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Agricultural managed aquifer recharge (Ag-MAR) – a method for sustainable groundwater management: A review

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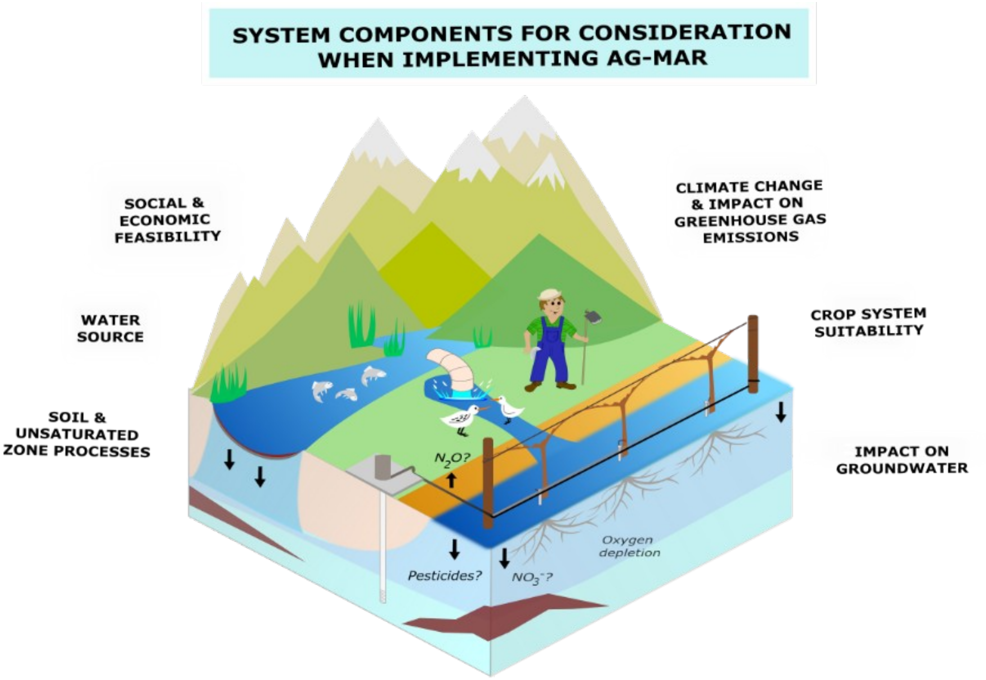
Abstract

More than two billion people and 40% of global agricultural production depend upon unsustainable groundwater extraction. Managed aquifer recharge (MAR), the practice of strategically recharging water to replenish subsurface storage, is an important subbasin scale practice for managing groundwater more sustainably. However, it is not yet reaching its full potential to counterbalance growing global groundwater demand. Agricultural managed aquifer recharge (Ag-MAR) is an emerging method for spreading large volume flows on agricultural lands and has capacity for widespread global implementation. Yet, knowledge gaps, synergies, and tradeoffs in Ag-MAR research still exist. We identify six key system considerations when implementing Ag-MAR: water source, soil and unsaturated zone processes, impact on groundwater, crop system suitability, climate change and impact on greenhouse gas emissions, and social and economic feasibility. We describe the present distribution, need for common terminology, and benefits of Ag-MAR including groundwater storage, increased environmental flows, and domestic wells support. We then outline major gaps, namely, water quality impacts, and crop health and yield. We showcase the multidisciplinary approach needed for communication and coordination of Ag-MAR programs with stakeholders and the public and provide a framework for implementation. Finally, we outline a vision for the path to Ag-MAR implementation. Ag-MAR is an important approach for achieving groundwater sustainability. However, it is one of many necessary solutions and does not offset the need for groundwater conservation.

Keywords: groundwater; managed aquifer recharge; soils; crops, water quality, vadose zone processes

Graphical abstract



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1. Introduction

More than 25% of the world population and 40% of global agricultural production depend upon unsustainable groundwater extraction (Connor, 2015). Population growth, rising living standards, and expansion of irrigated agriculture keep increasing demand for water and drive groundwater overdraft in many regions where surface water is scarce or only seasonally available (Pokhrel et al., 2021; Wada et al., 2010). Meanwhile, climate change models project increases in the magnitude and frequency of extreme precipitation events including multi-year droughts and floods (IPCC, 2014). For these reasons, the water resources community has emphasized the importance of sustainable groundwater management. Finding solutions for actualizing groundwater sustainability at multiple scales is crucial during the 21st century (Stokstad, 2020).

Managed aquifer recharge (MAR), the practice of strategically recharging water to replenish subsurface storage, is an important subbasin scale application for implementing sustainable groundwater management (Sprenger et al., 2017; Stefan & Ansems, 2018). Dillon et al. (2019) quantified MAR efforts from 15 countries with available data and found that between 1965 and 2015, MAR capacity increased from 1 to 10 km^3 year^{-1}. Effective MAR implementation requires careful tailoring according to local needs and constraints.

Although MAR technologies are implemented at increasing rates, recharge volumes of current MAR operations only replenish a fraction of the growing groundwater demand observed worldwide (Ross & Hasnain, 2018; Stefan & Ansems, 2018). This is likely due to factors such as availability of surplus water for recharge, lack of suitable and available recharge areas and delivery infrastructure (Niswonger et al., 2017), water rights limitations (Fuentes & Vervoort, 2020), economic feasibility (Ross & Hasnain, 2018), and institutional barriers (Miller et al., 2021b). Scientists and stakeholders largely continue to view MAR as a costly solution with high potential risk (Stefan & Ansems, 2018).

Agricultural managed aquifer recharge (Ag-MAR) is an emerging water spreading MAR method that has potential for widespread implementation (Bachand et al., 2014; Dahlke et al., 2018a). Ag-MAR, also referred to as agricultural groundwater banking, on-farm recharge, or flood-flow capture (Bachand et al., 2014), aims to transfer excess surface water during times of water availability (e.g., rainy season, snowmelt, reservoir releases) onto agricultural land for recharge to groundwater (Harter & Dahlke, 2014).

Ag-MAR differs in several ways from infiltration basins, a traditional MAR method which from the process perspective resembles Ag-MAR most closely (**Table 1**). The most significant difference is that MAR infiltration basins consist of land dedicated to a single purpose (Massuel et al., 2014; Prathapar et al., 2015), while Ag-MAR represents a secondary use of agricultural land that is primarily used for agricultural production (Dahlke et al., 2018a). With croplands and pastures comprising approximately 40% of the global land surface (Foley et al., 2005), agricultural land has the potential to recharge larger volumes (200 to 3200 Mm^3 year^{-1}) (Gailey et al., 2019; Kocis & Dahlke, 2017) of surplus (often surface) water to aquifers by flooding large agricultural areas (>500 ha) (Ulibarri et al., 2021).

Current knowledge gaps present challenges and concerns regarding Ag-MAR implementation. These include Ag-MAR effects on crop yield and health (including post-flooding effects such as pest management) (Dahlke et al., 2018a); leaching of legacy nitrogen, salts, pathogens, and inorganic geogenic contaminants (e.g., arsenic (As)) to groundwater (Bachand et al., 2014; Waterhouse et al., 2020); waterlogging of agricultural lands adjacent to Ag-MAR sites which may lead to hypoxic/anoxic conditions (Ganot & Dahlke, 2021b); short and long-term effects on in-stream flows (e.g., tradeoffs between ecosystem services) (Kourakos et al., 2019); economic feasibility (Gailey et al., 2019); water policy barriers; and methods for siting suitable Ag-MAR locations (O'Geen et al., 2015). Less apparent, yet equally important concerns include Ag-MAR effects on greenhouse gas (GHG) emissions and risk of soil compaction and reduced farm machinery trafficability after Ag-MAR (Devine et al., 2022). In addition, most published Ag-MAR research to date has been conducted in California and is scarce for other countries.

Increases in Ag-MAR research and stakeholder interest in implementation illustrate the need for a critical review on Ag-MAR that summarizes and synthesizes the current available knowledge. A SCOPUS search for peer-reviewed articles on Ag-MAR shows a steady increase in publications in the last 15-years (**Fig. S1**). However, to the best of our knowledge, a review paper specific to Ag-MAR – often not included in MAR reviews – does not yet exist.

The aim of this review is to synthesize past and current research related to Ag-MAR to showcase the current state of Ag-MAR knowledge, identify research gaps, describe possible synergies and tradeoffs, and offer a vision for the future of Ag-MAR. This review also provides a framework for understanding key components and mechanisms influencing Ag-MAR implementation. Accordingly, sources used herein include professional and committee reports in addition to academic research.

2. System components to consider for Ag-MAR implementation

During Ag-MAR, farmland is flooded with surplus water – often river water – to recharge the underlying aquifer (Kocis & Dahlke, 2017). Ag-MAR directly influences the atmosphere-crop-soil-groundwater continuum, and its implementation requires widespread socioeconomic coordination. Implementation of Ag-MAR requires careful consideration of several site conditions. As such, we focus and structure our state-of-the-science review on six system components (**Fig. 1**).

2.1. Water source

Several water sources can be considered for Ag-MAR including stormwater, recycled water, desalinated water, transferred water, conserved water, and surface water (Alam et al., 2020; DWR, 2015; Grinshpan et al., 2021). Among these sources, stormwater and high-magnitude streamflows (i.e., flood flows) are likely the most accessible and largest sources of water for expansion of groundwater banking programs worldwide (Harter & Dahlke, 2014; Scanlon et al., 2016), in part due to the intensification of the hydrologic cycle which predicts increases in flood magnitudes. Excess water availability is typically greatest when river levels are generally the highest in the middle of the rainy season (e.g., mid-winter, early-spring) or during monsoon or wet months due to precipitation and snow melt (Chowdhury et al., 2010; Niswonger et al., 2017). The use of reservoir releases can also be a source (e.g., releases for flood control) or extend the season of available water for recharge (Goharian et al., 2019).

Ag-MAR relies upon infrastructure to convey water from the source (e.g., river) to the agricultural recharge field. Common water conveyance systems use existing canals, ditches, creeks, turnouts, and pipelines (Marwaha et al., 2021; Ulibarri et al., 2021). Using unlined canals as water conveyance can generate additional groundwater recharge by seepage, which can be an order of magnitude smaller than the field recharge (Niswonger et al., 2017). Suitable conveyance infrastructure is one of the key challenges of Ag-MAR due to the need to transport high volumes of source water to the recharge basins during winter and spring months. For example, ~3.7×10^7 m^3 per day of river water was available for recharge in the Central Valley of California during February and March of 2017, but was concentrated in locations where conveyance limits were below needed capacity (Hanak et al., 2018). Additionally, different water sources require specific types of conveyance infrastructure, with stormwater requiring more than the other water sources (Perrone & Merri Rohde, 2016). Groundwater overdraft can increase land subsidence and damage conveyance infrastructure (land subsidence caused a 60% reduction in conveyance capacity in the southern part of the California Aqueduct; Hanak et al., 2018) and limit Ag-MAR potential. For much of California, and likely in many other places across the world, the capacity and structure of existing infrastructure needs to be evaluated and new infrastructure must be strategically located to facilitate Ag-MAR (Fitchette, 2017; Hanak et al., 2018). Costs for new infrastructure in areas where existing surface water conveyance is not available to transport high magnitude flows can further limit Ag-MAR feasibility (Gailey et al., 2019).

Water available for recharge depends on climatic conditions and site-specific regulations such as minimum in-stream flow requirements or surface water rights. The frequency of river water availability can vary considerably both intra- and inter-annually. Kocis & Dahlke (2017) found that in California only high magnitude storm flows – flows that are not legally apportioned in the water rights permitting process – provide a physically available surplus water source for Ag-MAR since most surface water is already fully allocated or over-allocated (Grantham & Viers, 2014). They recognized that environmental flow criteria must be considered when determining availability of flows for Ag-MAR implementation and recommended using the 90th percentile of daily streamflow during high magnitude flows for recharge. Using this criteria, Kocis & Dahlke (2017) showed that high magnitude streamflow was available 7 and 4.7 out of 10 years in the Sacramento and San Joaquin basins, respectively. Yang & Scanlon (2019) applied a similar approach with a threshold of the 95th percentile in the Texas Golf region (total of 10 rivers), which is subjected to extreme flooding events from hurricanes. They reported an average number of 2 to 15 high magnitude flow events per year over the past 50 years, and an average duration per event between 1 and 35 days. The 90th and 95th thresholds were motivated by the environmental flow community considering these as 'much above normal' flows. Using lower, less conservative thresholds is possible; however, high magnitude flows are also crucial for other environmental functions, such as sediment transport or riparian vegetation. Further discussion regarding these tradeoffs is given in Kocis & Dahlke (2017) and Yang & Scanlon (2019). Niswonger et al. (2017) found that climatic conditions in northwest Nevada, USA, supported sufficient river flows for Ag-MAR during 7 out of the 24 years (1990-2014) that were simulated. During these 7 years, annual runoff ranged between 130 and 220% of the average, out of which about 7% of the total annual runoff could be diverted for Ag-MAR.

Successful implementation of Ag-MAR requires that certain water quality standards of source water are met (Fakhreddine et al., 2021; Ghasemizade et al., 2019). High nutrient loads within source water can percolate from the land surface to the groundwater potentially contaminating the groundwater below and adjacent to the recharge field (Beganskas et al., 2018). Groundwater contaminants of concern are nitrate, salts, pesticides and metals (Dahlke et al., 2018b), however, these are often found at higher concentrations in the soil than the applied water.

Applied water can also vary widely in dissolved oxygen concentrations depending on water source. River water, in contrast to standing water (e.g., lakes, ponds), typically has higher dissolved oxygen concentration in the winter months due to lower microbial respiration and higher mechanical (e.g., turbulent or wind-driven) mixing. In general, water sources from upstream rivers or snow melt will have lower nutrient loads and higher dissolved oxygen than downstream sources or alternative water sources such as treated wastewater.

Another potential risk to human health associated with Ag-MAR operations (and MAR in general) is the presence of microbial pathogens in recovered groundwater (Dillon et al., 2010). Floodwater and stormwater may contain pathogenic microbes, such as viruses, bacteria, and protozoa. Bacteria and parasites are larger than viruses and would largely be removed during percolation (Regnery et al., 2017). However, viruses are generally considered to be of greatest risk because of their low infectious dose (Ward et al., 1986) and potential to travel long distances in the subsurface (Schijven & Hassanizadeh, 2000). The recent Ebola, SARS, MERS, and COVID-19 outbreaks are examples of viral infections with unprecedented impacts on public health (Elston et al., 2017) and the global economy (Orlik et al., 2020). Enteric viruses were found to travel within the soil to depths of several tens of meters, with most studies indicating a 1 to 5-log virus reduction during MAR (Betancourt et al., 2014; Gerba & Goyal, 1985). The survival rate of viruses is highly site-specific and field studies investigating viruses transport under MAR systems are needed (Regnery et al., 2017). Available studies focus mainly on treated wastewater as the water source for MAR; presently, there is no field data regarding virus transport under Ag-MAR. In agricultural settings, improperly treated or poorly contained waste, livestock, applied manure, and wildlife are a primary non-point source of microbial pathogens

(Benham et al., 2006; Bradford et al., 2006). During Ag-MAR operations, floodwater can incorporate these pathogens and contaminate the groundwater. In addition, flooding can pose a risk to crop production by favoring the development and spread of soil-borne pathogens such as phytophthora that depend upon wet soil conditions for growth, reproduction, and dissemination (Palti, 2012).

2.2. Soil and unsaturated zone processes

On its way to the groundwater table, recharge first needs to infiltrate into and percolate through the soil. To provide guidance on soil suitability for recharge, O'Geen et al. (2015) developed the Soil Agricultural Groundwater Banking Index (SAGBI) for California considering five soil factors: deep percolation, root zone residence time, topography, chemical limitations, and surface condition. Soil and hydrogeologic maps can be used in the assessment of a potential Ag-MAR site, based on the textural classification of the soil and subsurface sediments (Bouwer, 2002). Because Ag-MAR projects are planned for relatively large areas, lower infiltration rates are acceptable compared to conventional MAR sites, and generally the hydraulic conductivity (K_s) of most soils, excluding clayey soils (<0.1 m day^{-1}), should be sufficient (Ganot & Dahlke, 2021a). However, acceptable K_s are site-specific and dependent on applied water volumes and infiltration areas available for Ag-MAR. The presence of preferential flow paths, such as fractures or wormholes, can support percolation rates that are at least one-order of magnitude higher than the average modeled flow using K_s (Nimmo et al., 2021). Yet, estimating preferential flow is a complex, mostly unsolved problem, and therefore, K_s is still the best estimator to use when evaluating percolation rates at a potential Ag-MAR site.

Initial soil hydraulic properties can change during excessive flooding due to soil clogging at the soil-water interface. Soil clogging can reduce infiltration rates at the surface and is a primary operational concern in most MAR systems. There are three types of soil clogging: physical clogging due to the filtration of suspended solids in the recharge water, biological clogging resulting from bacterial activity and biofilm formation, and chemical clogging due to precipitation of particles and minerals (Pavelic et al., 2011; Zaidi et al., 2020). The degree of clogging depends on the particle size of the suspended material in the water, duration of flooding, the in-situ soil texture and chemical characteristics, and to a lesser extent on the ambient conditions, such as temperature (Ghazavi et al., 2010). In a column experiment, in which treated recycled water was used for recharge, soil clogging reduced infiltration rates 6-fold for sand and 8-fold for loam type soils, with physical clogging being the main process (Pavelic et al., 2011). A similar 4-fold reduction in infiltration rate was found in a recharge field study conducted on sandy loam soil using river water as source water (Ghazavi et al., 2010). Depending on the water source, clogging can occur in Ag-MAR operations, although no studies have been published investigating clogging in Ag-MAR to date (Beganskas & Fisher, 2017). Clogging during Ag-MAR with a high-quality source-water (low values of turbidity, organic matter, and total dissolved solids) is likely not a primary concern (Ganot et al., 2017). However, if the soil has a high silt or clay fraction, the clean water could detach particles from the soil surface and transport them with the flood water to the recharge field. Using high-magnitude streamflow with high sediment concentrations as the source water (Kocis & Dahlke, 2017) might require pre-treatment using a dedicated sedimentation basin to settle clay, silt, and other suspended solids (Beganskas & Fisher, 2017), or flooding only after the high volume of sediment has passed (e.g., use flows from the receding limb of the flood peak). The use of standard farm machinery to plow Ag-MAR recharge fields between seasons and the lower frequency at which Ag-MAR is practices could decrease potential long-term clogging effects on infiltration.

Ag-MAR can adversely impact groundwater quality at some sites and benefit water quality in others (Page et al., 2010; Schmidt et al., 2012). The risk of transporting contaminants to connected surface water or groundwater bodies depend largely on the source of the contaminant and biogeochemical processes within the vadose zone (**Fig. 2**).

Fertilizers are often found in elevated amounts in the vadose zone beneath agricultural fields (Böhlke, 2002; Walvoord et al., 2003). It is commonly assumed that fertilizers and salts lost from the root zone in agricultural areas will reach the groundwater through the vadose zone, which in most cases extend from several meters to several tens of meters (Gurevich et al., 2021). Nitrogen fertilizers are the major concern for groundwater contamination due to their wide, often excessive, spread in regions of agricultural development (Böhlke, 2002; X. Zhang, 2017). Important biogeochemical processes related to nitrogen cycle dynamics under Ag-MAR include denitrification (Gorski et al., 2019; Schmidt et al., 2012), mineralization (Cabrera, 1993; Harter et al., 2005), and nitrate leaching to groundwater, which is of most concern (**Fig. 2**). Nitrate leaching is expected to be highest at the onset of a flooding event, with the potential for dilution as additional flood water is applied.

If cropland is flooded for extended periods of time, an anaerobic environment ($O_2 < 5\%$) may develop in the root and vadose zone providing conditions for increased denitrification potential. Soil texture, infiltration rate, and ponding duration are the main parameters to determine the rate at which anaerobic conditions develop. In a recent recharge experiment conducted in two vineyards with fine sandy loam soil in the Central Valley (California, USA), anaerobic conditions (i.e., negative redox potential) occurred after ~1 day of flooding in one vineyard while the second vineyard maintained mostly aerobic (redox potential of ~400 Eh) conditions (Levintal et al., In prep.). The difference in oxygen status could be mainly attributed to differences in infiltration rate, which were 0.09 and 0.19 m day^{-1} in the anaerobic and aerobic vineyard, respectively. In cases of high frequency flooding (i.e., flooding every day for several hours), the soil between flooding cycles may be wet but aerobic, encouraging conditions favorable to mineralization and nitrification, which can increase the amount of mineral-N available for leaching in subsequent flooding applications (Murphy et al., 2021).

Nitrate leaching management is important because legacy nitrogen pools under intensively cultivated agricultural land have been documented globally (Harter et al., 2005; Van Meter et al., 2016). Nitrogen byproducts or nitrate from both fertilization and irrigation are often transported below the effective root zone, becoming unavailable for crop utilization. In general, more inefficient irrigation methods (e.g., gravity irrigation methods) result in a greater fraction of the applied nitrogen leaching from the root zone (Baram et al., 2016). The mobilization and transport of these legacy nitrate pools must be considered when establishing an Ag-MAR site. Bastani & Harter (2019) showed that if Ag-MAR is practiced in the source area of a domestic drinking water supply well, lowering the nitrate load while also increasing recharge in the well's source area simultaneously can reduce nitrate in the supply well by 80%.

Salts are distributed naturally in soil, but concentrations can be accelerated with the use of inappropriate irrigation regimes and sources (e.g., irrigating with brackish water) (Bachand et al., 2014; Pauloo et al., 2021; Zeng et al., 2014). Electrical conductivity and total dissolved solids

(TDS) are the two parameters used to determine soil and water salinity (Rusydi, 2018). The primary ions found in soils are Na^+, K^+, Ca^{2+}, Mg^{2+}, HCO_3^-, SO_4^{2-} and Cl^- (Zeng et al., 2014), with cation exchange and precipitation and dissolution being the controlling reactions for these ions (Schoups et al., 2005; Suarez & Šimůnek, 1997). Literature on salt leaching due to long-term flood irrigation suggests potential risk to groundwater (Dong et al., 2019; Schoups et al., 2005). Therefore, it is likely that salinity contamination, similar to nitrate, will occur as a result of Ag-MAR's high magnitude flows flushing mobile pools from the water source or/and vadose zone towards the groundwater table. However, after subsequent flooding events, groundwater quality might improve due to the dilution effect, depending on site-specific parameters, such as the number and magnitude of the flooding events, salt and other contaminant concentrations in the source water, and the residual contaminant loading of the flooded soil. The dilution effect is expected to be lesser if recovered, low-quality water from the same aquifer is used for subsequent irrigation. Bachand et al. (2014) developed a model to calculate the recharge volume needed to return the groundwater to its original background concentration, given the salt or nitrogen load present within the unsaturated zone. In their Ag-MAR field study conducted in California, they estimated that 12 m^3 m^{-2} of recharge water taken from a nearby river would be needed to displace the legacy soil salts (11 kg TDS m^{-2}) (groundwater level at 60 m below ground). Moreover, recent investigation of Ag-MAR conducted at six different sites in the San Joaquin Valley (California, USA) concluded that salt leaching under Ag-MAR is still not clear (Bachand et al., 2017). Thus, dedicated research on salt leaching under Ag-MAR is needed.

Leaching of pesticide residues are another concern under Ag-MAR, particularly under high recharge rates. Pesticide degradation occurs mainly in the upper soil and root zone, with the presence of high organic matter and increased microbial population abundance and activity (Youbin et al., 2009). However, pesticide residues below the root zone can be found years after their surface application (Rose et al., 2018). Pesticide fate and transport is influenced by various sorption and degradation processes, often quantified by their adsorption coefficients and the pesticide's degradation half-life (Youbin et al., 2009). Pesticide adsorption coefficients decrease with depth (Youbin et al., 2009), causing a greater leaching potential of pesticide residues below the root zone. Soil properties with the potential to affect pesticide sorption and degradation rates are clay content, organic matter content, total carbon, soil cation exchange capacity, temperature, moisture content, pH, redox conditions, residence time in the soil column, and the microbiological community (Rose et al., 2018; Youbin et al., 2009).

Studies on pesticides fate in flood-irrigated fields can be used as a simplified analogy to Ag-MAR application (Chokejaroenrat et al., 2020; Hildebrandt et al., 2008; Torrentó et al., 2018). The behavior of Metolachlor, a frequently used herbicide in Europe and the USA, was studied over a 12-year period in seven agricultural watersheds across the USA (Rose et al., 2018). The authors estimated that <0.02% of the annual applied Metolachlor leached to the groundwater after ~90% was degraded or taken up by the crop. To date, no dedicated Ag-MAR-pesticide research has been conducted. Thus, there is uncertainty regarding the behavior and transport of pesticides under high recharge rates.

Inorganic geogenic contaminants pose a challenge at MAR sites because they persist throughout large areas and do not decay like organic compounds (Fakhreddine et al., 2021; Schafer et al., 2021). Arsenic (As) is the most problematic geogenic contaminant for Ag-MAR sites due to its health threats, low regulatory limit in drinking water (maximum contaminant level in drinking water of 10 ppb; U.S. Environmental Protection Agency), ubiquity in sediments, and mobilization under recharge-induced shifts in redox conditions (Fakhreddine et al., 2021). The main mechanism for As mobilization during recharge is via oxidative dissolution of As-bearing pyritic minerals (Fakhreddine et al., 2021). Mobilization rates depend on the pyrite concentrations in the sediment, oxidant concentrations in the recharge water (primarily dissolved oxygen), concentrations of organic matter, and operational decisions (e.g., wetting and drying cycles) (Fakhreddine et al., 2021; Jones & Pichler, 2007). Geogenic contamination of groundwater from Ag-MAR operations is likely of lower risk compared to other MAR methods given the lower recharge rates and greater likelihood of similarities in redox potential between source water and the soil and unsaturated zone. Studies investigating geogenic contamination under Ag-MAR (e.g., mobilization of As and U under high nitrate load as potential oxidant (Nolan & Weber, 2015)) are needed.

Prior to flooding an Ag-MAR site, it is important to consider the historic land management practices to estimate the nitrate, salinity, or pesticide contamination potential of groundwater (Bastani & Harter, 2019). Possible remediation techniques of existing groundwater contamination include utilizing the dilution effect, or the addition of biomatter (e.g., wood chips, mulch, almond shells) to soil to promote the growth of microbes that remove contaminants (Beganskas et al., 2018; Stokstad, 2020). Through careful consideration of site-specific variables (soil properties, Ag-MAR site area, flooding timing and magnitude, crop stage), best management practices may be developed to minimize contaminant leaching potential. We note that these considerations should be carefully implemented due to the complexity of the system. For instance, promoting anaerobic condition through continuous flooding could increase desired denitrification (removal of nitrate), yet initiate undesired mobilization of As in soils with a neutral pH (Korte & Fernando, 1991) or increase dissolved concentrations of Fe and Mn due to dissolution of Fe oxides and Mn oxides (Fakhreddine et al., 2021). Although the potential for recharge to contaminate groundwater can be high, heavily irrigated areas with extensive overdraft can be at a greater risk when not pursuing Ag-MAR as ongoing depletion can degrade water quality in the aquifer (Beganskas & Fisher, 2017).

2.3. Impact on groundwater

The heterogeneity of an aquifer is important to consider when identifying locations for Ag-MAR projects to allow for infiltration to deeper aquifer layers and contain sufficient capacity for storage (Fuentes & Vervoort, 2020; Maples et al., 2019; Stokstad, 2020). Factors such as hydraulic conductivity, preferential flow paths, confined or partially confined layers, depth to groundwater, location within the groundwater system, and proximity to drinking water sources all affect successful Ag-MAR implementation. Characterizing subsurface heterogeneity is key for successful groundwater recharge, which can also affect denitrification rates (Waterhouse et al., 2021; Goebel & Knight, 2021). Ag-MAR over coarse-texture deposits is favorable compared to confining silt and clay units that limit recharge (see Maples et al. (2019) for further discussion). Goebel & Knight (2021) used a transient electromagnetic geophysical method to translate electrical resistivity to sediment type in effort to assess preferable locations for recharge, locations where pathways of hydraulically conductive sediments (sands and gravels) occur between the land surface and the groundwater table. Geophysical methods were also used to characterize perched aquifers adjacent to streams where Ag-MAR could potentially be used to support river baseflow (Kniffin et al., In prep). Using boreholes and geostatistical methods, Maples et al. (2019) found that interconnected, coarse-textured recharge pathways allow for rapid, high-volume MAR and propagate pressure responses in aquifers over multiple kilometers. In addition, a three-dimensional, variably saturated, integrated hydrologic modeling code, ParFlow, showed the importance of both course- and fine-textured sediment in alluvial systems: recharge was initially located within the course-texture facies, but was ultimately stored in fine-textured facies.

Agricultural lands are often sites of groundwater depletion due to high rates of groundwater pumping (Gleeson et al., 2012; Rodell et al., 2009). Ag-MAR applied in areas with reduced groundwater levels and storage can counteract groundwater depletion and associated consequences (e.g., degradation of groundwater dependent ecosystems, land subsidence) and/or promote recovery of depleted aquifers (Kourakos et al., 2019; Stokstad, 2020). Studies investigating Ag-MAR at the basin or regional scale using simulated numerical models in the southwestern USA over multiple decades found that Ag-MAR increased groundwater storage between 26 and 34% depending on aquifer characteristics (Ghasemizade et al., 2019; Kourakos et al., 2019; Niswonger et al., 2017). Model simulations showed that water level increases were sustained for at least three years above baseline conditions depending on the Ag-MAR regimen (Niswonger et al., 2017).

2.4. *Cropping system suitability*

Ag-MAR can reduce oxygen levels within the soil, potentially inhibiting root respiration and root growth, and thus can have a negative effect on crop yield. The oxygen levels in soils depend highly on the gas phase, since the oxygen concentration in atmospheric air is ~21% (210,000 mg l^{-1}) while water in equilibrium with the atmosphere contains dissolved oxygen of only ~8 mg l^{-1}.

Oxygen in the gas phase is supplied from the atmosphere to the soil mainly via diffusive transport (Ben-Noah & Friedman, 2018) and in some cases also by advective thermal, barometric, or wind transport (Ganot et al., 2014; Levintal et al., 2017, 2019; Massman, 2006). Upon flooding, ponding creates a barrier between the atmosphere and the soil root zone, which reduces diffusive rates by four orders of magnitude and blocks advective transport (Scott & Renaud, 2007). In addition, increase in soil water content reduces pore space connectivity, which also reduces oxygen gas diffusivity. The resulting depletion in soil oxygen will also depend on temperature and respiration activity, with lower depletion rates expected at low temperatures and low content of organic matter (Colmer & Greenway, 2005). Upon waterlogging, the decline in soil oxygen from ~21% to 0% can vary, ranging from one (Trought & Drew, 1980) to several days (Blackwell, 1983) or weeks. The effect of oxygen deficiency on crop health is mainly depended on the degree of oxygen shortage (partial – hypoxia, or total – anoxia) and its duration, crop stage (e.g., dormancy, blooming), crop flooding tolerance, microbial community and activity, salinity and temperature (Ben-Noah & Friedman, 2018). Root zone residence time, defined as the duration of saturated (or near saturated) conditions in the soil root-zone without crop damage or yield loss (O'Geen et al., 2015), is a key parameter for successful Ag-MAR implementation. It depends on both soil characteristics and plant tolerance to saturation, making its estimation a challenge (mainly because of lack of systematic data for flood-tolerant plants). Ganot & Dahlke (2021a) developed a model for estimating Ag-MAR flooding duration depending on root zone residence time for different crops and soil textures. Their model provides a first approximation of the amount of water that can be applied safely during Ag-MAR to avoid crop damage. According to the model it is, for instance, safe to apply water for 13 days on a vineyard during the dormancy stage, on loamy sand, and assuming an effective root depth of 1 m and a ponding level of 0.1 m.

If river water is used as source water for Ag-MAR, the dissolved oxygen of the applied water is expected to be around saturation values (~8 mg l^{-1} at 25 °C and 1 atm) with higher saturation values expected for cold, flowing surface water. Still, this dissolved oxygen amount is considered a negligible oxygen source for root respiration (compared to gas-phase) as respiration rates are higher than the dissolved oxygen replenishment rate of the infiltrating water (Hillel, 1998).

Beside oxygen depletion, flooding inhibits seed germination, vegetative and reproductive growth, changes plant anatomy, and ultimately can lead to plant mortality. In a review of the effects of flooding and salinity on woody plants, Kozlowski (1997) reported that under flooding conditions root growth is generally reduced more than shoot growth, and fruit growth is also inhibited resulting in lower fruit quality. Moreover, the combined effect of flooding and salinity decreases plant survival more than either stress alone. Prolonged flooding can also promote the growth of fungi, bacteria, and other pests that harm plant growth (Drew & Lynch, 1980).

When implementing Ag-MAR, prolonged flooding would generally occur on fallowed fields or during crop dormancy but damage in this phase can influence future productivity (Schaffer et al., 1992) and resilience to other stressors (e.g., root growth (Thompson & Fick, 1981); disease incidence (Drew & Lynch, 1980; Schaffer et al., 1992; Thompson & Fick, 1981); soil fertility (Kozlowski & Pallardy, 1984; Schaffer et al., 1992). For many crop types including pasture and alfalfa, grains, and almonds, the temperature of the applied water and the completely saturated root zone influences the extent of crop damage (Morales-Olmedo et al., 2015; Thompson & Fick, 1981; Zhou et al., 2003). Informed rootstock selection for fruit and nut trees (e.g., citrus, almonds) can help protect the plant from the risks of these saturated conditions, such as oxidative stress, ferric chlorosis, fungal infection, limited nutrient uptake (Bhusal et al., 2002; Morales-Olmedo et al., 2015; Schaffer et al., 1992).

Little research exists about crop tolerance and response to the prolonged flooding conditions required for Ag-MAR. Crop tolerance varies because crop type and growth stage have varying root depths and distribution which affect respiration and oxygen requirements throughout the root zone. Research done by Bachand et al. (2014, 2016) quantified the recharge capacity of fields located in California and timed flood flow diversions to not interfere with traditional crop management.

Bachand et al. (2014, 2016) found that vineyards displayed no damage to crop yield and quality after controlled flooding from April through May (Mediterranean climate, clay loam soil) and pistachios and alfalfa showed no significant yield penalties after controlled flooding in April when on sandy loams and loamy sands. Dahlke et al. (2018) recently investigated the effect of different Ag-MAR flooding schemes on established alfalfa fields in California (Mediterranean climate), and results suggest that there is no significant effect on yield when dormant alfalfa fields on highly permeable soils are subject to winter flooding. Appropriate crops for Ag-MAR implementation are summarized and discussed in O'Geen et al. (2015) and Ganot & Dahlke (2021a).

Crops that are normally subject to flooded conditions may allow for easier integration of Ag-MAR with traditional crop management. Kennedy (2015) found that flooding cranberries for an average of 33 days between late December and early February for groundwater recharge yielded four times greater recharge amounts than recharge conducted during harvest flooding. Winter flooding of rice fields is becoming increasingly common because of agronomic benefits (e.g., increasing straw decomposition rate, weed growth inhibition, limiting erosion), and when well informed, can also provide hydrologic and environmental benefits (Negri et al., 2020). To maintain higher groundwater levels until the beginning of the agricultural season (end of April through beginning of May; Mediterranean climate), winter flooding of rice likely needs to involve large, contiguous areas and should be continued for upwards of three months and/or end close to the beginning of the agricultural season (Mayer et al., 2019; Natuhara, 2013; Negri et al., 2020).

2.5. Climate change and impact on greenhouse gas emissions

Implementing Ag-MAR has potential implications for feedback mechanisms to climate change. The two main pathways include possible GHG emissions resulting from anaerobic conditions during long-term flooding and future water source changes resulting from changes in precipitation and snowmelt. The primary GHG concern associated with Ag-MAR is the potential emission of nitrous oxide, a long-lived stratospheric ozone-depleting gas (Tian et al., 2020) with a global warming potential 298 times greater than carbon dioxide (Verhoeven et al., 2017). Cultivated soils are the primary source for anthropogenic nitrous oxide emissions (Shcherbak & Robertson, 2019), with nitrification and denitrification being the biochemical processes controlling the production (Tian et al., 2020). Under continuous and prolonged flooding for Ag-MAR, sustaining anaerobic conditions for relatively long periods within the soil can stimulate higher denitrification rates, leading to higher production and emissions of nitrous oxide. Yet, results from a new field study of Ag-MAR implemented on two vineyards in California showed no observed emissions of nitrous oxide (or carbon dioxide or methane) during- and post-Ag-MAR flooding (Levintal et al., In prep).

Knowledge about water availability for Ag-MAR under future climatic conditions is limited and largely depends on existing climate and upland watershed models. Yet, most studies predict that the frequency and magnitude of floods across the world will increase due to climate change (Allan & Soden, 2008; Yang & Scanlon, 2019). Countries that are already facing widespread floods include: India, Bangladesh, and China (Yang & Scanlon, 2019), and the U.S. West Coast (Berg & Hall, 2015; Shields & Kiehl, 2016). Ag-MAR can utilize the increase in floodwater volume to recharge groundwater in depleted aquifers while also acting as a useful solution for flood control (Kourakos et al., 2019; Scanlon et al., 2016). Given these trends in climate, increasing groundwater recharge, could be a cost-effective tool to deal with climate change (Bachand et al., 2014) and could be considered for various carbon credit programs.

2.6. Social and economic feasibility

Water laws and regulations are one of the major barriers to pursuing Ag-MAR, even during times with surplus water (Fuentes & Vervoort, 2020). Water laws predominantly focus on volume and timing of water diversions to an implementation site (Fuentes & Vervoort, 2020). Although water laws vary across the globe, universal considerations used to determine regulatory feasibility of a site include historical water rights, environmental flows, Ag-MAR ecosystem services, and grower's water rights priorities (Ghasemizade et al., 2019; Niswonger et al., 2017). Case studies show that water laws often result in organizational challenges that impede successful Ag-MAR implementation (Miller et al., 2021b). Such challenges benefit from collaborative modeling (e.g., using a centrally coordinated model to communicate between organizations), public management and financing, and negotiation processes between stakeholders and local, state, and federal agencies (Miller et al., 2021a).

Economic costs are a second non-technical barrier for Ag-MAR implementation that comprise direct and indirect components (Tran et al., 2020). Direct components include project planning, building or maintaining conveyance infrastructure, and building physical barriers for ponding (e.g., berms). Indirect cost components include instrumenting monitoring systems to quantify the crop response and water volume and quality of recharge (Dahlke et al., 2018a), economic incentives for farmer participation compensating for perceived risks to crop health (Dahlke et al., 2018b; Gailey et al., 2019), and prior appropriation of water costs. Gailey et al. (2019) developed a hydro-economic approach for planning Ag-MAR projects, combining elements of recharge basin and groundwater hydraulics with economic considerations at a regional scale. In two sub-basins in California's Central Valley, they conclude that Ag-MAR was an economically feasible method with approximately 4.8 km^3 available for recharge over 20 years (1983-2003) at a 540 km^2 site. They indicate results are the "best-case scenario" because of three study limitations: fixed cropland rental price, uniform distribution of ponded water, and exclusion of water quality issues that could reduce available land surface for recharge.

In a case study focusing on a single farm, Ag-MAR cost was estimated to be \$0.03 per m^3 (over 25 years), which is much lower than the cost of engineered recharge basins (ranging between \$0.07 and \$0.89 per m$^{3)}$ (Bachand et al., 2014, 2016); for reference, the cost of groundwater for the farmer at that area was ~\$0.08 m^3. The Ag-MAR cost above included labor and farm-scale land preparation and infrastructure. Yet, there are additional cost considerations related to Ag-

MAR, such as development of large-scale infrastructure to convey source water (initial investment vs. maintenance), instrumentation and monitoring, and potential yield loss. Although the economic cost of Ag-MAR has not yet been investigated at the farm-scale, one can only assume that these factors will increase the cost of Ag-MAR in other locations.

Other studies have also shown that Ag-MAR is an economically viable method with a cost for one cubic meter of water that is one order of magnitude lower than other water storage and supply strategies, like seawater desalination or use of reservoirs (Dahlke et al., 2018b; Perrone & Merri Rohde, 2016). For example, Bachand et al. (2014) estimated Ag-MAR cost at \$0.03 per m^3, which is significantly lower than seawater desalination (\$1.54-\$2.43 per m^3) or large-scale surface water storage (\$1.38-\$2.27 per m^3).

Ag-MAR was estimated to be the most affordable option for groundwater dependent communities in the San Joaquin Valley, California, with an additional cost less than 10% of current rates (Bastani & Harter, 2019). The authors concluded that in cases of groundwater nitrate contamination, Ag-MAR can be a cost-effective alternative to existing solutions (e.g., well head treatment). They emphasize that low nitrogen emitting crops that can sustain high recharge rates during Ag-MAR may be economically advantageous in the long-term despite high conversion costs (e.g., converting almond orchards to vineyards), though additional studies are needed to validate this conclusion.

Many Ag-MAR benefits are externalities not presently considered in economic assessments. From a multi-generational, collective perspective, the cost of Ag-MAR implementation may be less than environmental degradation (e.g., land subsidence) and subsequent remediation.

However, mechanisms to incorporate multi-generational time horizons in land and water planning and implementation processes are lacking. It is important for policymakers to develop methods for valuing sustainable groundwater management, environmental justice, and environmental protection.

3. Discussion

3.1. Global Ag-MAR distribution

While not as globally prevalent as MAR projects, Ag-MAR practices have been around for several decades (Dokoozlian et al., 1987) and have increased, particularly in USA and Europe, in the last decade (Facchi et al., 2020). In Europe, winter flooding of rice paddies has been practiced since the late 1990s and recently northern Italy adopted Ag-MAR as part of the EU-Rural development program 2014-2020 (Facchi et al., 2020). However, the term Ag-MAR is a relatively new descriptor of several practices that involve excess irrigation or collection of flood water or surface runoff from farmland that have been practiced for decades or even centuries. In the web-based global database of MAR projects created by Stefan & Ansems (2018), excess irrigation is the type of MAR practice in the database that most closely resembles Ag-MAR, although other forms including flooding or infiltration ponds and basins could be grouped under the same term. To date, most Ag-MAR research is primarily conducted (and was defined) in the western USA – California and Nevada (Niswonger et al., 2017). Given human population growth and climate change predictions, Ag-MAR will likely expand throughout groundwater-dependent regions, particularly in arid and semi-arid areas with great pressure on groundwater resources such as the southwestern USA, India, Pakistan, the Middle East, the North China Plain, and North Africa.

The suitability of agricultural landscapes for Ag-MAR can be fairly easily assessed across the globe using existing geospatial datasets and Geographic information system (GIS)-based multi-criteria decision analyses (e.g. Russo et al. 2014, Sallwey et al., 2019; Marwaha et al. 2021). The key environmental variables to be considered in GIS multicriteria decision analyses (MCDAs) are soil type, land use (including crop type), topography, hydrogeology, and surface water conveyance infrastructure (Marwaha et al., 2021; O'Geen et al., 2015). Such GIS-based approached with the potential to incorporate Ag-MAR parameters are available, for instance, for northern Greece (Kazakis, 2018), Australia (Fuentes & Vervoort, 2020), India (Chowdhury et al. 2009), and South Africa (Zhang et al., 2019). A review of GIS-based MAR studies with the potential to delineate suitable locations for Ag-MAR across the globe is provided by Kazakis (2018) and Sallwey et al. (2019). From a site management perspective, Ag-MAR can easily be implemented where fields are flood-irrigated, because they already have the infrastructure to spread water in place. If flood irrigation infrastructure is in place, site suitability would have to be assessed based on soil type, land use, and water availability since some flood-resistance crops (e.g., rice) grow on soils that do not promote large recharge amounts. To date, information on global adoption and suitable areas for Ag-MAR is lacking, and there is a need for dedicated research examining the potential for Ag-MAR implementation worldwide.

In addition, institutional elements, which are often highly site-specific, present major barriers to nationwide or global Ag-MAR implementation. Economic feasibility and policy guidelines (e.g., water laws) are often not considered in GIS-based MCDAs, and therefore, overlooked. This is partially due to the dynamic nature of these institutional elements. Policy can change on a yearly basis compared to physical parameters such as soil texture or land use. A review of economic and policy guidelines is given by Dillon et al. (2019), focusing on Australia, USA, India, and Europe, and by Ajjur & Baalousha (2021) for the Middle East and North Africa. Although these reviews address traditional MAR systems, they could serve as a first step to guide the implementation of Ag-MAR from a local perspective.

3.2. Ag-MAR and ecosystem services

Ag-MAR has the capacity to support ecosystem services by transforming agricultural fields into multi-use, multi-functional landscapes. Ag-MAR ecosystem services include aquifer recharge and groundwater storage, environmental flows for groundwater dependent ecosystems, wildlife habitat, flood and drought mitigation, prevention of seawater intrusion, control of contaminant plumes, and prevention of land subsidence (Damigos et al., 2017). Alam et al. (2020) estimated that high magnitude flows allocated to MAR and applied throughout the California Central Valley can increase groundwater storage and recover 9 to 22% of existing groundwater overdraft, while supplementing 52 to 73% of Central Valley-wide low streamflows when simulated over a 56-year period (1960-2015). Kourakos et al. (2019) found that 66% of Ag-MAR applied to a sub-basin in the northern Central Valley discharged back to streams increasing environmental flows that support aquatic habitats over an 80-year simulation. Increases in groundwater storage, in turn, maintain groundwater levels important for groundwater pumping, particularly in preventing domestic well failure during high-risk drought periods (Pauloo et al., 2020). An average year with excess flows in the Central Valley exports approximately 3.2 km^3 of water to the Sacramento-San Joaquin Delta over a few storm events, and Ag-MAR can help mitigate these high magnitude flows (Kocis & Dahlke, 2017).

Balancing ecosystem services between stakeholders is a challenge given that water and land practices affecting ecosystem services are value-based and interactions between services are complex, often occurring as synergies or tradeoffs. Unlike conservation easements or retiring land for MAR, Ag-MAR is a multi-use land practice that requires consideration for agricultural production and other ecosystem services. An example of a synergy is when water spread on agricultural land mitigates floods, while contaminants are biodegraded in the soil substrate prior to reaching the groundwater table (Griebler & Avramov, 2015). Tradeoffs occur when pumping water from a river for Ag-MAR negatively impacts downstream groundwater dependent ecosystems by reducing flows or the necessary transport of nutrients, sediment, and freshwater to bay and estuary ecosystems (Kourakos et al., 2019). Tradeoffs also occur when long duration flooding events aimed to promote denitrifying conditions negatively impact crop health and yields (Gorski et al., 2019; O'Geen et al., 2015). While these interactions are complex, services commonly occur in groups on similar landscape types (Cord et al., 2017). Ag-MAR system designs may benefit from exploring ecosystem service literature focusing on systematic analyses of interactions that identify leverage points and maximize multi-functionality (Bennett et al., 2009; Cord et al., 2017; Howe et al., 2014).

3.3. Research gaps and future directions for Ag-MAR research

Optimization of synergies and trade-offs in Ag-MAR projects poses a complex problem as multiple goals and variables must be considered. The goal of Ag-MAR implementation is to maximize groundwater recharge quantity, while minimizing risks, such as contaminating groundwater through subsurface biogeochemical reactions that mobilize contaminants. Parameters affecting site selection (soil type, crop type, conveyance infrastructure) and best management practices (flooding frequency, flooding magnitude, timing between flooding events) impact the quantity and quality of water recharged to the underlying aquifer system. Additional research is needed to understand synergistic benefits, tradeoffs, and risks of Ag-MAR (Beganskas et al., 2018).

Future research should focus on investigating Ag-MAR mechanisms and economies of scale through integrated computational models paired with field studies. To date, few Ag-MAR studies have focused on Ag-MAR at regional rather than the site or farm scale (Alam et al., 2020; Ghasemizade et al., 2019; Kourakos et al., 2019). Computational models can help to: 1) identify locations for Ag-MAR sites/recharge locations (Behroozmand et al., 2019); 2) determine high magnitude flow volumes needed to maintain sediment transport and stream channel geometry (Yang & Scanlon, 2019); 3) assess the size of infiltration basins needed; 4) assess the fate and transport of water and contaminants through the subsurface; 5) evaluate increasing water tables in the root zone in response to Ag-MAR practices; 6) explore surface and groundwater interactions (Niswonger et al., 2017); and 7) determine potential impacts Ag-MAR can have on a system under future climate scenarios.

Additional research is needed to understand Ag-MAR's possible use as a soil aquifer treatment (SAT) system. The use of treated wastewater for agricultural irrigation is widespread and projected to grow due to precipitation variability and growing food demand (Poustie et al., 2020). Application of treated wastewater depends on its quality, the crop, hydrological vulnerability below the sites, and specific regulations of the region/state/country. Guidelines for the microbiological quality of treated wastewater are more restricted when applied through flood irrigation compared to sprinklers or drip irrigation to guarantee the safety of farmworkers

(Blumenthal et al., 2000). The composition of the applied water for Ag-MAR, combined with lithology and land use, will determine the quality of the water recharged to the aquifer below an Ag-MAR site, as the applied water undergoes biogeochemical transformations during deep percolation (Kass et al., 2005). Research needs to explore the possibilities of combined SAT/Ag-MAR applications. Combined applications may benefit from use of permeable reactive barriers to reduce contaminant loads including nitrate leaching (Gorski et al., 2019). On-going research is currently deployed at the SHAFDAN SAT site, Israel, where secondary effluents are used for Ag-MAR in citrus trees (Grinshpan et al., 2021, 2022).

Ag-MAR has the capacity to improve water security and the natural environment while supporting agricultural economies. However, communication and cooperation with and among stakeholders are essential for successful application (Hanak et al., 2018; Perrone & Merri Rohde, 2016). Efforts to include stakeholders in the research process are essential since solutions that deliver multiple ecosystem services to a range of stakeholders have higher chances of success (Hanak et al., 2018). Ag-MAR research can provide valuable information in discussions about future changes in land use and management when it is properly communicated to stakeholders and decision makers (Marwaha et al., 2021; O'Geen et al., 2015). Collaborative modeling can communicate complex scientific ideas across organizations and interest groups translating models from simulation to implementation on the landscape (Kniffin et al., 2020; Miller et al., 2021b). Integrative modeling frameworks that incorporate social, hydrological, and ecosystem factors of Ag-MAR have the capacity to inform multi-benefit projects that value groundwater sustainability, agricultural production, environmental justice, and environmental protection under multi-generational time horizons (Ghasemizade et al., 2019; Marwaha et al., 2021). It is then important to find methods for incorporating these findings into economic assessments.

3.4. Implementation of Ag-MAR at the site scale

Ag-MAR site selection and implementation demands a multidisciplinary knowledgebase and systematic decision-making process that involves stakeholders at all stages. Moreover, the process must recognize parameter tradeoffs and related risks along with ecosystem service tradeoffs. This is especially true since water sources for Ag-MAR can be relevant only once in several years, and therefore there is one chance to succeed. Yet, to the best of our knowledge, there is no published research describing the detailed steps, from planning to operation, for Ag-MAR.

To bridge this gap, we provide a framework of considerations for Ag-MAR implementation at the site scale (**Fig. 3**). The framework was divided into five chronological stages: preliminary regional investigations (stage #1), advanced site investigations (stage #2), site preparations (stage #3), flooding (stage #4), and post-flooding (stage #5). Each stage was divided into guidelines related to physical and socio-economic considerations. We acknowledge this is a simplified scheme, and therefore references of relevant studies were added within the scheme for each stage.

4. Summary and future vision

This paper provides a review of research on agricultural managed aquifer recharge (Ag-MAR) organized into six key system components affecting Ag-MAR implementation: water source, soil and unsaturated zone, groundwater, crop systems, climate change, and social and economic feasibility. We discuss the complexity of optimization of ecosystem service synergies, trade-offs and risks as well as Ag-MAR implementation considerations. We then provide a framework for Ag-MAR implementation at the site scale. For this method to be considered, first and foremost, excess water must be available at regular intervals to balance the infrastructure requirements and economic risks.

Ag-MAR implementation requires assessment of economic impacts and methods for overcoming organizational and institutional challenges. This will involve identifying suitable crops for Ag-MAR under different climate regimes and soil types to demonstrate that this method is economically viable and not detrimental to crop production. Economic costs currently limit Ag-MAR implementation and require public-private collaborations. Agricultural areas not producing high-cost, lucrative crops in particular, lack sufficient economic resources. Effective federal, state, and local government funds, incentive programs, and permitting processes are necessary and will need to support the private sector to effectively shift agriculture practices.

Additional studies should focus on groundwater quality impacts of Ag-MAR implementation, which is directly connected to soil health. Finding ways for soil health to be improved and reduce contaminant loading to groundwater is critical to ensure long-term groundwater quality. Future research should focus on combined benefits of improving soil health and on-farm recharge to allow for infiltration and water filtration via biogeochemical processes as water travels to the groundwater table.

Ag-MAR has been slower to develop compared to MAR likely because it employs a multi-functional land use approach requiring diverse knowledge bases and expertise for implementation. While stakeholder engagement is often included in grant proposals, in practice it is frequently implemented at the end of project timelines with minimal resources. The academic community needs to improve collaborative research by emphasizing the social science aspect of Ag-MAR, which can inform the theory and development of research methods and processes, management practices, policy infrastructure, incentives, and science communication and collaboration. Collaborative modeling is one approach for informing the theory and practice of Ag-MAR – linking stakeholder, technical, and process-based knowledge.

A vision for a successful Ag-MAR project relies on careful regional and site planning. Ag-MAR is not suitable for all locations – it is limited to agricultural areas with sufficient surface water resources. Ultimately, Ag-MAR must be placed in a larger context, recognizing that it is one approach in a portfolio of methods necessary for managing sustainable quantities and qualities of groundwater for generations to come.

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Table 1. Comparison between MAR infiltration basins and Ag-MAR sites

	MAR infiltration basins	**Ag-MAR sites**
Land use	Single	Integrated (agriculture and groundwater recharge)
Application time	Annual or seasonal	Seasonal (wet periods with fallow or dormant agriculture)
Flooded area	< 100 ha	> 500 ha
Water source	Surface, storm, recycled, or desalinated water	High volume surface water flows
Water volume	Between 12 and 70 Mm^3 $year^{-1}$	Between 200 and 3200 Mm^3 $year^{-1}$
Application frequency	Variable, depending on source	Periodic, weather dependent

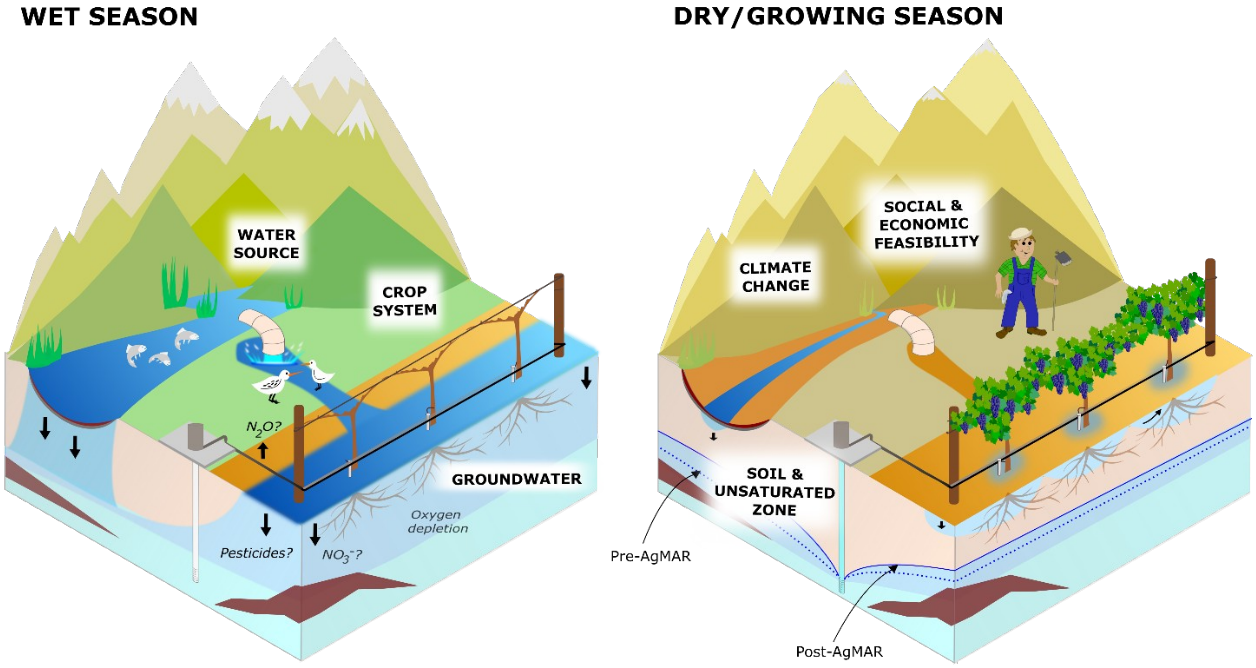


Fig. 1. Conceptual model of the six key system components influencing Ag-MAR implementation: 1) water source, 2) soil and unsaturated zone processes, 3) impact on groundwater, 4) crop system suitability, 5) climate change and impact on GHG emissions, and 6) social and economic feasibility.

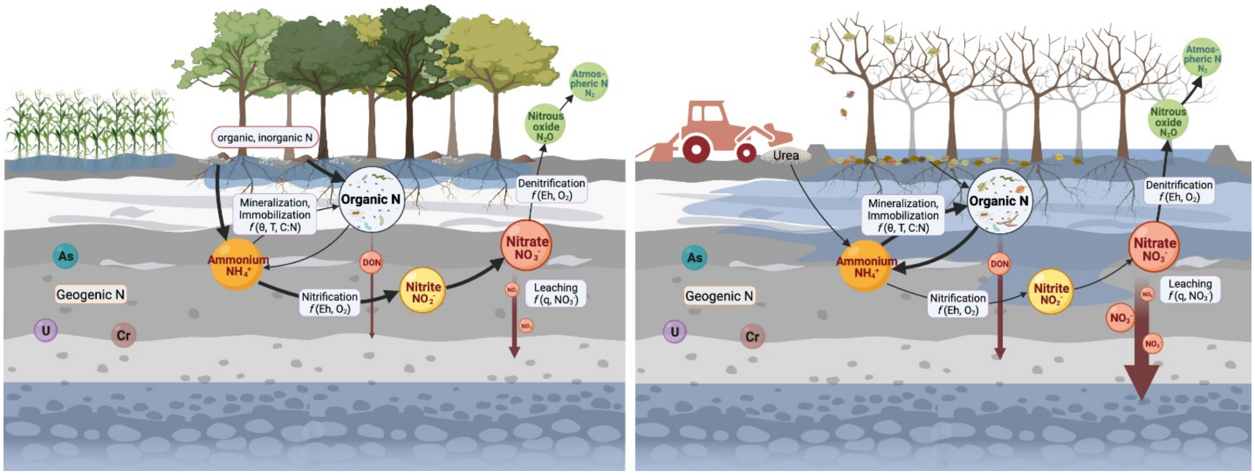
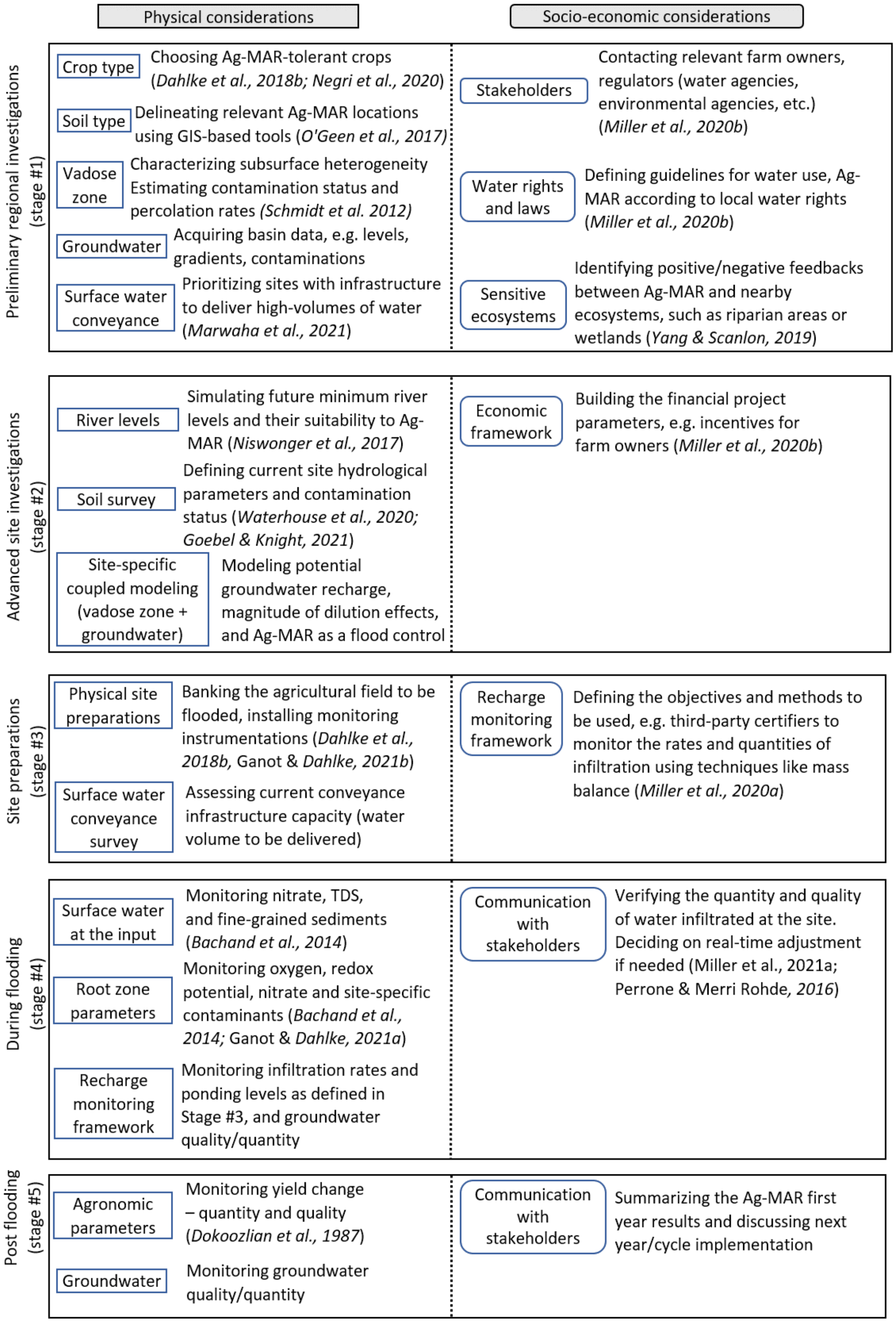


Fig. 2. Nitrogen cycle dynamics during a standard crop growing season (a) and under winter Ag-MAR flooding (b). Thickness of arrows indicates relative importance of contributing reactions. Figure created with Biorender.



Physical considerations

Socio-economic considerations

Preliminary regional investigations (stage #1)

- Crop type: Choosing Ag-MAR-tolerant crops (*Dahlke et al., 2018b; Negri et al., 2020*)
- Soil type: Delineating relevant Ag-MAR locations using GIS-based tools (*O'Geen et al., 2017*)
- Vadose zone: Characterizing subsurface heterogeneity Estimating contamination status and percolation rates *(Schmidt et al. 2012)*
- Groundwater: Acquiring basin data, e.g. levels, gradients, contaminations
- Surface water conveyance: Prioritizing sites with infrastructure to deliver high-volumes of water (*Marwaha et al., 2021*)
- Stakeholders: Contacting relevant farm owners, regulators (water agencies, environmental agencies, etc.) (*Miller et al., 2020b*)
- Water rights and laws: Defining guidelines for water use, Ag-MAR according to local water rights (*Miller et al., 2020b*)
- Sensitive ecosystems: Identifying positive/negative feedbacks between Ag-MAR and nearby ecosystems, such as riparian areas or wetlands (*Yang & Scanlon, 2019*)

Advanced site investigations (stage #2)

- River levels: Simulating future minimum river levels and their suitability to Ag-MAR (*Niswonger et al., 2017*)
- Soil survey: Defining current site hydrological parameters and contamination status (*Waterhouse et al., 2020; Goebel & Knight, 2021*)
- Site-specific coupled modeling (vadose zone + groundwater): Modeling potential groundwater recharge, magnitude of dilution effects, and Ag-MAR as a flood control
- Economic framework: Building the financial project parameters, e.g. incentives for farm owners (*Miller et al., 2020b*)

Site preparations (stage #3)

- Physical site preparations: Banking the agricultural field to be flooded, installing monitoring instrumentations (*Dahlke et al., 2018b,* Ganot & *Dahlke, 2021b*)
- Surface water conveyance survey: Assessing current conveyance infrastructure capacity (water volume to be delivered)
- Recharge monitoring framework: Defining the objectives and methods to be used, e.g. third-party certifiers to monitor the rates and quantities of infiltration using techniques like mass balance (*Miller et al., 2020a*)

During flooding (stage #4)

- Surface water at the input: Monitoring nitrate, TDS, and fine-grained sediments (*Bachand et al., 2014*)
- Root zone parameters: Monitoring oxygen, redox potential, nitrate and site-specific contaminants (*Bachand et al., 2014;* Ganot & *Dahlke, 2021a*)
- Recharge monitoring framework: Monitoring infiltration rates and ponding levels as defined in Stage #3, and groundwater quality/quantity
- Communication with stakeholders: Verifying the quantity and quality of water infiltrated at the site. Deciding on real-time adjustment if needed (Miller et al., 2021a; Perrone & Merri Rohde*, 2016*)

Post flooding (stage #5)

- Agronomic parameters: Monitoring yield change – quantity and quality (*Dokoozlian et al., 1987*)
- Groundwater: Monitoring groundwater quality/quantity
- Communication with stakeholders: Summarizing the Ag-MAR first year results and discussing next year/cycle implementation

Fig. 3. Physical and socio-economic considerations for different stages of implementing Ag-MAR.