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1 **Nitrogen fate during agricultural managed aquifer recharge: linking plant**
2 **response, hydrologic, and geochemical processes**

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12 **Abstract**

13 Agricultural managed aquifer recharge (Ag-MAR, on-farm recharge), where farmland is
14 flooded with excess surface water to intentionally recharge groundwater, has received increasing
15 attention by policy makers and researchers in recent years. However, there remain concerns
16 about the potential for Ag-MAR to exacerbate nitrate (NO_3^-) contamination of groundwater, and
17 additional risks, such as greenhouse gas emissions and crop tolerance to prolonged flooding.
18 Here, we conducted a large-scale, replicated winter groundwater recharge experiment to quantify
19 the effect of Ag-MAR on soil N biogeochemical transformations, potential NO_3^- leaching to
20 groundwater, soil physico-chemical conditions, and crop yield. The field experiment was
21 conducted in two grapevine vineyards in the Central Valley of California, which were each
22 flooded for 2 weeks and 4 weeks, respectively, with 1.31 and 1.32 $\text{m}^3 \text{m}^{-2}$ of water. Hydrologic,
23 geochemical, and microbial results indicate that NO_3^- leaching from the first 1 m of the vadose
24 zone was the dominant N loss pathway during flooding. Based on pore water sample and N_2O
25 emission data denitrification played a lesser role in decreasing NO_3^- in the root zone but
26 prolonged anoxic conditions resulted in a significant 29% yield decrease in the 4-week flooded
27 vineyard. The results from this research, combined with data from previous studies, are
28 summarized in a new conceptual model for integrated water-N dynamics under Ag-MAR. The
29 proposed model can be used to determine best Ag-MAR management practices.

30 **Keywords:** MAR, groundwater recharge, nitrate, denitrification, soil, crop tolerance

31 **1. Introduction**

32 One-quarter of the world's population and 40% of agricultural production relies on overdrafted
33 groundwater sources (Connor, 2015), with an expected increase in groundwater reliance due to
34 climate change and growing water demand (Gorelick and Zheng, 2015; Haddeland et al., 2014;
35 Siebert et al., 2010; Wada et al., 2010). Thus, reducing pressure on overdrafted groundwater
36 systems is crucial to increase global resilience of food and drinking water in response to growing
37 human population and climate change pressures.

38 Agricultural managed aquifer recharge (Ag-MAR, or on-farm recharge) is a form of managed
39 aquifer recharge (MAR) where farmland is flooded with excess surface water to recharge
40 groundwater intentionally (Grinshpan et al., 2021; Kocis and Dahlke, 2017; Waterhouse et al.,
41 2020) and it has been increasingly used across the globe to address groundwater overdraft
42 (Dillon et al., 2019; Levintal et al., 2022). The purpose of Ag-MAR, in comparison to more
43 traditional MAR methods, is to transfer large amounts of surplus surface water from rivers or
44 reservoirs to agricultural land (e.g., idle land, agricultural fields and orchards) that serve as
45 spreading grounds for the recharge (Dahlke et al., 2018).

46 Ag-MAR adoption has increased in recent years, particularly in the USA and Europe
47 (Grinshpan et al., 2021; Negri et al., 2020; Niswonger et al., 2017). In California, for example,
48 Ag-MAR is implemented as one of the methods to overcome ongoing groundwater depletion
49 (Kocis and Dahlke, 2017). However, using farmland as spreading grounds bears the risk to leach
50 contaminants from the water or soil to groundwater which can impact drinking water quality
51 (Bachand et al., 2014), waterlogging of the root zone for long periods that can reduce crop health

52 (Ganot and Dahlke, 2021a), and ecosystem service tradeoffs, such as short and long-term effects
53 on in-stream flows (Levintal et al., 2022). Out of the above, leaching of legacy nitrogen (N),
54 mainly in the form of nitrate (NO_3^-), is possibly the most widespread environmental risk of Ag-
55 MAR. Consumption of drinking water above the maximum contaminant level (MCL; 10 mg L^{-1}
56 NO_3^- -N in California) can increase the risk of cancers, birth defects, and other adverse health
57 effects (Weitzman et al., 2022). Globally, nitrate is the primary nonpoint source pollutant of
58 groundwater (Beganskas et al., 2018; Bishayee et al., 2022; Richa et al., 2022), whereby
59 agricultural lands serve as the main source for NO_3^- due to the buildup of legacy NO_3^- resulting
60 from years of fertilizer use inefficiencies (Van Meter et al., 2016; Waterhouse et al., 2020).

61 In comparison to the deep vadose zone, NO_3^- concentrations are highest either in the topsoil (0-
62 10 cm) or just below the root zone (~1-2 m) where NO_3^- is transported out of reach of roots with
63 irrigation (Waterhouse et al., 2021, 2020). NO_3^- in soils can originate from N-based fertilizer or
64 from N transformations, such as nitrification of ammonium (NH_4^+)-based fertilizer or nitrification
65 of NH_4^+ from mineralized soil organic-N (Stein and Klotz, 2016). In contrast, NO_3^- removal
66 pathways are denitrification and immobilization (controlled by microbial activity), leaching
67 (controlled by infiltration rate), and plant uptake (Kurtzman et al., 2021; Long et al., 2013;
68 Schmidt et al., 2011; Zhang et al., 2018). The governing processes during Ag-MAR are NO_3^-
69 leaching, mineralization, nitrification, and denitrification (Murphy et al., 2021; Schmidt et al.,
70 2012; Waterhouse et al., 2021). The latter is favored under suboxic to anoxic conditions, as
71 expected during soil saturation of Ag-MAR events (Ganot and Dahlke, 2021b). N transformation
72 processes, excluding leaching, occur in the soil dependent on electron donor availability (e.g.,
73 dissolved organic carbon (DOC)) and microbial community abundance and composition

74 (Peterson et al., 2013; Scarlett et al., 2021). Ambient conditions may also influence NO_3^-
75 transformations, including soil moisture, carbon/nitrogen ratio, pH, soil texture, temperature,
76 vegetation, oxidation-reduction potential (ORP), and water residence time in the topsoil (Kraft et
77 al., 2014; Stein and Klotz, 2016). The underlying assumption adopted by previous researchers is
78 that NO_3^- -related processes are negligible below the root zone; therefore, NO_3^- leached below the
79 root zone will eventually reach the groundwater (Baram et al., 2016b; Gurevich et al., 2021).
80 Thus, in Ag-MAR, the aim is to reduce legacy NO_3^- via denitrification before NO_3^- leaching
81 occurs below the root zone.

82 NO_3^- leaching is an environmental risk that is not unique to Ag-MAR, but can occur in any
83 MAR method (e.g., infiltration basins; Beganskas et al., 2018), since it mainly depends on the
84 source of NO_3^- in the recharge environment. For example, Beganskas et al. (2018), Gorski et al.
85 (2019), and Schmidt et al. (2011) recharged stormwater runoff from upslope agricultural fields in
86 an infiltration basin in the Pajaro Valley, California, which contained 22-25 mg L^{-1} NO_3^- -N. If
87 low-N source water is used in Ag-MAR (e.g., mountain runoff or snowmelt), elevated NO_3^- in
88 soil pore water often originates from the soil matrix from recurring fertilizer applications.

89 Only a few studies exist to date that have estimated soil NO_3^- biogeochemical transformations
90 and NO_3^- leaching under Ag-MAR. As one of the first, Bachand et al. (2014) investigated NO_3^-
91 fate in a large farm-scale experiment where they flooded alfalfa, wine grapes, tomatoes, and
92 fallow land for various periods ranging from 10 days to one month. They proposed a general
93 framework for NO_3^- fate under Ag-MAR, concluding that NO_3^- will be leached to groundwater,
94 but pore water and groundwater NO_3^- concentrations will be diluted following consecutive flood
95 applications (assuming the use of low- NO_3^- water). However, they did not investigate any N

96 transformations. Waterhouse et al. (2020) estimated the potential risk of NO_3^- leaching under Ag-
97 MAR using only data from 10 m deep soil cores from 12 fields (no flooding was performed),
98 focusing on three crops (almonds, processing tomatoes, and wine grapes) and two soil groups
99 (low- and high-infiltrating soils). They concluded that vineyards were the most suitable crop for
100 Ag-MAR due to low legacy NO_3^- resulting from the crop's deep roots and overall low N inputs.

101 Murphy et al. (2021) investigated the role of flooding frequency (i.e., three flooding events of
102 0.15 m depth each) and soil texture on N dynamics and potential NO_3^- leaching in laboratory
103 column experiments. Each flooding event was on the scale of hours followed by a drying period
104 of several days to two weeks. They observed that the majority of initially present soil NO_3^-
105 leached during the first few hours of the first water application when conditions for
106 denitrification (i.e., removal of NO_3^-) were unfavorable due to oxic conditions. NO_3^- leaching was
107 quantified only for the laboratory column experiments ranging between 0.028 and 0.072 g NO_3^- -
108 N m^{-2} for every 1 $\text{cm}^3 \text{ cm}^{-2}$ of applied water. Based on the combination of lab assay and field data
109 Murphy et al. (2021) showed that using only soil core data from pre- and post-flooding is
110 insufficient to quantify NO_3^- leaching, emphasizing the need for continuous field measurements
111 during Ag-MAR.

112 In a recent modeling study, Waterhouse et al. (2021) investigated Ag-MAR effects on
113 denitrification rates and NO_3^- leaching in a heterogeneous, layered deep vadose zone (~15 m).
114 They found that denitrification rates were highest in response to one continuous extensive water
115 application due to the development of suboxic conditions compared to small incremental
116 recharge events. However, this continuous water application scenario also leached NO_3^- deeper
117 into the vadose zone. None of the above studies measured N-related biogeochemical processes

118 during continuous, long (e.g., several weeks) flooding applications for Ag-MAR in different
119 soils. This suggests that understanding of N fate and NO₃⁻ leaching at the field scale is still
120 elusive in Ag-MAR; thus, further research is needed.

121 This study aims to investigate the environmental impacts of Ag-MAR using a large-scale field
122 experiment conducted in the Central Valley, California (semiarid Mediterranean climate). Two
123 vineyards, each with a different soil texture, were simultaneously flooded during late winter, one
124 for four weeks (V1) and the other for two weeks (V2). Combining hydrologic, geochemical, and
125 microbial process analyses, we quantified the effects of Ag-MAR on the soil N biogeochemical
126 transformations, potential NO₃⁻ leaching to groundwater, soil physico-chemical conditions, and
127 crop response. The results from this research, combined with data from previous studies, are
128 summarized in a new conceptual model for integrated water-N dynamics.

129 2. Materials and Methods

130 2.1. Description of site, water application, and measurements

131 A replicated field experiment was conducted at the Kearney Agricultural Research and
132 Extension Center (<http://kare.ucanr.edu/>) located 20 km southeast of the city of Fresno in the
133 Central Valley of California, USA. Two mature (>40-year-old) own-rooted ‘Thompson Seedless’
134 grapevine (*Vitis vinifera* L.) vineyards were flooded; the first vineyard for four weeks (V1) and
135 the second vineyard for two weeks (V2) (February–March 2020). V1 is on a very deep fine
136 sandy loam (*Hesperia*, coarse-loamy, mixed, nonacid, thermic Xeric Torriorthents) with a
137 saturated hydraulic conductivity (K_{sat}) of 0.02 m hr⁻¹ for the first 1 m depth (USDA-NCSS soil
138 survey data), and V2 is on a fine sandy loam (*Hanford*) with a K_{sat} of 0.1 m hr⁻¹ for the first 1 m

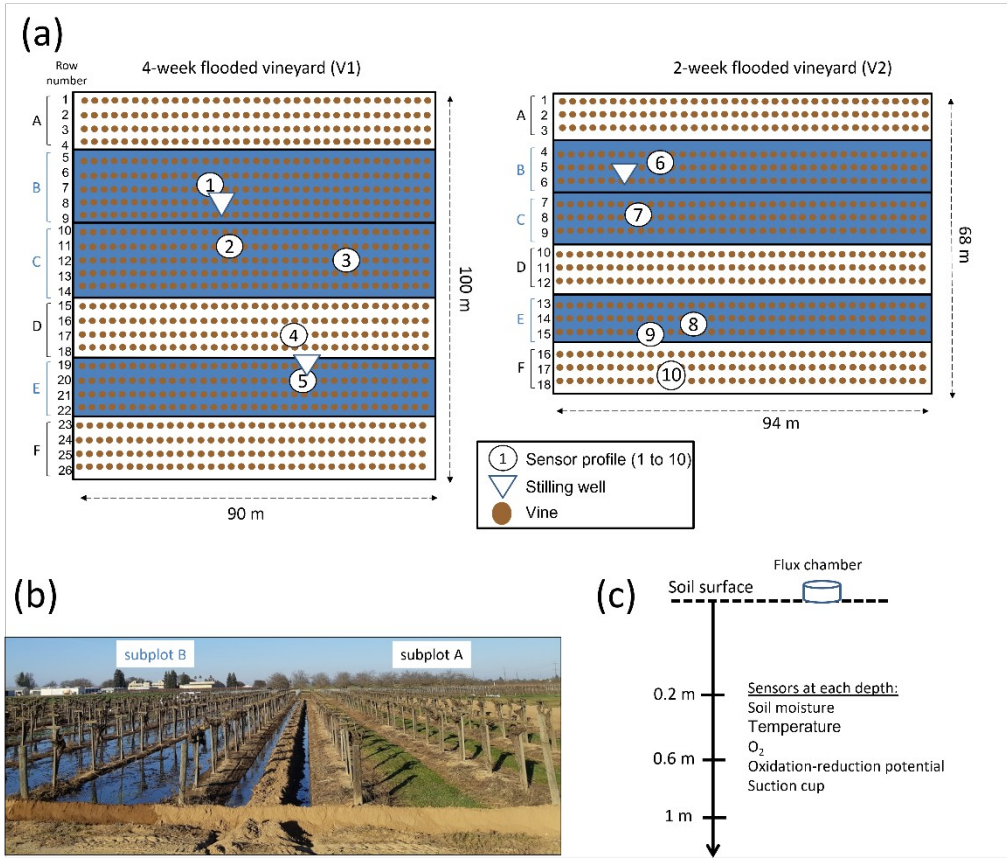
139 depth. Using a randomized complete block design, each vineyard was divided into six individual
140 subplots, of which three were artificially flooded and three were kept under the natural
141 precipitation regime (i.e., control) (**Fig. 1a**). Individual subplots were separated from one another
142 using berms approximately 0.5 m high and 0.8 m wide (see **Fig. 1b**). The groundwater table was
143 ~23 m below the surface measured in October 2019. The site has a semiarid, Mediterranean
144 climate (Onsoy et al., 2005). Mean air temperature and total precipitation during the flooding
145 period was 21.9 °C and 0.03 m, respectively.

146 Flooding started automatically at 06:00, 14:00, and 22:00 for 2-3 hrs at both vineyards each day,
147 except for the first four days during which manual operation was used to adjust flow rates to
148 prevent overflow to adjacent fields. Flooding of the vineyards started on 02/25/2020 and ended
149 on 03/10/2020 (V2) and 03/24/2020 (V1). Two flow meters were used at the water inlet point of
150 each vineyard to measure total applied water. Flooding was done using a single water outlet
151 located at the west side of each row within the flooded subplots. A total of 7298 and 4659 m³
152 were discharged at V1 and V2, respectively. Commonly Ag-MAR would be conducted using
153 surface water. However, due to drought during the winter of 2019-2020 no surface water was
154 available at our experimental site, therefore, pumped groundwater was used as an alternative
155 water source for Ag-MAR. Groundwater was determined as a suitable alternative to surface
156 water because N species concentrations in groundwater were found to be comparable to those of
157 surface water in this area, as previously reported by Bachand et al. (2014).

158 Ten monitoring profiles were instrumented for continuous measurements, five within each
159 vineyard (**Fig. 1a**). Out of the five profiles in each vineyard, only one profile was installed in the
160 control (i.e., the non-flooded subplots) to allow higher spatial measurement resolution in the Ag-

161 MAR treatment subplots, which are the focus of this study. Each profile was installed 0.7 m from
162 the vine row toward the furrow and included three sensor clusters at 0.2, 0.6 and 1 m depth. Each
163 cluster included measurements of soil moisture and temperature (TEROS12, Meter, WA, USA),
164 O₂ in gaseous phase (KE-25, Figaro, Japan), and ORP using constructed platinum electrodes
165 (following Owens et al., 2005) and commercial Ag/AgCl reference electrodes (Accumet, Thermo
166 Fisher Scientific, MA, USA). Additional measurements included three pressure transducers
167 within stilling wells to record ponding levels (CS451, Campbell Scientific, UT, USA). Sensors
168 were placed in hand augered holes that were backfilled with soil while compacting it at
169 approximately 0.2 m intervals. Data were logged (CR1000 and CR800, Campbell Scientific, UT,
170 USA) at a 10-min interval. The complete site and sensor descriptions are detailed in **Figs. 1a** and
171 **1c**, respectively, and a photo taken during the flooding is presented in **Fig. 1b**. Atmospheric
172 measurements (temperature, precipitation) at 60-min intervals were taken from the California
173 Irrigation Management Information System station (CIMIS; station 39 Parlier, CA) situated 400
174 m from the experimental site.

175 Crop management practices followed standard recommendations with grape harvest in
176 September and cane pruning on 20 January 2020. Both vineyards were fertilized once per year,
177 two months after the flooding (end of April 2020), when 336 kg ha⁻¹ of N fertilizer (ammonium
178 sulfate 21-0-0 with 24% sulfur) were applied. Vines were manually harvested on 23 September
179 2020. The entire amount of fruit harvested from each plot was recorded and used for statistical
180 analyses.



181

182 **Fig. 1.** Experimental setup. (a) Sensor locations in each vineyard. Flooded subplots are marked
 183 in blue. (b) A photo of the 4-week flooded vineyard (V1) showing subplots A and B during the
 184 flooding. (c) Sensor locations within a single profile.

185 *2.2. Pore water, gas emission, and soil sampling*

186 Pore water, greenhouse gas (GHG) emission, and soil sampling campaigns were carried out to
 187 better understand and quantify the N biogeochemical processes within the soil. Pore water
 188 samples were taken using suction cups (LT-DBL, Irrometer, CA, USA), installed at each of the
 189 ten profiles and at the same depths as the sensor clusters (0.2, 0.6, and 1 m). After installation,
 190 suction cup boreholes were backfilled with sieved soil slurry, followed by 0.15 m of soil, and
 191 sealed with 0.05 m bentonite; shallow 0.2 m suction cups were installed without bentonite. Pore

192 water was sampled on average twice a week during the flooding period and once a week at other
193 times, except for one continuous 5-day measurement campaign after fertilization to capture the
194 fertilization effect on N processes. Samples were stored on ice until storage in a 4 °C cold room.
195 Pore water was analyzed for NO_3^- , NH_4^+ , and DOC. Analytical protocols are detailed in the
196 supporting information.

197 *In-situ* GHG emissions were measured using the closed-flux vented chamber method (e.g.,
198 Garland et al., 2014). Twenty individual chambers were installed, each made from 0.26 m
199 diameter polyvinyl chloride (PVC) pipe, consisting of a bottom ring and a cap. Two chambers
200 were installed near each of the ten sensor profiles shown in **Fig. 1a** – one on the mount between
201 the vines and one in the furrow. Collars were inserted into the soil to a depth of 0.07 m and left
202 in the same location for the entire duration of the experiment. Chamber headspace was measured
203 after each sampling to account for any reduction in volume due to water within the chamber. At
204 sampling time, the chambers were sealed onto the collars with a rubber sleeve made from a tire
205 innertube. Gas samples were taken through a rubber septum at four times (0, 30, 60, and 80 min)
206 using a 20 mL air-tight syringe and injected into pre-evacuated 12 mL vials. Before each
207 sampling effort, a 12V fan (installed within each chamber) was activated for 20-30 seconds to
208 ensure a mixed representative headspace sample. Temperature was measured during each time
209 step in 10 out of the 20 chambers. Samples were analyzed using a gas chromatograph (Shimadzu
210 Trace Gas GC, Shimadzu Corp., Kyoto, Japan) for carbon dioxide (CO_2), methane (CH_4), and
211 nitrous oxide (N_2O). Fluxes were calculated according to Garland et al. (2011). Gas samples
212 were taken on similar days as the water samples and mostly around noon.

213 Soil samples were taken from four depths: 0-0.1, 0.1-0.2, 0.5-0.6, 0.8-1 m using a hand auger.
 214 All samples were taken from the same rows in which the sensor profiles were installed and at
 215 similar distances from the vines toward the furrow (~ 0.7 m). Samples were stored identical to
 216 pore water samples and taken during four campaigns according to the different experimental
 217 stages: 02/06/2020 (baseline data), 03/10/2020 (end of flooding at V2), 03/25/2020 (end of
 218 flooding at V1, and two weeks post-flooding at V2), and 04/15/2020 (three weeks post-flooding
 219 at V1). Soil samples were analyzed for NO_3^- , NH_4^+ , DOC, and texture. In addition, soil
 220 subsamples were stored at -80 °C and later used for incubation experiments (see details in
 221 section 2.4).

222 2.3. Nitrogen leaching estimates

223 The amount of NO_3^- leaching below 1 m (the root zone in the case of flooded-irrigated
 224 vineyards) during flooding was quantified using a vadose-zone-based water and N mass balance
 225 model (Baram et al., 2016b), in which M_A is the cumulative mass of NO_3^- per flooded area (as
 226 NO_3^- -N [g m^{-2}]) lost through leaching (**Eq. 1**) and M is the total mass of NO_3^- (as NO_3^- -N [g]) lost
 227 during the flooding (**Eq. 2**):

$$228 \quad M_A = \sum_{i=1}^n L_i C_i \Delta t_i \quad (1)$$

$$229 \quad M = \sum_{i=1}^n M_{A_i} A \quad (2)$$

230 where L is the amount of water leaching below 1 m estimated using the infiltration rate (section
 231 2.5) [m d^{-1}], C is the average NO_3^- -N concentration in the leaching water at 1 m depth [g m^{-3}], Δt
 232 is a given time period [d] between each i^{th} measurement, and A is the flooded surface area [m^2].

233 Because all pore water NH_4^+ samples measurements at both vineyards were $< 1 \text{ g m}^{-3}$, these
234 values were considered negligible in the quantification of total inorganic N leached for modeling
235 purposes.

236 2.4. *Potential denitrification*

237 Incubation experiments were conducted in the lab to assess denitrification rates during the
238 flooding period. Net denitrification rates were estimated using a modified method of Petersen et
239 al. (2012). Briefly, field-moist soil (15 g) collected one day before flooding and one day after
240 flooding was added to 100 ml serum bottle and sealed with rubber stopper and metal caps, and
241 then flushed with N_2 gas for 10 min to create anaerobic conditions and afterwards equilibrated
242 with atmospheric pressure using a syringe. Bottles were placed in an incubator for 7 days at
243 25°C . Incubated samples were analyzed for changes in NO_3^- concentrations on the initial and 7th
244 day to determine denitrification rates.

245 2.5. *Hydrological analysis*

246 Daily infiltration rates were calculated using two independent methods. In the first method, the
247 total applied water was divided by the flooded surface area and total number of flooding days for
248 each vineyard. The second method, also referred to as the falling head method, estimates the
249 infiltration rate from the slope of the decreasing ponding level in each flood plot after water shut-
250 off, which is then normalized for daily infiltration rates. Both methods provide bulk infiltration
251 rates.

252 Groundwater recharge was calculated using a one-dimensional vertical water balance model
253 (Eq. 1) (Dahlke et al., 2018). A single solution was solved for each of the eight flooded

254 monitoring profiles using input data averaged over an hourly interval. The fraction of applied
255 water going to deep percolation towards the groundwater table was calculated by accounting for
256 evapotranspiration and storage in pore space.

$$257 \quad R_t = I_t + P_t - ET_t - \Delta S_t - Q_t \quad (3)$$

258 where R_t [m] is recharge, I_t [m] is the amount of applied surface water, P_t [m] is precipitation,
259 ET_t [m] is evapotranspiration, ΔS_t [m] is the change in soil storage, and Q_t [m] is surface runoff at
260 time step t , which was assume to be negligible due to the use of berms. **Eq. 3** was solved using
261 the Thornthwaite-Mather procedure (Steenhuis and Molen, 1986) performed via Excel solver
262 (Office 365 ProPlus, 2020). This procedure was used in previous Ag-MAR studies (Dahlke et al.
263 2018). The procedure was solved for the first meter assuming this is also the maximum depth of
264 the major root zone (Araujo et al., 1995) in which evapotranspiration demand takes place.

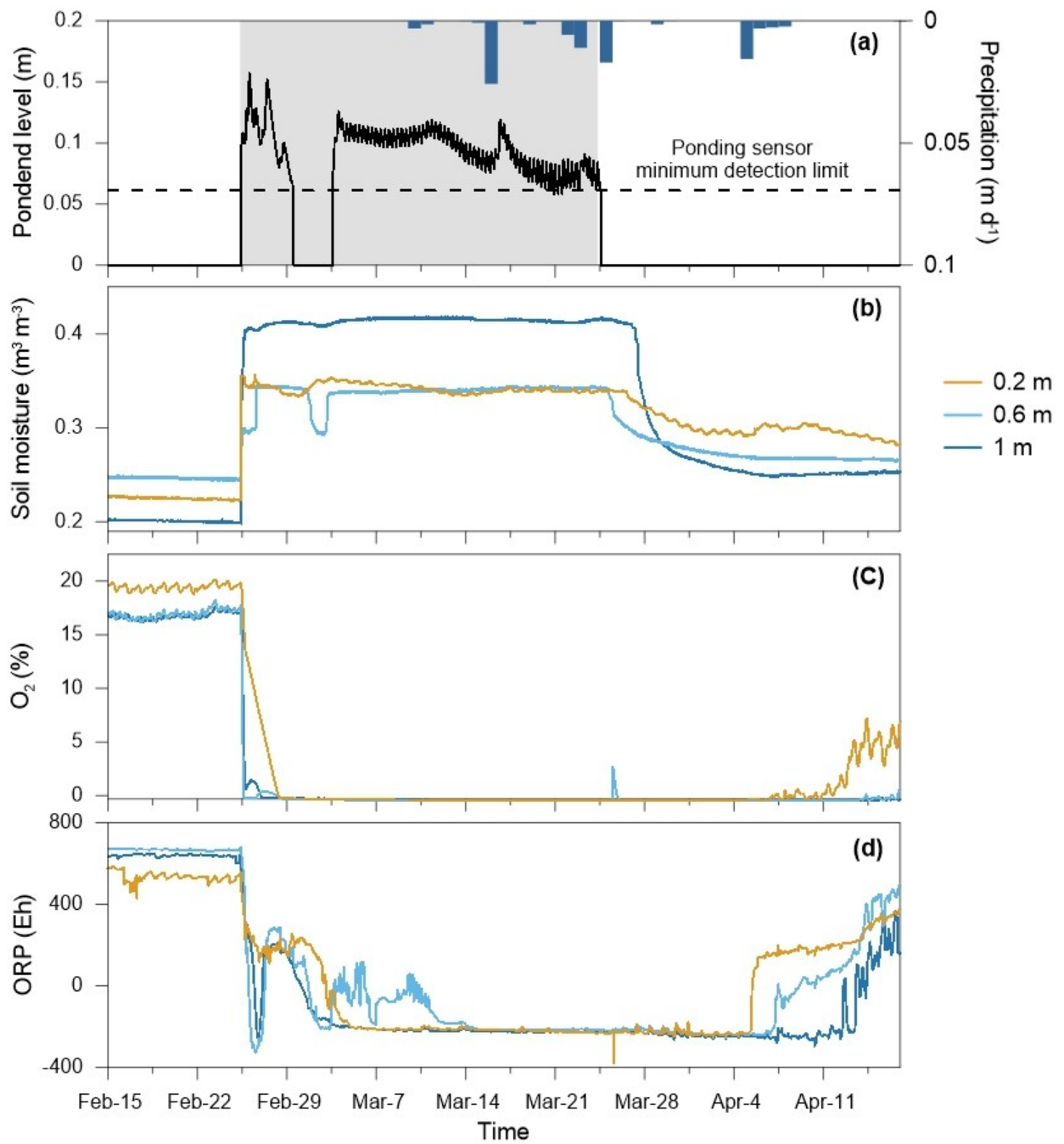
265 **3. Results**

266 *3.1. Recharge and soil physical characteristics*

267 The flood treatments in the two vineyards showed different hydrologic and biogeochemical
268 responses during and after the flooding for recharge (V1 - **Fig. 2** and V2 – **Fig. 3**). As a result of
269 field topography, flooding was not uniform in the flooded subplots. Dry areas occurred in the
270 eastern part of V1 (**Fig. 1a**; subplot C) and V2 (**Fig. 1a**; subplots C and E), effectively reducing
271 the flooded area to an average of 4613 m² (90%) and 3035 m² (75%) in V1 and V2, respectively.
272 Soil ORP and O₂ concentration decreased rapidly in V1, reaching anoxic conditions (ORP < 100
273 Eh) at 0.2, 0.6 and 1 m depth 1-3 days after flooding started (**Figs. 2c** and **2d**). Reducing, anoxic
274 conditions were sustained for 12-18 days after flooding ended, leading to ~40 days of continuous

275 reducing conditions within the soil profile in V1. In contrast to V1, V2 maintained good aeration
276 during the flooding and only experienced anoxic conditions at the shallowest depth of 0.2 m and
277 for a short duration (2-5 days) (**Figs. 3c** and **3d**). O₂ and ORP averaged ~20% O₂ and ~400 Eh
278 ORP at all other depths within the flooded soil profiles in V2. Although flooded plots at both
279 vineyards reached near or fully saturated conditions as indicated by the soil moisture (**Figs. 2b**
280 and **3b**), the observed differences in soil O₂ and ORP clearly indicate different environmental
281 conditions that impacted biogeochemical processes in each vineyard.

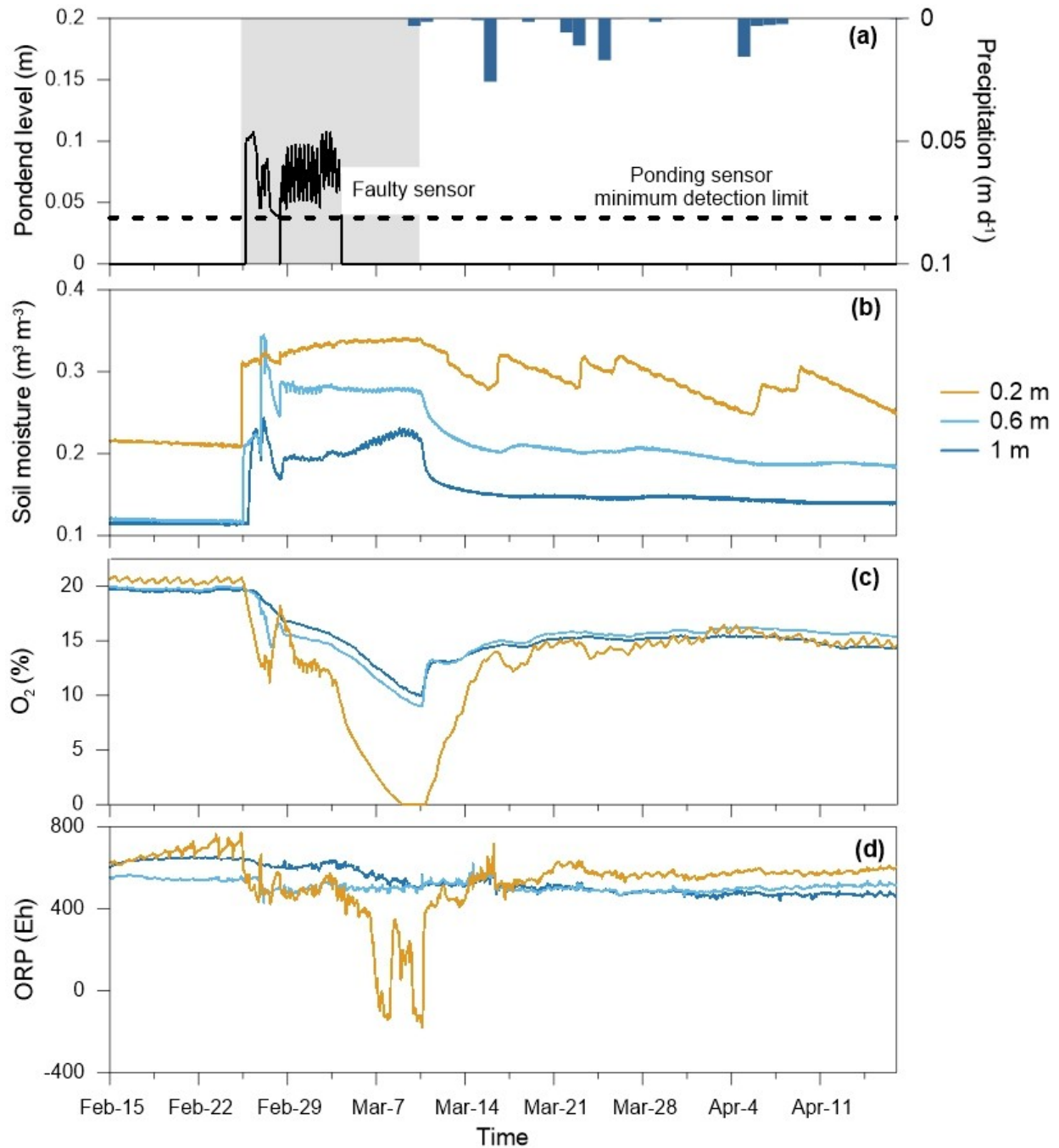
282 As expected, control plots of both vineyards showed predominantly oxic ORP conditions (i.e.,
283 O₂ ~20% and ORP > 500 Eh) for the duration of the experiment and only abrupt soil moisture
284 increases after rain events. The only exception was profile 4 in V1 (see **Fig. 1a** for location),
285 which showed a soil moisture increase at 1 m depth followed by a decrease in O₂. Since no soil
286 moisture increase was observed at 0.2 and 0.6 m, we attributed this change to the possibility of
287 lateral flow from the flooded subplots, impacting the oxygen status in the deeper soil profile.



288

289 **Fig. 2.** Time series data for the 4-week flooded vineyard (V1) for selected hydrologic and
 290 physico-chemical parameters from profile 1 (see **Fig. 1a** for profiles location). ORP – oxidation-
 291 reduction potential. The gray shaded area in (a) shows the time of water application. Blue

292 precipitation bars show the daily totals taken from a meteorological station located 400 m from
293 the experimental site.



294

295 **Fig. 3.** Time series data for the 2-week flooded vineyard (V2) for selected hydrologic and
296 physico-chemical parameters from profile 7 (see **Fig. 1a** for profiles location). ORP – oxidation-

297 reduction potential. Gray shaded area represents the flooding duration. Blue bars in plot a show
298 daily precipitation totals.

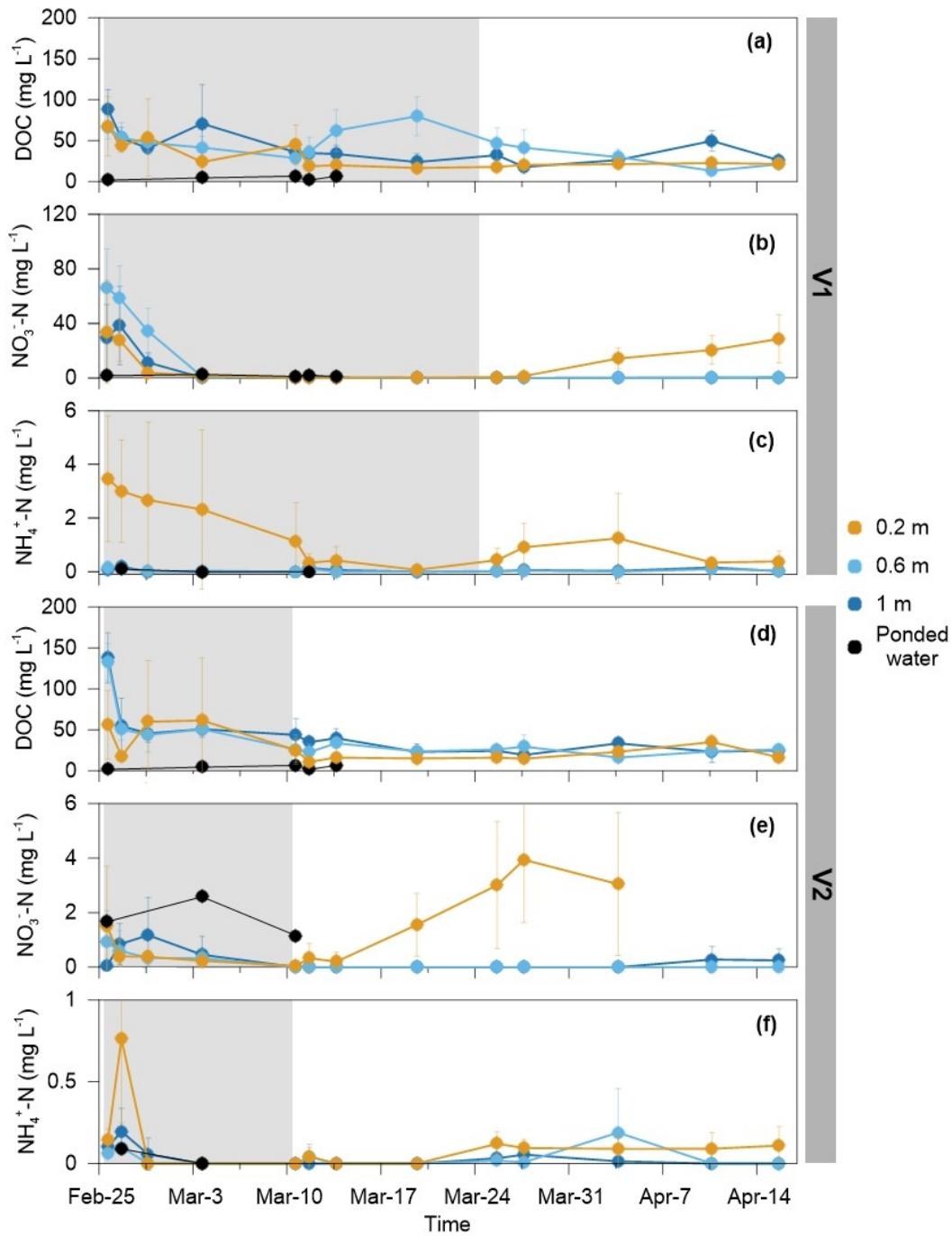
299 3.2. *Infiltration rates and recharge*

300 The experimental site received little precipitation prior and during the experiment (**Fig. 2a**), thus
301 providing an opportunity to study infiltration processes and water balance changes associated
302 with Ag-MAR more closely. The infiltration rate in V1 was 0.088 ± 0.031 m d⁻¹. Estimated
303 groundwater recharge was $83\% \pm 1.2\%$ of the applied water using **Eq. 3**. The V2 site had a
304 higher infiltration rate of 0.171 ± 0.025 m d⁻¹ with $86\% \pm 0.7\%$ of the applied water percolating
305 below 1 m. The higher V2 recharge rate is also reflected by the soil suitability ranking developed
306 by O'Geen et al. (2015) for California soils, which rates V2 as "excellent" for Ag-MAR whereas
307 the ranking of V1 is "moderately good" as supported by soil textural analysis, which showed a
308 higher sand fraction in V2 compared to V1 (**Fig. S1**). Upscaling the above values to each
309 vineyard indicates that 1.31 and 1.32 m³ m⁻² of water was recharged in the flooded areas in V1
310 and V2, respectively. Although V2 was only flooded for two weeks, the higher infiltration rate
311 (0.171 m d⁻¹ in V2 compared to 0.088 m d⁻¹ in V1) resulted in higher total recharge amounts.

312 3.3. *Pore water and residual soil chemistry*

313 Pore water NO₃⁻ and NH₄⁺ data from both vineyards are presented in **Fig. 4**. During the first
314 week of flooding, NO₃⁻ concentrations in both vineyards decreased to zero at all three depths
315 (**Figs. 4b** and **4e**). Pore water NH₄⁺ showed near-zero concentrations in the deeper soil profile
316 (0.6 and 1 m) before flooding and a clear reduction at 0.2 m depth (**Figs. 4c** and **4f**). DOC
317 concentrations in pore water remained approximately 20-60 mg DOC L⁻¹ in both vineyards and

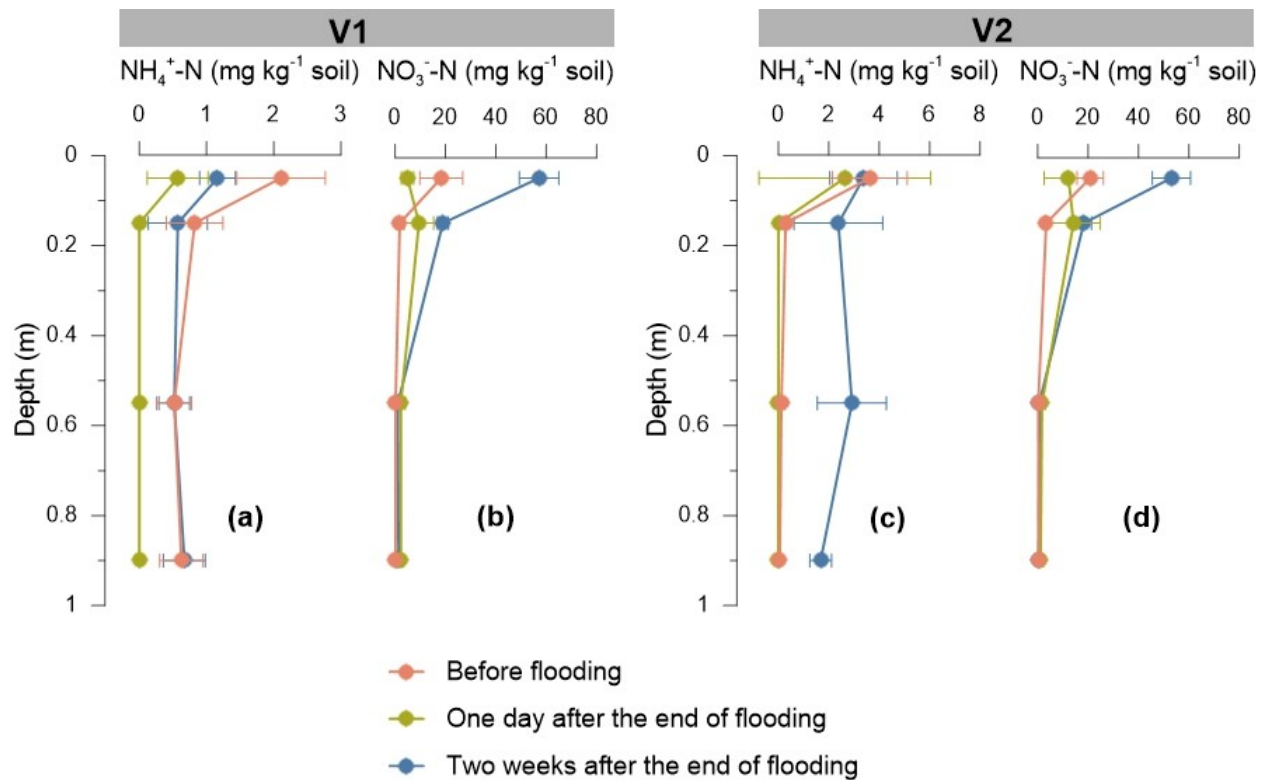
318 at all depths during flooding (Figs. 4a and 4d). Nitrite (NO_2^-) was not measured directly and
 319 assumed to be negligible following Bachand et al. (2014), which reported nitrite levels lower
 320 than $0.1 \text{ mg NO}_2^- \text{-N L}^{-1}$ during their Ag-MAR study.



321

322 **Fig. 4.** Pore water results for the 4-week flooded vineyard (V1) (a-c) and the 2-week flooded
 323 vineyard (V2) (d-f). Values are averages of the four flooded profiles in each vineyard with error
 324 bars representing the standard deviation. Gray shaded areas represent the flooding duration.

325 Residual soil NH_4^+ in V1 depleted from 1-2 to near 0 $\text{mg NH}_4^+\text{-N kg}^{-1}$ soil at all depths during
 326 the flooding (**Fig. 5a**). In V2, pre-flooding NH_4^+ concentrations were already low (e.g., ~ 0 mg
 327 $\text{NH}_4^+\text{-N kg soil}^{-1}$ below 0.1 m), with no significant change during flooding (**Figs. 5c and 5d**).
 328 Two weeks after the flooding ended, NH_4^+ levels in both vineyards increased uniformly at all
 329 measured depths to ~ 1 and ~ 3 $\text{mg NH}_4^+\text{-N kg soil}^{-1}$ for V1 and V2, respectively (**Figs. 5a and 5c**).
 330 Similar post-flooding trends of increasing concentrations were also observed for NO_3^- in both
 331 vineyards, however, only within the upper soil profile (0-0.2 m) where NO_3^- increased to 20-60
 332 $\text{mg NO}_3^-\text{-N kg}^{-1}$ soil (**Figs. 5b and 5d**).



333

334 **Fig. 5.** Residual NO_3^- and NH_4^+ concentration in soil prior and after flooding for the 4-week
335 flooded vineyard (V1) (a-b) and the 2-week flooded vineyard (V2) (c-d). Values are averages of
336 the four flooded profiles in each vineyard with error bars representing the standard deviation.

337 3.4. *Greenhouse gas emissions*

338 During flooding, no changes in N_2O and CO_2 emissions were observed in the flood or control
339 plots of V1 and V2 (**Fig. S2**). Post-flooding, N_2O emissions increased steadily for two weeks in
340 the flooded plots in V1 before they declined again (**Fig. S2b**, blue line). However, peaks in N_2O
341 and CO_2 emissions following the flooding were not as high as the spike observed two months
342 after flooding, shortly after both vineyards received 336 kg ha^{-1} of N fertilizer (ammonium
343 sulfate 21-0-0 with 24% sulfur; end of Apr-2020), which was the only fertilizer application
344 during that year. We note that CH_4 emissions were not detected during the experiments.

345 3.5. *Yield data*

346 Both vineyards were harvested in early Sep 2020. In V1, a significant 29% decrease in yield
347 was observed compared to the control (t-test, $p < 0.05$), whereas in V2, there was no evidence of
348 a significant difference in yield although yield was 14% lower in the flooded plots ($p = 0.24$)
349 (**Fig. S3**).

350 4. **Discussion**

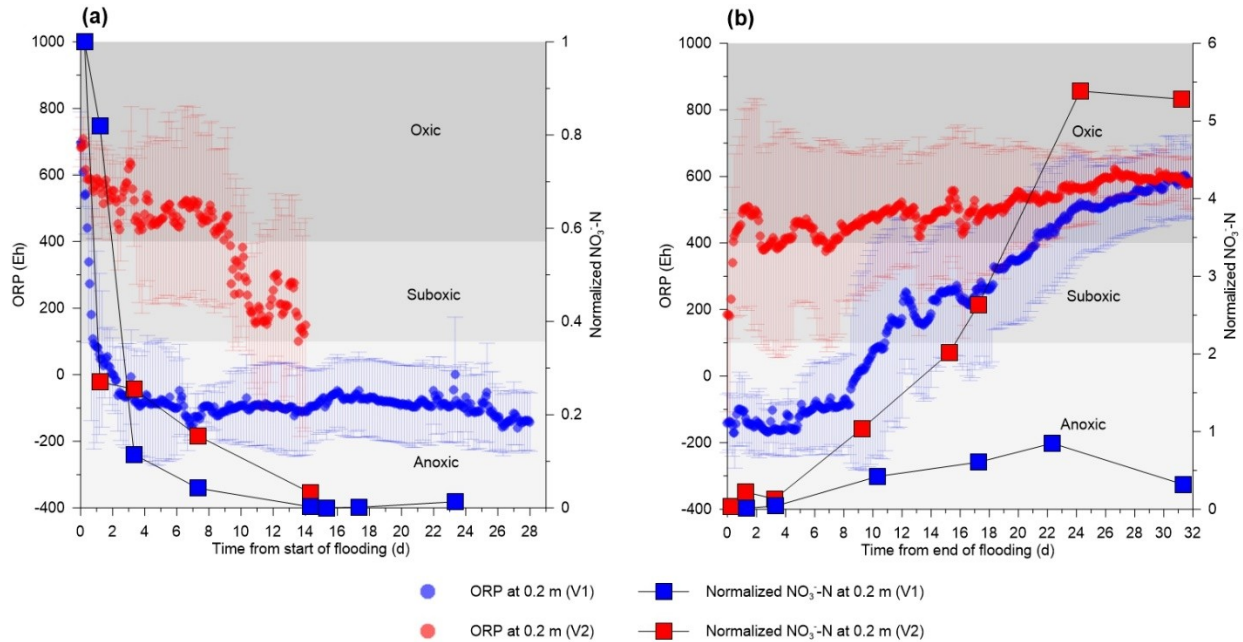
351 4.1. *Nitrate leaching vs. denitrification during flooding*

352 We observed a rapid decline in pore water NO_3^- concentrations within the first days of flooding
353 for Ag-MAR. Two possible processes can be attributed to the observed NO_3^- decrease: NO_3^-

354 removal due to denitrification, a microbial-controlled process, and/or leaching of NO_3^- , a solute
355 transport process. The former would be preferable during Ag-MAR (or any other MAR type)
356 since it can transform NO_3^- into the inert gas N_2 , thereby preventing NO_3^- from reaching the
357 groundwater (Levintal et al., 2022). Denitrification rates measured through incubation
358 experiments (Petersen et al., 2012; Verchot et al., 2001), showed the highest denitrification
359 activities in the top 0.1 m of the soil profile, decreasing to approximately zero below 0.6 m (**Fig.**
360 **S4**). Denitrification rates in the top 0.1 m were 1.4 and 1.35 $\text{mg NO}_3^- \text{-N kg soil}^{-1} \text{ d}^{-1}$ for V1 and
361 V2, respectively. These rates are considered maximum rates, since they were obtained under
362 optimal conditions (25 °C, $\text{O}_2 \sim 0\%$, C substrate addition), favoring denitrification over
363 nitrification. In both vineyards, oxic conditions dominated the root zone before flooding started
364 and during imbibition. Thus, it is expected that denitrification rates were lower than the rates
365 derived from the incubation experiments, at least until the root zone became fully
366 suboxic/anoxic.

367 Infiltration rates not only determined the transition from oxic to suboxic conditions in each
368 vineyard, but also the rate at which NO_3^- was transported below the active denitrification zone in
369 the topsoil (i.e., 0-0.2 m). **Fig. 6** describes the tradeoff between the development of
370 suboxic/anoxic conditions and the depletion of NO_3^- at 0.2 m. To allow comparison between the
371 vineyards, NO_3^- concentrations in each vineyard were normalized to the initial concentrations
372 observed in each vineyard prior to flooding. V1 reached suboxic conditions at 0.2 m (ORP < 400
373 Eh) after one day of flooding, compared to V2 which developed suboxic conditions only after 10
374 days. When the transition from oxic to suboxic conditions occurred in V1, more than 80% of the
375 initial NO_3^- was still available at 0.2 m depth (**Fig. 6a**, blue squares). In contrast, by the time V2

376 reached suboxic conditions only ~10% of the initial NO_3^- was available for denitrification (**Fig.**
 377 **6a**, red squares), indicating that 90% was leached.



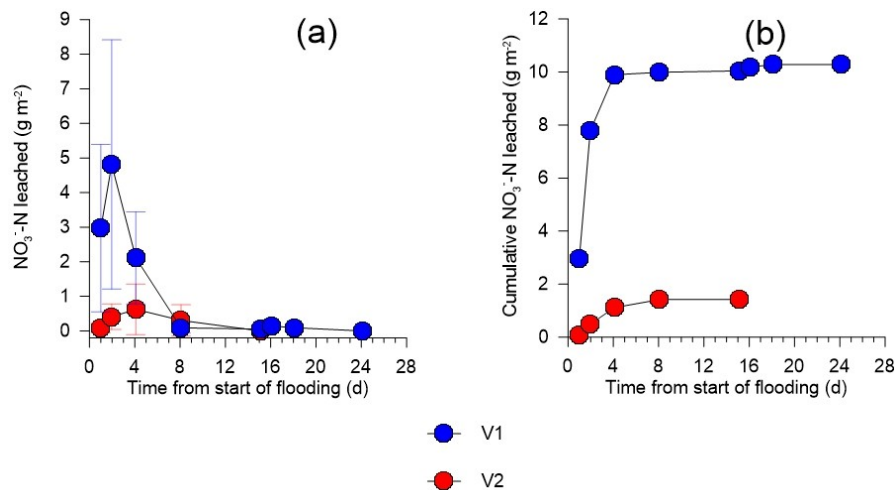
378

379 **Fig. 6.** Oxidation-reduction potential (ORP) and normalized pore water NO_3^- concentrations at
 380 0.2 m depth during the wetting (a) and drying phase (b) of Ag-MAR. Values are averages of the
 381 flooded profiles in each vineyard with error bars representing the standard deviation. NO_3^- was
 382 normalized to the initial pore water NO_3^- concentrations in each vineyard; values above 1
 383 indicate that NO_3^- exceeded the initial pre-flooding concentrations (i.e., V2 in subplot ‘b’).

384 The NO_3^- which was not consumed during the denitrification process in the topsoil was leached
 385 below the root zone and eventually transported towards the groundwater as denitrification is
 386 assumed to be negligible in the deep vadose zone (up to 3-5% per year) (Baram et al., 2016a). In
 387 both vineyards, the majority of NO_3^- was leached below 1 m during the first week of flooding,
 388 with a leaching peak of NO_3^- occurring during day 2 and 4 in V1 and V2, respectively (**Fig. 7a**).

389 The cumulative amount of NO_3^- leached (using **Eqs. 1 and 2**) was 10.3 and 1.6 g NO_3^- -N m^{-2} for

390 V1 and V2, respectively (**Fig. 7b**), which equates to a NO_3^- loading of 47.4 and 4.3 kg NO_3^- -N
 391 for V1 (flooded area of 4,613 m²) and V2 (flooded area of 3,035 m²) below 1 m, respectively.
 392 Averaging the cumulative NO_3^- leached per total applied water results in 8.6 and 1 g NO_3^- -N m⁻²
 393 for every 1 cm³ cm⁻² of water applied for V1 and V2, respectively (i.e., an average leaching
 394 concentration of 8.6 and 1 mg L⁻¹ NO_3^- -N for V1 and V2). These values are similar in magnitude
 395 to NO_3^- leaching amounts estimated in a previous Ag-MAR field experiment (Bachand et al.,
 396 2014) and in an Ag-MAR column experiment (Murphy et al., 2021). Yet, it is also possible to
 397 have different leaching amounts between sites due to site-specific factors influencing nitrate
 398 leaching (e.g., soil texture and soil pre-flooding NO_3^- concentration).



399

400 **Fig. 7.** Observed and cumulative amounts of NO_3^- leached below 1 m for (a) the 4-week flooded
 401 vineyard (V1) and (b) the 2-week flooded vineyard (V2).

402 The amount of NO_3^- leached differed by one-order of magnitude between the two vineyards.
 403 These differences can be attributed to the initial pore water and soil NO_3^- concentrations prior to
 404 flooding. In V1, the pre-flooding pore water NO_3^- concentrations ranged between 20-80 mg NO_3^- -

405 N L^{-1} (**Fig. 4b**), whereas in V2, concentrations were below $2 \text{ mg NO}_3^- \text{-N L}^{-1}$ (**Fig. 4e**). Therefore,
406 estimating legacy NO_3^- levels prior to Ag-MAR is essential for assessing leaching risks. It is
407 emphasized that the low initial NO_3^- concentrations in V2's topsoil does not necessarily indicate
408 a lower risk of groundwater contamination. Both vineyards were fertilized and irrigated
409 following the same protocols in the years prior to the Ag-MAR experiment, and therefore, it is
410 likely that elevated NO_3^- levels were also present in V2. Yet, the surplus NO_3^- was pushed deeper
411 into the soil profile with each flood irrigation and precipitation event due to the higher infiltration
412 rate of V2 compared to V1.

413 Once pre-flooding pore water NO_3^- depleted, concentrations in both vineyards and at all three
414 measured depths (0.2, 0.6, and 1 m) stayed zero for the remainder of the flooding even though
415 NO_3^- was added to the system with the floodwater ($2\text{-}3 \text{ mg NO}_3^- \text{-N L}^{-1}$; **Figs. 4b** and **e**, black
416 circles). The source water NO_3^- was depleted by denitrification in the first 0.1 m of the soil
417 profile. This conclusion is supported by the denitrification rates estimated through lab incubation
418 experiments. For example, the denitrification rate estimated for the top 0.1 m soil profile in V1
419 was $1.4 \text{ mg NO}_3^- \text{-N kg soil}^{-1} \text{ d}^{-1}$ (**Fig. S4**), which translates to a potential *in-situ* soil
420 denitrification of $10 \text{ mg NO}_3^- \text{-N}$ per day during flooding. This rate is ~5-fold higher than the
421 floodwater NO_3^- concentration, thus can explain the NO_3^- depletion during infiltration. At this
422 stage, water percolating below 1 m contributed to the dilution of the already leached residual soil
423 NO_3^- . This process is also referred as the dilution effect (Bachand et al., 2014). While Ag-MAR
424 with low-contaminant water (e.g., rainfall-runoff or snowmelt) can cause not only NO_3^- leaching
425 from the root zone, but also dilution of NO_3^- in pore water, its final impact on groundwater
426 quality is a long-term process, which depends on several factors including the legacy NO_3^- load

427 in the vadose zone, the NO_3^- concentration of groundwater, geogenic sources, and groundwater
428 flow velocity (Dahan et al., 2014; Levintal et al. 2022). Therefore, quantifying the dilution effect
429 from a single year is not feasible; further discussion is given below in section 4.3.

430 4.2. *Post-flooding nitrogen dynamics*

431 Owing to the legacy of N in agricultural plant systems, we investigated the post-flooding N
432 cycling processes during the growing season. In V1, soil NH_4^+ levels recovered to pre-flooding
433 levels in the first two weeks after the flooding ended (**Fig. 5a**). An increase was also observed for
434 NO_3^- , however, only in the topsoil (**Fig. 5b**, 0-0.2 m). Similar post-flooding $\text{NH}_4^+/\text{NO}_3^-$
435 concentrations increases were observed in V2 (**Figs. 5c** and **5d**), while NH_4^+ increased at all
436 depths, NO_3^- increased only in the topsoil, indicating that mineralization occurred throughout the
437 1 m profile, whereas nitrification was mainly limited to the topsoil. NH_4^+ concentrations were an
438 order-of magnitude lower than NO_3^- for both pore water and soil. The accumulation of NO_3^- in
439 the top of the soil profile (0-0.2 m) rather than at deeper layers is consistent with previous studies
440 (e.g., Dahan et al., 2014) who concluded that this accumulation pattern is preferable for both
441 crop uptake and reducing leaching risks.

442 NO_3^- in pore water is considered to be more mobile in the vadose zone compared to the
443 immobile NH_4^+ cations attached to soil particles and organic matter (Subbarao and Searchinger,
444 2021). In terms of NO_3^- leaching risk, the chemical composition of the pore water has a more
445 immediate and significant effect on groundwater, rather than contaminants adsorbed onto
446 sediment in the immobile phase. Thus, prioritization of pore water data over soil data is
447 potentially recommended when assessing NO_3^- leaching risk to groundwater. We emphasize that
448 additional research is needed to validate this statement as previous studies also suggested that

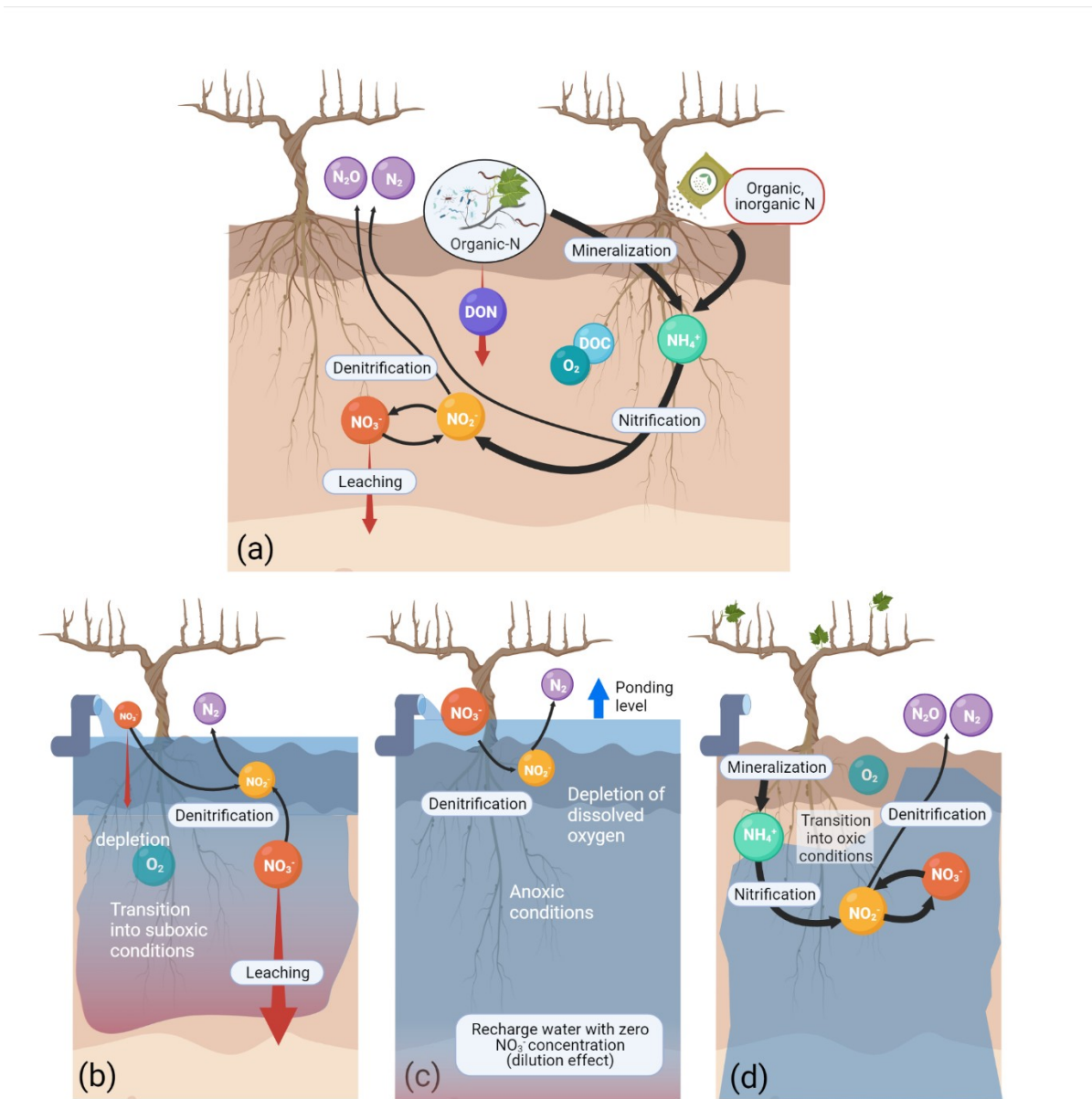
449 nitrogen can be released from the soil to the pore water at later stages of the flooding and in the
450 deep vadose zone (Mienis and Arye, 2018; Xin et al., 2019). In general, the relationship between
451 pore water and soil samples is complex (Darrouzet-Nardi and Weintraub, 2014; Rimon et al.,
452 2011). Studies suggested equations to convert between the two (Zhu et al., 2021). Yet, to date, no
453 empirical relationship has been validated for different soils or soil moisture variations as
454 expected during flooding.

455 Anoxic conditions continued in V1 for about 3 weeks after the flooding ended, while V2 stayed
456 mostly in oxic conditions throughout and after the flooding (**Fig. 7b**). The topsoil (0.2 m) in V1's
457 flooded area stayed in anoxic/suboxic conditions for 20 consecutive days once flooding ended,
458 promoting denitrification. We observed elevated N₂O emissions, which can be a byproduct of
459 denitrification, in the 20 days of anoxic/suboxic conditions following the flooding of V1 (**Fig.**
460 **S2b**, blue circles during the start of April). We note that N₂O emissions can also be attributed to
461 nitrification, however, under near-saturated conditions denitrification dominates N₂O production
462 (Bateman and Baggs, 2005; Zhu et al., 2013). N₂O emissions started to decrease in V1 after the
463 20 days due to developing oxic conditions following flooding. N₂O emissions started to increase
464 again after the vineyard's fertilization and first irrigation of the growing season on 4/29/2021.
465 The measured N₂O emissions were similar in magnitude to reported fertilization-related
466 emissions from vineyards under a similar Mediterranean climate (Garland et al., 2014, 2011).

467 4.3. *Conceptual model – coupling nitrogen and water dynamics under Ag-MAR*

468 We developed a conceptual model for N fate and transport in agricultural soils flooded for
469 extended periods of time with low-contaminant water for Ag-MAR (**Fig. 8**). The model was
470 developed based on data from the two vineyards investigated in this study and from recent Ag-

471 MAR studies conducted on alfalfa (Dahlke et al., 2018), almonds (Ganot and Dahlke, 2021b,
 472 2021a), and in controlled soil column experiments (Murphy et al., 2021). We identify four stages
 473 that determine N fate during Ag-MAR: pre-flooding, start of flooding, quasi-steady flooding, and
 474 post-flooding. In each stage, the N fate is determined by the soil physico-chemical characteristics
 475 and the initial N concentration.



476

477 **Fig. 8.** Conceptual model for N fate under long-term flooding conditions at the topsoil. (a) pre-
478 flooding, (b) start of flooding, (c) quasi-steady flooding, and (d) post-flooding. Detailed
479 description for each stage is given in section 4.3. Figure created with Biorender.

480 *Pre-flooding stage:* In the pre-flooding stage (**Fig. 8a**), the root zone is unsaturated and
481 predominantly oxic. The soil matrix and pore water contain variable amounts of legacy inorganic
482 N species (i.e., NH_4^+ and NO_3^-) from previous fertilization events and from mineralization of
483 organic-N, which are either transported with the infiltrating water or transformed through various
484 microbial processes.

485 *Start of flooding stage:* With the start of flooding (**Fig. 8b**) the root zone saturates with water
486 thereby decreasing oxygen and ORP levels. This is the most dynamic stage during the flooding
487 event, yet with high environmental significance as this stage will define the magnitude of the
488 NO_3^- leached. The amount of NO_3^- leached below the root zone and the amount of NO_3^-
489 denitrified in the topsoil is dependent on site-specific physico-chemical parameters (K_{sat} ,
490 infiltration rate, ORP, available NO_3^- and carbon, and temperature) and the abundance and
491 activity of both nitrifying and denitrifying microbial communities in the topsoil. NO_3^- leaching is
492 the dominating loss process in soils with high K_{sat} (or infiltration rate), while denitrification plays
493 a larger role in soils with lower K_{sat} where the residence time of water in the topsoil is longer. Of
494 course, denitrification is the preferred loss pathway for NO_3^- to reduce NO_3^- contamination to
495 groundwater, however, in the time it takes to reach suboxic conditions that promote
496 denitrification, high K_{sat} soil might already leach > 80% of the legacy NO_3^- present in the profile.

497 *Quasi-steady flooding stage:* After the highly dynamic NO_3^- leaching and N transformation
498 phase at the start of flooding, the system transitions into the quasi-steady flooding stage (**Fig. 8c**),
499 which is characterized by saturated, suboxic/anoxic conditions and more stable soil physico-
500 chemical parameters. If anoxic conditions are reached in the root zone, it can be assumed that
501 dissolved oxygen is likewise depleted as reported in other MAR studies (Turkeltaub et al., 2022).
502 Under these conditions, denitrifiers in the topsoil consume NO_3^- in the infiltrating water. This
503 NO_3^- removal will eventually contribute to the dilution of pore water NO_3^- in the vadose zone.
504 Sustaining the quasi-steady flooding stage for as long as possible is preferred because it increases
505 the amount of ‘clean’ recharge while reducing NO_3^- levels in the pore water transported to
506 groundwater. However, the dilution effect only takes effect if NO_3^- concentrations in the source
507 water are relatively low and the retention time of the infiltrating water in the denitrifying soil
508 layer is sufficiently long to allow complete NO_3^- consumption. For example, in this study,
509 infiltration rates of $0.1\text{-}0.2\text{ m d}^{-1}$ were sufficient to allow denitrification of NO_3^- in the source
510 water ($2\text{-}3\text{ mg NO}_3\text{-N L}^{-1}$) in the first 0.2 m of the soil profile. If recharge water is sourced from
511 pristine streamflow, it is typically low in NO_3^- concentrations. Hence, there is a high probability
512 for the dilution effect to occur during the quasi-steady flooding stage when implementing Ag-
513 MAR for periods of more than several days. However, flooding fields for long periods of time
514 might negatively impact crop performance, as shown for V1, which needs to be considered and
515 balanced (Levintal et al., 2022). A possible solution is to use a new model that predicts crop
516 damage as a function of the duration of saturated conditions in the soil root-zone, soil texture,
517 and crop tolerance to waterlogged conditions (Ganot and Dahlke, 2021a).

518 *Post-flooding stage:* In the post-flooding stage (**Fig. 8d**), oxygen and ORP will gradually
519 increase back to pre-flooding oxic levels. In soils with higher infiltration rates (sand or loamy-
520 sand) this process can take several days compared to several weeks in soils with lower
521 infiltration rates (e.g., sandy loam or loam). Soil moisture will also decrease from near- or full-
522 saturation to pre-flooding levels. However, this process is relatively slow (on the scale of weeks
523 to months) and controlled also by precipitation or irrigation events that will temporarily increase
524 soil moisture in the root zone. A post-flooding increase in NO_3^- and NH_4^+ is also expected,
525 mainly in the topsoil (0-0.2 m) as a result of mineralization and nitrification. Denitrification is
526 another post-flooding process that can occur, yet it will be more pronounced in soils with slow
527 infiltration rates when soil moisture is still high during the first stage of drainage when
528 anoxic/suboxic conditions prevail (V1 in this study).

529 Ag-MAR includes three distinct differences compared to other MAR types (Goren et al., 2014;
530 Gorski et al., 2020, 2019; Mienis and Arye, 2018): (1) the nitrogen source is mainly in the soil
531 rather than in the recharge water, (2) recharge (i.e., the flooding) will be primarily seasonal
532 during wet periods and not yearly with continuous flooding and drying cycles, and (3) the
533 infiltration basin is not necessarily on high-infiltrating land, such as sandy soil. Therefore,
534 caution should be taken when adapting the four model's stages for long-term flooding of other
535 MAR types. For example, nitrogen fate under soil aquifer treatment (SAT; a MAR form where
536 treated wastewater is recharged) showed highly complex biogeochemical processes due to
537 changing nitrogen and carbon loading (e.g. NH_4^+) in the source water (Mienis and Arye, 2018).
538 This resulted in varying nitrate concentrations in the groundwater during the recharge, which our
539 model cannot adequately explain.

540 4.4. *Implications for Ag-MAR operation*

541 Understanding the N-related dynamics at each stage of Ag-MAR as presented in the conceptual
542 model (**Fig. 8**) is essential for the development of best management practices. Taking V1 as a
543 case study, a better practice to decrease the risk of NO_3^- leaching would be to irrigate the
544 vineyard initially for 1-2 days (before the continuous flooding for Ag-MAR) until anoxic
545 conditions are developed, which would promote denitrification and minimize NO_3^- leaching.
546 Maintaining anoxic conditions is also important for the inhibition of the nitrification process that
547 can increase inorganic NO_3^- concentrations between individual recharge (i.e., flooding) events.
548 The predicted outcome of the suggested recharge practice will be a decrease in the overall
549 amount of NO_3^- leached. However, this practice is unlikely to be applicable in high infiltrating
550 soils in which nearly all the initial NO_3^- will be leached in the first few days before the
551 development of anoxic conditions (V2 in this study, infiltration rate of 0.171 m d^{-1}). A more
552 detailed discussion on the trade-offs between leaching and mineralization–denitrification
553 processes under repeated flooding events is provided by Murphy et al. (2021).

554 Our study was conducted in a Mediterranean climate, and therefore, adjustments are needed if
555 adapting the findings to other locations. For instance, lower ambient soil temperatures will
556 decrease the rate at which nitrification/denitrification occurs and to a lesser extent also influence
557 the infiltration rate (Grinshpan et al., 2022). Thus, Ag-MAR implementation in colder climates
558 will decrease denitrification rates during infiltration, potentially increasing the risk for leaching
559 of soil residual NO_3^- . Other considerations for Ag-MAR implementation are non-point source
560 contaminants with high leaching risks, such as salts, pesticides, and inorganic geogenic
561 contaminants. A comprehensive review of these contaminants with potential solutions is

562 provided in Levintal et al. (2022). Still, high NO_3^- concentration in the aquifer is the main
563 concern at global scale (Beganskas et al., 2018; Dahan et al., 2014). With respect to yield,
564 several reasons can explain the yield difference between the vineyards investigated in this study,
565 such as differences in soil texture and the resulting K_{sat} , infiltration rates, O_2 /redox levels,
566 duration of flooding, movement of available N below root zone, and timing of flooding with
567 respect to bud break in March. Therefore, possible effects on yield should be investigated in
568 more detail, and until then, caution is advised in using Ag-MAR in late winter-early spring, in
569 vineyards with less-than-ideal soil properties.

570 **5. Conclusions**

571 An agricultural managed aquifer recharge (Ag-MAR) experiment was conducted in the Central
572 Valley of California on two vineyards managed similarly but differing in soil texture and
573 hydraulic properties. One vineyard was flooded for four weeks (V1) and the other for two weeks
574 (V2). Although the flooding for groundwater recharge started on the same day, different soil
575 biogeochemical outcomes occurred. Suboxic conditions, favoring denitrification in the
576 microbially active topsoil (0-0.2 m), developed in the vineyard with the low-infiltration rate
577 (0.088 m d^{-1} ; V1) in less than one day, compared to 10 days in the high-infiltration vineyard
578 (0.171 m d^{-1} ; V2). Nevertheless, in both vineyards, NO_3^- leaching was the dominant N loss
579 process while denitrification played a lesser role in decreasing NO_3^- in the root zone. The amount
580 of NO_3^- leached below the 1 m root zone was mainly determined by the amount of residual NO_3^-
581 present in the root zone prior to the flooding for groundwater recharge, showcasing the
582 importance of estimating legacy NO_3^- pools before establishing an Ag-MAR site.

583 A conceptual model for water-N dynamics under Ag-MAR was developed based on hydrologic,
584 geochemical, and microbial process analyses. Four stages were identified: pre-flooding, start of
585 flooding, quasi-steady flooding, and post-flooding. Out of the above, the start of flooding is the
586 most dynamic stage that will define the magnitude of NO_3^- leached. The proposed model can be
587 used to determine best Ag-MAR management practices. Adoption of Ag-MAR, as one approach
588 in a portfolio of MAR methods, is desirable as population growth and expansion of irrigated
589 agriculture contribute to unsustainable groundwater extraction and a growing need to replenish
590 groundwater resources to buffer growing water supply variability.

591 **CRedit authorship contribution statement**

592 **Elad Levintal:** Methodology, Writing – original draft, Formal analysis. **Laibin Huang:**
593 Writing - review& editing, Formal analysis. **Cristina Prieto García:** Methodology, Writing -
594 review& editing, Formal analysis. **Adolfo Coyotl:** Writing - review& editing, Formal analysis.
595 **Matthew W. Fidelibus:** Writing - review& editing, Formal analysis. **William R. Horwath:**
596 Writing - review& editing, Funding acquisition. **Jorge L. Mazza Rodrigues:** Writing - review&
597 editing, Funding acquisition. **Helen E. Dahlke:** Supervision, Methodology, Writing - review&
598 editing, Funding acquisition.

599 **Declaration of competing interest**

600 The authors declare that they have no known competing financial interests or personal
601 relationships that could have appeared to influence the work reported in this paper.

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