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ORIGINAL ARTICLE

Modeling nitrate leaching risk from flood managed aquifer recharge in California's agricultural lands

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Abstract

Flood managed aquifer recharge (Flood-MAR) is an emerging practice to enhance groundwater sustainability through recharge of aquifers. However, in agricultural systems Flood-MAR may harm groundwater quality, given residual soil nitrate (NO_3^-). This research evaluated Flood-MAR NO_3^- leaching risk across a precipitation and soil textural gradient in a globally important agricultural region, California's Central Valley, and whether Flood-MAR timing strategies could mitigate the risk. Using multi-decadal root zone water quality model simulations of irrigated and fertilized maize (250 kg N ha^{-1} year^{-1}) on well-drained soils as a representative case-study, results suggest Flood-MAR can be used with near-negligible additional NO_3^- leaching in locations with median annual precipitation >400 mm year^{-1}. In those locations, wet-year precipitation leached most residual NO_3^- without practicing Flood-MAR. At drier locations, Flood-MAR NO_3^- leaching risk increased most clearly in loamy soils. Additional NO_3^- leaching risk increased in drier climates because minimal precipitation-driven deep percolation maintained residual NO_3^- accumulation across growing seasons. In fine-texture soils, NO_3^- leaching risk was mitigated by denitrification, preventing residual NO_3^- accumulation. Flood-MAR practices diminished denitrification, leaching NO_3^- rapidly when soils were colder and biogeochemically inactive, and thereby decreased growing season denitrification when soils were warmer and denitrification rates higher. Effects of Flood-MAR timing strategies (January Flood-MAR vs. March Flood-MAR), combined with variable pauses among applications (3- vs. 7- vs. 21-day intervals) were negligible. Infrequent Flood-MAR should be practiced with care in arid climates and especially after prolonged droughts coinciding with more limited irrigation water supplies that constrain salt-leaching practices all favoring residual NO_3^- accumulation.

Abbreviations: BAU, business-as-usual; CERES-Maize, crop environment resource synthesis model; CIMIS, California Irrigation Management Information System; Flood-MAR, Flood Managed Aquifer Recharge; MCL, maximum contaminant level; PRISM, parameter-elevation regressions on independent slopes model; RZWQM2, root zone water quality model; SIC, simulated initial conditions; SOM, soil organic matter; SSURGO, Soil Survey Geographic database; UAN, urea ammonium nitrate.

Plain Language Summary

Agricultural management of floodwater (Flood-MAR) is a new practice where floodwaters are applied to agricultural fields to recharge groundwater. Flood-MAR may harm groundwater quality by leaching soil nitrate (NO_3^-) into groundwater. This modeling study evaluated the Flood-MAR NO_3^- leaching risk in different climates and soil textures in California. It evaluated whether Flood-MAR timing strategies (early- vs. late-season irrigation application strategies) influenced risk. Flood-MAR had near-negligible NO_3^- leaching risk in locations with rainfall >400 mm year^{-1}. At drier locations, Flood-MAR NO_3^- leaching risk was highest (especially in loamy soils) because low rainfall allowed NO_3^- to build up in soils from year to year. Different Flood-MAR timing strategies (early season vs. late season), combined with variable pauses in water applications (3- vs. 7- vs. 21-day intervals), showed no difference in NO_3^- leaching. Flood-MAR should be practiced with care in arid climates.

1 | INTRODUCTION

Globally, irrigated landscapes have more than doubled in extent since the 1960s to 342 million ha in 2019, helping boost agricultural production at a rate faster than the total area under cultivation (FAO, 2021). However, when practiced in semiarid and arid climates with a natural climatic water deficit, irrigated agriculture requires enormous freshwater withdrawals to meet irrigated crop evapotranspiration demands; for example, in California, 81% of groundwater use has been attributed to irrigated agriculture (Nelson, 2012). This intense reliance on groundwater has led to numerous occurrences where groundwater extraction rates greatly exceed recharge rates underlying irrigated regions (Herbert & Döll, 2019; Siebert et al., 2010). In California, where globally important agriculture has revenues now exceeding $60 billion annually, multiyear droughts in recent decades have highlighted groundwater depletion as an existential threat to the long-term viability of irrigation and even access to groundwater in overlying communities (Liu et al., 2022).

The agricultural management of floodwaters (Flood-MAR) is of broad interest as a tool to recharge aquifers, offset groundwater pumping, and mitigate downstream flood risk (CDWR, 2018). This practice involves flooding agricultural fields during the winter and spring, and in some cases early summer, when rivers can reach high magnitude flows during wet years as a result of atmospheric river events and snowmelt (Kocis & Dahlke, 2017). Novel approaches are necessary to sustain irrigated agriculture in the face of multiple challenges. These include new public policy constraints on groundwater use, such as the Sustainable Groundwater Management Act requiring groundwater sustainability plans in overdrafted basins (CDWR, 2022), and additional challenges imposed by climate change, such as more intense precipitation whiplash (Swain et al., 2018) and decreased mountain snowpack, a natural reservoir for California's irrigated agriculture that will diminish with warming (Qin et al., 2020).

However, there are concerns to be analyzed before Flood-MAR can be safely implemented, such as contamination of groundwater by flushing out residual soil nitrate (NO_3^-) during crop dormancy or fallowing at times that coincide with high magnitude river flows. Our research assumed an overarching hypothesis that Flood-MAR would enhance NO_3^- leaching compared to no Flood-MAR (business-as-usual [BAU]), following the concern that Flood-MAR would mobilize residual NO_3^- in fertilized agroecosystems (Levintal et al., 2022). For example, in relatively deep vadose zone samples under California citrus orchard fertilizer trials, Harter et al. (2005) reported residual nitrate levels of 218–477 kg NO_3^--N ha^{-1} in 16-m deep cores. Bachand et al. (2014) measured residual NO_3^- in western Fresno County fields (San Joaquin Valley, California) where Flood-MAR had been practiced for one season and observed a residual NO_3^- quantity less than 15%, on average, of the soil profile NO_3^- (0–3 m) in wine grape control fields. In a deep soil coring study (0–30 m) across 36 study sites (12 fields) in the same region, Waterhouse et al. (2020) found relatively high residual NO_3^- across tomatoes and almonds (737–966 and 420–1240 kg NO_3^--N ha^{-1}, respectively) with a lower range for wine grapes where some growers tend to apply less fertilizer (75–1371 kg NO_3^--N ha^{-1}), but overall still demonstrating the risk for mobilizing residual NO_3^- with Flood-MAR.

We evaluated scenarios of contrasting seasonal timing and frequency of Flood-MAR as strategies to possibly minimize NO_3^- leaching risk by leveraging the root zone water quality model (RZWQM2), a widely validated tool developed and maintained by a team of USDA Agricultural Research Service scientists. We explored the primary controls (climate and soil properties) influencing the interaction of Flood-MAR with the N-cycle. Laboratory-based soil column leaching experiments demonstrated that the frequency of wetting in the field can potentially influence NO_3^- leaching. For example, when 15-cm water pulses were applied every 1–2 weeks, total NO_3^- leached exceeded initial residual NO_3^- by 30%–50% because N was mineralized from the organic pool between pulses. When 15-cm water pulses were applied every 3 days, total NO_3^- leached was less than the initial residual NO_3^- (Murphy et al., 2021). Following this finding, we hypothesized that early-winter Flood-MAR timing would leach less NO_3^- than late-winter or early-spring Flood-MAR timing over the long term, due to lower rates of N mineralization when soils are cooler. Additionally, we hypothesized that higher frequency Flood-MAR pulses (shorter interval between water applications) would leach less NO_3^-, since there would be less time for soils to generate NO_3^- from mineralization of soil organic matter (SOM) between Flood-MAR applications. As a counteracting process to the effect of leaching, Waterhouse et al. (2021) demonstrated that prolonged Flood-MAR may enhance some NO_3^- removal via denitrification. Following this, we also hypothesized that Flood-MAR NO_3^- leaching risk would be partially offset by denitrification in finer textured soils, which would require more extended periods of soil saturation to achieve similar recharge depths compared to coarser soils, leading to anaerobic conditions. We expected this mitigation to be amplified when Flood-MAR was repeatedly practiced in the latter part of the wet season when soils are warmer and temperature-driven denitrification rates are potentially higher. RZWQM2 was used to simulate a multi-decadal BAU scenario and five different Flood-MAR approaches in the context of a fertilized (250 kg N ha^{-1} $year^{-1}$) and irrigated maize agroecosystem with winter fallows across different representative soils and climates, as described below.

Core Ideas

- Modeling suggests NO_3^- leaching risk during flood managed aquifer recharge (Flood-MAR) varies by texture and climate.
- NO_3^- leaching risk diminished from Flood-MAR in locations with annual precipitation >400 mm $year^{-1}$.
- Flood-MAR should be practiced with care in arid climates and especially after prolonged droughts.
- Flood-MAR timing strategies (early vs. late season) and durations (3- vs. 7- vs. 21-day intervals) were negligible.

2 | MATERIALS AND METHODS

2.1 | Overview of RZWQM2

RZWQM2 is a comprehensive process model that simulates fluxes of soil water and heat, organic matter cycling, and biogeochemical transport of N, linked to crop growth and nutrient and water uptake models, all paired to an agriculture management module (Ahuja et al., 2000). A primary use of RZWQM2 has been to calibrate and then predict the hydrologic response of crop–soil–climate combinations to various agricultural management systems, including effects on groundwater or surface water quality through runoff, leaching, or tile drainage (Kisekka et al., 2017; Ma et al., 2006). In California, for example, RZWQM2 has been used to model the application of dairy manure to cropland and its effects on various greenhouse gas emission components (Geisseler et al., 2012) and NO_3^- leaching to groundwater from an intensively monitored almond orchard site (Domagalski et al., 2008).

The RZWQM2 model embedded with DSSAT-CSM v4.0 (decision support systems for agrotechnology transfer cropping system model) CERES-Maize (crop environment resource synthesis model) crop growth module was used in this study (Ma et al., 2011). Infiltration and runoff are simulated by RZWQM2 using a modified form of the Green–Ampt equation (Green & Ampt, 1911). Unsaturated soil water flow and soil water redistribution are modeled using a one-dimensional (1D) Richards equation (Richards, 1931). Macropore flow is governed by the Poiseuille's law. When rainfall or irrigation exceeds the soil's infiltration capacity, RZWQM2 generates runoff. The potential evaporation and transpiration demand of the atmosphere were computed using the Shuttleworth–Wallace ET model (Ahuja et al., 2000). The Shuttleworth–Wallace ET model in RZWQM2 is an extension of the Penman–Monteith ET model, but the former considers incomplete canopy cover and plant height in potential evaporation and transpiration estimations (Kisekka et al., 2017). Actual root water uptake described by the sink term in Richards's equation was calculated numerically using the Nimah and Hanks (1973) procedure. Actual root water uptake, potential evaporation, and potential transpiration were used in computing water stress factors used in CERES-Maize to modulate plant growth processes like leaf growth as soil water gets depleted. Carbon–nitrogen (N) cycles are driven by microbial biomass and environmental factors such as temperature and soil water content.

RZWQM2 simulations were performed on soil profiles (152- to 218-cm deep) with properties defined by horizon based on the Soil Survey Geographic database (SSURGO). RZWQM2 then automatically discretizes these user-defined soil horizons into multiple numerical nodes per horizon to simulate 1D soil water and heat transport (i.e., nodes placed every 1–5 cm with intervals between nodes increasing with depth). Dynamic (transient) simulations spanned a 37-year simulation period with user-defined daily weather and management data, but variable time steps depended on numerical solution convergence. Simulated initial conditions (SIC) for RZWQM2 state parameters (water content, nitrate, SOM, etc.) were obtained by taking the end-of-simulation-time state of these parameters in an initial 37-year "spin-up" scenario.

2.1.1 | Parameterization of RZWQM2

For this study, RZWQM2 modeling included 33 Central Valley soil series representing distinct taxonomic family particle-size classes (see Section 2.3) and five different 37-year climate records obtained from the California Irrigation Management System (Table 1; see Section 2.2), spanning a precipitation gradient from relatively wet to dry in space and time (n = 165 unique soil–climate scenarios). The default RZWQM2 biogeochemical rate constants affecting N dynamics (e.g., nitrification, denitrification, and SOM decay) were used across all scenarios. The RZWQM2 GUI (specifically, the "Soil Hydraulics" tab of the "Site Description" input window) was used to define Brooks–Corey parameters for each modeled soil horizon by inputting SSURGO estimates for saturated hydraulic conductivity (cm h^{-1}) and water content at saturation, field capacity (1/3 bar) and wilting point (15 bars) (Rawls et al., 1982). Soil thermal properties are estimated by calculating heat capacity and thermal conductivity based on soil texture and soil water content.

Only residue and SOM inter-pool transfer coefficients, which control the relative proportion of organic matter that is transferred amongst the two residue and three SOM pools during decay, were modified from default RZWQM2 values as modeled by Cameria et al. (2007) and determined experimentally by Ma et al. (1998). This was done to minimize the accumulation of soil organic nitrogen in low SOM soils, observed during SIC, which influenced N mass balance and leaching, following suggestions in Ma et al. (1998, 2011) about the relative importance and appropriateness for calibration of SOM inter-pool transfer coefficients to achieve reasonable results. In this case, it was not viewed as reasonable to accumulate SOM rapidly in low SOM, sandy soils. But like other first-order biogeochemical rate constants, SOM inter-pool transfer coefficients were still assumed not to vary across soil, climate, and scenario combinations. Thus, the biogeochemical processes simulated in this study vary as a function of soil physical properties and climate.

Initial values for SOM pool sizes, microbial populations, crop litter, and soil profile NO_3^- were allowed to vary by soil × climate combination and were established by using end-of-run values from the 37-year SIC run of each unique soil × climate modeling combination (33 soils × 5 climates = 165).

TABLE 1 Thirty-seven-year climate summary for five California Irrigation Management System stations used to input climate data for root zone water quality model (RZWQM2) simulations. See Section 2.2 for quality control check, error-correction, and gap-filling procedures. Water balance components were summarized by water year (October–September), with evaporation and transpiration summarized from RZWQM2 business-as-usual median runs among all soils by station.

Station	Wind (m s^{-1})	RH (%)	Insol (kWh m^{-2})	Jan	Mar	MAT	Jul	E_d	E_g	T_c	P_{min}	P_{med}	MAP	P_{10}	P_{max}	P_{dorm} (%)	K-G_{arid}
				Temperature (°C)				Water balance components (mm $year^{-1}$)									
Durham	1.85	65.6	4.82	8.0	12.4	16.1	24.4	189	242	502	298	540	569	648	1099	82.5	1.77
Davis	2.55	62.6	4.96	8.0	12.6	16.0	23.5	165	260	667	221	404	444	518	781	87.7	1.39
Parlier	1.65	64.9	4.94	8.4	13.5	17.1	26.5	131	255	605	127	279	280	345	557	84.0	0.82
Five Points	2.51	56.8	5.19	8.3	13.4	17.0	25.9	114	289	758	76	193	211	278	400	83.9	0.62
Shafter	1.64	61.8	5.27	8.4	13.7	17.1	26.4	101	244	593	60	143	159	182	411	84.7	0.46

Note: wind = mean annual; RH = relative humidity, mean annual; Insol = insolation, mean annual; Jan = January, mean; Mar = March, mean; MAT = mean annual temperature; Jul = July, mean; E_d = evaporation, dormant season (October–March), median; E_g = evaporation, growing season (April–Septmeber), median; T_c = crop transpiration, median; P_{min} = minimum precipitation; P_{med} = median precipitation; MAP = mean annual precipitation; P_{10} = 10th wettest water year precipitation (≈Q3); P_{max} = maximum precipitation; P_{dorm} = dormant season (October–March) precipitation as annual percentage; K-G_{arid} = Köppen–Geiger climate definition of aridity for climates where >70% of MAP falls during the winter months (October–March for northern hemisphere): MAP/(MAT × 20), where MAP is in mm and MAT is in °C. If the resulting aridity metric is <0.5, the climate is arid desert; 0.5–1 is arid steppe—sometimes referred to as semiarid.

TABLE 2 Summary of conditions simulated using the root zone water quality model.

Scenario	Flood-MAR application frequency	Season
BAU (business-as-usual)	NA	NA
High frequency-late	15 cm H_2O applied every 3 days	March
High frequency-early	15 cm H_2O applied every 3 days	January
Low frequency-late	15 cm H_2O applied every 7 days	March
Low frequency-early	15 cm H_2O applied every 7 days	January
Extended frequency	15 cm H_2O applied every 21 days	January through March

Note: Each Flood-MAR scenario consisted of four applications of 15 cm of water performed during the 10 wettest years of the 37-year simulation period. BAU received no Flood-MAR treatment, thus information for application frequency and season was not applicable (NA).

This approach was used because soil survey data reflect soil properties of representative pedons from locations with natural conditions rather than irrigated fields, which have higher biomass inputs and a wetter moisture regime. Specifically, SOM pools were first parameterized by soil horizon using data from the SSURGO and initially assumed to be distributed across the SOM pools as follows: the slow-cycling pool was assumed to represent 80% of total SOM; the intermediate-cycling pool 15%; and the fast-cycling pool 5% across all horizons, following Ma et al. (1998) and Cameira et al. (2007), but then allowed to equilibrate during the 37-year SIC run, saving the end-of-run values as the SIC. This produced unique initial biogeochemical conditions for each of the soil × climate combinations to be tested again under another 37-year BAU run (without Flood-MAR treatments) and five 37-year model runs with contrasting Flood-MAR strategies, as described next.

In RZWQM2 simulations, Flood-MAR was practiced during the 10 wettest water years (October–September) of each specific 37-year climate record, applying 600-cm additional water (60 cm year^{-1}) through Flood-MAR. During a Flood-MAR year, four 15-cm water applications were made in either January or March, using an application frequency of either 3- or 7-day intervals (Table 2). A fifth scenario tested applications from January to March with 21-day intervals. For fine-textured soils with slower saturated hydraulic conductivities, Flood-MAR was practiced up to nine continuous days to achieve 15-cm applied water in each event, depending on the soil's depth-weighted hydraulic conductivity, using 3-, 7-, or 21-day periods between these continuous periods of Flood-MAR, depending on the scenario. RZWQM2 can mimic surface ponding by adjusting the maximum surface storage parameter. In this study, in order to realistically represent FloodMAR, time periods to achieve 15-cm infiltrated water had to be determined a priori as shown in Table 2.

2.1.2 | Simulated cropping system details

The DSSAT-CSM v4.0 CERES-Maize model was used to represent crop growth and development in the RZWQM2 model (Ma et al., 2006) as continuous maize with winter fallow. The CERES-Maize model is a radiation-based mechanistic crop model that predicts maize growth and development based on weather (precipitation, solar radiation, maximum and minimum temperature, and to a lesser extent photoperiod), management practices (plant population, row spacing, planting date, and irrigation management), and six cultivar coefficients (P1, P2, P5, G2, G3, and PHINT) as described in Hoogenboom et al. (2019). More details about the RZWQM2 can be found in Ahuja et al. (2000).

The objective for the project here was to have a stable crop growth model with reliable, consistent plant growth and N accumulation each year across all soil × climate scenarios, accepting that some minor differences would be expected. Ideally, similar yields and thus N removal through harvest would result in similar potentials for residual nitrate buildup due to less removal than applied across all simulated scenarios, to test the possible effects of Flood-MAR on additional leaching risk. For the model, planting occurred on April 15 every year, fertilization was applied as urea ammonium nitrate (UAN) in four splits with 50 kg total N applied in a preplant on April 15, and the remainder was divided evenly as 66.7 kg N applications on June 1, July 1, and July 21. UAN applications were entered into RZWQM2 input files as appropriate proportions of NH_4, NO_3^-, and urea. Irrigation was managed using the automated option in RZWQM2, where irrigation is triggered when plant-available water is depleted to some allowable percentage threshold, called "allowable depletion," and then the soil is refilled by irrigation back to field capacity. Allowable depletion was set at 60% during the first 20 days of crop growth, then reduced to 50% until harvest. This typically resulted in an irrigation frequency of several days during initial stand establishment and then every 5–14 days during the summer, depending on the soil's water holding capacity. Irrigation depths varied dynamically with rooting depth as the crop developed, reaching 5–8 cm per irrigation mid-summer. An irrigation application of 1.25 cm at planting was also made to stimulate germination at a uniform date. The simulated tillage depth was 15 cm with a tandem disk on April 15 before planting and then on September 15 and 16 after harvest with a bedder ridge operation on September 17. Harvest date varied dynamically across years and scenarios and was defined as the removal of aboveground biomass with 10 cm stubble height and occurring at growth stage 0.95 (0–1) with 95% efficiency.

2.2 | Climate data

Five stations from the California Irrigation Management Information System (CIMIS) were chosen on a wet-to-dry precipitation gradient (569–159 mm $year^{-1}$) with at least a 37-year, largely intact record to provide climate data as input to RZWQM2 simulations (Table 1). Daily aggregated weather data were downloaded using the *cimir* package in R software (Koohafkan, 2021) and then subjected to in-house R software functions for visualizing anomalies and also automated quality control checks, error correction, and gap-filling procedures. Specifically, thresholds for possible minimum and maximum temperatures, relative humidity, insolation, wind, and precipitation were defined for each station based on official Fresno and Sacramento, CA, National Weather Service records and then corrected when necessary, using calculated daily means for the station. In some cases, missing precipitation or likely erroneous precipitation data were gap-filled using daily PRISM (parameter-elevation regressions on independent slopes model) data downloaded for the 4 km cell within which each CIMIS station was located. Additionally, some anomalous growing season precipitation values at the Five Points and Parlier stations, attributed to malfunctioning irrigation systems used to maintain a reference green grass surface surrounding the CIMIS weather stations, were corrected with PRISM data. For the Shafter station, erroneous precipitation data from 2004 to 2005 and multi-month missing data in 2012–2013 were provided by the nearby Delano CIMIS station 182. Finally, all temperature data were additionally corrected using daily means if a daily maximum or minimum temperature exceeded 3.5 standard deviations from its respective daily mean.

2.3 | Soil data

Note that 33 soil profile datasets from SSURGO were used to parameterize the 990 RZWQM2 scenarios (one BAU and five Flood-MAR scenarios across 165 soil × climate conditions). The soil data objective was to represent widespread soils without restrictive horizons, spanning a broad textural gradient, and to generally test whether differences in inherent soil physical properties affected NO_3^- leaching response to Flood-MAR. The SSURGO sampling strategy was based initially on a soil classification study that defined seven generalized soil regions in California. Of these seven, three typically well-drained soil regions varying in texture were used as sampling targets, including coarse soils with no restrictions, loamy soils with no restrictions, and a fine-textured endmember, shrink-swell soils (Devine et al., 2021). The most widespread soil map unit components within each class were selected from the SSURGO with at least 80% assignment to one of these three soil regions used to create a sampling subset. Then, one representative soil profile dataset was selected randomly for each of the 11 most widespread soil components in each of the three soil regions. Only soil profiles with standard horizon nomenclature were sampled (A, B, C, etc.), since some SSURGO soil components from vintage survey areas have H1, H2, H3, and so forth nomenclature that indicates that soil data were entered without regard to a soil's typical horizonation, due to database constraints at the time of data entry.

The soil regions above were also found to correspond very well to generalized USDA-NRCS taxonomic family particle-size definitions (Soil Survey Staff, 1999). These USDA-NRCS soil classification definitions were ultimately used for the purpose of aggregating the results of this study as follows: coarse soils included those in the "coarse-loamy" and "sandy" classes, loamy soils included those in the "fine-loamy" and "fine-silty" classes, and fine soils included those in the "fine" and "very fine" classes. However, the other four soil regions in the seven-region soil classification approach, including soils with restrictive horizons and salt-affected soils (Devine et al., 2021), would also fall under these USDA-NRCS taxonomic particle-size classes, highlighting that not all of the Central Valley soil diversity has been represented in this study.

2.4 | Definition of "additional NO_3^- leaching risk"

For each unique Flood-MAR scenario (33 soils × 5 climates × 5 Flood-MAR timing strategies = 825 Flood-MAR scenarios), an "additional NO_3^- leaching risk" was calculated by subtraction of the total nitrate leached from its respective BAU scenario that used the same soil and climate (BAU for 33 soils × 5 climate combinations = 165 BAU scenarios). This relative "additional NO_3^- leaching risk" term is used throughout Section 3 and acknowledges that the study is not attempting to predict any absolute amount of NO_3^- leaching for a given place or time. The purpose of the BAU scenario is to serve as a control to measure the extent to which Flood-MAR impacts NO_3^- leaching. In other words, the interpretation of this study's results is meant to ascertain the relative risks of practicing Flood-MAR across different soils and climates in the Central Valley. The simulated maize cropping system with winter fallow is meant to serve as an analogue for any cropping system in California with a substantial dormant or fallow season, receiving ample N fertilizer applications with the potential for residual NO_3^- accumulation.

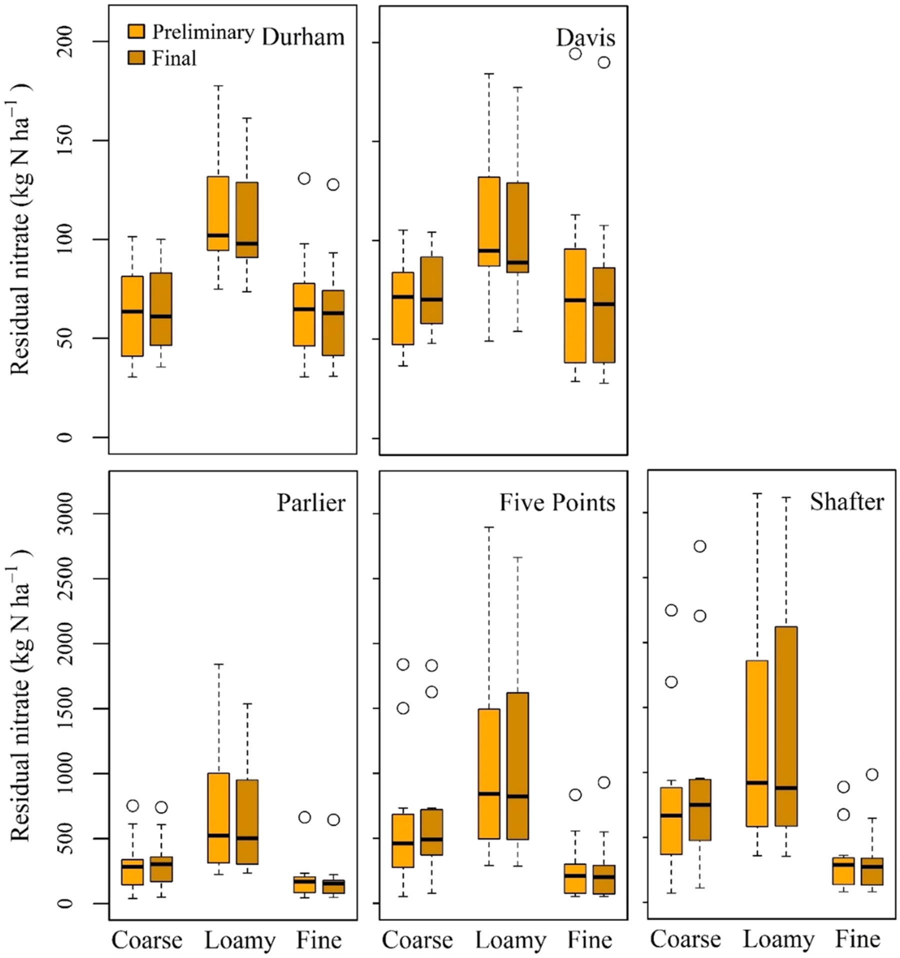

FIGURE 1 Preliminary (end of simulated initial conditions [SIC]) and final residual soil nitrate (in the business-as-usual scenario, no Flood-MAR) by three generalized taxonomic family particle-size classes: coarse, loamy, and fine ($n = 11$ soils per class). Each plot shows the different climates modeled from wettest (Durham, 537 mm year^{-1}) to driest (Shafter, 143 mm year^{-1}). Preliminary is the end of a 37-year preliminary run defining SIC. Final is the end of a subsequent 37-year run, using the SIC end-of-run values for initialization. Note the difference in *Y*-axis scale in wet versus driersites. Box plots display the median and interquartile range. Whiskers display the distribution of data extending 1.5 times the upper and lower quartiles. Open circles are outliers.

3 | RESULTS AND DISCUSSION

3.1 | Wetter climates prevent accumulation of residual NO_3^- in business-as-usual scenarios

Under BAU scenarios, the end-of-run residual soil NO_3^- largely stabilized across generalized taxonomic family particle-size classes in each climate modeled (Figure 1). This is inferred by the similarity in residual nitrate levels between the SIC (after 37 years of "spin-up") and at the end of the 37-year BAU simulation (Figure 1). This suggests that, for each unique soil and climate combination, the SIC represent the appropriate conditions for the dynamic steady-state residual NO_3^- in the modeled agroecosystem, thus allowing for a robust test of the effect of Flood-MAR on additional NO_3^- leaching risk. Whole soil residual NO_3^- in wetter climates (Durham and Davis) were typically 60–100 kg N ha^{-1} in the BAU scenario (Figure 1). Thus, climates with typical precipitation amounts of >400 mm year^{-1} (Table 1) were sufficient to leach soils in this simulated, fertilized agroecosystem, suggesting that Flood-MAR practiced in wetter Central Valley climates is of relatively lower additional NO_3^- leaching risk (Figure 2). Modeled residual NO_3^- levels and the relationship with mean annual precipitation generally corresponded with levels found in the literature, although we were unable to find data for California. In Minnesota, as an example, residual NO_3^--N under corn production ranged from around 100 kg N ha^{-1} at locations with relatively low precipitation (400 mm year) to 42 kg N ha^{-1} at locations with high precipitation (891 mm year) (Yadav, 1997). In addition, a 4-year trial in corn in eastern Colorado showed gradual buildup (23 kg ha^{-1} NO_3^--N) of residual nitrate in plots that were suboptimally

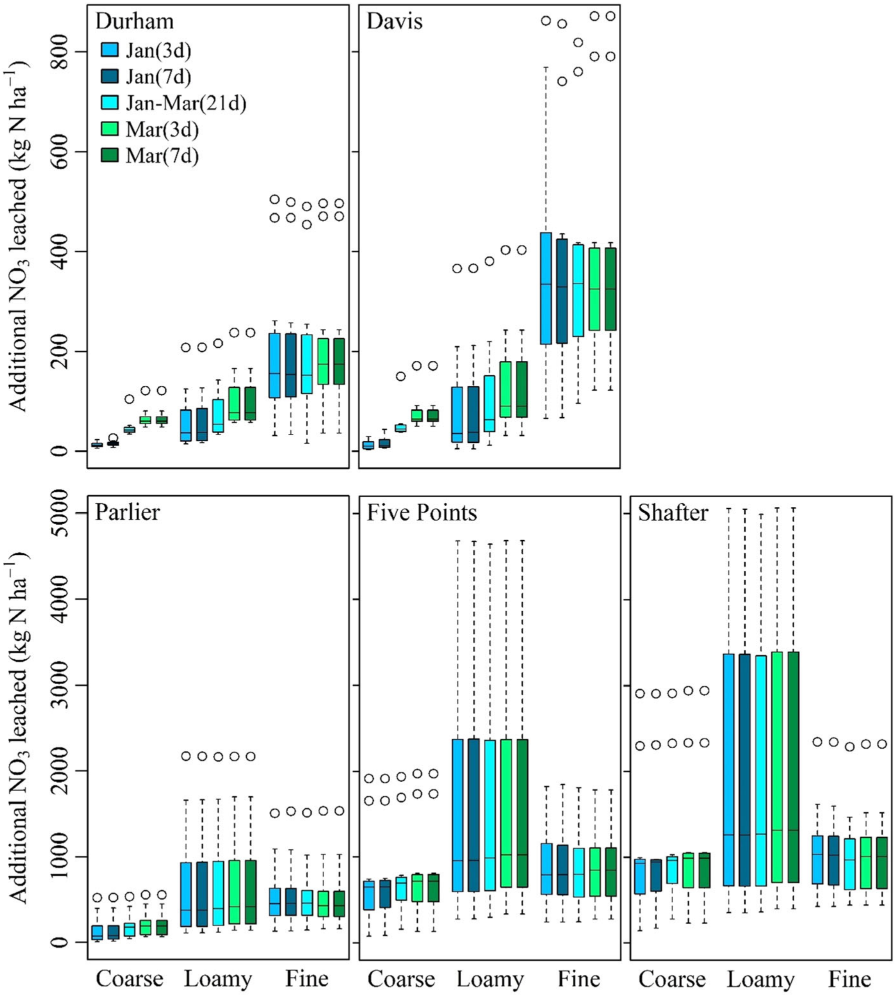

FIGURE 2 Effect on 37-year, cumulative nitrate leached of different Flood-MAR strategies, including 3-day and 7-day frequency events practiced either in January or March or 21-day frequency events spanning early January through late March. Additional nitrate leached is relative to each soil's business-as-usual run (no Flood-MAR). Each plot is grouped by three generalized taxonomic family particle-size classes: coarse, loamy, and fine ($n = 11$ soils per class), and shows the different climates modeled from wettest (Durham) to driest (Shafter). See Table 1 for descriptions of climates. Note the difference in Y-axis scale in wet versus drier sites. Box plots display the median and interquartile range. Whiskers display the distribution of data extending 1.5 times the upper and lower quartiles. Open circles are outliers.

irrigated (46 cm of applied water) compared to half as much nitrate buildup (13 kg ha^{-1} NO_3^--N) in plots that were over irrigated (85 cm of applied water) (Ludwick et al., 1976).

3.2 | Accumulation of residual NO_3^- in dry climates drives additional NO_3^- leaching risk

Residual soil NO_3^- increased by an order of magnitude in BAU coarse and loamy soils, but not in fine soils from wettest to driest climates (Figure 1). Compared to coarse soils, loamy soils tended to accumulate even more residual soil NO_3^- and displayed more variance from wet to dry. In the second driest climate (Five Points), BAU residual NO_3^- exceeded 500 kg NO_3^--N ha^{-1} in 5 of 11 coarse soils (interquartile range: 277–686 kg NO_3^--N ha^{-1}), 8 of 11 loamy soils (interquartile range: 498–1493 kg NO_3^--N ha^{-1}), and 2 of 11 fine soils (interquartile range: 77–301 kg NO_3^--N ha^{-1}). In the driest climate (Shafter), BAU residual NO_3^- exceeded 500 kg NO_3^--N ha^{-1} in 7 of 11 coarse soils (interquartile range: 370–882 kg NO_3^--N ha^{-1}), 10 of 11 loamy soils (interquartile range: 586–1860 kg NO_3^--N ha^{-1}), and 2 of 11 fine soils (interquartile range: 137–342 kg NO_3^--N ha^{-1}). In the driest climate, extreme values of residual NO_3^- accumulated in 2 of 11 coarse soils (1693–2249 kg NO_3^--N ha^{-1}) and 4 of 11 loamy soils (1831–3145 kg NO_3^--N ha^{-1}), but the most extreme fine soil accumulated far less (887 kg NO_3^--N ha^{-1}).

Field studies in the region have demonstrated similar findings to this modeling exercise where soil profile nitrate was shown to accumulate during dry years and leach into the vadose zone during wet years (Liang et al., 1991; Raij-Hoffman et al., 2024). Soil coring studies from the southern portions of the San Joaquin Valley, California, which correspond with the driest sites in this study, show residual NO_3^- masses similar to but slightly lower (180–269 kg ha^{-1}) than the range in values derived from these RZWQM2 simulations of fertilized maize (Bachand et al., 2014; Harter et al., 2005; Waterhouse et al., 2021). These studies sampled sites growing almonds, nectarines, tomatoes, grapes, and wheat, and did not span the gradient in texture modeled here.

In drier climates, loamy soils tended to present the greatest possibility of risk of additional NO_3^- leaching with Flood-MAR (Figure 2). In the driest climate (Shafter), 4 of 11 loamy soils leached >3000 kg additional NO_3^--N ha^{-1} over 37 years under the 21-day frequency Flood-MAR scenario, while the median flux was 1270 kg additional NO_3^--N ha^{-1} (interquartile range: 665–3347 kg additional NO_3^--N leached ha^{-1}). Although coarse- and fine-textured soil groups only leached marginally less NO_3^- (median) compared to loamy soils during Flood-MAR treatments (Figure 2), the interquartile ranges of coarse and fine textural groups were much narrower: 695–1007 and 622–1214 kg additional NO_3^--N leached ha^{-1}, respectively. In the second driest climate (Five Points), 3 of 11 loamy soils leached >3000 kg additional NO_3^--N ha^{-1}, with median fluxes of 990 kg additional NO_3^--N ha^{-1} (interquartile range: 609–2360 kg additional NO_3^--N leached ha^{-1}). Similar to the driest climate, interquartile ranges of coarse and fine soil groups were much narrower: 499–764 and 532–1103 kg additional NO_3^--N leached ha^{-1}, respectively.

Although there are several facets of the N-cycle, which can complicate this risk assessment (Figures 3 and 4; Figures S1–S4), the most direct mechanistic explanation for additional NO_3^- leaching risk from loamy soils in the drier climates is due to their moderate level of microporosity and capacity to accumulate NO_3^- due to high water storage capacity. Loamy soils require more percolating water to leach effectively compared to coarse soils, explaining their conduciveness to residual NO_3^- accumulation in drier settings relative to coarse soils (Figure 1). These model findings are supported by deep core analysis from California field crops where a significant negative relationship was found between clay content in the soil profile (particle size control section) and NO_3^- concentration in the vadose zone (1.8–8 m; Lund et al., 1974). The study found low residual NO_3^- concentrations (<5 μg g^{-1}) in the upper 1.8 m of sandy soils underlain by higher concentrations (20 μg g^{-1}) in the deep vadose zone. In contrast, comparatively high amounts of NO_3^- were found in the upper 1.8 m of loamy soils (peaking at 15 μg g^{-1}) with lower concentrations (5 μg g^{-1}) in the underlying deep vadose zone. Onsoy et al. (2005), in a 15 m deep, highly heterogeneous vadose zone, reported the lowest NO_3^--N concentrations in coarse-textured sediment facies relative to finer textured sediments. Similarly, lab-based core leaching experiments have shown finer textured soils retained more NO_3^--N compared to sandy soils (Gaines & Gaines, 1994).

While fine-textured soils also have high water storage capacity, residual N buildup was less (Figure 1) due to substantially higher denitrification (Figures 3 and 4). Fine-textured soils have the potential to experience higher denitrification rates and/or longer episodes of denitrification due to longer durations of elevated water-filled pore space (Groffman & Tiedje, 1988; Schindlbacher et al., 2004).

The RZWQM2 irrigation module uses an idealized irrigation schedule assuming perfect foresight by the farmer and uniform irrigation conditions: irrigation was triggered to apply water when a set threshold of plant-available water was depleted. Upon triggering, RZWQM2 applied a depth of water equal to refilling the soil's real-time, dynamic rooting depth to field capacity. This highly efficient irrigation decision approach had the effect of minimizing deep percolation, thus overestimating the accumulation of residual nitrate. To some extent, the higher NO_3^- mass leaching in the drier modeled climates would have been mitigated if this ideal BAU irrigation scheme was less efficient or if winter water applications were used to mitigate salt buildup, and thus lessen the impact of Flood-MAR on NO_3^- leaching concentrations.

Under real, practical conditions, conventional irrigation management approaches recommend applying more water than necessary to avoid creating soil moisture stress. Furthermore, irrigation systems do not apply water uniformly, and infiltration rates are inherently variable (Letey, 1985). For example, if 5 cm of applied water is needed to offset evapotranspiration, then a system with 80% irrigation distribution uniformity would need a 6.25-cm water application to prevent soil moisture stress. The consequence of this irrigation strategy is that most portions of a field will actually receive a depth of water greater than the amount necessary to refill the depleted soil moisture, resulting in deep percolation and NO_3^- leaching below the root zone as a consequence. A similar outcome is practiced in areas prone to salinity, where leaching fractions are used to manage the salinity of root zones (Rhoades et al., 1973), now often postharvest to avoid reducing nitrogen use efficiency. Applying more water to address salinity or irrigation distribution uniformity could complicate the interpretation of this study's results. However, while such real-world practices may mean more deep percolation than this study suggests, it would also mean the Flood-MAR risk for additional NO_3^- leaching in this study is a conservatively high "book-end" estimate. Our results present a relatively conservative risk assessment, and Flood-MAR would more clearly be a safe groundwater-quality mitigation tool in drier climates as well. Conversely, widespread use of subsurface drip irrigation throughout California has been shown to limit root zone leaching and/or minimize leaching to concentrated areas surrounding the drip line (Hanson et al., 2008), leading to conditions more closely reflecting the simulated BAU here.

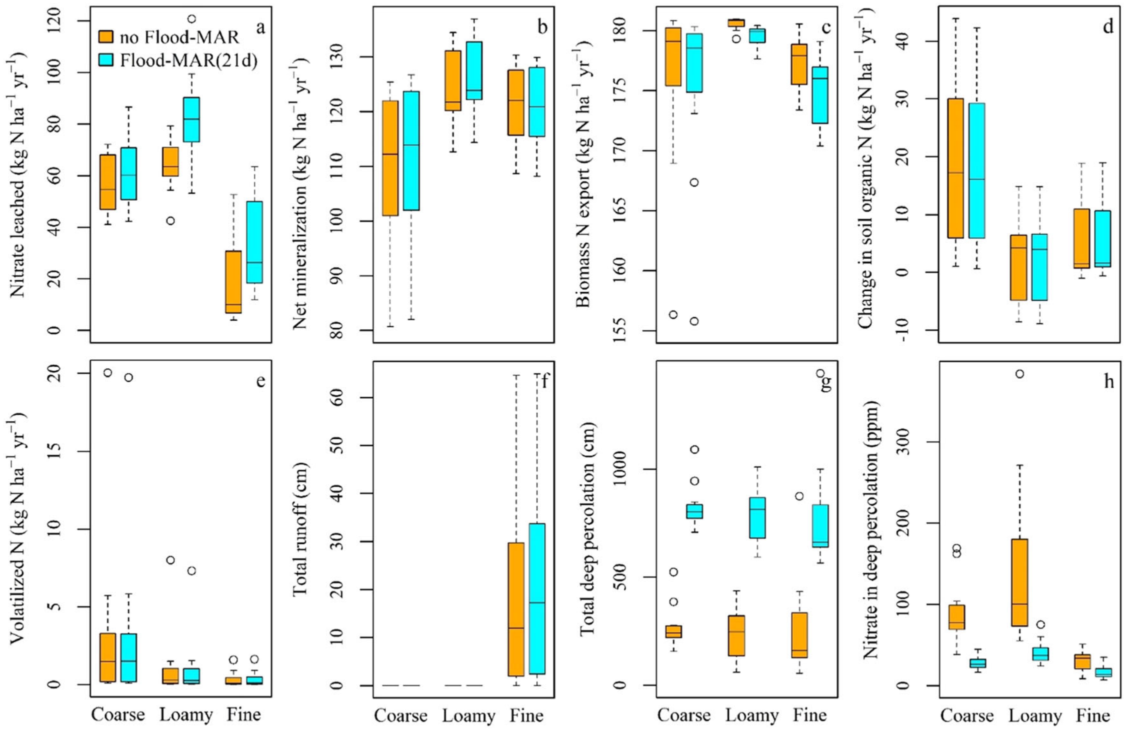

FIGURE 3 (a–h) RZWQM2 simulated effects, comparing agricultural management of floodwaters (Flood-MAR) with business-as-usual (Control) on annualized N mass balance, hydrology, and water quality: (a) nitrate leached mass, (b) net mineralization, (c) harvested crop biomass N, (d) soil organic matter change affecting N balance, (e) volatilized N, (f) runoff, (g) deep percolation, and (h) NO_3^- in deep percolation. Each plot is grouped by generalized taxonomic family particle-size classes: coarse, loamy, and fine ($n = 11$ per class). The crop system is 250 kg N ha^{-1} year^{-1} fertilized and irrigated maize in Parlier, CA (median precipitation = 278 mm year^{-1}). Box plots display the median and interquartile range. Whiskers display the distribution of data extending 1.5 times the upper and lower quartiles. Open circles are outliers.

3.3 | Additional NO_3^- leaching risk in drier climates is mitigated by denitrification in finer textured soils and relative ease of leaching in coarse soils

The residual NO_3^- interquartile ranges of coarse and loamy texture groups tended to increasingly overlap from dry to driest (Figure 1). This convergence partly reflected the denitrification capacity of loamy soils under high residual NO_3^- conditions (Figure 4), which contributed to mitigate additional NO_3^- leaching risk of loamy soils in drier climates (Figure 2). However, loamy soils maintained an environment less favorable for denitrification compared to fine-textured soils (D'Haene et al., 2003), as simulated by RZWQM2 (Figure 4).

In fine-textured soils, denitrification limited the accumulation of residual soil NO_3^- to a greater extent, especially when dry climates favored buildup of NO_3^- due to less deep percolation (Figures 1, 3g, and 4; Figures S3g and S4g). Coarse soils displayed only negligible denitrification capacity, even under the driest climate (Figure 4). In fact, their capacity to accumulate some additional residual NO_3^- in the driest climate was reflected in the BAU run compared to the SIC run that assumed BAU conditions (Figure 1). In addition to denitrification, the greater microporosity of fine soils physically limited BAU NO_3^- leaching, especially in drier climates, making the notably lower total NO_3^- leached in Flood-MAR scenarios from fine soils more impactful on additional NO_3^- leaching risk due to Flood-MAR (Figures 3a and 5–7; Figures S1a–S4a). This is in spite of the fact that deep percolation depths from fine soils were not markedly different compared to coarse and loamy soils in drier climates (Figure 3g; Figures S3g–S4g). The retention of NO_3^- in soil under BAU, drier climate scenarios (Figures 5 and 7) helps explain the similar relative responsiveness of coarse and fine soils to Flood-MAR in terms of additional NO_3^- leached (Figure 2). In the BAU scenario under the driest climate, coarse soils typically leached 1779 kg NO_3^--N ha^{-1} (interquartile range: 1305–1917 kg NO_3^--N ha^{-1}), compared to 117 kg NO_3^--N ha^{-1} (interquartile range: 37–327 kg NO_3^--N ha^{-1}) from fine soils (Figures 5 and 7). Thus, this directly offset the Flood-MAR effect on leaching in coarse soils but exacerbated the NO_3^- leaching risk from fine soils, in spite of their high levels of denitrification and lower residual NO_3^-. Denitrification losses can be very high in fields with clay texture (Pratt et al., 1972), and the relationship between texture and denitrification has been documented where denitrification potential has been shown to be highest in fine-textured soils, moderate in loamy

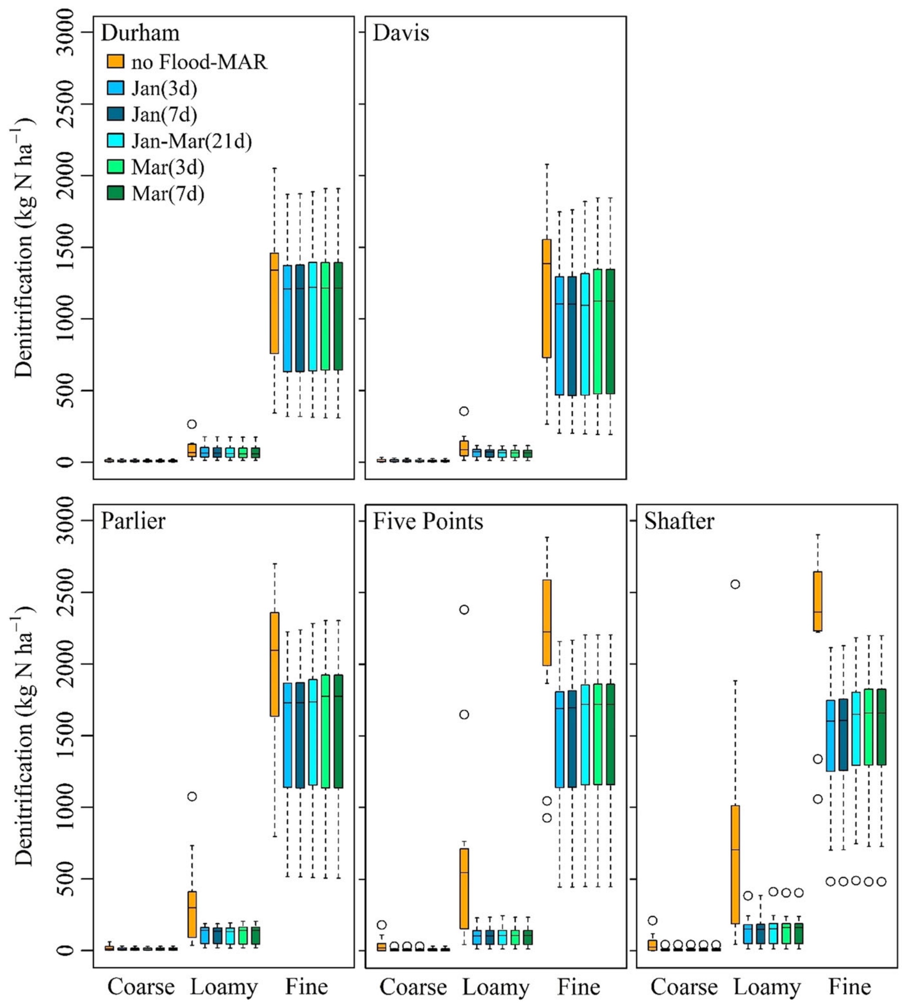

FIGURE 4 Effect on 37-year, cumulative denitrification of business-as-usual (no Flood-MAR) and various Flood-MAR strategies, including 3-day and 7-day frequency events practiced either in January or March or 21-d frequency events spanning early January through late March. Each plot is grouped by three generalized taxonomic family particle-size classes: coarse, loamy, and fine (n = 11 soils per class), and shows the different climates modeled from wettest (Durham) to driest (Shafter). See Table 1 for descriptions of climates. Box plots display the median and interquartile range. Whiskers display the distribution of data extending 1.5 times the upper and lower quartiles. Open circles are outliers.

soils, and low in coarse soils when water-filled pore space exceeds 60% (D'Haene et al., 2003). A study of deep cores across a texture gradient in California found residual nitrate concentrations decreased with increasing clay content. The study suggests that while clay-rich soils have the potential to build up in N via incomplete leaching, this N pool is removed by denitrification (Lund et al., 1974). Greater losses of NO_3^- via denitrification were simulated in fine (silt loam)-textured compared to coarse (sandy loam)-textured vadose zones during Flood-MAR using a reactive transport model (Waterhouse et al., 2021).

3.4 | NO_3^- leaching risk "tipping point"

Comparing BAU deep percolation rates versus additional NO_3^- risk highlights a clearly nonlinear relationship to climate and, specifically, precipitation (Figure 8). The inflection or "tipping point" of this relationship is perhaps best exemplified by the Parlier climate (median annual precipitation: 278 mm year^{-1}), where coarse soils showed less additional NO_3^- leaching risk compared to loamy and fine soils (Figure 2). The Parlier climate had ample precipitation to effectively leach most coarse soils in the initial BAU scenario, preventing accumulation of residual NO_3^-, but not enough for most loamy soils (Figure 1). Accordingly, loamy soils leached more additional NO_3^- during Flood-MAR compared to coarse soils in the Parlier climate (Figure 2). Despite substantial denitrification limiting residual NO_3^- accumulation in fine soils in the Parlier climate (Figures 1 and 4), the typical fine-textured soil leached slightly more additional NO_3^- (460 additional kg NO_3^--N ha^{-1}) compared to loamy soils (395 additional kg NO_3^--N ha^{-1}) (Figure 2). However, loamy soils arguably still had the greater additional NO_3^- leaching risk in

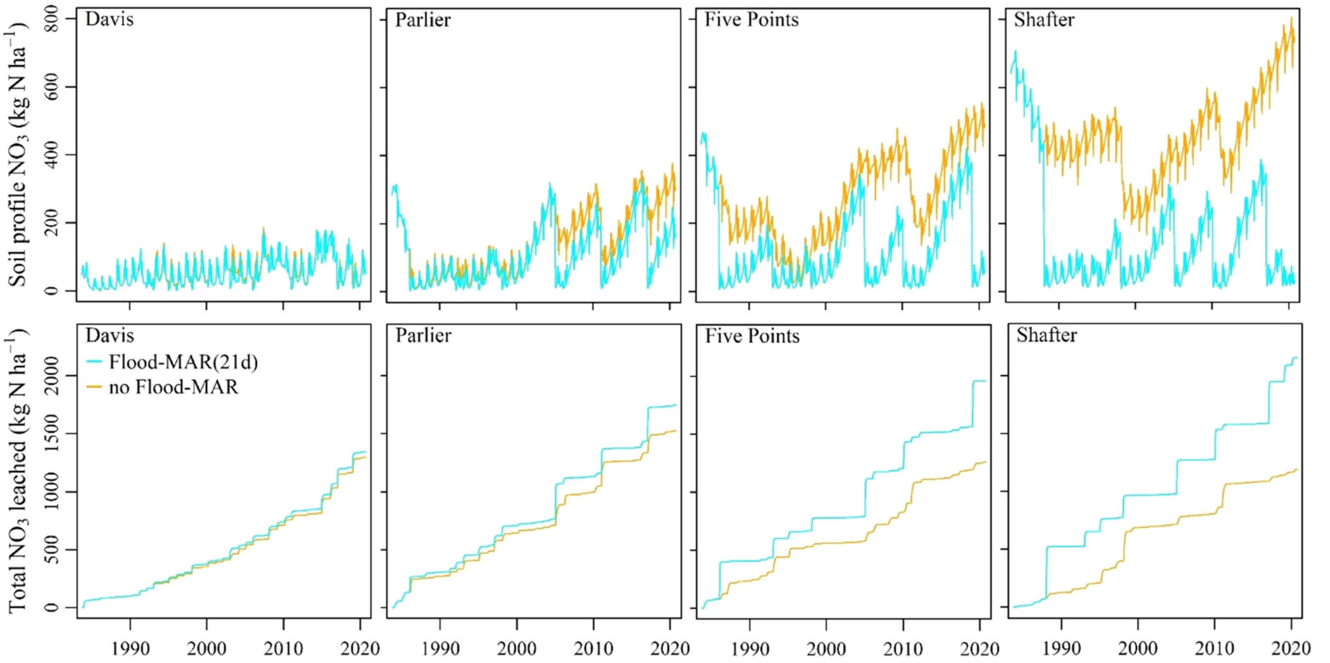

FIGURE 5 Typical soil profile NO_3^- (top row) and total NO_3^- leached (bottom row) for a *coarse* (taxonomic family particle-size class) soil, comparing business-as-usual (no Flood-MAR) versus managed groundwater recharge (Flood-MAR(21d) = 60 cm year^{-1} across four applications on 21-day intervals), practiced during the 10 wettest winters from Davis (left column), the second wettest climate modeled, through Shafter (right column), the driest climate modeled. Note the difference in the *Y*-axis scale in wet versus drier sites.

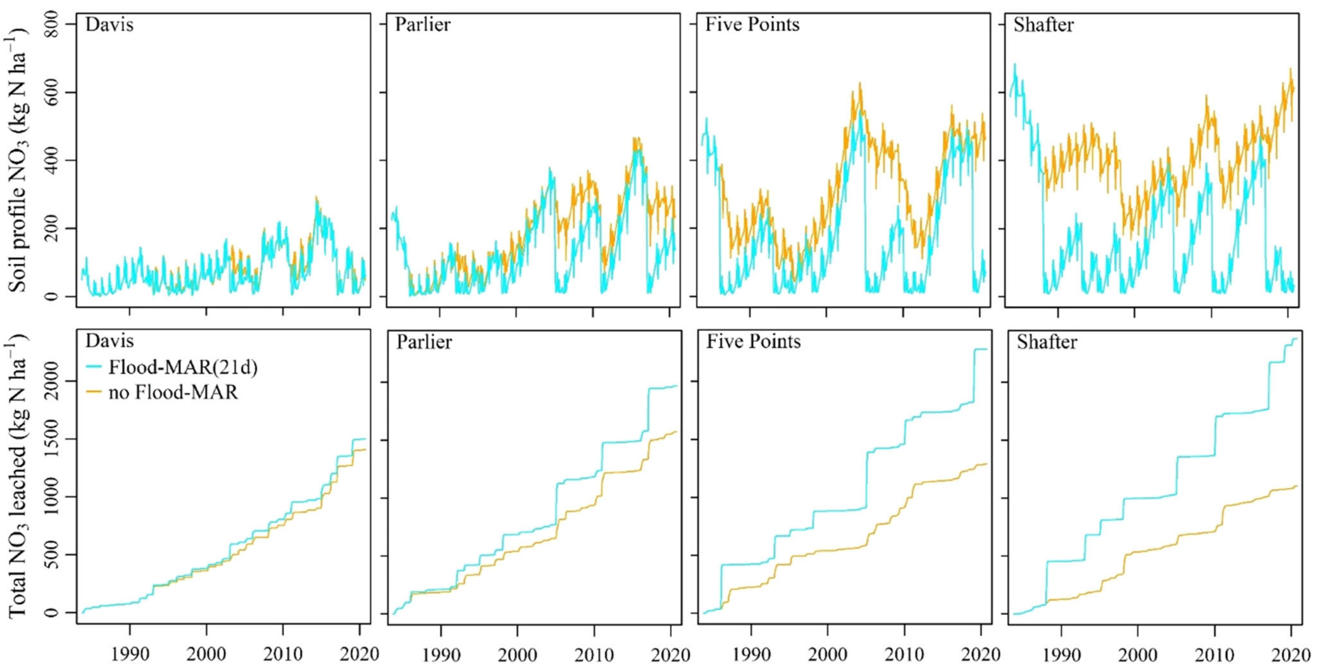

FIGURE 6 Typical soil profile NO_3^- (top row) and total NO_3^- leached (bottom row) for a *loamy* (taxonomic family particle-size class) soil, comparing business-as-usual (no Flood-MAR) versus managed groundwater recharge (Flood-MAR(21d) = 60 cm year^{-1} across four applications on 21-day intervals), practiced during the 10 wettest winters from Davis (left column), the second wettest climate modeled, through Shafter (right column), the driest climate modeled. Note the difference in the *Y*-axis scale in wet versus drier sites.

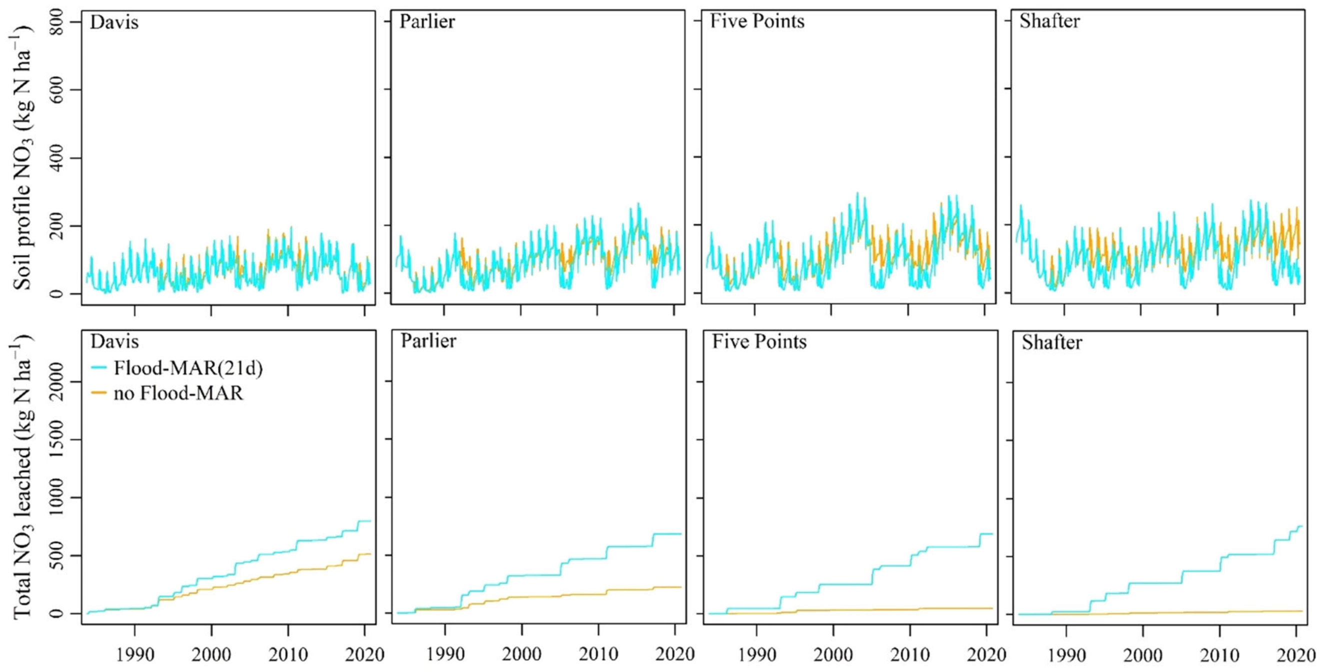

FIGURE 7 Typical soil profile NO_3^- (top row) and total NO_3^- leached (bottom row) for a *fine* (taxonomic family particle-size class) soil, comparing business-as-usual (no Flood-MAR) versus managed groundwater recharge ((Flood-MAR(21d) = 60 cm year^{-1} across four applications on 21-day intervals), practiced during the 10 wettest winters in Davis (left column), the second wettest climate modeled, through Shafter (right column), the driest climate modeled. Note the difference in the *Y*-axis scale in wet versus drier sites.

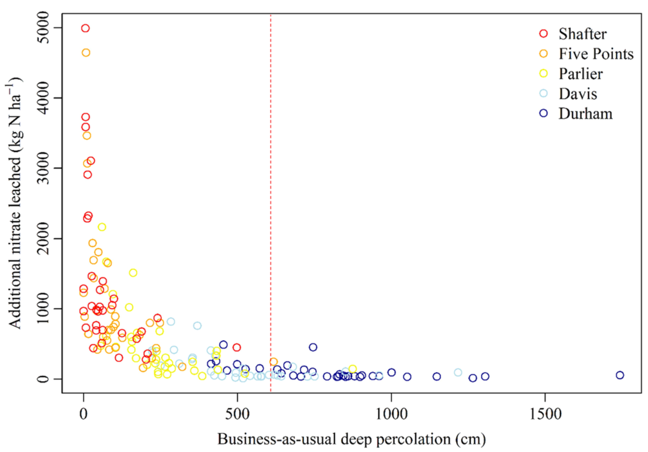

FIGURE 8 Relationship between total, 37-year deep percolation in the business-as-usual scenario (no Flood-MAR) and additional nitrate leached due to Flood-MAR practiced during the 10 wettest water years (October–September), applying 60 cm^{-1} water year^{-1} across four applications on 21-day intervals. The vertical red line indicates the total Flood-MAR water applied over the 37-year period. Data shown across all climates modeled (n = 5) and soils (n = 33), symbolized by climate. Additional nitrate leached is relative to each soil's business-as-usual simulation in a particular climate. See Table 1 for description of climates.

the Parlier climate due to their propensity for more extreme residual NO_3^- accumulation, despite the relatively high levels of BAU NO_3^- leaching (3 of 11 soils developed >1000 kg residual NO_3^--N ha^{-1} in Parlier; Figure 1). This was directly linked to additional NO_3^- leaching with Flood-MAR that was skewed toward more extreme fluxes in loamy soils (Figure 2). Close examination of RZWQM2 breakthrough curves for these more extreme loamy soils indicated that they did not generate substantial deep percolation until at least the second 15-cm Flood-MAR event. This explains the propensity of loamy soil to accumulate residual NO_3^- in BAU scenarios in the just-dry-enough Parlier climate with not enough precipitation during even the wettest years to generate substantial deep percolation.

3.5 | Effect of Flood-MAR timing

Despite the marked increase in additional NO_3^- leaching with Flood-MAR in drier climates, no timing strategy was effective at appreciably mitigating this risk. However, there were consistent effects of late-season versus early-season Flood-MAR timing across different soils. Specifically, coarse and loamy soils experienced more additional NO_3^- leached during later season Flood-MAR scenarios (Figure 2, March 3-day and 7-day). Early-season Flood-MAR experienced reduced additional NO_3^- leached because mineralization rates are subdued at cooler temperatures (Miller & Geisseler, 2018). Moreover, Flood-MAR had the effect of reducing annual denitrification compared to BAU (Figure 4) by decreasing residual soil NO_3^- through leaching before the growing season when denitrification potential is greatest on account of warmer, more biogeochemically active soils. In addition to the amounts of O_2 and available C in soil, residual NO_3^- concentration is a primary control of denitrification (Myrold & Tiedje, 1985).

3.6 | Flood-MAR following multiyear droughts presents some risk

Across the 37-year simulation with Flood-MAR during the 10 wettest winters, median Flood-MAR scenarios tracked BAU scenarios closely in wetter climates: the typical coarse soil leached just 43 and 45 kg additional NO_3^--N ha^{-1} in Durham and Davis, respectively (Figure 5); the typical loamy soil leached just 54 and 63 kg additional NO_3^--N ha^{-1} in Durham and Davis (Figure 6); and the typical fine soil leached 152 and 336 additional NO_3^--N ha^{-1} in Durham and Davis, respectively (Figure 5).

However, in drier climates, residual NO_3^- accumulated during periodic droughts in the climate record (e.g., 2012–2016), which was then flushed as deep percolation during subsequent wet years (e.g., 2017) in the Flood-MAR scenarios on typical coarse and loamy soils (Figures 5 and 6). Thus, the timing of Flood-MAR following multiyear droughts is of special concern in drier climates (median precipitation < 400 mm) and especially on more difficult-to-leach soils where high levels of residual NO_3^- accumulation are possible. It is during these dry years that residual NO_3^- would be accumulating, but it is relatively rare in these locations that wet-year precipitation is sufficient to initiate substantial deep percolation capable of leaching soils effectively. If precipitation whiplash continues, strategies will be needed to deplete residual nitrate either via improved fertigation methods (higher nitrogen use efficiency), crop rotations, or use of cover crops (Delgado et al., 1999).

In California, nutrient management plans are used as tools to help growers reduce nitrogen export from irrigated agriculture into groundwater and surface waters of the State. Nutrient management plans are designed to help growers understand the nitrogen cycle and the N budget, especially the amount of N left in soil after harvest. N budgeting may partially mitigate concerns about residual NO_3^- buildup in soil, which drives the outcomes of this study. Improved N management in the future may reduce the risk of leaching when Flood-MAR is practiced. However, this study simulated ideal irrigation and N application rates, which is a conservative estimation. Greater adoption of N budgeting would only make this study more relevant to actual field practices.

3.7 | Simulated runoff behavior in wetter climates partially influences results

The greater microporosity of higher clay content soils also explained why fine soils had the highest levels of additional NO_3^- leached with Flood-MAR in the wetter climates of Durham and Davis, though the risk was relatively low compared to drier climates (Figure 2). The precipitation regimes of wetter climates were not completely effective at leaching the fine soils. However, the leaching response of two fine outlier soils in wetter climates was explained by their simulated high runoff, reducing effective precipitation and deep percolation in BAU scenarios (Figures S1–S3). Irrigation as applied in RZWQM2 cannot induce runoff (all applied water is assumed to infiltrate), which was the mechanism by which water was applied in the Flood-MAR scenarios. This was accepted as a realistic assumption because farmers practicing Flood-MAR would be expected to control runoff and manage for maximizing infiltration. In fact, to represent a comparable BAU scenario by minimizing the effect of runoff in RZWQM2 in wet climates, daily precipitation data from CIMIS were assumed to occur in 12-h storms with variable precipitation intensities of 0.05 (min) – 5.0 (max) cm h^{-1}, creating low runoff conditions in BAU scenarios for most soils (Figure 4; Figures S1–S4). On the other hand, if an actual "business-as-usual" scenario involved a field with, for example, surface sealing and infiltration issues, resulting in appreciable runoff and reduced deep percolation, then a transition to Flood-MAR management aimed at improved infiltration and enhanced deep percolation could present additional NO_3^- leaching risk not represented in this study.

3.8 | Simulated soil organic matter accumulation partially influences additional NO_3^- leaching risk, especially in coarse soils

RZWQM2 simulations consistently projected greater SOM accumulation in the coarse soil scenarios and hence, greater soil organic N accumulation (Figure 3d; Figures S1–S4). The change in organic N was above 15 kg N ha^{-1} in coarse soils compared to less than 5 kg N ha^{-1} in loamy and fine soils. This phenomenon reduced the total N pool available for leaching from coarse soils across growing seasons. This also partly explained the higher additional NO_3^- leaching risk under loamy soils compared to coarse soils.

3.9 | Implications for groundwater and well water nitrate concentrations

Results shown here are relevant due to the potentially long-term implementation of various Flood-MAR regimes as climate resiliency and groundwater (supply) sustainability practices are expected to expand in the future (Marr et al., 2018). Potentially widespread implementation of these practices is being considered to maximize the retention of floodwaters in aquifers rather than their loss to the ocean

(Marr et al., 2018). Simulations here pertain to losses of NO_3^- from the most dynamic and reactive portion of the unsaturated zone, the upper 2 m of soil. Conservatively, assuming relatively low reactivity in the vadose zone below soil and furthermore assuming laterally widespread application of these simulated conditions, the NO_3^- concentration in leachate from soil would be similar when it becomes recharge at the water table. And, if that recharge has the same concentration across a relatively large lateral extent, it would be expected that shallower (predominantly domestic) wells also experience concentrations of that magnitude, albeit after some period of delay corresponding to the travel time from the root zone to the water table and from the water table to the well (typically a few to tens of years). In this section, we address two questions: what concentration of NO_3^- do the aforementioned additional losses of NO_3^- mass due to Flood-MAR represent? And, second, how is that concentration different from those under BAU conditions?

The difference in mass losses from soil ranges from a few tens to several thousand kg N ha^{-1} over the simulated period (Figures 5–7). The main difference in water percolation over that same period is approximately 600 cm across all scenarios (the amount of additional Flood-MAR applied). The drinking water limit of 10 mg NO_3^--N L^{-1} is equivalent to 1 kg N/(ha·1 cm) or 10 kg N ha^{-1} in 10 cm of water (10 kg N in 0.1 ha-m of water) or 600 kg N ha^{-1} in 600 cm of water. From basic mass balance considerations, the *additional* water percolating from Flood-MAR does not exceed the nitrate maximum contaminant level (MCL), where the additional N leaching over the simulation period does not exceed 600 kg N ha^{-1} (Figure 8: almost all wet climate scenarios in Durham and Davis, some scenarios in the intermediate climate in Parlier, and few scenarios in the dry climate). Under dry climate conditions, where median N losses are on the order of twice 600 kg N ha^{-1}, the *additional* water percolating from Flood-MAR may be on the order of twice the MCL concentration, in some cases five times the MCL concentration (where the additional leaching is 3000 kg N ha^{-1}). Clearly, under the drier climate conditions, without adding additional practices such as improved nitrogen use efficiency, crop rotation, cover crops, or higher Flood-MAR applications than 60 cm in the wettest winters, the percolation due to the additional Flood-MAR water cannot meet drinking water standards. In fact, modeled NO_3^- concentration in deep percolation waters receiving Flood-MAR treatment was reduced to equal the drinking water standard for most fine soil groups, while being slightly to moderately above the standard for loamy and coarse texture groups depending on location (Figure 3h; Figures S1–S4).

The NO_3^- concentrations of deep percolation water in the BAU scenarios were higher in dry regions (Figure 3h). Moreover, NO_3^- concentrations of deep percolation water in the BAU scenarios can be estimated from the total mass of N leached under BAU conditions (no Flood-MAR; Figures 5–7) and the corresponding total BAU percolation (Figure 8). The total BAU NO_3^- leached was highest in coarse soils, ranging from 1000 kg N ha^{-1} under dry climates to over 1500 kg N ha^{-1} in wet climates (Figure 5). The corresponding percolation rates over the simulation period range from less than 20 to 250 cm in drier climates and from 300 to over 1200 cm in wet climates (Figure 8). Hence, under BAU conditions of drier climate, nitrate-N concentrations in (very slowly) percolating water exceed the MCL by nearly one to nearly two orders of magnitude (1500/250–1500/20 kg N/(ha·cm water)), while, under BAU conditions of wetter climates, nitrate-N concentrations in (rapidly) percolating water range from at the MCL to five times above the MCL (1500/300–1500/1200 kg N/(ha·cm water)) under the simulated conditions.

Hence, under all climate conditions, the NO_3^- concentration in the *additional* percolating water due to Flood-MAR is significantly lower than the NO_3^- concentration in percolating water from BAU conditions. Due to the mixing of these two waters in the root zone and further in the deeper unsaturated zone, shallow groundwater, and along the vertical well screen of an extraction well, the net long-term effect of Flood-MAR is a significant reduction of NO_3^- concentration in shallow groundwater and, hence, in domestic and shallow public water supply water under the dominant influence of agricultural return water recharge. Despite the fact that Flood-MAR leads to additional nitrate *mass* leaching, especially under dry climate conditions, the amount of Flood-MAR in the simulated scenarios is sufficient to dilute this additional nitrate mass to less than or slightly above the NO_3^- MCL depending on soil type and climate (Figure 3h). Pratt et al. (1972) arrived at a similar finding showing low $[NO_3^-]$ (below MCL) in deep leachate from orange orchards in the Central Valley due to the application of excess irrigation water to maintain low salt levels in soils.

The above considerations, while hypothetical, are highly relevant for consideration of the long-term impacts of Flood-MAR on well water NO_3^- concentrations but will be further modified due to the fact that stream and incidental canal recharge with typically negligible NO_3^- concentrations plays an important role particularly in the drier climate conditions of the southern Central Valley, leading to further dilution of irrigated lands return flow recharge NO_3^- in some wells. We note that, under the typical percolation and recharge rates of the drier climate scenarios (50 cm/37 years) when compared to the wet climate scenarios (500 cm/37 years; Figure 8), the source area of a well, at any given pumping rate, would be an order of magnitude larger in dry climate locations than in wet climate location, assuming the entire landscape was subject to the simulated scenario conditions.

4 | STUDY LIMITATIONS

This study evaluated the nitrate leaching risk of 33 different soil series with contrasting physical properties representative of commonly mapped soils across the region. The broad scale of the study domain, the magnitude of time evaluated, and the large number of Flood-MAR scenarios investigated made it impossible to calibrate and validate the model uniformly with field data. Thus, for the most part, model default conditions were used. Model calibration is site-specific, and this study lacked representative locations for this purpose. As a result, results reflect state factor controls (climate and soil properties that influence the water balance) on additional NO_3^- leaching risk through a theoretical representation of N cycling in response to Flood-MAR scenarios. Future studies could explore improvements in model simulations resulting from site-specific calibrations.

Similarly, the study used soil survey data to parameterize physical and hydraulic model conditions. RZWQM2 uses the Brooks and Corey parameters to characterize soil water retention. This approach has been widely tested in diverse settings and soil types (Ahuja et al., 2000; Ma et al., 2006). More recent studies, however, have shown that optimization procedures to calibrate hydraulic parameters in the model using Latin hypercube sampling and gradient-based optimization improved the model performance (Fang et al., 2010; Ma et al., 2011). The results of our approach could have higher uncertainty compared to in-depth parameterization procedures. Moreover, the depth of soils studied here is not equal, varying randomly between 150 and 218 cm, which may have influenced some of the variability seen among texture classes.

The study did not assume surface ponding during Flood-MAR scenarios, which could occur. Ponding has the potential to increase the rate of deep percolation when the soil is saturated. Ponding is likely to have minimal impact on denitrification rates because the process is occurring across a time frame spanning soil water states from saturation to field capacity when air-filled pore space is low (Bateman & Baggs, 2005; Mekala & Nambi, 2017). In addition, the fate of NO_3^- in soil during the dormant season is largely controlled by temperature and the amount of water applied, not the differences in flow rate as influenced by ponded versus non-ponded conditions (Ludwick et al., 1976; Miller & Geisseler, 2018; Or et al., 2007). Moreover, our simulations, which reflect different durations of saturation, show no difference in denitrification when comparing 3-, 7-, or 21-day Flood-Mar treatments. This indicates that denitrification during Flood-MAR periods is controlled by low temperature and the initial amount of NO_3^- in the soil rather than the duration of saturation. Moreover, ponding is likely to be discouraged during Flood-MAR due to its potential harm to crops and surface soil condition (O'Geen et al., 2015). We perceive and recommend Flood-MAR to resemble water application strategies similar to surface irrigation procedures in order to limit the negative effects of standing water.

This study only modeled one of the over 200 crops grown in California. Various crops will inevitably have different N requirements, irrigation strategies, rooting depths, and N use efficiencies, which will influence residual nitrate buildup. Although the residual N accumulation will differ among crops, this study is generalizable to any crop that has a dormant period over the winter months and some degree of residual nitrate accumulation. Clearly, absolute amounts of residual N buildup will differ by fertilization rate and crop uptake, and, most importantly, irrigation efficiency, but the climate and soil are what drive the buildup. Corn as a test crop is least generalizable to crops that have low N demand such as wine grapes and alfalfa.

Despite these limitations, this study has value in presenting new testable research questions and management considerations such as the likelihood of denitrification during the dormant season, the potential buildup of residual nitrate in response to drought, the effects of irrigation management, the potential impact of climate whiplash on residual NO_3^-, and the relevance of the N-cycle process in the context of Flood-MAR. Findings are not intended to prescribe the ideal conditions for Flood-MAR; rather, they justify the importance of documenting residual nitrate concentrations when considering this practice.

5 | CONCLUSION

Coarse soils are considered of the greatest risk to NO_3^- leaching into groundwater when N-fertilizer applications exceed N-removal processes in an agroecosystem: export in crop biomass, accumulation in SOM, volatilization, and denitrification (Figure 4). However, this truism did not hold up to evaluations of the effect of agricultural management of floodwater (Flood-MAR) on additional NO_3^- leaching risk. Simply put, except in the driest climates, precipitation is sufficient in California's Central Valley to leach residual NO_3^-, such that the additional NO_3^- leaching risk presented by Flood-MAR is typically lower in coarse soils compared to loamy soils.

Overall, this research suggests that Flood-MAR timing strategies, specifically seasonality and intervals between Flood-MAR events, are generally negligible in their effects on NO_3^- leaching risk in comparison to the risk of accumulating residual nitrate in dry climates. Nevertheless, the results point to some broader timing concerns for using Flood-MAR as a general strategy to recharge depleted aquifers. Most importantly, growers are advised to manage nitrogen carefully in dry years and monitor residual NO_3^- following successive dry years, especially in locations where natural deep percolation is already typically limited by low rainfall. Residual NO_3^- may accumulate most rapidly during droughts, especially when growers face limited water resources that may typically result in lower-than-expected growing season soil flushing due to irrigation and leaching practices. However, under all scenarios, Flood-MAR is likely to noticeably reduce NO_3^- concentration in groundwater relative to BAU over the long run.

AUTHOR CONTRIBUTIONS

Scott Devine: Conceptualization; data curation; formal analysis; investigation; methodology; validation; visualization; writing—original draft; writing—review and editing. **Helen Dahlke**: Conceptualization; funding acquisition; writing—review and editing. **Thomas Harter**: Writing—review and editing. **Isaya Kisekka**: Writing—review and editing. **Majdi Abou Najm**: Writing—review and editing. **Anthony T. O'Geen**: Conceptualization; funding acquisition; investigation; project administration; resources; supervision; writing—original draft; writing—review and editing.

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CONFLICT OF INTEREST STATEMENT

The authors declare no conflicts of interest.

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