## Title
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# Managed Aquifer Recharge as a tool to enhance sustainable groundwater management in California: examples from field and modeling studies

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## 1. Introduction

A growing population and an increased demand for water resources have resulted in a global trend of groundwater depletion. Arid and semi-arid climates are particularly susceptible, often relying on groundwater to support large population centers or irrigated agriculture in the absence of sufficient surface water resources [1]. For example, it is estimated that 43% of global consumptive water use for agricultural irrigation comes from groundwater, with the most agricultural land irrigated with groundwater in China, India, and the United States [2]. Natural recharge is inherently limited in arid and semi-arid climates and the anticipated effects of climate change on recharge in these regions are largely uncertain [3]. In an effort to increase the security of groundwater resources, managed aquifer recharge (MAR) programs have been developed and implemented globally [4]. Managed aquifer recharge is the approach of intentionally harvesting and infiltrating water to recharge depleted aquifer storage (Figure 1).

California is a prime example of this growing problem, with three cities that have over a million residents [6] and an agricultural industry that was valued at $47 billion dollars in 2015 [7]. As a result of the ongoing depletion of groundwater reserves in California, groundwater aquifers currently have the capacity to store an additional 44 $km^3$ to 80 $km^3$ of water above the natural groundwater reservoir capacity, for a total storage capacity three times the amount currently provided by surface water reservoirs [8, 9, 10, 11]. California is marked by having the largest climatic variability in the United States, challenging water resource managers' ability to meet water supply needs and mitigate flood risks [12]. The present day groundwater overdraft of over 100 $km^3$ (since 1962) indicates a clear disparity between surface water supply and water

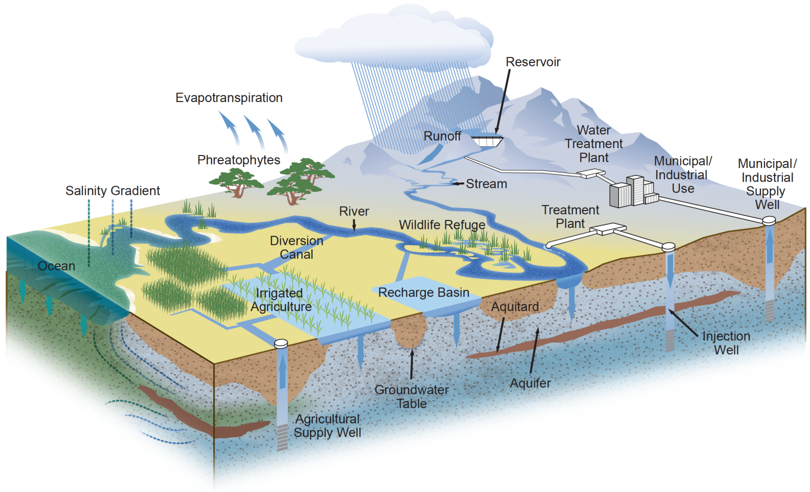


Figure 1: Groundwater management schematic including MAR methods ([5]).

demand within the state. Climate change models predict an increase in aridity and the occurrence of droughts, which could exacerbate groundwater overdraft in the state [13]. However, while total annual precipitation is expected to decrease, precipitation frequency and magnitude is expected to increase, potentially leading to greater surface runoff from precipitation in excess of infiltration, reduced groundwater recharge, and more extreme flood events during wet years [12, 14, 15, 16]. Exacerbating California's climatic variability, and therefore the variability in surface water availability, climate change poses a serious concern for the future management of surface and groundwater supplies. In the face of groundwater overdraft and the anticipated effects of climate change, many new MAR projects are being constructed or investigated throughout California, adding to those that have existed for decades [17]. California therefore provides an excellent case study to look at the historical use and performance of MAR, ongoing and emerging challenges, novel MAR applications, and the potential for expansion of MAR.

Effective MAR projects are an essential tool for increasing groundwater security, both in California and on a global scale. In order for MAR projects to be effective they must be appropriately tailored to the local needs and constraints. There are many existing types of managed aquifer recharge, which vary in land availability requirements, source water, project objectives, and other factors. Some common MAR types utilized in California include injection wells, infiltration basins (also known as spreading basins, percolation basins, or recharge basins), and low-impact development (Table 1). An emerging MAR type that is actively being investigated is the winter flooding of agricultural fields using existing irrigation infrastructure and excess surface water resources, known as agricultural MAR. Many of these MAR types can be considered through the lens of conjunctive use, which is the coordinated management of surface water and groundwater supplies to maximize the sustainable yield of the overall water resource[18]. When surface water is used to recharge groundwater, MAR can be viewed as an expansion of conjunctive use[19], and vice versa.

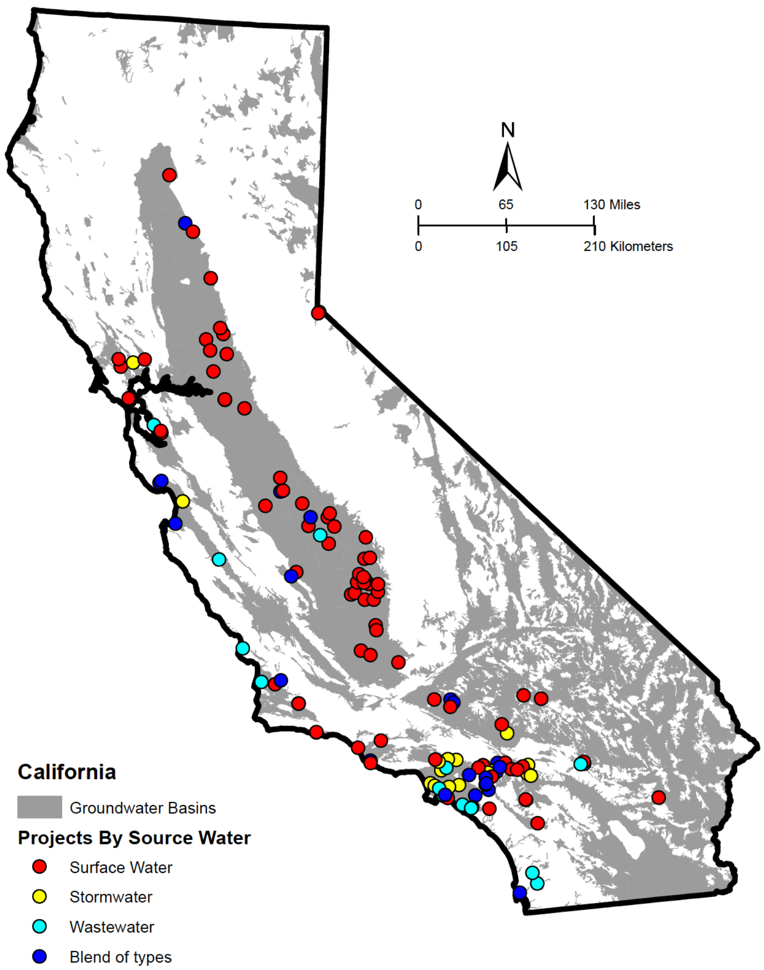


Figure 2: Proposed and funded MAR projects in CA since 2000 ([17]).

This chapter aims to provide an overview of the most common MAR types and applications within the State of California and neighboring semi-arid regions. Based on differences in project constraints and project objectives, this chapter reviews both traditional and new, promising MAR approaches in urban, agricultural, and coastal areas, respectively (Figure 2). Urban areas typically have limited land availability and may rely on injection wells, infiltration basins, or low-impact development, and utilize developed surface water, run-off, or recycled water. Agricultural areas have extensive land surfaces for spreading water and can utilize existing irrigation infrastructure, but are also limited by sporadic surface water availability depending on location. Coastal regions differ from agricultural and urban areas in that prevention or mitigation of seawater intrusion is often the primary MAR objective. Each section introduces the most common MAR types found in urban, coastal, and agricultural regions within California and discusses their strengths, limitations, and future implications. This chapter concludes with a discussion of environmental benefits of MAR in the context of California's new groundwater legislation, opportunities for future expansion of MAR, and potential concerns or barriers to the expansion of MAR.

Table 1: Managed aquifer recharge (MAR) types and source water quality regulations for California.

| MAR type | Context | Source water type | Water quality requirements | Regulations |
|---|---|---|---|---|
| All MAR types | Urban Coastal Agricultural | all | • Federal Endangered Species Act<br>• Federal Clean Water Act (1972)<br>• Porter Cologne Water Quality Control Act (1969)<br>• California Environmental Quality Act | 33 U.S. Code § 1251, 14 CCR § 15000-15387 |
| Infiltration basins | Urban Coastal | Recycled | 2 month minimum retention time (if determined by added tracer) | 22 CCR § 60320.124 |
| | | Surface water | California Code of Regulation<br>• General federal and state water quality regulations | |
| Injection wells | Urban Coastal | Recycled | California Code of Regulation<br>• 2 month minimum retention time (if determined by added tracer)<br>• Treatment by reverse osmosis and oxidation | 22 CCR § 60320.224, 22 CCR § 60320.201 |
| | | Surface water | U.S. Code on Public Health and Welfare<br>• Must comply with Safe Drinking Water Act program for Underground Injection Control (administered in California by the U.S. EPA) | 42 U.S. Code § 300f |
| Low-impact development | Urban | Stormwater | • National Pollutant Discharge Elimination System (NPDES) stormwater permits | 33 U.S. Code § 1342 |
| ag-MAR | Agricultural | Surface water | None specifically for MAR, but agricultural lands must comply with:<br>• Porter-Cologne Water Quality Control Act (incl. Irrigated Lands Regulatory Program, Central Valley Salinity Coalition, Dairy Order) | California Water Code Division 7 13000-16104 |

## 2. Managed Aquifer Recharge in urban settings

California has some of the oldest and largest urban MAR projects in the United States to secure urban water supply, improve groundwater quality, and mitigate negative impacts of groundwater overdraft (i.e., subsidence)[20]. Sources and pathways for groundwater recharge in urban environments are more numerous and unique compared to rural environments [21], which provide both opportunities and challenges for MAR implementation. MAR projects that provide flood protection have been practiced as early as 1910 in Los Angeles (LA) [22, 23], while water quality focused urban MAR projects were introduced later in the 20th century (e.g. 1990s), when the U.S. Environmental Protection Agency (EPA) began regulating stormwater quality after passage of the U.S. Clean Water Act in 1972 [24, 25]. MAR programs in California's urban centers have changed in size, purpose, and benefits over the past century. While enhancing water supply was the primary goal of urban, *centralized* MAR projects (i.e. large footprint, $> 1$ ha in area, $> 1{,}000{,}000$ $m^3$/yr recharge volume) prior to the 1980s, recently implemented *decentralized* (i.e. small footprint, $<1$ ha in area, $< 10{,}000$ $m^3$/yr recharge volume) MAR projects are found to bring diverse benefits such as conjunctive use, flood protection, stormwater quality management, and groundwater recharge [17].

### *2.1. Centralized MAR approaches*

In Los Angeles and Orange County, surface reservoirs for flood control (e.g. Ivanhoe and Silver Lake reservoirs) and infiltration basins (e.g. Prado Dam) were built by federal and local agencies in response to significant flooding between 1900 and 1950 [22, 26]. These projects represent some of the best studied centralized urban MAR projects in California today, characterized by infiltration volumes on the order of more than 100,000 $m^3$/yr and infiltration areas on the order of tens of hectares [27]. Infiltration basins are a relatively low cost, simple technology that have been implemented extensively to recharge groundwater in California. Infiltration basins require land and dedicated facilities constructed solely for recharge. Compared to the more maintenance-intensive dry wells and injection wells, infiltration basins are often preferred because of their relatively low capital cost and low annual operation and maintenance costs [27, 28, 29]. However, a primary drawback of infiltration basins is their large land area requirements compared to well technologies, which can become a capital cost factor in areas where property prices are high [29].

Since its inception in the 1930s, the Orange County Water District has employed a variety of technologies to secure water supply to its population, which has grown from 120,000 in the 1930s to 2.4 million today [26]. Early MAR efforts in Orange County began with increasing the natural percolation capacity of the Santa Ana River [26]. As natural recharge proved insufficient to offset increasing water demand, imported water from the Colorado River was purchased starting in 1949 and recharged in the 26 ha Anaheim Lake (Figure 3) [30] since 1958, the Orange County Water District's [26] first infiltration basin. Since then, treated Colorado River water has been delivered to 25 infiltration basins (including Anaheim Lake) within Orange County. However, decreasing reliability and increasing costs of imported water led water agencies in Southern California look at alternative water sources, particularly recycled wastewater. In 1962, Los Angeles County implemented the first large scale infiltration project of secondary-treated wastewater in California using the Montebello Forebay; in 1976 the Orange County Water Factory 21 became the first facility permitted by California's Department of Public Health and Regional Water Quality Control Board to tertiary treat, blend, and inject wastewater into drinking water aquifers [31]. The Water Factory 21 was replaced by the Groundwater Replenishment System in 2008, a larger wastewater treatment plant, which now feeds the Miraloma Basin, a 4 ha infiltration basin, at a rate of 36,990,000 $m^3$ (30,000 acre-feet) annually.

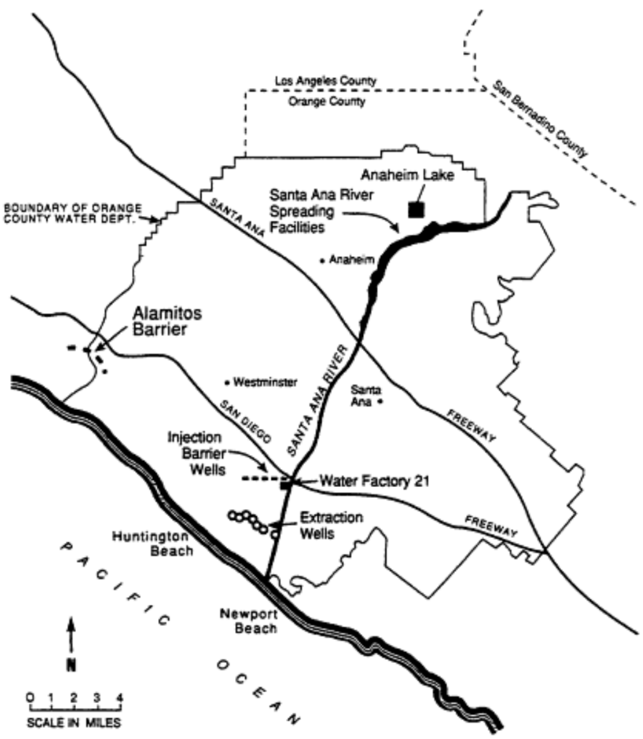


Figure 3: The location of OCWD, its recharge facilities, and geological gaps ([17]).

Los Angeles water managers have shifted from local, to imported, to recycled and

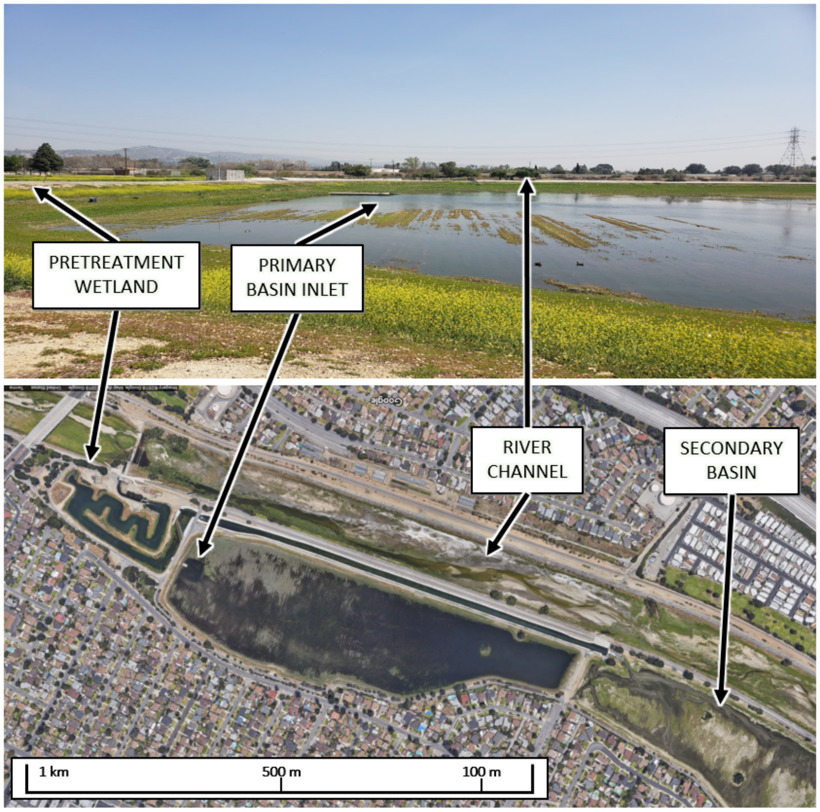


Figure 4: San Gabriel River channel and infiltration basin recharging stormwater, treated wastewater and imported water to the Los Angeles Central groundwater basin.

stormwater sources over the last 80 years [23], [31], while updating the infrastructure of infiltration basins to match the changing water sources. In LA, large centralized flood control structures (e.g. surface reservoirs, lined stormwater flow channels) were first engineered through federal and regional projects to capture large but infrequent runoff to reduce flood risk [22]. As groundwater supply diminished, flood control structures were altered to capture more runoff during rain events, resulting in recharge of 0.09 km$^3$ of stormwater (71,144 acre-feet) county-wide in 2016 [32]. In addition, implementation of flexible infrastructure such as in-channel inflatable dams at the San Gabriel infiltration project has increased infiltration throughout the basin by replacing sand and gravel levees that would wash out during high flows [33]. A new, 20-year plan expects to produce a two-fold increase in recharge, bringing the city's annual recharge from 0.03 to 0.08 km$^3$ (26,671 acre-feet to 64,022 acre-feet) by 2035 [23]. The bulk of the recharge increase is expected to come from 19 centralized stormwater capture projects at various scales that combine flood control and groundwater recharge (Figure 4).

The use of infiltration basins in urban settings has also raised questions about the impact of infiltration basins on groundwater quality in settings where groundwater flow velocities are high, potentially increasing the risk of groundwater contamination with surface water or stormwater contaminants [34]. O'Leary et al. [34], for example, observed groundwater flow velocities of 13 m/d in an alluvial aquifer near Stockton, CA. However, water quality monitoring in the aquifer near the recharge site showed that concentrations in dissolved solids, dissolved organic carbon, and arsenic in the groundwater decreased, indicating that the recharged surface water had a diluting effect on groundwater quality. At the same time they observed low concentrations in herbicides typically found in stormwater runoff, indicating that the risk of groundwater contamination with pollutants present in the recharged surface water was low [34, 35].

#### *2.1.1. Conjunctive use and in-lieu recharge*

In addition to innovations in infrastructure, urban water agencies across California have found it necessary to enhance recharge management strategies through soft technologies such as conjunctive use and in-lieu recharge of groundwater. The Santa Clara Valley Water District was among the first agencies to implement a conjunctive use program [36] to support local water supply reliability dating back to the 1930s. In response to declining groundwater levels and resulting land subsidence in the 1960s, the district began importing and treating surface water to significantly reduce the direct use of groundwater, also known as in-lieu recharge [36]. In a modeling study, Hanson [37] used MODFLOW-2000, the USGS three-dimensional finite-difference model, to determine groundwater flow in the Santa Clara Valley, a region characterized by complex aquifer layering, faults, and stream channels. The model determines the supply and demand components of the water inflows and outflows of the valley for six climate cycles (i.e. dry, wet periods) since 1800. The study highlights the need to optimize where groundwater is pumped in the valley depending on water demand and groundwater management goals.

Despite its clear benefits, implementation of conjunctive use programs is often dependent on political and institutional factors [38]. In a recent example, the San Francisco Public Utilities Commission (SFPUC) and its partner agencies engaged in a formal collaboration to coordinate surface and groundwater supply beyond city boundaries (Figure 5) [39]. In wet years, SFPUC would supply the partner agencies with surface water to promote in-lieu recharge of the the Southern Westside Basin [40], resulting in approximately 0.08 $km^3$ (61,000 acre-feet) of groundwater that remains stored in the basin [41]. In dry years, up to 16 new recovery wells, with an average pumping capacity of 0.01 km$^3$/yr (8100 acre-feet/yr), would provide a secure water supply to the city of San Francisco [42]. In other cases, economic incentives have been proven as a useful tool to promote in-lieu water use. For example, to promote in-lieu recharge within the Orange County Water District (OCWD), a financial incentive program was developed between 1977-2007. The OCWD in-lieu program paid the price difference between the more expensive imported water and the less expensive local groundwater to replace groundwater pumping with imported surface water, resulting in 1.1 km$^3$ (900,000 acre-feet) of net recharge over the next 30 years. On average, the in-lieu program in OCWD only contributed to 3% of total groundwater recharge, however, during wet years, in-lieu recharge reached a similar magnitude (e.g. 0.04 km$^3$ in 2011) as other water sources within the district (e.g. direct recharge with Santa Ana River, imported, or recycled water).

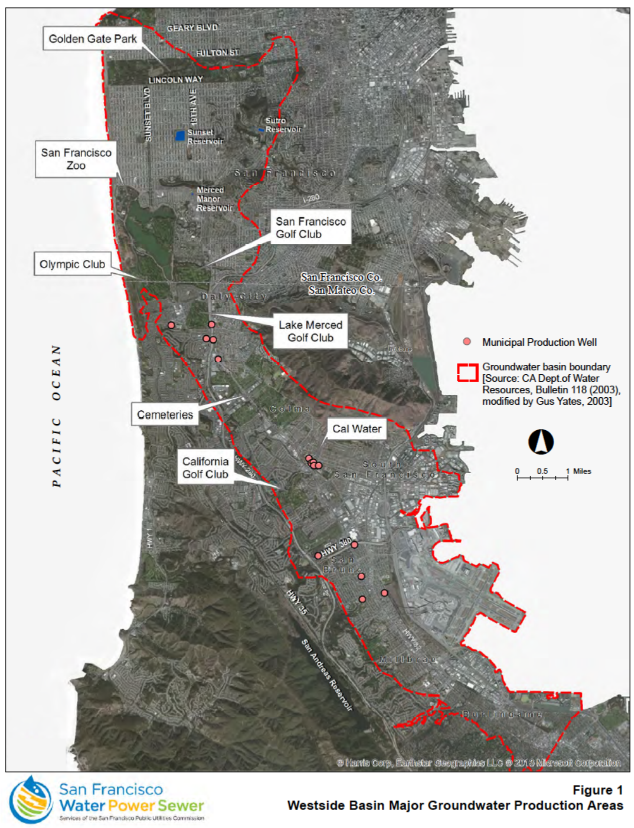


Figure 5: Westside basin of the San Francisco Public Utilities Commission (SFPUC) district area and locations of 16 recovery wells used by the SFPUC for water supply during drought years.

*2.1.2. Use of treated wastewater in centralized urban MAR*

Over the last several decades treated wastewater (also referred to as recycled water) has become an increasingly important water source for urban areas. One of the earliest treated wastewater reuse projects in the U.S. was created in Los Angeles County in 1929 to provide irrigation water for public parks. Since then, improvements in treatment technology have allowed use of recycled water to expand. Estimates within the last decade state that approximately 7-8% of total wastewater in the U.S. is reused [43]. Recycled water has the potential to provide a reliable water supply source for recharge, although water quality concerns exist related to potential pathogen presence and disinfection byproducts from chlorine treatment [44]. Research on pathogen presence in recharge projects has shown that bacterial pathogens have limited survival rates (T90 <3 d) in aquifers of sand or limestone, but enteric viruses such as the adenovirus have been found to survive much longer (T90 = >200 days) in the same conditions [45]. While Sidhu et al. [45] found persistence of viruses in aquifers, another study across the States of California, Arizona, and Colorado using natural treatment riverbank filtration and soil-aquifer treatment found that a 99% removal of adenovirus could be achieved within about 15 days residence time [46]. These differing results support the hypothesis that pathogen survival and attenuation in aquifers is influenced by site-specific geochemical factors, as well as the particular species of pathogen [45]. This is especially important in urban aquifers where limited space can result in short hydrogeologic travel times, as is the case for the Los Angeles Montebello Forebay MAR operation where infiltration basins lie within 150 m or less than 10 weeks travel time of groundwater supply wells, failing to meet California regulations from 2006 that require at least 150 m or 6 months of travel time for recharge facilities using recycled water (Table 1 ) [47]. MAR projects using recycled water require differing levels of pretreatment depending on the final intended use; in California for example, groundwater recharge regulations require advanced treatment including reverse osmosis and advanced oxidation [48]. In addition, California is one of only four U.S. states that has treatment regulations for groundwater recharge for non-potable uses, such as prevention of land subsidence, and one of only three U.S. states with regulations for indirect potable reuse, which includes recharge of recycled water for potable reuse [49].

Orange County's Groundwater Replenishment System (GWR System) in Southern California provides an example of MAR using high quality advanced treated wastewater. The GWR System was designed to produce advanced treated recycled water through a process involving microinfiltration, reverse osmosis, and advanced oxidation treatment with hydrogen peroxide and ultraviolet light exposure [50]. Because the purification process removes nearly all minerals from the water, lime is introduced to stabilize the pH of the final product. The treated water is then used to recharge seawater intrusion barriers as well as local infiltration basins. The final product from the treatment system has been found to remain within all state and federal drinking water standards, with a final total dissolved solids (TDS) concentration of approximately 45 mg/L, which is well below the typical TDS of imported surface water to the region [51]. While California's requirements for recycled water recharge are considered cautious from an international perspective [52], other governmental regulations may require less stringent treatment, depending on the application.

### *2.2. Decentralized MAR approaches*

As space and economic resources for large scale centralized infiltration projects have diminished over the last 100 years, regionally distributed or decentralized programs have become more attractive to urban planners [53, 54]. Decentralized projects focus on infiltrating smaller volumes of water, on the order of 10-100 $m^3$ per rain event, through small projects with a footprint of 10 $m^2$ to 1 ha [23, 55]. Recent studies on decentralized groundwater infiltration in urban settings have focused primarily on the implementation of Low Impact Development (LID), an approach piloted in Maryland, U.S., that is designed to mitigate the negative effects of urbanization (e.g. an increase in impervious surfaces) on surface runoff [56, 57, 58]. LID practices include pervious pavement, vegetated swales, bioretention basins, and small-scale infiltration basins [58, 59, 60]. In addition to the above mentioned LID practices, many urban areas in California and neighboring states such as Arizona use drywells, rainwater capture, reuse projects, and rooftop runoff infiltration to increase urban infiltration. The Los Angeles metropolitan area serves as a leader in California for LID planning and implementation. In 2010 the Los Angeles & San Gabriel Rivers Watershed Council conducted a modeling study to determine the amount of regional groundwater that could be augmented through decentralized stormwater management and groundwater recharge methods [61]. The 2015 urban water management plan of the Los Angeles Department of Water & Power, for example, estimated that about 0.04 to 0.08 $km^3$/yr of recharge could be captured through decentralized projects in addition to the existing incidental decentralized capture projects (0.04 $km^3$/yr) [23].

Under the umbrella of LID projects, bioretention systems use vegetation, such as shrubs or trees, in low-lying areas in the landscape to treat contaminated water through physical, chemical, and biological processes [58]. Vegetated swales or bioswales are similar to bioretention basins, however, they generally use grass instead of diverse vegetation and they have a shallower topographic profile and therefore smaller capacity to capture stormwater [60]. Bioretention basins and vegetated swales are often used in combination with other decentralized measures such as dry wells, cisterns, or infiltration basins [59]. They typically do not support capture of large volumes of stormwater because infiltration rates depend on local soil properties. However, they provide several benefits such as slowing stormwater runoff, removing pollutants, and settling out suspended solids. Studies on the pollutant removal efficacy of bioretention basins have shown significant reductions in heavy metals such as copper (43 - 97%), lead (70 - >95%), and zinc (64 - >95%) [62], and nutrients such as total nitrogen (31 - 69%) (Table 2) [63].

Table 2: Reported bioretention pollutant retention from various studies (modified from Table 1 from [58]).

| Location | TSS | NO3–N | NH3–N | TKN | TP | TN | ON | Cu | Pb | Zn |
|---|---|---|---|---|---|---|---|---|---|---|
| Connecticut | | | | | | | | | | |
| Haddam | – | 67 | 82 | 26 | 108 | 51 | 41 | – | – | – |
| Maryland | | | | | | | | | | |
| Greenbelt | – | 16 | – | 52 | 65 | 49 | – | 97 | >95 | >95 |
| Largo | – | 15 | – | 67 | 87 | 59 | – | 43 | 70 | 64 |
| New Hampshire | | | | | | | | | | |
| Durham | 96 | 27 | – | – | – | – | – | – | – | 99 |
| North Carolina | | | | | | | | | | |
| Greensboro | –170 | 75 | –1 | –5 | –240 | 40 | – | 99 | 81 | 98 |
| Chapel Hill | – | 13 | 86 | 45 | 65 | 40 | – | – | – | – |

Bioretention basins and vegetated swales tend to remove high levels of metals and nitrogen, while often having varied effects on other contaminants such as suspended solids, phosphorus, salts, and pathogens as a result of the organic matter or legacy pollutants contained in the basins [58]. Results from monitoring a bioretention basin in Los Angeles showed reductions in copper ( 33%), lead ( 60%), and total suspended solids ( 15%) [64], which agree with removals reported in other literature [58]. Infiltration and recharge of untreated stormwater could potentially have adverse effects on the receiving groundwater. However, Dallman and Spongberg [65] looked at stormwater infiltration sites in industrial, commercial, and residential areas in Los Angeles County, and found no increases in metals and fecal coliform concentrations in groundwater and no evident buildup of contaminant concentrations in soils, with the exception of a metal recycling plant, which saw slight increases in copper (8%) and zinc (8%) [65]. Collecting runoff from rooftops presents an additional decentralized water source for groundwater recharge in urban areas, which can be implemented without the need for significant infrastructure or retrofitting. A notable concern of using rooftop runoff for groundwater recharge, however, is water quality, since rooftop runoff can contain contaminants such as pathogens, metals, and other materials either leached from rooftop materials or deposited from airborne pollution. An investigation of rooftop runoff in rural New Zealand found the presence of lead, copper, zinc, and arsenic above national drinking water standards, as well as the presence of potential microbial pathogens such as *Salmonella*, *Aeromonas*, and *Cryptosporidium* [66]. In industrial or commercial areas, runoff from metal-roofed buildings may be a significant source of elevated metal concentrations in runoff. For old metal rooftops in acidic rainwater conditions, metal concentrations in runoff have been found as high as 2,230 $\mu$g/L for zinc and 1,510 $\mu$g/L for copper [67]. Rooftop runoff quality has been shown to be affected by roof material and rainwater quality [67], thus proper management is necessary to prevent contamination risks from this potential water source.

Recharge using deep infiltration techniques such as drywells (i.e. infiltration galleries) offers additional options for urban MAR portfolios. Drywells are wells drilled for the purpose of groundwater recharge, which stop short of the water table. The general design of a drywell including pretreatment is included in Figure 6. There is a perceived risk that drywells offer more direct passage of contaminants to groundwater aquifers, because they bypass the unsaturated zone and soil filtration processes [68]. Therefore, drywells are often combined with LID structures to provide pretreatment of the source water before infiltration [59]. In California, drywells have been implemented since the 1950s to augment agricultural groundwater sources [69]. Urban use, however, has only received promotion through demonstration projects since the late 1990s and local ordinances in the last 10 years [70, 71, 72]. Drywells are a common MAR practice in the neighboring state of Arizona, which has installed a high percentage of the total drywells present in the U.S. [68]. A study in Arizona examined four drywells receiving water from either residential, industrial, or commercial sites to test whether the drywells caused groundwater contamination [73]. The drywells were not found to be a major source of groundwater pollution for the study region, although some organic pollutants such as ethylbenzene and toluene were detected in drywell sediments [73]. A broader review of drywell effects on groundwater quality in the U.S. found that reported cases of groundwater contamination from drywells is often the result of contaminant spills in the vicinity of the drywells or inappropriate use of drywells, rather than deficiencies in the well construction itself [68]. Monitoring of groundwater quality up- and downgradient of two drywells near Elk Grove, CA revealed that the groundwater contained lower concentrations of some metals (aluminum and manganese) and higher concentrations of others (arsenic and chromium) compared to the infiltrated stormwater, which raised some concerns about desorption of metals present in the soil [74].

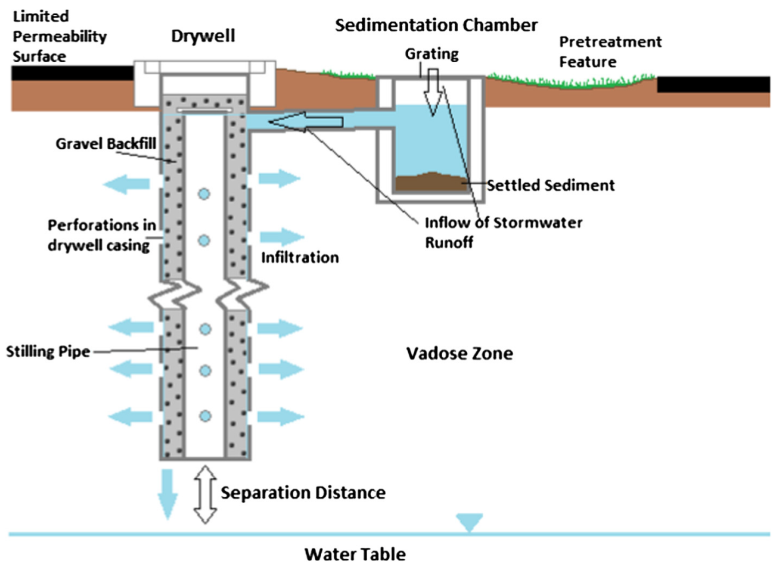


Figure 6: General design of a drywell, including pretreatment with a grass swale and sedimentation chamber [68].

Other decentralized MAR approaches are so-called capture and reuse or on-site direct use projects. Capture and reuse projects encompass a wide variety of water storage techniques (e.g. constructed aquifer storage and recovery systems, modular underground storage tanks, rain barrels) that are designed to capture precipitation, hold it for a period of time, and reuse the stored water or slowly release it over time for irrigation or groundwater recharge [60]. Often the rainwater storage systems consist of cisterns constructed above or below ground that can generally hold about 1 $m^3$ in household applications to 1000 $m^3$ in public applications such as parks. TreePeople in Los Angeles installed a 14 $m^3$ cistern at a typical house and a 416 $m^3$ cistern at a school as part of a demonstration project in 1998 and 2005, respectively [70]. The scale of each project leads to varying treatment needs for the captured rainwater: at the house, a first-flush system was installed to divert the low-quality initial runoff of each storm, while at the school, a swirl-concentrator was installed to provide sedimentation and removal of floating pollutants, and chlorination was added to disinfect the stored water. Capture and reuse projects using cisterns have become popular in recent years, however, alternative designs have been proposed, such as the use of constructed aquifer storage and recovery systems (also known as geostorage systems), which are preferred to capture runoff at sites with poor soil infiltration [75]. A modeling study conducted by Taylor et al. [75] compared the cost and benefits (e.g. runoff volume that could be captured, end use of water) of a geostorage system and a modular storage tank system for a 34 ha site in Riverside County. The capture and reuse project had the goal to retain the 85th percentile rainfall-runoff event (a common standard in urban water management in California and known as the water quality volume) on site. Both capture systems were modeled using the EPA (Environmental Protection Agency) model SWMM. The geostorage system was simulated as an open aquifer system allowing evaporation under pervious pavement while the below-grade modular tanks were simulated as closed conduit system. The results showed that a geostorage system with a capacity of 22,700 $m^3$ provided the more cost-effective solution, capturing 61% of the total rainfall-runoff volume, providing 38% of the property's irrigation needs and meeting the local water quality volume requirements (88% of the water quality events that occurred over the 17 year simulation period were captured) [75]. In contrast, the modular storage tanks could not meet the water quality volume requirements since it only captured 44% of the total runoff volume but instead it met 91% of the irrigation demand of the property. This study illustrates that stormwater runoff reduction goals can sometimes be at odds with water quality goals.

Fresno, California has successfully used decentralized infiltration basins to recharge groundwater since the 1970s. The city's recharge management includes more than 100 stormwater recharge basins infiltrating imported surface water from the Sierra Nevada Mountains as well as stormwater runoff from the city's industrial, residential, and commercial areas [56]. One of the recharge systems named Leaky Acres has been used to recharge water from the nearby Kings River since 1970. Over its first ten years of use, Leaky Acres achieved recharge rates of 12.1 cm/day and an average efficiency of 0.86, defined as the ratio of number of days of water availability to number of days of recharge [76]. An extensive study conducted by the USGS in 1986-1987 examined sediment, soil, and groundwater quality impacts from a recharge basin near Fresno, CA draining an urban industrial site [55]. While the study found a wide range of organic and inorganic compounds from urban runoff, these constituents were primarily trapped in the upper 4 cm of the basin's sediment. The shallow sediment concentrations of certain elements were much greater than background concentrations, particularly for zinc (3,800% above background levels), copper (2,500%), and lead (900%) [55]. Despite the high constituent loadings found in the sediments of the infiltration basin, the report concluded that there was no impairment to groundwater quality.

#### *2.2.1. Water quality considerations in decentralized urban MAR*

Water quality in stormwater runoff is highly variable, although highest pollutant loads are often observed during the first flush of the wet season, when pollutants accumulated on impervious surfaces over the dry season become mobilized in the first storm events of the wet season. This first flush phenomenon is often observed in urban areas of Mediterranean climates such as California that have distinctive wet and dry seasons [77]. In California, pollutant loads from the first part of the wet season have been found to be 1.2-2.0 times higher than loads near the end of the season [77]. Pollutants in urban stormwater reflect the variety of land use activities that occur in cities and include sediments and metals accumulated on roads, construction site runoff, organics such as animal wastes and decaying vegetation, pesticide and fertilizer runoff from landscaping, and trash [78]. On California highways, heavy metals such as copper, lead, and zinc have been identified as main pollutants, with average edge-of-pavement concentrations equaling 33.5 $\mu$g/L, 47.8 $\mu$g/L, and 187.1 $\mu$g/L, respectively [79]. Fecal contamination from the urban dog and cat population is a common problem in stormwater runoff that may even lead to human health impacts when contact with the polluted water occurs, as is the case with reuse of captured stormwater for landscaping [80]. Levels of fecal coliform bacteria have been found to exceed California state standards by as much as 500% in stormwater runoff draining southern California urban areas [81]. Consequently, groundwater contamination is a common concern when designing recharge projects using urban stormwater runoff.

## 3. Managed Aquifer Recharge in agricultural settings

### *3.1. Background*

In semi-arid regions with intensively irrigated agriculture, such as California, groundwater overdraft is a pervasive problem that threatens the long term sustainability of the agricultural industry. Over the past 100 years a combination of factors including changing climate, changing land use (from annual to more water intensive perennial tree and vine crops), widespread adoption of high-efficiency irrigation systems (e.g. sprinkler and drip systems), and the conversion of rangeland into cropland have led to increasing demand in surface and groundwater resources and groundwater depletion in the Central Valley of California since the 1960s [13, 82, 83, 54]. Bringing groundwater basins back into sustainability necessitates capitalizing on excess surface water during wet years to actively recharge groundwater. Agricultural managed aquifer recharge (ag-MAR) is a water management approach whereby excess surface water is diverted onto agricultural fields to recharge the underlying aquifer for later use during times of drought. California has over 7 million ha of agricultural land with an extensive water conveyance delivery system that could be used to transfer excess water to farm fields [11, 84, 85]. While dedicated infiltration basins or injection wells to capture excess surface water are expensive to build, leveraging agricultural lands for on-farm recharge presents an opportunity for MAR at minimal cost [84, 86]. However, feasibility of ag-MAR depends on many interrelated and site-specific factors such as water availability for recharge, infrastructure to convey surface or source waters to fields, associated economic costs, water laws and permits, the physical and biochemical properties of the soil, the crop's tolerance to water inundation, the capacity of the aquifer to store and recover the recharged water, and the effect of the practice on groundwater quality (Figures 7, 8).

Figure 7: Application of storm water on an almond orchard for groundwater recharge.

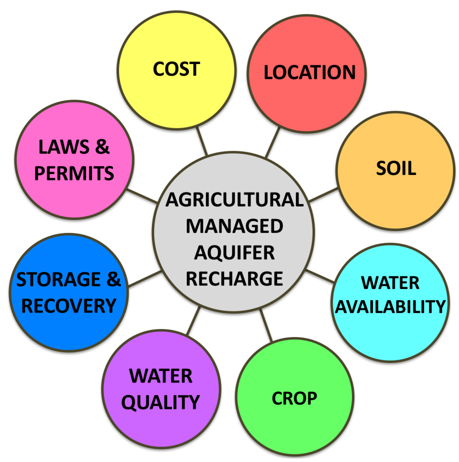


Figure 8: Factors influencing the feasibility of ag-MAR implementation.

### *3.2. Feasibility*

#### *3.2.1. Water Availability*

Although the Sustainable Groundwater Management Act passed in 2014 by the California legislature aims to bring critically overdrafted groundwater basins back into balance (i.e. sustainable yield) by 2040, water managers question what alternative water resources will be made available to meet statewide water demand while reducing groundwater depletion. Although MAR can be conducted with any available water (e.g. stormwater, recycled water, desalination, surface water), most water sources (e.g. recycled water, desalination) do not provide the water volumes needed to sustain agricultural water demand within the state [87, 11]. However, flood flows (i.e. high magnitude flows) or flows that occur during large storm events (e.g. atmospheric rivers [12]) likely represent the most accessible and largest source of water available for future expansion of groundwater recharge [82, 10, 11]. High-magnitude flows (HMFs) are an appealing source because agricultural demand for surface water during the winter months, during which the majority of these events occur, is relatively low. Research has found that mean HMFs (i.e. flows above the 90th percentile) may provide an average of 3.2 $km^3$ of surface water in years when HMFs occur [11]. The frequency at which HMFs occur in different parts of California's Central Valley include 7 out of 10 years in the Sacramento River basin, 4.7 out of 10 years in the San Joaquin River basin, and 2-3 out of 10 years in the Tulare Lake basin [11]. Recent groundwater overdraft estimates by the California Department of Water Resources range from 0.6 - 3.5 $km^3/yr$, meaning that utilization of these high magnitude flows could play a significant role in offsetting groundwater overdraft as a result of extensive managed aquifer recharge projects (Figure 9) [11, 54]. It is important to consider the limitations of utilizing surface water resources for groundwater recharge projects, including post-diversion environmental in-stream flow regulations and the diversion capacity of infrastructure [88].

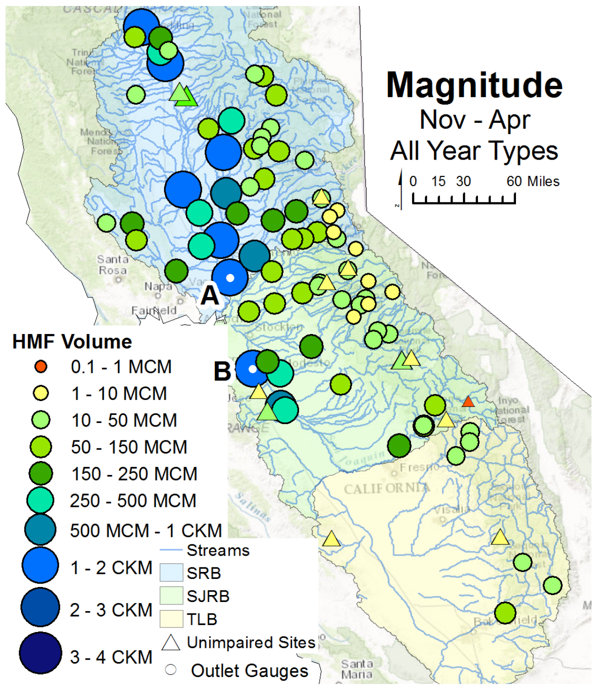


Figure 9: Average volume estimates of high-magnitude flow (HMF) occurrence (flow >90th percentile) between November and April over the full period of record for 93 stream gauges located within the Central Valley watershed. A and B denote the locations of the two outlet gauges. MCM and CKM stand for million $m^3$ and $km^3$, respectively.

### *3.3. Infrastructure*

It is important to acknowledge that the existing water conveyance structure may be unsuitable to transport high magnitude flood flows to recharge areas [89]. Bachand et al. [84] found field preparation to allow for infiltration on existing farmland to be relatively rapid and inexpensive when compared to large-scale surface storage or even dedicated infiltration basins, however, the capacity of existing conveyance equipment (e.g. pipes and pumps) can limit flood flow applications (Figure 10). In fact, the California Department of Water Resources identifies infrastructure transport capacity as a limiting factor for groundwater banking projects [88]. This limiting factor may be overcome with further implementation of the Sustainable Groundwater Management Act, which promotes more groundwater recharge within the state, and increased availability of public funds such as the California Water Quality, Supply and Infrastructure Improvement Act of 2014, providing about $2.7 billion for the improvement of water storage and infrastructure [11].

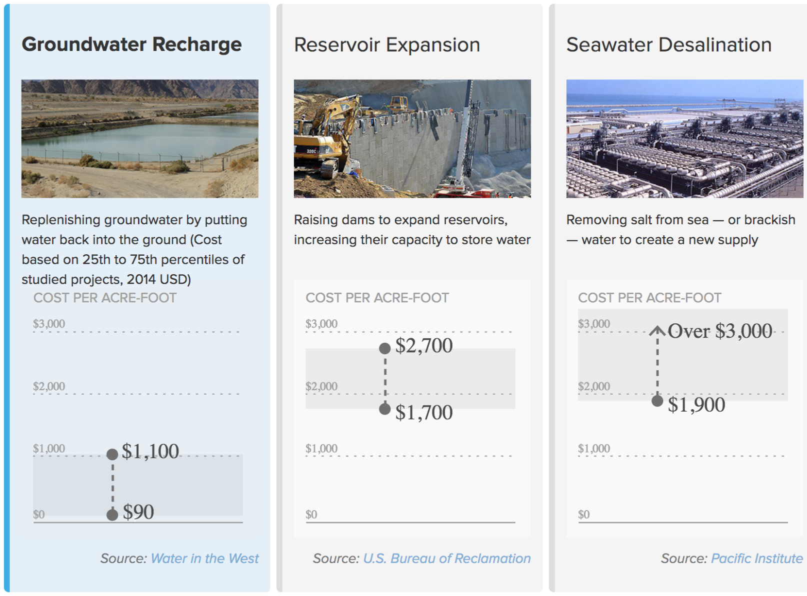


Figure 10: Cost comparison of water projects in California [90].

### *3.3.1. Soil suitability*

Although agricultural fields present a promising opportunity for managed aquifer recharge, the suitability of each site must be evaluated on a number of factors. Recent soil suitability research for agricultural groundwater banking used national soil survey data and identified five factors that are critical to successful on-farm recharge when selecting locations for ag-MAR across agricultural land in California [89]. The Soil Agricultural Groundwater Banking Index (SAGBI) considers deep percolation, root zone residence time, topography, chemical limitations, and soil surface condition.

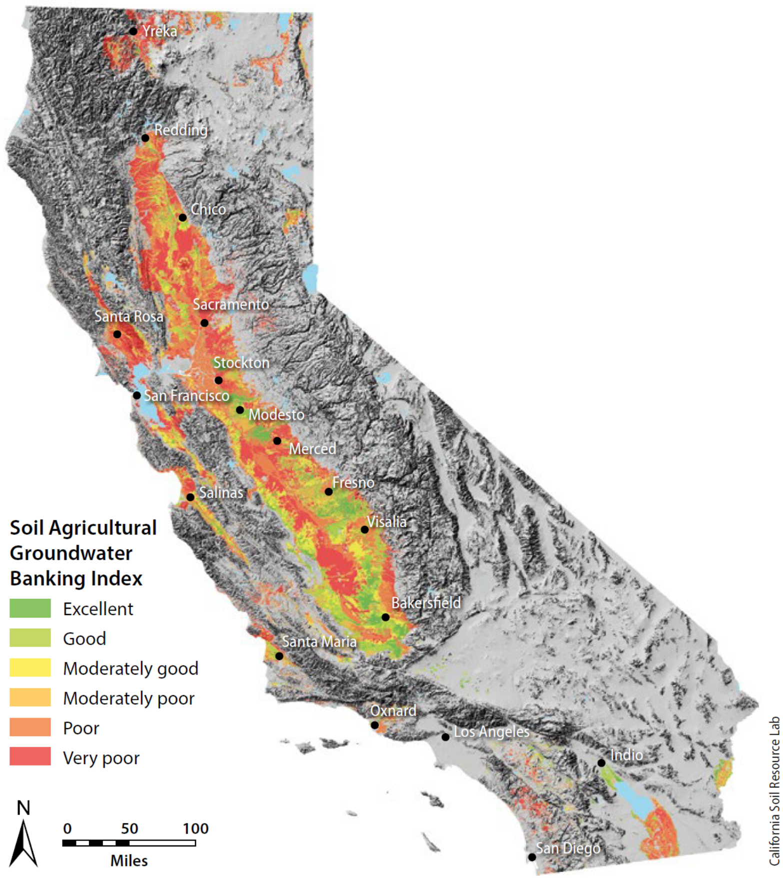


Figure 11: Soil Agricultural Groundwater Banking Index (SAGBI). Ratings of California soils based on their suitability for ag-MAR (Figure 5 from [89]).

The *deep percolation* factor captures the ability of a site to transmit water through the soil profile (top 1.5m) and is determined by the soil horizon with the lowest saturated hydraulic conductivity ($K_{sat}$). This factor becomes important when utilizing large amounts of water such as flood flows for ag-MAR, which are only available for sporadic but short periods of time during winter storm and spring snowmelt events [89]. *Root zone residence time* is a measure of the duration of saturated or near saturated conditions in the soil profile and derived from the harmonic mean of $K_{sat}$ of all horizons in the soil profile, soil drainage class and shrink-swell properties. Near-saturated conditions have the potential to negatively impact the root health of crops, reduce yields or cause undesirable anoxic conditions in the root zone. Both the deep percolation factor and the root zone residence time are often controlled by the presence of less permeable clay layers. A confining or semi-confining clay layer with low hydraulic conductivity can impede the percolation of water towards the groundwater table. Deep percolation is a consideration of how much water will actually reach the groundwater table, while root zone residence time considers how crop health will be affected by prolonged ponding conditions associated with flooding events.

SAGBI's *chemical limitation factor* considers the salinity and leaching potential of a site's soil. In California, salts from the marine sediments along the coastal range, as well as irrigation management practices, have led to the accumulation of salts in the soil, which may pose a contamination threat to groundwater resources. Further research is ongoing concerning other chemical contamination factors in agricultural fields, including nitrate and pesticide transport processes. The last factors considered are the *topography* (slope of the field site) and the *soil's susceptibility to physical change*, such as erosion or compaction [89]. SAGBI weighs the five factors according to their relative importance for ag-MAR, with deep percolation and root zone residence time ranked as the most important ones. In many parts of the Central Valley of California, low permeability layers (often clay-rich or consisting of precipitated carbonates) lie below the root zone, impeding deep percolation and root zone residence time. Some of these restricting features can be temporarily alleviated by deep tillage practices, using machinery that plough the soil to a depth of 0.5-0.6 m, prior to planting. Deep tillage can result in significant increases in the amount of land suitable for ag-MAR [89]. In California, about 2.03 million ha of agricultural land, mainly found on the alluvial fans on the east side of the Central Valley (Figure 11), were rated as excellent, good, and moderately good for groundwater banking, or 28% of the agricultural land throughout the state. However, when considering land that has been deep tilled, the area suitable for groundwater banking increased to 2.25 million ha, or 31% of the total agricultural land area, and could potentially be used to bank up to 1.5 $km^3$ of water per day on grape, alfalfa, or fallowed land [89]. This preliminary estimate assumes that the infrastructure to deliver water to all available agricultural land is in place and that 0.3 m per day of water is available and infiltrated. However, field trials assessing the infiltration rates of varying soils are needed.

#### *3.3.2. Crop tolerance*

A concern for implementing ag-MAR on a large scale is the potential adverse effect that ag-MAR could have on crop health and yields, which is largely dependent on the crop's ability to tolerate flooding or saturated conditions in the root zone, and the local soil properties. The effects of prolonged flooding on root health, specifically anoxic conditions in the root zone must be evaluated. A decrease in root health may result in lower nutrient uptake, impacting annual average yields. Recently, repeated experimental flooding events for groundwater recharge on test plots of alfalfa have shown minimal yield loss when water was applied during the winter months (e.g. crop dormancy) on highly permeable soils [85]. Although reduced oxygen conditions were observed in the root zone during flooding events, soils return to pre-flooding conditions within several days after water applications for recharge ceased [85]. Other research studies have corroborated the results, finding no significant yield decreases in pistachio or alfalfa orchards, and no observable root damage to pistachio trees or wine grapes [84]. To avoid injury of perennial crops on less suitable soils (e.g. soils with a SAGBI rating of moderately good or less) cropland could be flooded when it is fallow, reducing the risk of root damage or yield decrease. So far, ag-MAR has not had any significantly negative effects on root health of almonds or crop yields of alfalfa in soils with high percolation rates [85]. In order to ensure this, it may be advisable to implement ag-MAR on fields with relatively low root zone residence times (i.e. prioritize highly rated soils from the SAGBI index).

#### *3.3.3. Cost*

During times of drought, when surface water allocations are reduced, farmers turn to a combination of groundwater and land fallowing to meet irrigation needs. However, long-term groundwater depletion threatens the groundwater's capacity to serve as a buffer during times of drought. During the 2012-2016 drought, even with a five-fold increase in groundwater pumping, an estimated 228,242 ha were fallowed in California, with farm revenue losses of  \$1.8 billion [91, 92, 8]. Costs of groundwater pumping are increasing as water tables are falling, as indicated by an average increase of 39% in groundwater pumping costs during the 2012-2016 drought [91, 92, 93, 94]. As farmers in California shift towards high-value, perennial cropping systems, which harden water demand, groundwater reserves will become increasingly important during times of decreased surface water availability because these systems cannot be temporarily idled. Thus, economic incentives for farmer participation in ag-MAR are needed.

In comparison to other water storage and supply strategies such as seawater desalination or surface water storage, ag-MAR has emerged as a more economical method. Costs for ag-MAR are estimated to be about \$0.03 per $m^3$ compared to \$1.54 to \$2.43 per $m^3$ for seawater desalination, \$1.38 to \$2.27 per $m^3$ for large-scale surface water storage, and \$0.07 to \$0.89 per $m^3$ for dedicated recharge basins (Figure 10) [84, 17, 95]. Costs associated with ag-MAR include labor, land preparation, fuel, and farm-scale infrastructure improvements [86]. Furthermore, if excess surface water is used for in-lieu recharge (using surplus surface water to irrigate rather than groundwater), the costs of pumping groundwater for irrigation can be avoided or partially offset depending on how much of the crop's demand is met with in-lieu recharge. Finally, if flood flows are diverted, costs associated with downstream flood damage can also be mitigated. Since 1983, there have been three years (1983, 1995, 1997) where flood damage has occurred along the Kings and San Joaquin Rivers causing \$1.2 billion in damage [96]. Bachand et al. (2011) [96] estimated that if approximately 14 $m^3/s$ of water had been diverted from the Kings River during those three years and applied to the entire study area (404 ha), a total of 1.23 $km^3$ would have been diverted and the entire costs from flood damage could have been avoided.

### *3.3.4. Impact on water quality*

Despite the increased interest in ag-MAR in California, the potential for groundwater contamination with nitrate, salts and pesticides as a result of agricultural flooding must be assessed before widespread implementation occurs. Nitrate levels in public supply wells in California are already increasing at an average rate of 2.5 mg/L per decade in large portions of the Central Valley, and many wells exceed the maximum contaminant level (45 mg/L) set by the California Department of Public Health [97]. Agricultural groundwater banking has the potential to flush contaminants, including nitrate, out of the root zone towards the groundwater table. The time it takes for nitrate to be transported from the land surface to the groundwater table can range anywhere from a sub-annual to decadal scale, depending on factors such as depth to groundwater, hydraulic conductivity of the soils and sediments of the underlying vadose zone, and the hydrologic regime (e.g. annual precipitation, irrigation efficiency) of the region [98, 99, 100]. Build-up of nitrate in the soil and unsaturated zone above the groundwater table occurs under agricultural lands as a result of over-fertilization and inefficient irrigation practices. The use of NPK (nitrogen, phosphorus, potassium) fertilizer in California's agricultural production systems is ubiquitous and may continue to increase in the future as population growth demand greater food and agricultural production. However, research shows that crops only use up to $\sim$50% of the applied nitrogen fertilizer [101]. This low nitrogen use efficiency leaves nitrate in the root zone, where it can undergo denitrification processes and degas into the atmosphere as nitrous oxide ($N_2O$), nitrogen gas ($N_2$) or nitric oxide (NO), or leach under inefficient irrigation practices deeper into the vadose zone, towards the groundwater table (Figure 12) [102].

Nitrate transport and nitrate contamination of groundwater have been an important research topic in recent years, as the effects of long-term agricultural production on groundwater resources are beginning to be realized ([98, 100, 103]. Studies in the Central Valley of California have looked into the effects of nitrate leaching from almond orchards as a function of fertilization and irrigation timing and practices [100]. The authors found that nitrate leaching was minimized when fertilizer applications occurred at the end of irrigation events, and maximized when flooding events occurred pre-bloom or post-harvest [100].

In California, irrigated agriculture is identified as the greatest source of nitrate contamination of groundwater in the southern parts of the Central Valley [104]. For example, research using a modified version of the University of California's Groundwater Pollution Hazard Index has been developed using characteristic soil parameters, types of irrigation systems in place (e.g. sprinkler or drip) and nitrogen use efficiencies for different crops to identify high risk areas for nitrate leaching due to agricultural practices [105]. Ag-MAR uses amounts of water orders of magnitude greater than typical sprinkler or drip irrigation systems, potentially decreasing the transit time of nitrate transport through the vadose zone and allowing mobilization of nitrate previously bypassed by preferential flow [98]. Although implementing ag-MAR will likely result in an initial downward pulse of nitrate from the root zone, it is proposed that subsequent flooding events on a dedicated field site may result in a dilution effect [96]. This is where the initial nitrate pulse is offset by higher quality water traveling down the same pathways to recharge groundwater. The amount required for this effect to occur will depend on the amount of nitrate present in the unsaturated zone and porous media characteristics such as hydraulic conductivity, porosity, and the degree to which preferential flow occurs during flooding events.

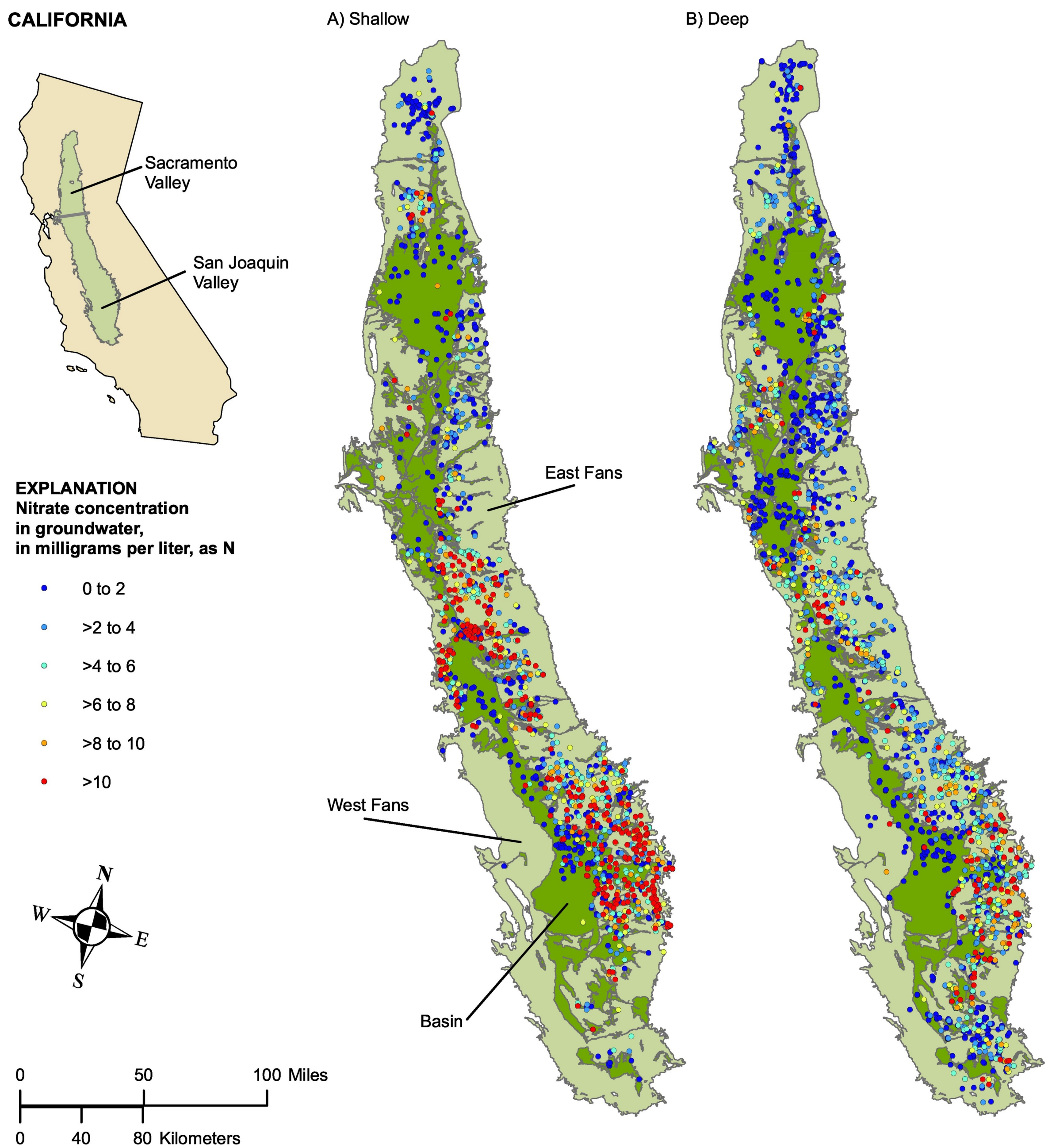


Figure 12: Modeled Nitrate Concentrations for Central Valley of California (EPA Water Standard for $NO_3$-N is 10 mg/L). Dark green shading indicates the central basin while light green shading indicates the western and eastern alluvial fans (Figure 1 from [102]).

### *3.3.5. Ag-MAR Modeling*

To the best of our knowledge, few studies have been conducted in relation to modeling ag-MAR. However, Niswonger et al. ([106]) conducted a comprehensive modeling study to evaluate and constrain the regional and long-term benefits or consequences of ag-MAR for both groundwater and surface water sustainability. The study coupled MODSIM, a linked-network optimization and operations/planning model that determines surface water diversions and reservoir releases within the constraints of the overarching water laws, operations, and demands, with MODFLOW-NWT, a distributed hydrologic model that simulates groundwater flow, surface water-groundwater interactions, and unsaturated flow. The modeling study focused on the Carson Valley of California and Nevada, a semiarid agricultural basin, with a two-tiered water priority rights system that includes aminimum in-stream requirement, and three varying aquifer hydraulic conductivity values ($K_h$) of $K_h$= 2, 4, and 8 m per day [106]. A more generalized physiography of the valley was employed to create a simplified model that can be applied to other semiarid settings. Over a 24 year period, between 1990 and 2014, seven years had enough excess surface water to implement ag-MAR. Modeling results show an increase in total annual volumetric recharge of 0.23 $km^3$ (12%), 0.18 $km^3$ (10%), and 0.17 $km^3$ (9%) for the $K_h$ values of 2, 4 and 8 m/day, respectively. Furthermore, groundwater levels increased on average by as much as 7 m with increases in storage being the greatest in areas where groundwater pumping was most severe. Consecutive years of ag-MAR provided the greatest increases in groundwater storage, with levels 1.5–2.5 m higher for six years after recharge water application compared to modeled scenarios without ag-MAR. A single year of ag-MAR provided three years of sustained elevated groundwater levels of 2.5 m across $K_h$ values, even during subsequent drought years. Lower $K_h$ values had more significant sustained groundwater storage increases compared to higher $K_h$ values due to lower groundwater discharge rates, however, lower conductivity aquifers were more negatively impacted by groundwater overdraft in times of drought due to the increased storage capacity.

Water flow and transport of constituents are highly influenced by the hydrogeology of the vadose zone [98], thus, modeling exercises are limited by the knowledge and characterization of the underlying stratigraphy. To date, point measurements have been used to describe the vadose zone, with limited ability to capture the variability. New methods for describing the vadose zone include remote sensing methods such as Interferometric Synthetic Aperture Radar (InSAR) [107], and geophysical imaging techniques such as Electric Resistivity Tomography (ERT) [108, 109]. These non-intrusive methods are able to characterize, with a considerable amount of detail, the textural variability in the subsurface across large scales. These advances in characterizing the vadose zone will further our understanding of water flow and constituent transport to the underlying aquifers under normal irrigation practices and ag-Mar.

#### *3.3.6. Ag-MAR case study*

A case study in the King's River Basin examined the infiltration rates of floodwater diverted from the river onto an adjacent 405 hectare ag-MAR test field to estimate the amount of land needed to capture the available flood flows [84, 86]. Like much of California's Central Valley, the Kings River Basin is characterized by an annual overdraft of 0.20 $km^3$ and groundwater levels 60 m below the land surface. Flood flows from the King's River ranged from 14 to 160 $m^3/s$ over the studied 42 year period and exceeded the flood capacity of the Kings River channel on a seven year recurrence interval. Bachand et al. [84, 86] conducted ag-MAR on three cropping systems (grapes, alfalfa and pistachio) and fallow land (prior to spring row crop planting) on soils that ranged from sandy loams to loamy sands of which most were considered to have limited infiltration rates. Flows diverted in this study ranged from 0.06 to 0.6 $m^3/s$, with 3.8 x $10^6$ $m^3$ of water diverted. Infiltration rates ranged from 6.8 cm/d on sandy loams to 40 cm/d on loamy, coarse sands, with a mean of 10.7 cm/d. Total water applied in this case study ranged from 0.5 m to 3 m reaching depths of 3 to 36 m, with higher volumes positively correlating to the number of days flooded. The study found that 1.6 to 4 ha are needed to capture 0.03 $m^3/s$ of diverted water [84]. Although soil surveys indicated these sites to be of lower infiltration potential, soil preparation including deep tillage of the underlying confining layer, allowed for higher infiltration rates. Thus, while soil survey is helpful in the initial targeting of potential sites for recharge, site specific anomalies and soil management practices should be taken into consideration.

#### *3.3.7. Inefficient irrigation and canal seepage*

Pumping groundwater for irrigation represents a major discharge component of the water budget of an aquifer. However, inefficiencies in irrigation lead to losses of water below the root zone which, in turn, contribute to groundwater recharge [110]. In arid agricultural regions, percolation of excess irrigation water (water applied in excess of crop demand) can contribute more to the recharge of underlying aquifers than for example mountain-block recharge, with one study finding 0.04 to 0.08 $km^3/yr$ of groundwater recharge from excess irrigation water and only 0.002 $km^3/yr$ of recharge from mountain-block recharge [111]. Regional irrigation efficiencies averaged over a 22 year period (1984–2009), are 70% of crop demand with 30% recharging underlying aquifers, which is similar to the irrigation efficiency range of 40 to 80% given for gravity fed systems in the Encyclopedia of Water science [10, 112, 113]. Since 2000, many California farmers have switched from flood irrigation systems to high-efficiency irrigation technologies (e.g. pressurized micro-sprinkler and drip systems), which generally have efficiencies ranging from 70 to 95% [112]. While the high-efficiency irrigation practices seem to have a positive effect on surface water reservoirs (of up to 4.5 m in lake stage gains), evidence is mounting that high-efficiency systems can reduce the amount of excess water leached below the root system and therefore decrease groundwater recharge [114, 115, 110, 116].

In the Central Valley of California, 50% of crops are now irrigated with micro-irrigation systems as opposed to flood irrigated systems [117]. It is believed that increased irrigation efficiency (the ratio of water used by plant evapotranspiration to water diverted from the river or canal system) leads to water savings. However, an increase in irrigation efficiency has been shown to increase total water use by allowing for more intensive use of the irrigation water (increasing yields per hectare as well as water use per hectare) and expansion of irrigated farmland [117, 116, 118]. In a case study in the arid Southwest, Ward and Velazquez [116] found that by increasing drip irrigation subsidies from 0 to 100% of the capital, total water applied to agricultural fields decreased by 0.05 $km^3$ and groundwater pumping decreased by 0.04 $km^3$, however, groundwater recharge was reduced by 0.03 $km^3$ and total water use increased by 0.04 $km^3$. This result is attributed to drip irrigation causing higher total crop evapotranspiration and higher crop yields and less excess irrigation water leaching below the root zone to groundwater. Furthermore, water savings can be used to expand irrigation area of a farm operation or applied to more water-intensive crops, and therefore less of the water contributes to groundwater recharge [117]. The switch to high-efficiency irrigation systems also has the undesirable result that more farmers use only groundwater for drip/micro irrigation (because of the better water quality) even at times when surface irrigation water is available [114], leading to increased groundwater use and depletion. Based on a survey of 21 water districts in California, Burt and Monte [119] found that the main factor for the use of groundwater for drip/micro irrigation was the lack of flexible water delivery service to fields.

Other sources of groundwater recharge in agricultural areas include leaky surface water conveyance systems (e.g. unlined canals, ditches, leaky pipelines). Carrol et al. ([111]) found that surface water delivery canals can lose on average 20% of the diversion water to groundwater via leakage and that in wet years, groundwater recharge from canal leakage can account for 33% of groundwater inflows. This study estimated 0.03 to 0.05 $km^3$/yr of groundwater recharge via canal leakage. In some areas of California, water managers intentionally release surface water from reservoirs into canals to recharge groundwater [120]. However, canals that are constructed over highly permeable soils are usually lined with concrete to reduce seepage and increase lateral surface water conveyance and therefore are not sources of groundwater recharge [120].

## 4. Managed Aquifer Recharge in coastal areas

### *4.1. Coastal Managed Aquifer Recharge in California: Overview*

Managed aquifer recharge in California's coastal regions differs from agricultural and urban MAR in that it has the primary goal of preventing seawater intrusion while also enhancing groundwater storage, improving water quality, preventing subsidence, or protecting groundwater-dependent ecosystems. Seawater intrusion was recognized in the early 1900s in the Mission Valley of San Diego (1906), the West Basin of Los Angeles County (1912), Orange County (1925), the Pajaro Valley of Santa Cruz and Monterey Counties (early 1940s), and Ventura County (1951) [121] (Figure 13). Efforts to locally or regionally raise groundwater levels and slow or halt seawater intrusion have relied principally on injection wells (also called barrier wells) [122, 123, 124, 125] and infiltration basins [126, 127, 128, 129, 15, 130, 131] (Figure 13).

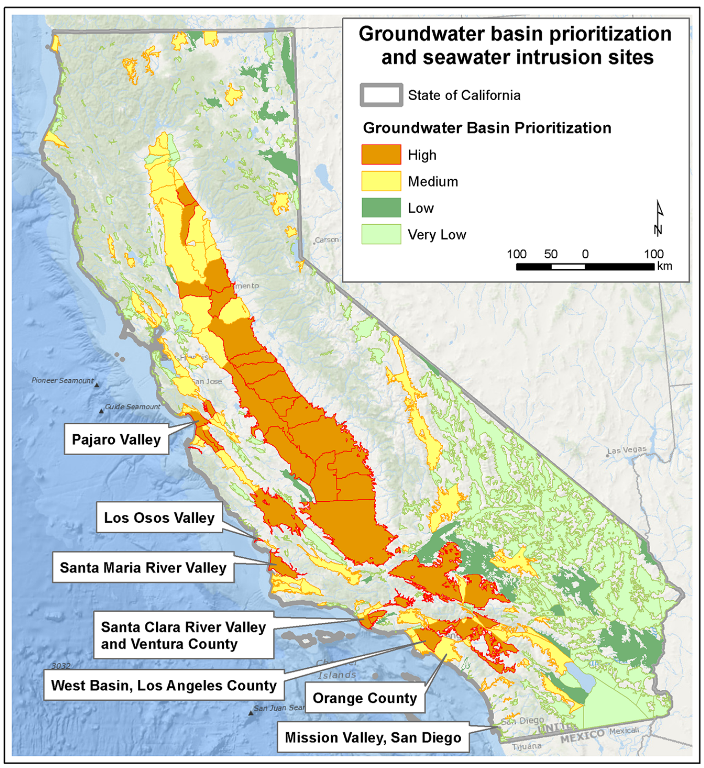


Figure 13: Seawater intrusion and basin prioritization of groundwater basins in California. Basins with high or medium priority account for approximately 96 percent of groundwater use in California and 88 percent of the state's population.

### 4.2. Injection wells

Injection wells are used to place fluids underground into porous geologic formations [132]. In the context of MAR, they recharge water directly into an aquifer through abandoned wells [121] or wells constructed specifically for that purpose [122, 123, 124]. Seawater intrusion was a significant problem in nine California groundwater basins by 1958, with Los Angeles County's West Coast Basin and the Coastal Plain of the Orange County Groundwater Basin being the most severely affected [121]. Hence, these areas were some of the first basins to utilize injection wells in California [124]. Test injections of freshwater in an abandoned well were conducted at Manhattan Beach, Los Angeles County in 1950 [121], a test barrier was completed in 1953, and the West Coast Basin Seawater Barrier and the Dominguez Gap Barrier were completed in 1969 and 1971, respectively [124]. The mean annual recharge from the Los Angeles County injection wells is 0.04 $km^3$/yr (35,000 acre-feet/yr), and particle tracking analysis using the USGS MODFLOW model has shown that most of the injected water moves inland at a speed of about 800 m per decade [123]. Furthermore, the model shows that while seawater intrusion has been halted along the majority of the coastline, it continues in some areas despite the injection well barriers, especially near the Dominguez Gap Barrier in Long Beach, CA [123]. It has been suggested that in-lieu delivery of surface water to reduce groundwater pumping would be more cost-effective than injection of surface water in this area, as injected water is more than three times the price of in-lieu surface water, largely due to pumping costs and the requirement that the water supply for injection wells be uninterrupted [123]. Source water for these projects shifted from Colorado River water and water from the California State Water Project to blending of these sources with recycled water beginning in 1995 [122, 124]. Source water is a particularly important consideration for injection wells, as unlike some other types of MAR (e.g. infiltration basins, bank infiltration), there is little natural filtration to remove sediment or contaminants. Source water is typically treated to drinking water quality standards (i.e. tertiary treatment) prior to injection, regardless of whether surface water or recycled water is used [133]; however, for recycled water advanced treatment (beyond tertiary treatment) is required, involving reverse osmosis and oxidation processes [134]. The West Coast Basin Seawater Barrier, Dominguez Gap Barrier, and Alamitos Gap Barrier (a joint project between Los Angeles County and Orange County) all use source water that has received advanced treatment [122] (Figure 14). While this water treatment largely eliminates the potential for biological and chemical contamination of drinking water, it may be insufficient to maintain the performance of injection wells due to clogging resulting from chemical precipitation caused by geochemical incompatibility of the source water and the groundwater [133]. Pumping water from an injection well daily for short periods of time can be an effective strategy to mitigate clogging issues [133].

Injection wells have a major advantage over other forms of MAR, in that they offer more flexibility in determining appropriate locations (since they have a very small footprint compared to infiltration basins); this allows injection wells to be sited where they will create the most effective barrier against seawater intrusion. The exception to this flexibility is the California Department of Public Health requirements that mandate injection wells using recycled water be situated far enough from production wells to provide a minimum 2-month residence time [135]. In addition, to the injection well projects described above, Orange County Water District also maintains the Talbert Seawater Intrusion Barrier using 100% recycled water from the Groundwater Replenishment System, an advanced water purification facility designed to produce about 3800 $m^3$ per day [125, 136]. According to the U.S. Environmental Protection Agency, there were already 308 documented seawater intrusion barrier injection wells in California by 1999, and the number has continued to grow since [132]. The projects discussed above utilize at least 327 injection wells combined [124, 125].

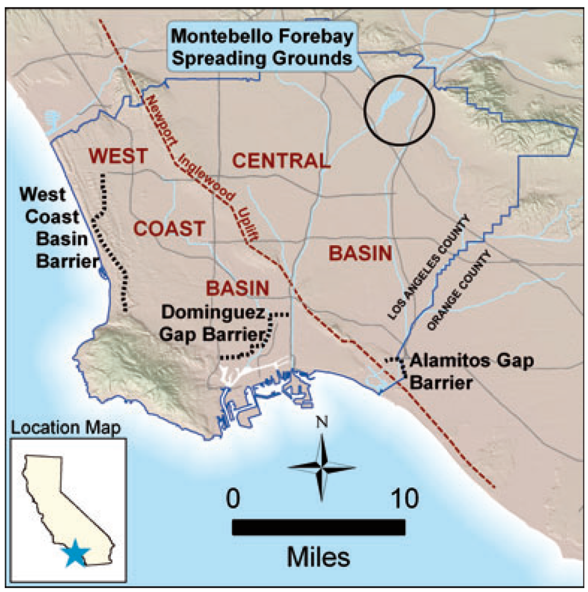


Figure 14: Location of injection well barriers (black dotted lines) for seawater intrusion control in Los Angeles County.

### *4.3. Infiltration basins*

Although injection wells have proven successful in managing seawater intrusion, traditional infiltration or surface water spreading basins were likely the first form of

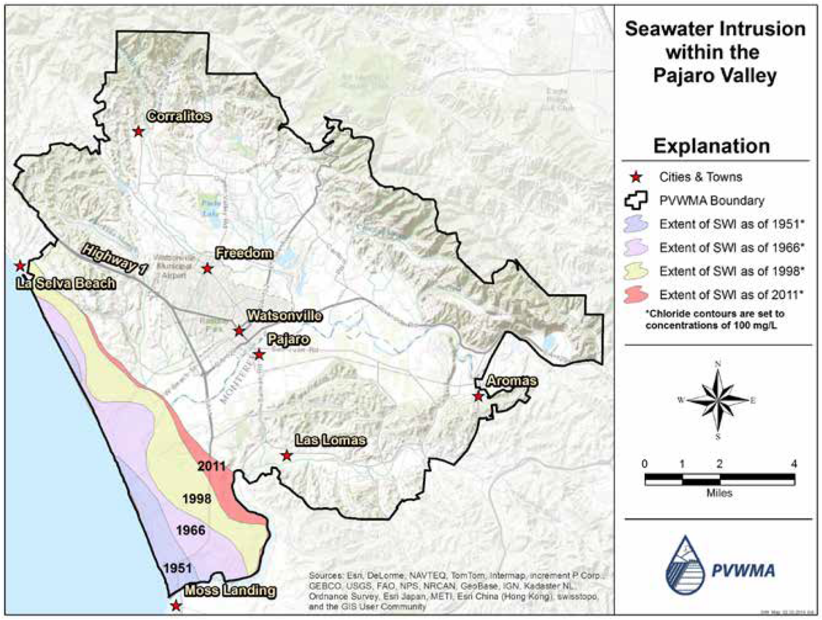


Figure 15: Seawater Intrusion within the Pajaro Valley, California (Figure ES-2 from [138])

MAR practiced in California. Infiltration basins are still an important tool to raise groundwater levels and combat seawater intrusion. Infiltration basins have been used since 1917 to recharge groundwater in Los Angeles County, though the injection wells mentioned above have become the principal defense against seawater intrusion since their installation in the 1960s and 1970s [137]. However, other areas experiencing seawater intrusion, like the Oxnard Plain in Ventura County and the Pajaro Valley in Santa Cruz County (Figure 15), do not have injection well barriers and rely on infiltration basins to raise groundwater levels and reduce or eliminate seawater intrusion [126, 127, 128, 129, 138, 15, 130].

Infiltration basins differ significantly from injections wells in the factors that must be considered to ensure maximum benefits. Site selection must consider the soil infiltration capacity, slope, connection to the underlying aquifer, land use, vadose zone thickness, and aquifer storage, not to mention the potential for conveyance of source water and myriad legal and political issues [15]. Site selection has been greatly aided by GIS tools, such as those used to identify suitable sites for infiltration basins in Santa Cruz County [15], which parallels similar efforts in the agricultural sector discussed earlier in this chapter. The appropriate scale for an infiltration basin depends on the source water availability, the extent of the project goals, and the financial resources available to a project. The scale of projects and size of infiltration basins vary widely: for example, infiltration basins supplied by distributed stormwater collection (DSC) may range in size between 0.4 – 4 ha with a catchment area between 40 – 400 ha [130]. In contrast, centralized infiltration basins supplied by developed surface water may be much larger, like the El Rio spreading grounds in Ventura County, which covers approximately 40 ha [126]. A DSC-supplied 1.7 ha infiltration basin in Santa Cruz County infiltrated 8.8 x $10^4$ $m^3$/yr on average over six years (Figure 16), while the centralized 40 ha El Rio spreading grounds infiltrated an average of 4.0 x $10^7$ $m^3$/yr during the 1990s [126, 130].

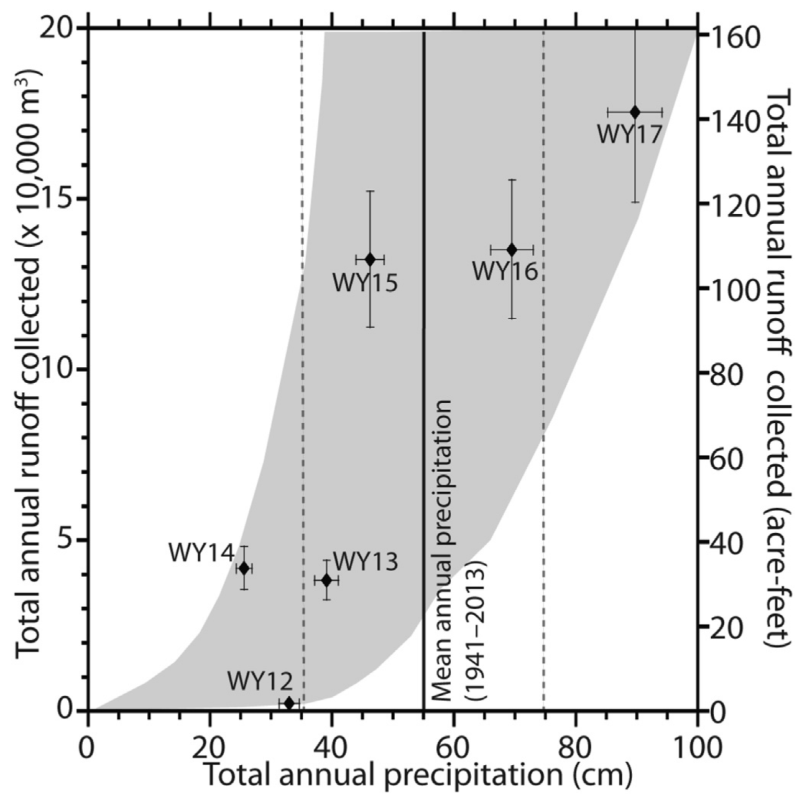


Figure 16: Runoff collected in a MAR project supplied by distributed stormwater collection in Santa Cruz County (from Figure 3 from [130]).

Source water for infiltration basins varies from developed surface water to recycled water to distributed stormwater collection (DSC), with the appropriate water source depending on its availability and the scale desired for the project. Source water quality considerations for infiltration basins differ from those for injection wells because passage through the vadose zone will allow physical filtration or transformation of some contaminants and alter the geochemical composition of the water; nonetheless, clogging can still be a major issue [133]. Infiltration basins are scraped routinely to remove accumulated sediments and restore high infiltration rates, but infiltration rates can decline more than an order of magnitude even during a single (albeit season-long) infiltration event [129]. Sediment detention basins can allow settling time for surface water sources with high sediment loads. Nevertheless, an infiltration basin supplied by DSC in Santa Cruz County accumulated up to 8 cm of sediment per season, despite the use of a sediment detention basin, resulting in a significant decrease in the effective hydraulic conductivity [130]. A study conducted in Orange County showed that bank infiltration, a MAR technique not commonly used in California, can effectively reduce suspended solids in river water prior to its use in infiltration basins, thus maintaining high percolation rates [131]. More research is needed on corresponding methods to reduce suspended solids in source water from DSC. Although the sediment load of source water is one of the primary water quality concerns given its impact on infiltration basin performance and maintenance costs, biological and chemical water quality also need to be considered. For projects using recycled water, the mandated residence time in the aquifer has been reduced from 12 months for injection wells and 6 months for infiltration basins to 2 months for both surface and subsurface applications of recycled water [122, 139, 135]. The transport time of introduced gas tracers has been shown to be a reliable indicator of aquifer residence time and is one potential method to document that required residence times are met in coastal California infiltration basins [140, 126]. Whereas direct injection into the aquifer requires advanced treatment of recycled water (reverse osmosis and oxidation), specific treatment processes are not prescribed for the use of recycled water in infiltration basins, provided that the required reductions in pathogenic microorganisms and other water quality requirements are met [141, 134]. Infiltration basins using developed surface water or DSC don't have these same regulatory requirements, but like in ag-MAR, nitrate leaching can still be an important consideration [127, 128]. Whereas residual nitrate in the soil may be the dominant nitrate source in ag-MAR, nitrogen-rich source water can be an important nitrate source for infiltration basins [127]. It has been shown that 30–60% of the original nitrate load may be removed from source water during infiltration, predominantly by denitrification processes [127, 128]. Schmidt et al. ([127, 128]) further showed that denitrification may be enhanced with the addition of labile carbon sources that increase the organic carbon concentrations in the infiltrating soil layer [127, 128]. It has also been suggested that the reduction of nitrate loads by denitrification is reduced at high infiltration rates, and that an optimal infiltration rate may be identified by taking into account both water quality and quantity goals [127]. In addition to the challenges of site selection, sediment accumulation, and potential nitrate leaching, the cost of infiltration basins is an important consideration. Proponents of DSC-MAR argue that it can represent a more cost-effective option compared to large-scale centralized infiltration basins, especially since it takes advantage of natural precipitation rather than developed water sources [130]. However, unlike centralized infiltration basins, DSC-MAR likely requires the cooperation of private landowners and a mechanism for incentivizing landowner cooperation. To this end, the Pajaro Valley Water Management Agency has recently launched a Recharge Net Metering program in which recharge from infiltration basins on a landowner's property generates a rebate for groundwater pumping fees [142].

## 5. Discussion and Conclusions

### *5.1. Undesirable results and environmental benefits of MAR*

California's Sustainable Groundwater Management Act (SGMA) requires Groundwater Sustainability Agencies (GSAs) to assess the sustainability of their basin using six critical parameters or sustainability indicators. The six indicators include i) lowering of groundwater levels, ii) reduction of groundwater storage, iii) seawater intrusion, iv) groundwater quality degradation, v) land subsidence, and vi) depletion of interconnected surface water. Every GSA must assess the current condition of their basin using these six parameters and then establish minimum thresholds and measurable objectives for each one. Managed aquifer recharge can be used to address one or many of these undesirable results of groundwater overdraft.

Agricultural managed aquifer recharge can be implemented to increase groundwater elevation and storage, improve groundwater quality, mitigate land subsidence, and reduce surface water depletion of interconnected groundwater and surface water systems [84, 106, 11]. Capturing flood flows for ag-MAR can increase groundwater elevation in a fully allocated river basin without negatively impacting other water users or minimum in-stream flow requirements, although consideration of the timing of diversion of the flood flows is needed [106, 11]. High magnitude flows (HMFs) are important for the geomorphology and ecology of a river, including transportation of sediment, channel formation, dispersal of native riparian organisms, and creation of spawning grounds for fish [143, 144, 145]. Kocis and Dahlke ([11]) suggest that HMF events after dry periods could be reserved for channel formation or environmental flows since the majority of sediment is usually transported early in the wet season, and HMFs later in the season could be diverted for ag-MAR so as not to negatively affect riverine ecosystems. The historical hydrologic condition of the Sacramento-San Joaquin River Delta, which provides water to the Central Valley of California, has been in excess of surface water allocations for urban, agricultural, and environmental needs 41% of the days since 1976, suggesting the joint utilization of HMFs for groundwater banking and environmental flows is possible [11]. This mutually beneficial situation would allow basin managers to address SGMA sustainability indicators using MAR, while preserving ecosystem functioning.

Excessive groundwater pumping is the primary cause of subsidence in California and in the San Joaquin Valley (the southern two thirds of the Central Valley); it is the single largest human alteration of the earth's surface, affecting 13,468 $km^2$ [146]. Subsidence is an undesirable effect of groundwater overdraft and causes damage to infrastructure, such as buildings, bridges, roads, and California's surface water conveyance systems [147]. Subsidence also increases the risk of flood damage to low-lying areas, permanently decreases the capacity of fine-grained aquifers to store water, and can negatively impact sensitive environments such as wetlands and groundwater dependent ecosystems (GDEs). The aquifer system of California's Central Valley is made up of confined and unconfined parts. Unconfined coarse grained sediment aquifers are able to be easily extracted from and recharged, experiencing recoverable subsidence from elastic deformation. However, finer grained aquitards can experience both elastic and inelastic deformation. Inelastic subsidence occurs when hydraulic heads drop below preconsolidation heads, which can occur from excessive groundwater pumping. Inelastic subsidence is permanent and irreversible, often caused by the collapse of clay minerals, thus reducing the capacity of the aquifer to store water for the future. More than 50% of the alluvial aquifer system in California is made up of fine-grained sediments that are susceptible to compaction when the preconsolidation stress is exceeded [148, 149]. Smith et al. ([150]) used Interferometric Synthetic Aperture Radar (InSAR) to find that between 2007–2010, during a drought period, groundwater extraction in California's San Joaquin Valley resulted in 0.78 m of permanent compaction and that 98% of all subsidence measured was permanent [150]. Groundwater pumping during this time resulted in historically low groundwater levels, with hydraulic head measurements of wells dropping below preconsolidation heads, causing the inelastic deformation. A more recent study conducted by the National Air and Space Administration (NASA) with data from 2006-2016 found that several spots within the San Joaquin Valley have experienced continuous subsidence, with rates up to 0.6m/yr [147]. The report found that subsidence in the San Joaquin Valley has affected the California Aqueduct, the largest water conveyance canal of California's State Water Project, reducing its efficiency by 20% [147]. Figure 17 shows subsidence in the San Joaquin Valley between May 7, 2015 and September 10, 2016, and where major aqueducts intersect with the subsidence zones [147]. There are areas in California, however, where improved groundwater management is now replenishing aquifers and in some cases even causing small amounts of land uplift. Figure 18 shows the Santa Clara Valley in California's southern San Francisco Bay Area, which has experienced uplift of up to 2.5 cm between March 2015 and March 2016 [147]. As discussed in sections 2 and 3 on conjunctive use and in-lieu recharge, Santa Clara Valley Water District has recently implemented a number of heightened groundwater recharge efforts using recycled water and surface water imports, which may contribute to the region's slight uplift.

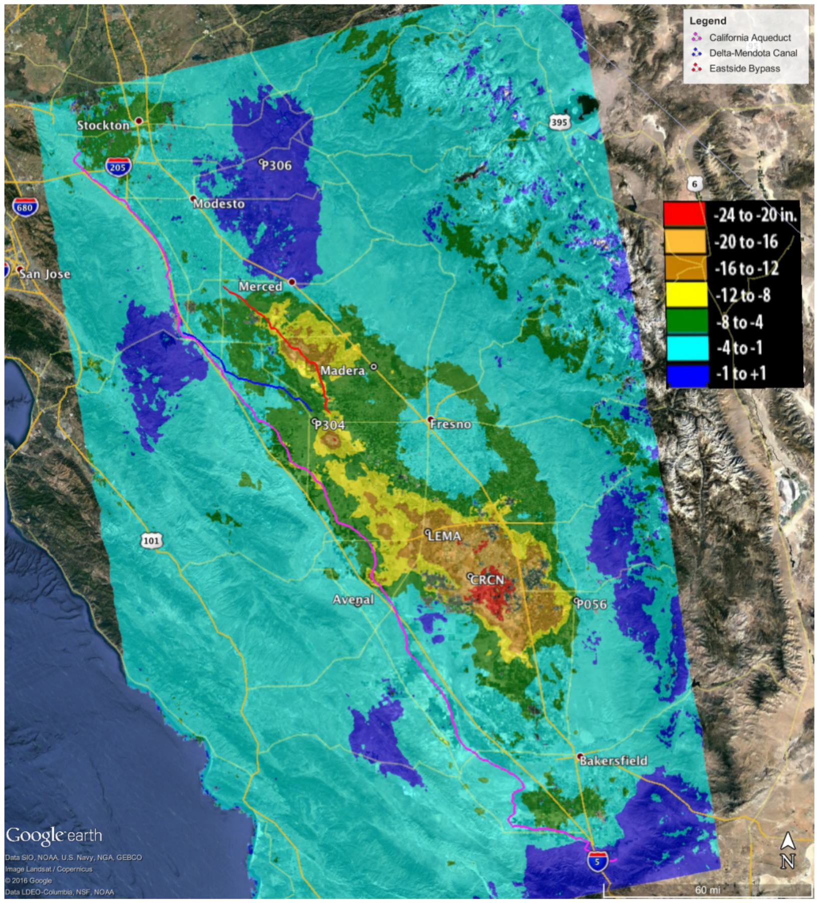


Figure 17: Subsidence in the San Joaquin Valley of California between May 7, 2015 and Sept. 10, 2016. (from Figure 1 from [147]). Original Sentinel-1 data courtesy of ESA.

Groundwater dependent ecosystems are ecosystems in which the species' survival is dependent on groundwater [152]. Unsustainable groundwater pumping can lower groundwater elevation to the point that surface-groundwater interactions become disconnected, which adversely affects GDEs and can threaten species that are endemic to these ecosystems [111, 117, 153]. California's SGMA is the only groundwater legislation in the United States that explicitly considers GDEs in its water management plans [152]. While ag-MAR, and MAR in general, can increase baseflow and benefit groundwater dependent ecosystems, its efficacy depends on the dominating process of

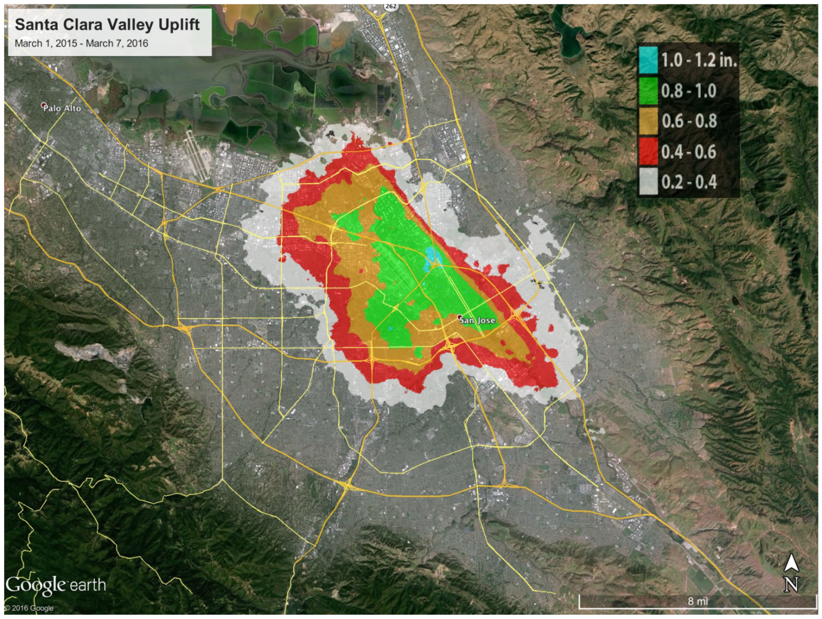


Figure 18: Subsidence in the Santa Clara Valley of California between March 1, 2015 - March 7, 2016 (from Figure 21 from [147] and adapted from [151]). Original Sentinel-1 data courtesy of ESA.

groundwater discharge. Niswonger et al. [106] found only a minimal (1%) increase in baseflow to streams after winter season ag-MAR was implemented [106]. After aquifer mounding subsided and groundwater pumping activities were re-initiated, groundwater discharge to river baseflow was negligible. The authors concluded that the distribution of groundwater discharge from ag-MAR primarily went to fulfill the evapotranspiration needs of overlying crops and adjacent phreatophyte vegetation, instead of contributing to baseflow [106]. They further suggested that if the ag-MAR sites were closer to river channels, the benefits to baseflow may have been more evident.

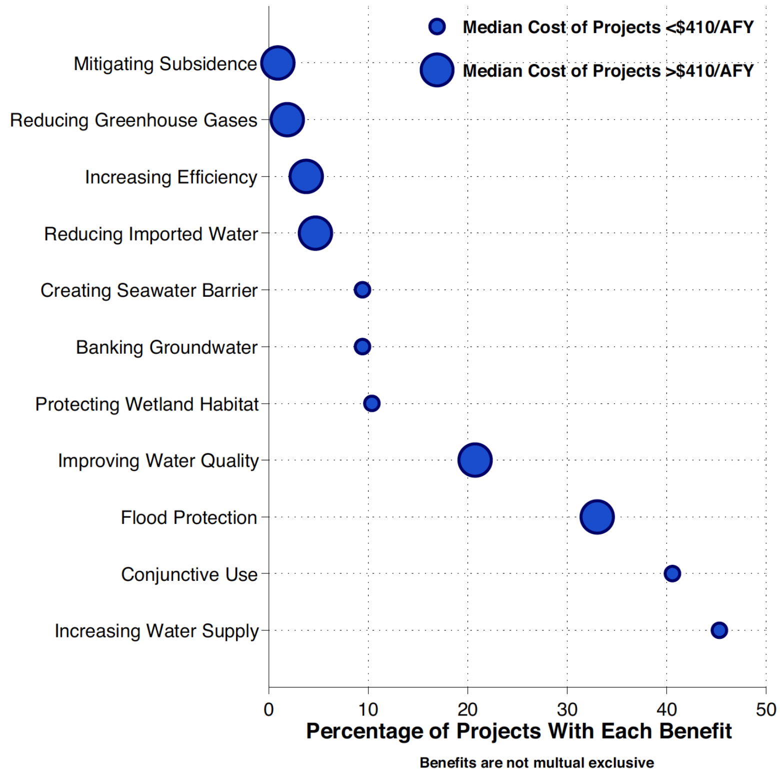


Figure 19: Analysis of MAR project benefits with cost information. The size of the dot indicates whether the median costs for projects within each benefit category is below or above the median cost of all of the projects (i.e., \$0.33 per $m^3$/yr [\$410/acre-feet/yr]) (from Figure 3 from [17]).

Wetlands are a specific category of GDEs with high ecological significance in California. Wetlands provide essential ecosystem services such as naturally improving water quality and flood buffers. In California, 90% of original wetlands have been lost due to land conversion[154]. However, this provides many mitigation opportunities, in which an adverse environmental impact such as habitat destruction for economic purposes can be mitigated by creating or improving habitat elsewhere (Figure 19). Mitigation may even be achieved with minimal effort; research has found that wetland reestablishment can occur spontaneously in degraded areas if lowered groundwater levels are restored to natural levels and intensive uses such as agriculture are halted [155]. While urban MAR may require a change in land use, sometimes leading to a loss of habitat, mitigation through wetland habitat restoration may help to offset the environmental impacts of MAR.

MAR can cause loss of habitat by converting natural areas into infiltration basins.

However, agricultural and urban MAR should be considered separately because the impact of MAR depends on its context and the condition of the land before its conversion to MAR. Ag-MAR, for example, does not necessitate new land conversion and therefore does not directly cause a loss of habitat (although water being used to recharge aquifers may be diverted from natural ecosystems). Conversely, in the urban center of Orange County, the construction of a new recharge basin called Burris Basin required that vegetation and wildlife habitat be removed[26]. This loss was mitigated by removing non-native invasive trees and shrubs elsewhere and replacing them with 650 native trees, 2,900 shrubs, and 1,000 mulefat plants, an important riparian species. The Burris Basin also required the creation of new habitat for wetland-dependent bird species, in which storage water from a local dam was used to create new wetland habitat. A small freshwater wetland was also created on the basin's edge using native sedges to improve the basin's habitat value. In addition to land use consideration, MAR projects can also mitigate their environmental impact by using alternate water sources, instead of diverting river flows needed to support river ecosystems. MAR can use recycled water or stormwater runoff for example, meaning that less water must be diverted from natural habitats.

### *5.2. Potential for future expansion of MAR in California*

MAR is well poised to increase in use in California, due to pressing needs for high-quality water to meet competing agricultural, urban, and environmental demands. MAR infiltration methods offer strong benefits such as significantly lower capital costs than other storage methods for use in unconfined aquifers, and lower land surface requirements for injection-based MAR types, as evidenced in Tables 3, 4 [156].

Table 3: Costs of storage (AUS$ in 2008) and land area requirements of managed aquifer recharge projects in relation to costs of alternative storages in Australia (from Table 1 from [29]). (1ML = $10^3 m^3$).

| Type of storage | Storage size range costed (ML) | Unit capital cost of storage[1] ($'000/ML) | Land surface area required ($m^2$/ML) |
|---|---|---|---|
| Rainwater tankpolyethylene | 0.002–0.010 | 200 | 500 |
| Concrete tanktrafficable | 1–4 | 1,000 | 200 |
| Pre-cast concrete panel tank | 4–8 | 250 | 250 |
| Lined earthen dam impoundment | 4–8 | 12 | 600 |
| Large damgravity or concrete | 350–200,000 | 4–10 | 100–200 |
| Pond infiltration/soil aquifer treatment[2] | 200–600 | 1–2 | 20–60[3] |
| Aquifer storage and recovery[2] | 75–2,000 | 4–10 | 1[4] |

[1] Excluding land cost.
[2] Storage size used here for MAR is the mean annual recharge volume. Actual storage volume of recoverable water may be many times this amount, however in brackish aquifers recoverable volume from earlier years will depreciate due to mixing.
[3] For hydraulic loading rates of 17 to 50m/yr.
[4] $1m^2$/ML for ASR system, but if detention storage is required to capture stormwater, size may be 20 to $100m^2$/ML depending on runoff from catchment and capture efficiency.

As demonstrated in this chapter, in the Central Valley of California, one of the world's most productive agricultural regions, a history of groundwater pumping for agriculture has led to critical overdraft and land subsidence. However, deep water tables and past groundwater depletion leave ample subsurface storage capacity to support future expansion of MAR, especially in the southern part of the Central Valley [10]. MAR projects in the Central Valley are indeed increasing in popularity [149], but expansion of MAR in California must consider source water in the context of over-allocated surface water and increasing environmental water demand. As discussed earlier, high magnitude flows (HMFs) that exceed environmental flow requirements can be a promising water source for MAR projects in California's wet years [11]. This possibility, however, may require confronting political barriers in California, arising from water rights and regulatory restrictions that involve a wide variety of stakeholders. Although literature on HMFs for MAR is primarily from California, the method is also being considered in New Zealand, on the Te Arai River in the Poverty Bay area [157]. This potential MAR project would use flows from the ecologically significant Te Arai River for MAR when flows exceed 220 L/s, in a watershed dominated by agriculture.

Source water will determine future applications of MAR, especially in arid regions where conventional water sources such as streamflow and groundwater are already fully

Table 4: Cost summary of groundwater recharge and habitat restoration measures considered by the U.S. Army Corps of Engineers and the Stockton East Water District for eastern San Joaquin County, California (modified from Table ES-1 from [28]).

| Measure | Capital Cost ($1,000$)[3] | Annual O&M Costs ($1,000) | Annual Cost ($/ha) | Potential Ecosystem Benefits |
|---|---|---|---|---|
| Flooded Fields (32 ha site) | $517[1]– $531[2] | $32[1]– $40[2] | $69[2]– $124[1] | • Water depths from zero to 12 inches<br>• Most desirable waterfowl habitat |
| Spreading Basins (32 site) | $1,966 | $33 | $289 | • Large areas of ponded water with gradually sloped sides<br>• Desirable habitat for waterfowl |
| Excavated Recharge Pits (16 ha site) | $909 | $23 | $1,021 | • Smaller areas of ponded water with steeply sloped sides<br>• Fair habitat for waterfowl |
| Unlined Flat Canal | $15,819 | $84 | $603 | • Similar to excavated pits<br>• Opportunity for continuous corridor |
| Dry Wells | $1,651 | $220 | $680 | • Would not create waterfowl habitat<br>• If combined with surcharge ponds, benefits would be similar to spreading basins |
| Injection Wells (4 wells) | $4,510 | $646 | $427 | • Would not create waterfowl habitat |
| Enhance Recharge through Streams | $2,657 | $32 | $294 | • Broadened floodplain areas along streams would provide additional riparian habitat |
| Flood Detention Basins | $500[4] | $38 | $119 | • Similar to flooded fields for shallow flooding<br>• Similar to excavated pits during flood events |
| In-Lieu Delivery (agricultural delivery program) | $7,098 – $14,195[5] | $177 | $554 | • Would not create waterfowl habitat |

[1] Assumes infiltration rate of 0.08 m/d.
[2] Assumes infiltration rate of 0.15 m/d.
[3] Capital costs include all first costs including land acquisition, construction, PED, contingency, etc.
[4] Cost does not include conveyance modifications that may be necessary to support recharge.
[5] Low and high cost estimates assume a pipeline length of 8 and 16 km, respectively.

exploited. Looking to the future, additional water sources may include recycled water, desalinated water, and even oil processed water. MAR using recycled water is a growing water security strategy in California and globally. In regions such as California where wastewater effluent discharge standards require expensive tertiary or advanced treatment, it becomes increasingly cost-beneficial for municipalities to reuse their effluent rather than discharge it to surface waters [158]. However, there are barriers to implementation such as public acceptance [159]. In an Australian survey, for example, researchers found evidence of opposition to the use of recycled water for consumption, with 61% of responders stating that they had health-related concerns about drinking recycled water [160]. Nevertheless, recharge using recycled water is being practiced and promoted in Australia [161, 162], as well as in California, as discussed earlier in section 2.

Countries in the arid Middle East and Northern Africa region have also turned to recycled water for added water security, in some cases using it for MAR. Israel, a world leader in water reuse, irrigates a large fraction of its agriculture with recycled water, using a process in which secondary treated effluent is recharged to infiltration basins (i.e. soil aquifer treatment), then recovered later in wells for irrigation use [163]. In Muscat, Oman, 94% of municipal water is sourced from desalinated water, and 46% of wastewater is treated and reused for non-potable purposes such as landscaping [164]. The city is now considering implementing MAR with recycled water produced in excess during the low-irrigation winter months, which would otherwise be discharged to the ocean. An analysis of the proposed project found it economically appealing to implement MAR with recycled water, although public acceptance of blending recycled water with the existing public supply was highlighted as a primary barrier to implementation [164]. Additional concerns arise given the growing body of knowledge on emerging contaminants, such as pharmaceuticals and personal care products, that have been found to pass through wastewater treatment processes and may persist in the environmental for extended periods of time [165].

In Shanghai, China, MAR has been used for decades for the dual benefits of preventing land subsidence and providing water cooling for industrial plants. Urban MAR began in Shanghai in the 1960s to halt land subsidence when excessive groundwater extraction occurred due to population migration from rural to urban areas [166]. Tap water was injected via wells and it was observed that the water maintained cool temperatures for a long period of time. Subsequently, the cold water was exploited as a cheap option for industrial cooling, with nearly 500 cold storage wells being deployed in China by 1984 [167]. However, these storage wells have not actually resulted in significant volumes of aquifer recharge, due to well clogging [168]. Some parts of China, however, are now considering implementation of MAR to restore groundwater supplies. The

Northern China Plain region is considered a global hotspot for groundwater depletion, experiencing high rates of overdraft and issues such as land subsidence and seawater intrusion [169]. Here, MAR has been proposed as a strategy to reduce groundwater depletion, using urban recycled water and diversion flows from upstream reservoirs, but these proposals have not yet been implemented [169].

### *5.3. Barriers and concerns to expansion of MAR*

Although there is significant potential for expansion of MAR in California, several challenges and concerns must be addressed for MAR to be successful. Source water quality, for example, may impact MAR project performance in terms of infiltration capacity and groundwater quality [127, 128, 129, 130, 131]. Sediment accumulation in infiltration basins can significantly reduce the saturated hydraulic conductivity and thus the infiltration capacity of a basin [129, 130]. In Southern California, the Orange County Water District controls for sediment accumulation in its system of over 23 recharge basins by routing recharge water from the Santa Ana River into a series of desilting ponds [26]. The recharge basins still develop clogging layers of silt over time, so the water district will periodically drain and scrape the bottom of the basins with bulldozers. Figure 20 shows the accumulated clogging layer from a recharge basin operated by OCWD. More research is needed to better understand the dynamics of sediment accumulation and to further investigate methods to reduce the sediment load of source water, such as bank infiltration or sediment detention basins [130, 131].

Figure 20: Accumulated clogging layer from a recharge basin operated by OCWD (from Figure 5-12 [26]).

Nitrate leaching has the potential to negatively affect groundwater quality, either from nitrate loads in source water or residual nitrate in the soil, and is a major concern for some infiltration basins and especially for ag-MAR [84, 127, 128]. Denitrification in the anaerobic zone created by the perched water table (the saturated soil layer immediately under the infiltration basin) can significantly reduce nitrate leaching and more research is needed to determine how denitrification can be enhanced in infiltration basins [127, 128]. One potential strategy to promote denitrification that is currently being investigated is the addition of reactive carbon sources to infiltration basins [170, 171]. In coastal areas, there is concern about the effect of sea-level rise associated with climate change on the continued effectiveness of current MAR projects. Many modeling and laboratory studies have attempted to determine how sea-level rise will affect seawater intrusion, although the results of these studies show significant variability, ranging from no effect on seawater intrusion to migration of seawater several km further inland [172]. Analytical models generally suggest that the effect of sea-level rise on seawater intrusion will be small compared to the effects of continued overdraft of groundwater [3]. Werner et al. ([172]) provide a detailed description of the research on sea-level rise and seawater intrusion.

Lastly, there are several legal and institutional barriers that need to be overcome in the next few years to ease the process of implementing new MAR projects (particularly ag-MAR) statewide. Given that groundwater recharge is not considered a beneficial use of water in the California Water Code [173], and landowners or water districts planning on implementing new MAR programs will likely have to obtain a new surface water right or change an existing water right, the legal use of excess surface water remains questionable for the near future. The California State Water Resources Control Board (SWRCB) currently calculates surface water availability for a new appropriative surface water right using a method similar to the Rational Runoff Method [174, 175], which estimates the average annual unimpaired runoff at a diversion point of interest only considering contributing area, average annual precipitation, and the land use within the watershed [175]. This conservative method is used to ensure that there is "unappropriated water available to supply the applicant" (California Water Code section 1375(d)), while accounting for "...the amounts of water needed to remain in the source for protection of beneficial uses... (California Water Code section 1243), such as recreation and the preservation of fish and wildlife habitat.

However, as indicated by Grantham and Viers [176], in many areas of California, mainly the Central Valley, surface water has been over-allocated to the extent that surface water rights account for nearly 1,000% of natural surface water supplies. This, theoretically, precludes any additional appropriation of surface water. However, over-appropriation is, to a large extent, an artifact of the water availability analysis conducted by the SWRCB, which is based on average annual flows and does not take into account the large variability in streamflow. Hence, new permitting approaches that would legally permit the use of high-magnitude flow for groundwater recharge are needed.

Allowing a water-right permit for the diversion of High Flows could potentially bridge the gap between policy requirements (such as the need for a temporary or permanent water right for surface water diversions), legal requirements (stream reaches that are already legally over-appropriated), and physical surface water availability for groundwater recharge (in the form of flood flows during above normal or wet years). Such permits would have to agree on legally acceptable high flow thresholds at the point of diversion to ensure that high flow diversions for groundwater recharge do not cause injury to existing water-right holders or environmental flow considerations. However, permits could be restricted to the winter period only (e.g. November-March) and define strict instream flow requirements (e.g. the passage of channel forming flows or fall flushing flows for sediment and nutrient transport). Solving these regulatory challenges to groundwater recharge will open new avenues to greater water security in California.

## 6. List of acronyms and abbreviations

| | |
|---|---|
| Ag-MAR | agricultural managed aquifer recharge |
| CA | California |
| CCR | California Code of Regulation |
| CV | Central Valley |
| DSC | Distributed stormwater collection |
| EPA | Environmental Protection Agency |
| GDE | Groundwater dependent ecosystem |
| GSA | Groundwater Sustainability Agency |
| GWR | Groundwater Replenishment |
| HMF | High-magnitude flow |
| InSAR | Interferometric Synthetic Aperture Radar |
| $K_{sat}$ | Saturated hydraulic conductivity |
| $K_h$ | Hydraulic conductivity |
| LA | Los Angeles |
| LID | Low-impact development |
| MAR | Managed aquifer recharge |
| NASA | National Air and Space Administration |
| OCWD | Orange County Water District |
| SAGBI | Soil Agricultural Groundwater Banking Index |
| SFPUC | San Francisco Public Utilities Commission |
| SGMA | Sustainable Groundwater Management Act |
| TDS | Total dissolved solids |

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