounts to \$0.22/m^3 of water pumped from the public supply well. The cost is significantly lower than the conversion to alfalfa, but much higher than regular winter recharge. The costs compare favorably to well head treatment, another alternative available to the public supply well operator: Ion exchange treatment for nitrate is a common approach with total costs in the range from \$0.19 to \$0.57/m^3 (Jensen et al., 2014).

Another source area management alternative, foregoing both conversion cost and profit losses, would be to implement rigorous nutrient management in the almond orchards within the source area. Given the high nitrogen input to almond orchards, it may be difficult to achieve N losses not exceeding 30 kg N/ha/yr at the water table even under very efficient management practices. However, LIC scenario results indicate that nitrate concentrations at the public supply well would stabilize below the MCL even if nitrate leaching under the almond orchard is twice the amount simulated (that is, 60 kg N/ha/yr), which is still far less than under the BAU scenario. Adding WR to such nutrient management in the source area, would provide further relief and additional insurance for keeping water quality in the supply well below the MCL over the long term.

3.5. Implications for region-wide groundwater quality protection

Our findings can be used at regional or state scale for nitrate removal plans. The total number of community water systems in California that rely on groundwater as a primary source of drinking water is around 2584 which use 8396 active wells serving 4.1 million people (Department of Water Resources, 2013). Many of these groundwater-reliant community water systems are in rural areas throughout the state (State Water Resources Control Board, 2013). Nitrate was detected above MCL in 451 wells associated with 205 community water systems (DWR, 2013). Therefore, approximately 8% of community water systems in California have nitrate problem (SWRCB, 2017), which serve drinking water to 328,000 people statewide, many of them in the Central Valley. The water supplied to these communities is treated and delivered at a cost typically on the order of \$0.30–0.60/m^3 (Honeycutt et al., 2012). If we assume a per capita water consumption of 0.7 m^3/d (Mount et al., 2014), total groundwater pumping in these wells is on the order of 84 MCM/yr.

If addressed through source area management including winter recharge, the case study presented here suggests that recharging nearly the same amount of water pumped under the LICWR-Regular scenario

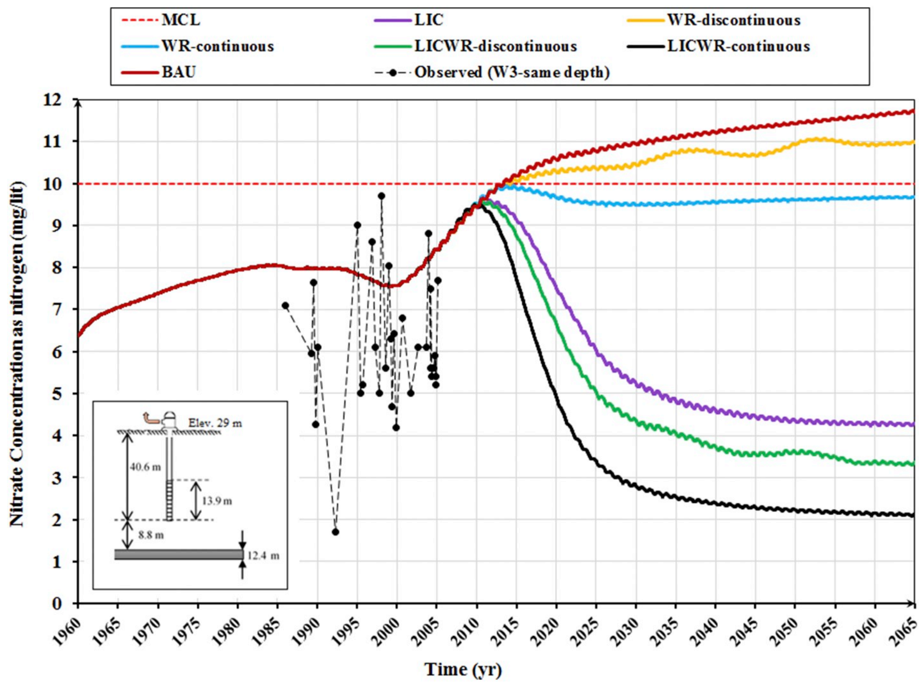


Fig. 10. Nitrate timeline in the public supply well for various source area management practice scenarios and concentration data available at an observation well (i.e., W3) near the study PSW

provides significant groundwater quality improvements. Statewide, that amounts to 84 MCM per year of winter recharge focused in the source area of at-risk public supply wells. As the number of community water supply systems that will exceed the nitrate MCL will likely rise significantly over the next twenty years (Honeycutt et al., 2012), the need for such recharge may increase to as much as 200 MCM. For comparison, the estimated amount of unused flood water available in the Central Valley during an average or wet year is 3200 MCM (Kocis and Dahlke, 2017). A significant fraction of the amount of flood water available for recharge may be needed to implement WR or LIC-WR type scenarios as long-term alternatives to drinking water treatment. On the other hand, winter recharge would not suffice to address much of the domestic well water pollution due to the wide-spread occurrence of domestic wells and high nitrate groundwater concentration (Ransom et al., 2017).

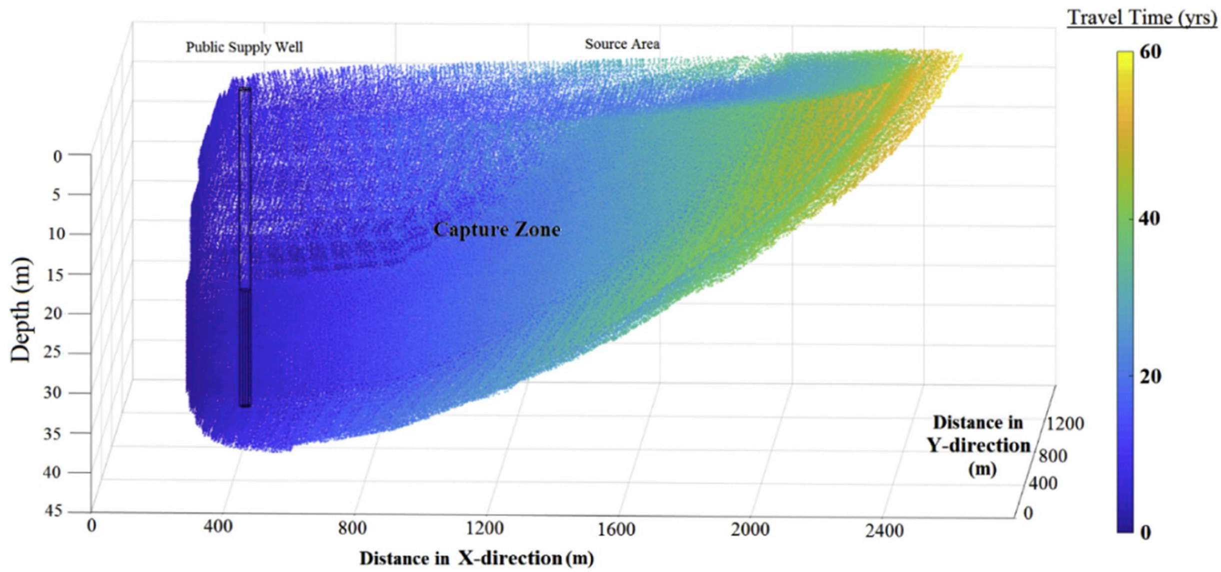
3.6. Limitations and uncertainties

Through the process of developing the flow and transport simulations presented in this study, some conceptual assumptions and a set of numerical parameters were used which are sources of uncertainty in the expected nitrate concentration values. While the underlying, transient flow model was informed by detailed hydrogeologic architecture data and calibrated against measured water levels and stream flows (Phillips et al., 2015) and while modeled nitrate concentrations and trends agree with measured data in the region for 1960 to 2005, the transport model does not reflect some source and aquifer variability affecting measured nitrate in wells (Section 3.1). Hydraulic properties, porosity, and hence pore water velocities are among the most variable and uncertain parameters, resulting in uncertainty about travel time, but also about the exact location of the source area (Henri and Harter, 2019), despite the significant amount of aquifer heterogeneity captured by the underlying geologic framework. Historic and current nitrate loading is based on published land use maps, statistics, and reported average nutrient management practices and average harvest rates. Soil nitrogen accumulation, local denitrification, and other processes may significantly affect the actual nitrate leaching rate. Ransom et al. (2018), developed N loading estimates based on groundwater nitrate measurements, confirming the overall magnitude of groundwater nitrate loading used here, but also demonstrating that rates vary widely and may be lower for nut crops than estimates based on field N surplus. For actual project implementation, this will require improved hydrogeologic characterization of the likely source area of a supply well, or alternatively, the designation of a larger source area zone, within which targeted management practices are applied. Monitoring of water quality near the source zone would provide early confirmation of potential improvements in water quality.

Unsaturated zone travel time, complex root zone processes, including denitrification, N mineralization, N uptake into soil organic matter may vary widely within the Central Valley or other agricultural regions with similar climatic and hydrogeologic conditions. Unsaturated zone travel time may be particularly long in (semi-)arid regions with highly efficient irrigation systems, minimal annual recharge rates much smaller than 300 mm, and in regions with deep vadose zones (> 20 m) (Harter et al., 2005). On the other hand, there is significant evidence for preferential flow and rapid vadose zone transport under irrigation (Botros et al., 2011; Turkeltaub et al., 2014). Under certain climate and soil conditions, fertilizer N undergoes significant storage in the soil carbon pool (Sebilo et al., 2013) and increase N storage in the root zone (Geisseler and Scow, 2014), before being released. In the Central Valley, therefore, considering unsaturated variations to the simulations could increase the modeled nitrate values and lengthen the response time in the system.

Here, soil and deep vadose zone denitrification processes were considered by assuming a 10% loss of the N surplus prior to recharge (Harter et al., 2017). Groundwater denitrification was neglected, thus leading to conservatively high concentration predictions. For the early time predictions (2005–2020), actual denitrification rates in the oxic aquifer system are typically sufficiently small enough to not justify the assumptions. Longer transport may in fact be significantly affected by denitrification, especially when transport occurs through the clayey fractions of the alluvial aquifer system architecture (McMahon et al., 2008).

This work also neglected seasonal variations in nitrate loading and relied on an average annualized loss rate. Under the Mediterranean climate conditions in the Central Valley, leaching losses during the



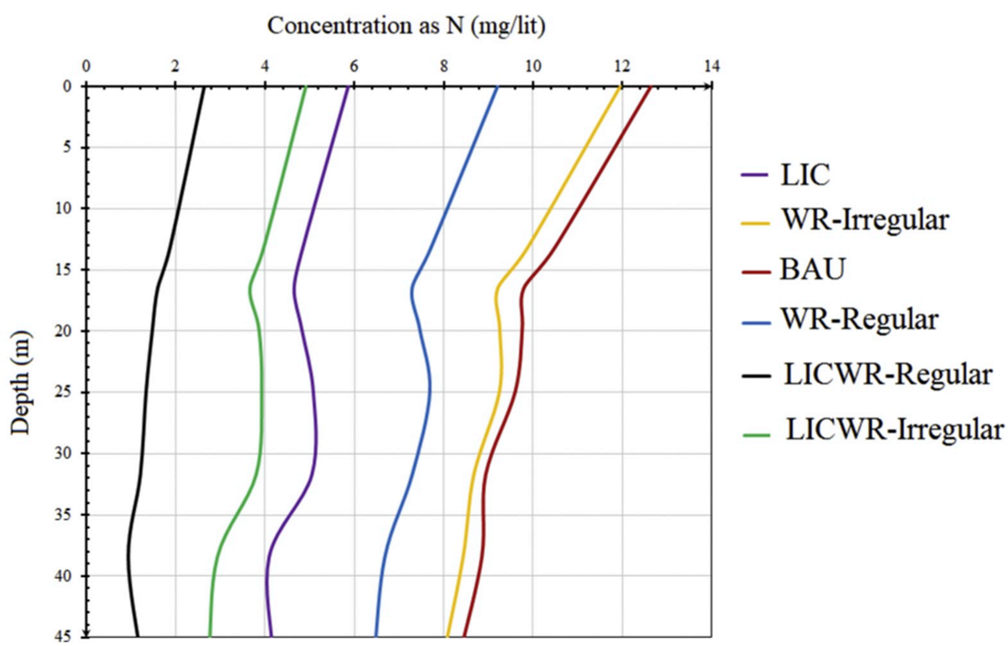


Fig. 11. (top) Three-dimensional capture zone of the well, and (bottom) the average nitrate profile throughout the capture zone of the public supply well after 60 years

Table 5
Winter recharge volume, rate, and land area, without and with land conversion of almonds to alfalfa.

Scenarios	ΔV (m^3)	ΔR (m^3/yr)	Δr (m/yr)	ΔD (m)	A (m^2)
WR-Irregular	9.5×10^6	1.6×10^5	0.11	6.7	1.44×10^6
WR-Regular	31.8×10^6	5.3×10^5	0.37	22.1	1.44×10^6
LIC	0	0	0	0	1.28×10^6
LICWR-Irregular	23×10^6	3.8×10^5	0.26	15.8	1.44×10^6
LICWR-Regular	77.9×10^6	13×10^5	0.90	54.2	1.44×10^6

ΔV: total volume of recharge water applied to source area over 60 years.
ΔR: average volumetric recharge rate added to the alfalfa / almond land use.
Δr: average recharge rate added to the alfalfa/almond land use.
ΔD: total depth of extra water replenished in the alfalfa / almond lands after 60 years.
A: area over which scenario changes apply

growing season are largely controlled by irrigation. Efficient water and nutrient management practices may lead to minimal water and nitrogen losses during the growing season, even in crops with high N demand, if properly managed (Baram et al., 2016). Residual nitrogen remaining after harvest is subject to root zone flushing by winter rains, a major source of recharge under effective management. Under more traditional

BAU management practices, significant recharge occurs during the growing season combined with N losses occur due to irrigation non-uniformity and low irrigation efficiency (Alva et al., 2006). While these seasonal and daily dynamics significantly affect leaching of nitrate, applying an annualized, average nitrate leaching rate was considered appropriate here, not only because of the lack of more detailed input data, but also because of the mixing of water of a range of age in supply wells. Additional research is needed to better understand this assumption on nitrate dynamics in water supply wells of interest.

The study here focuses on a proof of concept and on the overall qualitative outcome of WR and LIC type management scenarios in the source area of public and domestic water supply wells. The study findings, relative to the BAU scenario, are significant, despite the uncertainties about site specific conditions. To the degree that the setting is typical of many semi-arid, irrigate agricultural systems overlying unconsolidated sedimentary aquifer systems in Mediterranean climates, the findings here generally apply not only to the Central Valley, but to regions around the globe, although time- and spatial scales will vary depending on site conditions.

4. Conclusions

In this study, the numerical modeling approach was used to assess changes in nitrate concentration of a public supply well and in ambient groundwater due to implementation of protective agricultural practices in the well's source area. The conceptual scenarios include increasing recharge and decreasing nitrate mass load to groundwater.

Simulated breakthrough curves at the public supply well and profile distributions of nitrate in the capture zone indicate that the by far most aggressive management practice is to apply both techniques simultaneously: (Alva et al., 2006) achieve lower nitrate losses in the source area by replacing high nitrogen demand crops with low nitrate leaching crops (e.g., alfalfa or vineyards) or by aggressive nutrient management improvements, and (Bachand et al., 2014) add more clean water to the source area by diverting stored floodwater to fields suitable for winter recharge (Ag-MAR) in the source area. This scenario, LICWR-Regular, was found to achieve the quickest and most reliable results. First improvements were obtained in the public supply wells within less than a decade and full attenuation was achieved within 40 years. Nitrate levels were 80% lower than under the BAU scenario in 2065.

In the study area analyzed here, other scenarios, some involving low nitrate mass loading to groundwater only (LIC), some focused on winter recharge only (WR) were able to reverse the trend of increasing nitrate and stabilizing nitrate concentrations at levels below the MCL. WR scenarios led to a slight increase in nitrate prior to nitrate improvements being observed in the public supply well. This was due to the faster travel of legacy nitrate in groundwater following an increase in hydraulic head gradients due to the recharge.

The two scenarios that failed to provide safe drinking water are the BAU scenario and a winter recharge scenario that relies on the highly irregular occurrence of flood water (WR-Irregular). At the study site, the latter scenario provided relative improvements to the BAU with respect to nitrate concentrations in the supply well, but not sufficient to meet regulatory requirements.

Aggressive nitrate mass reduction was here achieved by replacing current high nitrate leaching crops (almond orchards) with alfalfa or wine grape vineyards, both of which have low N fertilizer requirements and can be managed to low nitrate leaching losses. Changing lands to alfalfa or vineyard has two benefits: (Alva et al., 2006) it increases the capacity of lands to tolerate extra water and therefore allows for more winter recharge than the almond orchard, (Bachand et al., 2014) it decreases the leaching mass of nitrate to groundwater. Combining nutrient management, even in a high fertilizer use crop like almonds, with winter recharge may provide sufficient improvements in groundwater quality within the source area to meet water quality at the public supply well after an initial lag time.

Given the cost of recharge water, assuming that it does not exceed current high-end rates in the study region, regular winter recharge would be the most affordable option, adding < 10% to current water rates of groundwater dependent community water system users. Given that almonds make up about one-third of the source area land use, conversion of these orchards to alfalfa would significantly improve long-term water quality at the public supply well, but at costs that are at or above alternative treatment costs, due to the loss in economic productivity. Converting to higher value, low nitrogen emitting crops such as vineyards may economically be more advantageous than well head treatment, in the long term, despite significant conversion costs. A combination of significantly improved nutrient management in the almond orchards and winter recharge may also represent a promising, less costly alternative to achieving good water quality in the water supply well.

These results provide promising alternatives to well head treatment and other technical solutions, at least for achieving long-term results or to prevent further degradation, where nitrate levels do not already exceed the MCL. Results are scalable to the entire region, given the similar climate and the overall water supply conditions.

This study does not address additional questions regarding the ability to store and redistribute available flood water in the Central Valley to source areas on a regular, if not continuous basis. Further work is also needed to identify incentive programs that public water supply agencies may pursue in their source areas, in collaboration with growers and grower organizations, to incentivize appropriate land use changes, winter recharge (Ag-MAR), and improved nutrient management practices.

Acknowledgements

This work was funded by California State Water Resources Control Board grant agreement No. 15-062-250. Also, we would like to thank Steven Phillips for kindly providing us MODFLOW-OWHM transient model of the study area.

References

Alva, A.K., Paramasivam, S., Fares, A., Delgado, J.A., Mattos Jr., D., Sajwan, K., 2006. Nitrogen and irrigation management practices to improve nitrogen uptake efficiency and minimize leaching losses. J. Crop Improv. 15 (2), 369–420. https://doi.org/10.1300/J411v15n02_11.

Bachand, P., Horwath, W.R., Roy, S.B., Choperena, J., Cameron, D., 2014. Implications of using on-farm flood flow capture to rechargegroundwater and mitigate flood risks along the Kings River, CA. Environ. Sci. Technol. 48 (23), 13,601–13,609. https://doi.org/10.1021/es501115c.

Baram, S., Couvreur, V., Harter, T., Read, M., Brown, P.H., Kandelous, M., Smart, D.R., Hopmans, J.W., 2016. Estimating nitrate leaching to groundwater from orchards: comparing crop nitrogen excess, deep vadose zone data-driven estimates, and HYDRUS modeling. Vadose Zone J. 15. https://doi.org/10.2136/vzj2016.07.0061.

Bexfield, L.M., Jurgens, B.C., 2014. Effects of seasonal operation on the quality of water produced by public-supply wells. Groundwater 52, 10–24 (Burow, K.R., Nolan, B.T., Rupert, M.G., Dubrovsky, N.M., Nitrate in groundwater of the United States, 1991-2003, Environ. Sci. Technol., 44 (2010), pp. 4988–4997).

Botros, F.E., Onsoy, Y.S., Ginn, T.R., Harter, T., 2011. Richards equation–based modeling to estimate flow and nitrate transport in a deep alluvial vadose zone. Vadose Zone J. 11. https://doi.org/10.2136/vzj2011.0145.

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. In: Report for the State Water Resources Control Board Report to the Legislature. Center for Watershed Sciences, University of California, Davis.

Burow, K.R., Shelton, J.L., Dubrovsky, N.M., 2008. Regional nitrate and pesticide trends in ground water in the eastern San Joaquin Valley. California J. Environ. Qual. 37 (5S-249-S-263).

Karen, Karen R., Nolan, Bernard T., Rupert, Michael G., Dubrovsky, Neil M., 2010. Nitrate in groundwater of the United States, 1991-2003. Environ. Sci. Technol. 44 (13), 4988–4997. https://doi.org/10.1021/es100546y.

California Department of Food and Agriculture, 2017. California Agricultural Statistics Review 2016–2017. United States Department of Agriculture.

California Public Utilities Commission, 2016. What Will be the Cost of Future Sources of Water for California?.

Dahlke, H., Brown, A., Orloff, S., Putnam, D., O'Geen, T., 2018. Managed winter flooding of alfalfa recharges groundwater with minimal crop damage. Calif. Agric. 72 (1),

65–75. https://doi.org/10.3733/ca.2018a0001.
Department of Water Resources, 2013. "California's Groundwater Update 2013: A Compilation of Enhanced Content for California Water Plan Update 2013", California, April 2015.
Department of Water Resources, 2018. "Flood-MAR: Using Floodwater for Managed Aquifer Recharge to Support Sustainable Water Resources", California, June 2018.
Dzurella, K.N., Darby, J., De La Mora, N., Fryjoff-Hung, A., Harter, T., Hollander, A.D., Howitt, R., Jensen, V.B., Jessoe, K., King, A.M., Lund, J., Medellín-Azuara, J., Pettygrove, G.S., Rosenstock, T.S., 2012. "Nitrogen Source Reduction to Protect Groundwater. Technical Report 3 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. Center for Watershed Sciences, University of California, Davis.
Eberts, S.M., Thomas, M.A., Jagucki, M.L., 2013. The quality of our Nation's waters - Factors affecting public supply well vulnerability to contamination - Understanding observed water quality and anticipating future water quality: U.S. Geological Survey Circular 1385. pp. 120. Available online at. https://pubs.usgs.gov/circ/1385/.
Frate, C., Mueller, S., Campbell-Mathews, M., Canevari, M., Klonsky, K., de Moura, R., 2008. Sample Costs to Establish and Produce Alfalfa. 2008 University of California Cooperative Extension, Davis, CA, USA.
Gailey, R.M., 2017. Inactive supply wells as conduits for flow and contaminant migration: conditions of occurrencen body and elytra large, less dense, sub-ovate and offwhi and suggestions for management. Hydrogeol. J. 25 (7), 2163–2183. https://doi.org/10.1007/s10040-017-1588-y.
Geisseler, D., Scow, K.M., 2014. Long-term effects of mineral fertilizers on soil microorganisms: a review. Soil Biol. Biochem. 75 (2014), 54–63.
Gejl, R.N., Rygaard, M., Henriksen, H.J., Rasmussen, J., Bjerg, P.L., 2019. Understanding the impacts of groundwater abstraction through long-term trends in water quality. Water Res. https://doi.org/10.1016/j.watres.2019.02.026.
Ghasemizade, M., Asante, K.O., Petersen, Ch., Kocis, T., Dahlke, H.E., Harter, Th., 2019. An integrated approach toward sustainability via groundwater banking in the southern Central Valley, California. Water Resour. Res. https://doi.org/10.1029/2018WR024069.
Gu, B., Ge, Y., Chang, S.X., Luo, W., Chang, J., 2013. Nitrate in groundwater of China: sources and driving forces. Glob. Environ. Chang. 23 (5), 1112–1121.
Hansen, B., Thorling, L., Schullehner, J., Termansen, M., Dalgaard, T., 2017. Groundwater nitrate response to sustainable nitrogen management. Sci. Rep. (7), 8566.
Harter, T., Onsoy, Y., Heeren, K., Denton, M., Weissmann, G., Hopmans, J., Horwath, W., 2005. Deep vadose zone hydrology demonstrates fate of nitrate in eastern San Joaquin Valley. Calif. Agric. 59 (2), 124–132. https://doi.org/10.3733/ca.v059n02p124.
Harter, T., Dzurella, K., Kourakos, G., Hollander, A., Bell, A., Santos, N., Hart, Q., King, A., Quinn, J., Lampinen, G., Liptzin, D., Rosenstock, T., Zhang, M., Pettygrove, G.S., Tomich, T., 2017. Nitrogen fertilizer loading to groundwater in the Central Valley. In: Final Report to the Fertilizer Research Education Program, Projects 11-0301 and 15-0454. California Department of Food and Agriculture and University of California Davis (333 pp.). http://groundwaternitrate.ucdavis.edu.
Henri, C., Harter, T., 2019. Stochastic assessment of non-point source contamination: joint impact of aquifer heterogeneity and well characteristics on management metrics. Water Resour. Res (in review).
Hiscock, K.M., Lovett, A., Saich, A., Dockerty, T., Johnson, P., Sandhu, C., Sünnenberg, G., Appleton, K., Harris, B., Greaves, J., 2007. Modelling land-use scenarios to reduce groundwater nitrate pollution: the European Water4All project. Q. J. Eng. Geol. Hydrogeol. 40 (4), 417–434.
Honeycutt, K., Canada, H.E., Jenkins, M.W., Lund, J.R., 2012. Alternative water supply options for nitrate contamination. Technical report 7 in: 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. Center for Watershed Sciences, University of California, Davis.
Jensen, V.B., Darby, J.L., Seidel, Ch., Gorman, C., 2014. Nitrate in potable water supplies: alternative management strategies. Crit. Rev. Environ. Sci. Technol. 44 (20), 2203–2286. https://doi.org/10.1080/10643389.2013.828272.
Jordan, T.E., Weller, D.E., 1996. Human contributions to terrestrial nitrogen flux. BioScience 46, 655–664.
Jurgens, B.C., Burow, K.R., Dalgish, B.A., Shelton, J.L., 2008. Hydrogeology, water chemistry, and factors affecting the transport of contaminants in the zone of contribution of a public-supply well in Modesto, eastern San Joaquin Valley, California. In: U.S. Geological Survey Scientific Investigations Report 2008-5156.
Jury, W.A., Nielson, D.R., 1989. Nitrate transport and leaching mechanisms. In: Follett, R.F. (Ed.), Nitrogen Management and Ground Water Protection. Elsevier Sci. Pubs, Amsterdam, pp. 139–157.
King, A., Jensen, V., Fogg, G.E., Harter, T., 2012. Groundwater remediation and Management for Nitrate. Technical report 5 (50 pp.) 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. Center for Watershed Sciences, University of California, Davis.
Kirchmann, H., Johnston, A.E., Bergstrom, L.F., 2002. Possibilities for reducing nitrate leaching from agricultural land. AMBIO J. Hum. Environ. 31 (5), 404–408.
Kocis, T.N., Dahlke, H.E., 2017. Availability of high-magnitude streamflow for groundwater banking in the Central Valley, California. Environ. Res. Lett. 12 (8).
Lee, J.Y., Kwon, K.D., Park, Y.C., Jeon, W.H., 2017. Unexpected nationwide nitrate declines in groundwater of Korea. Hydrol. Process. 31, 4693–4704.
Liu, C.-W., Sung, Y., Chen, B.-C., Lai, H.-Y., 2014. Effects of Nitrogen Fertilizers on the Growth and Nitrate Content of Lettuce (Lactuca sativa L.). Int. J. Environ. Res. Public Health 11 (4), 4427–4440. https://doi.org/10.3390/ijerph110404427.
Luhdorff & Scalmanini Consulting Engineers (LSCE), Larry Walker & Associates (LWA), 2016. CV-SALTS Alta Irrigation District Management Zone: Aggressive Restoration Alternative Modeling Scenario Results, Technical Memorandum Phase II Report.
Mayzelle, M.M., Viers, J.H., Medellín-Azuara, J., Harter, T., 2015. Economic feasibility of irrigated agricultural land use buffers to reduce groundwater nitrate in rural drinking water sources. Water 2015 (7), 12–37.
McMahon, P.B., Burow, K.R., Kauffman, L.J., Eberts, S.M., Böhlke, J.K., Gurdak, J.J., 2008. Simulated response of water quality in public supply wells to land-use change. Water Resour. Res. https://doi.org/10.1029/2007WR006731.
Medellín-Azuara, J., Rosenstock, T.S., Howitt, R., Harter, T., Jessoe, K., Dzurella, K.N., Pettygrove, S.G., Lund, J.R., 2012. Economic analysis of crop nitrate source reductions. In: International Water Resources: Challenges for the 2151 Century and Water Resources Education.
Mount, J., Freeman, E., Lund, J., 2014. Water Use in California. Public Policy Institute of California (PPIC).
Niswonger, R.G., Morway, E.D., Triana, E., Huntington, J.L., 2017. Managed aquifer recharge through off-season irrigation in agricultural regions. Water Resour. Res. 53. https://doi.org/10.1002/2017WR020458.
Nolan, B.T., Hitt, K.J., Ruddy, B.C., 2002. Probability of nitrate contamination of recently recharged groundwaters in the conterminous United States. Environ. Sci. Technol. 36, 2138–2145.
Phillips, S.P., Green, C.T., Burow, K.R., Shelton, J.L., Rewis, D.L., 2007a. Simulation of multiscale ground-water flow in part of the northeastern San Joaquin Valley, California. In: U.S. Geological Survey Scientific Investigations Report 2007-5009, (43 pp.).
Phillips, S.P., Burow, K.R., Rewis, D.L., Shelton, J., Jurgens, B., 2007b. Hydrogeologic settings and ground-water flow simulations of the San Joaquin Valley Regional Study Area, California, section 4 of Paschke, S.S., ed., Hydrogeologic settings and ground-water flow simulations for regional studies of the transport of anthropogenic and natural contaminants to public-supply wells—Studies begun in 2001: Reston, Va. In: U.S. Geological Survey Professional Paper 1737–a, (pp. 4–1 – 4–31).
Phillips, S.P., Rewis, D.L., Traum, J.A., 2015. Hydrologic model of the Modesto Region, California, 1960–2004. In: U.S. Geological Survey Scientific Investigations Report, (69 pp.).
Pollock, D.W., 2012. User guide for MODPATH version 6 - a particle-tracking model for MODFLOW. In: U.S. Geological Survey Techniques and Methods 6–A41, (58 pp.).
Ransom, K.M., Nolan, B.T., Traum, J.A., Faunt, C.C., Bell, A.M., Gronberg, J.M., Wheeler, D.C., Rosecrans, C.Z., Jurgens, B., Schwarz, G.E., 2017. A hybrid machine learning model to predict and visualize nitrate concentration throughout the Central Valley aquifer, California, USA. Sci. Total Environ. 601, 1160–1172.
Ransom, K.M., Bell, A.M., Barber, Q.E., Kourakos, G., Harter, T., 2018. A Bayesian approach to infer nitrogen loading rates from crop and land-use types surrounding private wells in the Central Valley, California. Hydrol. Earth Syst. Sci. 1607–7938. https://doi.org/10.5194/hess-22-2739-2018.
Rosenstock, T.S., Liptzin, D., Dzurella, K., Fryjoff-Hung, A., Hollander, A., Jensen, V., King, A., Kourakos, G., McNally, A., Pettygrove, G.S., Quinn, J., Viers, J.H., Tomich, T.P., Harter, T., 2014. Agriculture's contribution to nitrate contamination of Californian groundwater (1945–2005). J. Environ. Qual. 43, 895–907. https://doi.org/10.2134/jeq2013.10.0411.
Rudolph, D.L., Devlin, J.F., Bekeris, L., 2015. Challenges and a strategy for agricultural BMP monitoring and remediation of nitrate contamination in unconsolidated aquifers. Groundw. Monit. Remediat. 35, 97–109. https://doi.org/10.1111/gwmr.12103.
Schmid, W., Hanson, R.T., 2009. The Farm Process Version 2 (FMP2) for MODFLOW-2005—Modifications and Upgrades to FMP1: U.S. Geological Survey Techniques and Methods 6-A-32. (102 pp.).
Schulze-Makuch, D., 2005. Longitudinal dispersivity data and implications for scaling behavior. Ground Water 43 (3), 443–456.
Sebilo, M., Mayer, B., Nicolardot, B., Pinay, G., Mariotti, A., 2013. Long-term fate of nitrate fertilizer in agricultural soils. Proc. Natl. Acad. Sci. U. S. A. 110 (45), 18185–18189. https://doi.org/10.1073/pnas.1305372110.
Somura, H., Goto, A., Matsui, H., Ali Musa, E., 2008. Impacts of nutrient management and decrease in paddy field area on groundwater nitrate concentration: a case study at the Nasunogahara alluvial fan", Tochigi Prefecture, Japan. Hydrol. Process. 22, 4752–4766. https://doi.org/10.1002/hyp.7089.
Sprague, L.A., Hirsch, R.M., Aulenbach, B.T., 2011. Nitrate in the Mississippi River and its tributaries, 1980–2008: are we making progress? Environ. Sci. Technol. 45, 7209–7216.
State Water Resources Control Board, 2013. "Communities That Rely on a Contaminated Groundwater Source for Drinking Water", California, January 2013.
State Water Resources Control Board, 2017. "Groundwater Information Sheet: Nitrate", GAMA Program, November 2017.
Tesoriero, A.J., Duff, J.H., Saad, D.A., Spahr, N.E., Wolock, D.M., 2013. Vulnerability of streams to legacy nitrate sources. Environ. Sci. Technol. (8), 3623–3629. https://doi.org/10.1021/es305026x.
Turkeltaub, T., Dahan, O., Kurtzman, D., 2014. Investigation of groundwater recharge under agricultural fields using transient deep vadose zone data. Vadose Zone J. 13. https://doi.org/10.2136/vzj2013.10.0176.
Van der Schans, M.L., Harter, T., Leijnse, A., Mathews, M.C., Meyer, R.D., 2009. Characterizing sources of nitrate leaching from an irrigated dairy farm in Merced County, California. J. Contam. Hydrol. 110, 9–21.
Van Grinsven, H.J.M., Bouwman, L., Cassman, K.G., van Es, H.M., McCrackin, M.L., Beusen, A.H.W., 2015. Losses of Ammonia and nitrate from agriculture and their effect on nitrogen recovery in the European Union and the United States between 1900 and 2050. J. Environ. Qual. 44, 356–367. https://doi.org/10.2134/jeq2014.03.0102.
Verdegaal, P., Klonsky, K., de Moura, R., 2012. Sample Costs to Establish a Vineyard and

Produce Winegrapes. University of California Cooperative Extension, Davis, CA, USA.

Viers, J.H., Liptzin, D., Rosenstock, T.S., Jensen, V.B., Hollander, A.D., McNally, A., King, A.M., Kourakos, G., Lopez, E.M., De La Mora, N., Fryjoff-Hung, A., Dzurella, K.N., Canada, H.E., Laybourne, S., McKenney, C., Darby, J., Quinn, J.F., Harter, T., 2012. Nitrogen sources and loading to groundwater. Technical report 2 in: 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. Center for Watershed Sciences, University of California, Davis.

Wang, L., Stuart, M.E., Lewis, M.A., Ward, R.S., Skrivin, D., Naden, P.S., Collins, A.L., Ascott, M.J., 2016. The changing trend in nitrogen concentrations in major aquifers due to historical nitrogen loading from agricultural land across England and Wales from 1925 to 2150. Sci. Total Environ. 542, 694–705.

Zhang, H., Hiscock, K.M., 2016. Modelling response of groundwater nitrate concentration in public supply wells to land-use change. QJEGH 49, 170–182.

Zheng, C., 1990. MT3D: A modular three-dimensional transport model for simulation of advection, dispersion and chemical reactions of contaminants in groundwater systems. In: Report to the U.S. Environmental Protection Agency, Ada, Oklahama, (170 pp.).

Zheng, C., Bennett, G.D., 2002. Applied Contaminant Transport Modeling, second ed. Wiley, New York (621 pp.).