PAPER

Sixty years of global progress in managed aquifer recharge

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Abstract

The last 60 years has seen unprecedented groundwater extraction and overdraft as well as development of new technologies for water treatment that together drive the advance in intentional groundwater replenishment known as managed aquifer recharge (MAR). This paper is the first known attempt to quantify the volume of MAR at global scale, and to illustrate the advancement of all the major types of MAR and relate these to research and regulatory advancements. Faced with changing climate and rising intensity of climate extremes, MAR is an increasingly important water management strategy, alongside demand management, to maintain, enhance and secure stressed groundwater systems and to protect and improve water quality. During this time, scientific research—on hydraulic design of facilities, tracer studies, managing clogging, recovery efficiency and water quality changes in aquifers—has underpinned practical improvements in MAR and has had broader benefits in hydrogeology. Recharge wells have greatly accelerated recharge, particularly in urban areas and for mine water management. In recent years, research into governance, operating practices, reliability, economics, risk assessment and public acceptance of MAR has been undertaken. Since the 1960s, implementation of MAR has accelerated at a rate of 5%/year, but is not keeping pace with increasing groundwater extraction. Currently, MAR has reached an estimated 10 km^3/year, ~2.4% of groundwater extraction in countries reporting MAR (or ~1.0% of global groundwater extraction). MAR is likely to exceed 10% of global extraction, based on experience where MAR is more advanced, to sustain quantity, reliability and quality of water supplies.

Keywords Managed aquifer recharge · Artificial recharge · Review · Water banking · History of hydrogeology

Introduction

Over the last half of the twentieth century, rotary drilling, submersible pumps, electricity distribution, population growth and concentration in urban areas, the need for increased food production, pursuit of rural incomes and avoidance of famine have all conspired to elevate the value of groundwater as an essential resource (OECD 2015).

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Groundwater exploitation has grown at a rapid rate, and has challenged human capability to sustain the resource, and where climate is drying the challenge has intensified. Managed aquifer recharge (MAR), used to enhance the quantity and quality of groundwater, is a term conceived by the British hydrogeologist Ian Gale, who was the founding co-chair of the International Association of Hydrogeologists (IAH) Commission on Managing Aquifer Recharge from 2002 to 2011 (IAH-MAR 2018a). Managed aquifer recharge refers to a suite of methods that is increasingly used to maintain, enhance and secure groundwater systems under stress. River-bank filtration for drinking water supplies was well established in Europe by the 1870s and the first infiltration basins in Europe appeared in 1897 in Sweden and in 1899 in France (Richert 1900; Jansa 1952). However, 60 years ago, at the time of the formation of the IAH, human intervention to increase the rate of groundwater recharge such as drainage wells for flood relief, disposal of sewage water via septic tanks

or seepage beneath surface-water irrigated crops, was generally unmanaged or incidental. Intentional recharge, then called artificial recharge, was rare but soon began being adopted at large scale in urban areas of California and New York in the USA, to arrest declining water levels.

Although hand dug wells and percussion drilling have been used for more than 2,000 years, the rotary drilling rig was first used in the 1880s and has evolved considerably, including reverse circulation, introduced in 1946. Also in the 1880s, the development of the AC transformer led to electrical energy distribution in the USA and Europe and ultimately reaching rural areas in developing countries through the course of the twentieth century. Then in 1928, Armais Arutunoff invented the electric submersible pump for the oil industry, whereby in the mid-1960s, this was adapted to pump water from deep wells and a disruptive technology had emerged. Until then, groundwater extraction for irrigation had been constrained by the rate at which oxen or mules could draw water from a dug well, the strength of the wind, or by the depth to which a centrifugal pump or extended shaft turbine pump could be installed. The combined availability of deep wells, electric power and electric submersible pumps radically escalated water withdrawal from aquifers and quickly reduced groundwater in storage. Between 1900 and 2008, 4,500 km^3 of depletion had occurred globally (Konikow 2011). Alarmingly, the depletion rate is still accelerating, reaching 145 km^3/year between 2001 and 2008 (Konikow 2011).

Although there is considerable uncertainty in estimates of annual groundwater exploitation and recharge, Margat and van der Gun (2013) report annual exploitation of groundwater of ~980 km^3/year in 2010, which is less than 8% of estimated global mean natural recharge (which exceeds 12,000 km^3/year; Margat 2008), but nonetheless causes substantial depletion in some areas. Hence, combining this information,

groundwater storage depletion in aggregate constitutes only about 15% of groundwater extraction. The balance is composed of enhanced "natural" recharge due to steeper gradients in intake areas and reduced natural groundwater discharge with adverse consequences for surface-water resources and groundwater dependent ecosystems (Burke and Moench 2000). For comparative purposes, the global storage volume of modern groundwater is estimated at 0.8–1.9 million km^3 for groundwater aged 25–100 years (Gleeson et al. 2015) and constitutes less than 6% of the estimated total volume of groundwater. Residence time depends more on the natural discharge than groundwater extraction, but the minimum estimate for the global mean exceeds 250 years.

By contrast, the total surface-water storage in dams and lakes is two orders of magnitude smaller, 12,900 km^3 (from The World's Water 2002–2003 Data, Pacific Institute 2018), with residence times of typically a few years (average <3.3 years), giving an annual turnover of the order of 4,000–6,000 km^3. The decline in new large dams (i.e. typically >3 Mm^3 capacity; ICOLD 2018), since the 1970s (Fig. 1), represents increasing saturation and diminishing prospects as well as concerns over ecological impacts of dams, siltation of reservoirs and equity of benefits of communities downstream, particularly across political borders. It may also in part reflect the availability of alternative supplies such as desalination, which by 2005 had reached a capacity of 55 Mm^3/day (20 km^3/year; Pacific Institute 2018). In the USA, use of recycled water was reported in 2000 to be 3.6 km^3/year (7% of the sewage treated) and reuse was growing at 15%/year, and with locally high rates of recycling in Australia, Europe and the Middle East (Miller 2006); FAO estimated that 2,212 km^3/year is released into the environment as wastewater in the form of municipal and industrial effluent and agricultural drainage water, with 80% of this untreated (UN Water

Fig. 1 Evolution of the global number of large dams built during the 20th Century (from The World's Water 2002–2003 Data; Pacific Institute 2018)
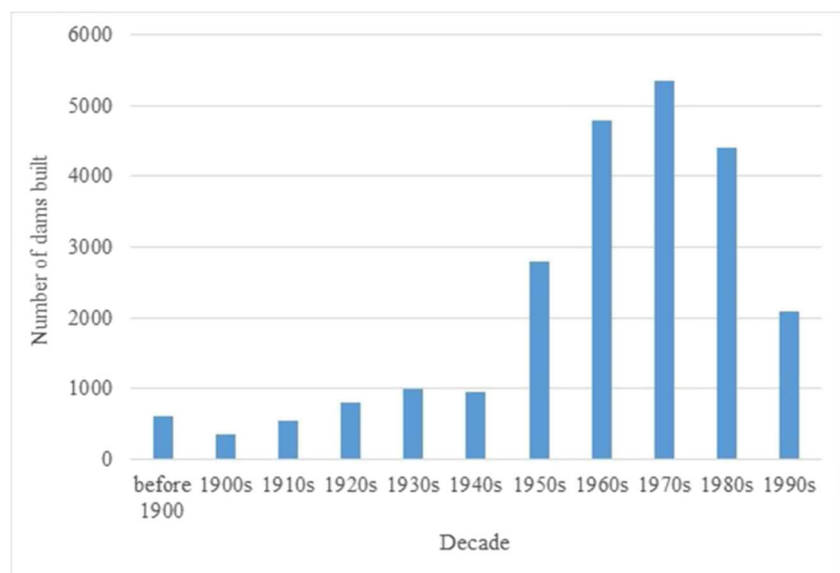
2017). The opportunities for improved treatment for more and safer reuse are very significant. Managed aquifer recharge downstream of existing dams, including through recharge releases, would offer conjunctive storage of water and the opportunity to increase dams' benefits with considerably lower financial and environmental costs than raising their height, which would also increase efficiency of water storage particularly in arid environments with low relief (Dillon 2016).

It is clear that for sustainable-water-resource utilization, stabilization of storage decline is important and there are only two means of accomplishing this for groundwater: reducing demand (through increased water use efficiency or conjunctive use with other water sources) or increasing replenishment (Dillon et al. 2012). In most locations, it is unreasonable to expect groundwater replenishment alone to reverse the impacts of excessive groundwater extraction (Dillon et al. 2009a). Managed aquifer recharge is a term for a wide and growing range of measures to support active management of groundwater resources at the local and basin level, to make more efficient use of water resources, assist conjunctive management of surface and groundwater resources (Gale 2005; Evans et al. 2012; Evans and Dillon 2018), to buffer against increasing intensity of climate extremes, particularly drought, and to protect and improve water quality in aquifers. While a few of these measures have been in use for millennia, many more have developed over the last 60 years, supported by a growing body of scientific knowledge and practical experience, fanned by the increasing pressures on groundwater systems.

This paper contains nationally aggregated estimates of annual recharge volumes and annual groundwater use. In addition, it includes global estimates of natural groundwater recharge, annual groundwater exploitation, annual volumes of desalinated and recycled water, accumulated groundwater depletion and total surface-water storage in dams and lakes. None of these quantities is subject to simple direct measurement, but the estimates rather are derived as the sum of a mix of data acquired in very different ways (including correlations and guesses) and finding different versions of the same statistic reported is not uncommon. Therefore, it has to be emphasized that the numbers shown, although being "best estimates", are subject to considerable uncertainty. The reason to show them nonetheless is that they help put the quantities of water involved in MAR in proper perspective.

Evolution of the practice of recharge augmentation

Over millennia, human endeavor has resulted in significant unintentional increases in recharge of aquifers. Typically, when forests or jungles were cleared for soil tillage, or crops irrigated with surface water, these actions have inadvertently increased groundwater recharge. Irrigation began in Egypt and Mesopotamia around 6000 BC by diversion of water from rivers, with dams and canals used from 3100 BC (Irrigation Association 2016). Watershed management interventions such as contour bunds and check dams have been used for millennia in the Middle East, Asia and South America to detain monsoon runoff, to defend against soil erosion and conserve water, and as a by-product, groundwater recharge increased. In the last half century, as agriculture has come to increasingly depend on groundwater, the resource value of the additional water has taken over as a significant driver for implementing these watershed measures, with soil conservation regarded as a co-benefit. In cities, unwanted leakages from water pipes and sewers have also recharged aquifers, since the time of the piping of the first water to cities (Sedlak 2014). Unintended and undesirable consequences of these deliberate actions include waterlogging, land salinization or groundwater pollution.

Unmanaged recharge describes where there is human intent to discharge waters into soil or aquifers but without consideration of the resource value of the disposed water, and often no thought of the impacts on groundwater quality. Stormwater drainage wells for example have been used since ~2000 years BC initially in ancient Persia (Burian and Edwards 2002). These were still being installed until the mid-twentieth century around the world, particularly in towns and cities sited on clay overlying karstic aquifers. Septic tanks are still being installed today, as a first step in village sanitation, but potentially concentrating pathogen and nutrient loads to aquifers and undermining public health where shallow wells are a source of drinking supplies. Similarly, municipalities and industries that dispose of wastewaters to sumps, injection wells or by irrigation without adequate pretreatment also pose a pollution threat.

Managed aquifer recharge is the intentional recharge of water to aquifers for subsequent recovery or environmental benefit (NRMMC, EPHC, and NHMRC 2009). The management process assures adequate protection of human health and the environment. Whereas formerly, the term "artificial recharge", has been used when the focus had been on augmenting the quantity of recharge, but with much less attention given to managing water quality, MAR means that both quantity and quality are managed effectively. As in many countries, in India where artificial recharge has been undertaken by government agencies since the 1970s, the focus has been on quantity with scant thought to water quality. It is proposed that those projects are termed "artificial recharge" until water quality is evaluated and groundwater is shown to be safe for its uses, or competently deemed so. Examples include where water recharged to unconfined aquifers is of the same quality as natural recharge, or where water quality is managed before recharge or on recovery to ensure public health and the environment are protected (Dillon et al. 2014a) and then such sites can be termed MAR (Table 1).

Table 1 Examples of groundwater recharge augmentation, showing evolution from unintentional to unmanaged and now MAR (adapted from NRMMC, EPHC, and NHMRC 2009)

Unintentional recharge enhancement	Unmanaged recharge (for disposal)	MAR (for recovery and/or environmental benefit)
• Clearing of deep rooted vegetation, or soil tillage	• Stormwater drainage wells and sumps	• Streambed channel modifications
• Spate irrigation	• Septic tank leach fields	• Bank filtration
• Irrigation deep seepage	• Mining and industrial water disposal to sumps	• Water spreading
• Leakage from water pipes and sewers	• Drainage water from construction pits	• Recharge wells and shafts
		• Reservoir releases

Much of the development of MAR over the last 60 years has been in managing previously unmanaged recharge to improve water quality and to ensure recovered water is fit for use. In developing new techniques, fit for a wide variety of hydrological, hydrogeological and societal conditions, both quantity and quality have been improved. Initially source waters were natural waters in streams, lakes and other aquifers. These remain, despite anthropogenic influences on water quality, the largest source of water worldwide, recharged via streambed structures in monsoonal catchments in India or released from large water-supply dams in the USA for recharge via infiltration basins.

Since the 1990s, urban stormwater has been extensively harvested in South Australia, recharged and recovered via wells for public open-space irrigation, even though the storage aquifer was originally brackish. Risk assessment has been completed to enable stormwater drainage wells (unmanaged recharge) to be accepted as MAR to safely sustain groundwater-fed drinking-water supplies (Vanderzalm et al. 2014). Another source of water is dewatered groundwater, a by-product of mineral extraction, which is rapidly increasing in importance in some countries, with one example being the separate recharge of brackish and saline dewatering water for several iron ore mines in NW Western Australia to provide future mine water supplies and to protect a salina ecosystem (Fortescue Minerals Group 2011).

Since the 1970s, in California, treated sewage effluent has been stored and further treated in aquifers for subsequent use (Mills 2002). Similarly, since 2013 in Queensland, groundwater from dewatered aquifers in coal where coal seam gas (natural gas, methane) is produced, has been treated and stored in aquifers (APLNG 2012). Managed aquifer recharge gave the opportunity for these otherwise wasted waters to be considered as water resources, and in some cases paved the way for direct reuse. Evolution of treatment technologies has provided a springboard for new MAR applications. Aquifers previously too brackish for beneficial uses have been transformed into productive resources (e.g. Dillon et al. 2003).

Thermal desalination of seawater commenced in Kuwait in the mid-1950s and research investment during 1952–1982 by the US Office of Saline Water ($2 billion research in 2008 terms) facilitated the establishment of the reverse osmosis membrane industry. This matured further with continuing government and private research such that, between 1978 and 2006, improved permeability, membrane life and reduced membrane and energy costs were noted. These have increased productivity by a factor of 480 and have also advanced flash distillation and electro-dialysis techniques (UNESCO 2008). Additionally, advances in membrane technologies (Amy et al. 2017) have been a major factor in the increase in installed seawater desalination capacity to 20 km^3/year by 2005 with 75% of this occurring in the Middle East, where energy is cheap and freshwater is scarce. Groundwater recharge of the excess of supply over demand for desalinated seawater, notably in flash distillation as co-generation with electricity production, allows accumulation of reserves and improves resilience of water supply in areas with high evaporation rates. The Liwa groundwater storage reserve near Abu Dhabi, UAE, is a 50-Mm^3 example, for which Stuyfzand et al. (2017) report on water quality management.

Wastewater treatment to protect river water quality since the 1970s in USA, Europe and Asia, has made advances both in the number of plants and the level of treatment. In the early 1960s, Loeb and Sourirajan invented a cellulose acetate membrane for reverse osmosis (Visvanathan et al. 2000) enabling membrane bioreactors (MBR) to become viable in the early 1990s (Hai and Yamamoto 2011). In 2003, 66% of the worlds MBR plants were in Japan, 16% in North America, 11% in Europe, and 7% between Korea and China, with the largest plant then in Beijing producing at 80,000m^3/day. Growth in water reuse via aquifers has been primarily motivated by the need to cost-effectively secure high-quality water supplies by accumulating and drawing on buffer storages in aquifers in off-peak and peak times, seasonally or over years. The aquifer integrates existing wastewater and water infrastructure. Membrane treatments are generally well suited to maintaining flow rates in injection wells and infiltration basins and galleries, contribute to the range of pre-treatments for MAR, and complement the treatments that aquifers provide (Kazner et al. 2012).

Managed aquifer recharge overlaps with aquifer thermal energy storage (ATES) when water is seasonally recharged and recovered from aquifers via wells. There are many thousands of these systems in the Europe. Gao et al. (2017)

reviewed the performance of recent ATES systems and found energy savings of 40–90% compared with conventional sources and payback times were typically less than 5 years. The hydrogeological factors affecting efficiency were discussed by van Elswijk and Willemsen (2002) and Miotlinski and Dillon (2015). Interesting examples at municipal scale are found in the cities of Sapporo and Sendai in Japan where water from warm deep aquifers are pumped through pipes beneath footpaths and roads to melt snow and ice. As a result of groundwater depletion, the cooled water is now injected into shallow aquifers and in the summer this cool water is recovered and used in heat exchangers for air conditioning in buildings, then the warm water is reinjected into the deeper aquifer making the system sustainable (Yokoyama et al. 2002).

Quantifying the recent growth of MAR

The historical quantity of MAR is summarized in Table 2 as a result of most authors of this paper each taking responsibility for a geographic area. Generally, these estimates were produced by reference to documentation of individual projects with known starting dates and volumes, and closing dates when known, and aggregating these for incorporation into Table 2. Recharge volumes are reported as the average annual volumes for each decade to smooth out climatic variability. In most instances, recharge capacity is recorded, as relatively few sites publically reported actual annual volumes of recharge. The annual volume recharged is reported rather than volume recovered, as for water-banking systems, recovery is infrequent in comparison with recharge, and it is assumed that recharge and recovery are related over the long term.

India, the country with the most MAR capacity, has several million recharge structures (more than 500,000 in Gujarat alone; R. C. Jain, CGWB, personal communication, 2014) and 11 million more are planned (CGWB 2017), but has less than 30 structures where recharge has actually been measured and documented (Dashora et al. 2018). Information on aggregate detention capacity of streambed recharge structures and rainwater harvesting was found for Gujarat in 2012 (CGWB 2013). From studies that quantified recharge for structures with known capacities, the average ratio of mean annual recharge volume to detention capacity ranged from 1 to 2 (Dashora et al. 2018), and a conservative estimate of 1 was adopted here. For several other Indian states, aggregate numbers of recharge structures of different scales were recorded, and recharge volume was estimated using a triangular frequency distribution (that is, maximum frequency at the lower margin of each size range tailing linearly to zero frequency at the upper margin) and the same ratio for mean annual recharge to detention capacity. For states where capacities were not identified, the stated costs (CGWB 2013) of establishing

recharge structures were compared with states where both capacities and costs were known and recharge volumes were estimated assuming the same ratios for detention capacity to cost and recharge to capacity. Indian programs to establish recharge structures commenced in the 1960s and government expenditure over different planning periods was known. It was assumed that the five reported states followed the national pattern, taking into account nongovernment programs that have continued since the 1960s.

Hence, the current and historical recharge estimates for India in Table 2 are for only five states that had sufficient documentation—Andhra Pradesh (includes Telangana which became independent in 2014), Gujarat, Jharkhand, Karnataka and Uttarakhand and are conservative. These states contain 18% of the national population and in 2009 accounted for 16% of national groundwater extraction (CGWB 2014). The total recharge for India could potentially be between 2 and 5 times the five-state estimated amount in 2015 (3.1 km^3/year), considering the sustained extensive but unquantified investment in recharge structures and rainwater harvesting in other states such as Haryana, Maharashtra, Madhya Pradesh, Rajasthan and Tamil Nadu (which, combined, have a further 32% of national groundwater extraction) but in the absence of factual data such estimates are currently excluded. Similarly in the USA, it is expected that MAR is under-reported in Table 2, because detailed data are limited to California and Arizona (Scanlon et al. 2016), although historical recharge in Florida, New York and Texas is included. In Germany, the Federal Statistical Agency provides summary information for public water supplies and about every 3 years since 1979 has identified the sources of water, which enables MAR including bank filtration to be quantified.

In other countries, MAR volumes are considerably smaller and estimates are derived from government reports, or more commonly by accumulating documentation of known MAR projects. The Global MAR Inventory Working Group (IAH-MAR 2018b) has consolidated information on the scope of 1200 MAR projects (Stefan and Ansems 2018) and anyone with quantitative information on other sites is invited to submit this via the MAR Portal (IGRAC 2018) to allow improved and more complete estimates in future.

Table 2 draws on many national sources of information. For some countries a national summary was produced by co-authors of this paper; the reports have been uploaded on the IAH-MAR web site (IAH-MAR 2018b) and provided with this paper as electronic supplementary material, ESM 1. These reports indicate the types of recharge, source waters used, purposes (such as water supply), subsidence prevention, and water quality improvement (such as in river-bank filtration), and describe novel practices. The table is not comprehensive, as a number of countries (reported and unreported) have known MAR facilities for which quantitative information was unavailable. Bank filtration is also accounted for

Table 2 Reported volume of MAR in 34 countries, areas or regions from the 1960s to 2015 (million cubic metres/year)

Country/region	Average annual MAR volume in the decade centred on date (Mm³/year)						Annual gw use (Mm³/year)[b]	MAR as % gw use	MAR as % of global reported capacity[c]	MAR vol growth (%/year[d]) to 2015	gw use as % global use
	1965	1975	1985	1995	2005	2015[a]	2010	2015	2015		2010
Austria	–	–	–	–	–	56	1,120	5.0%	0.6%	–	0.11%
Australia	79	144	185	213	257	410	4,960	8.3%	4.1%	3.6%	0.51%
Belgium	–	–	–	–	–	2.5	650	0.4%	0.0%	–	0.07%
China	20	23	23	24	56	106	112,000	0.1%	1.1%	3.6%	11.41%
Croatia	–	–	42	48	48	46	600	7.7%	0.5%	0.3%	0.06%
Czech Republic	–	–	–	–	–	22	380	5.8%	0.2%	–	0.04%
Denmark	–	–	–	–	–	0.25	650	0.0%	0.0%	–	0.07%
Finland	<1	30	35	50	55	65	280	23.2%	0.7%	9.3%	0.03%
France	20	21	26	30	31	32	5,710	0.6%	0.3%	1.0%	0.58%
Germany	–	867	766	875	765	870	3,080	28.2%	8.7%	0.0%	0.31%
Greece	–	–	–	–	–	0.3	3,650	0.0%	0.0%	–	0.37%
Hungary	–	–	–	–	–	335	370	90.5%	3.4%	–	0.04%
India (5 states only)	154	430	706	1,020	1,739	3,070	39,800	7.7%	30.9%	6.6%	4.05%
Israel	87	91	127	132	144	134	1,250	10.7%	1.3%	0.9%	0.13%
Italy	178	294	301	348	391	461	10,400	4.4%	4.6%	2.0%	1.06%
Jordan	4	9	9	13	19	20	640	3.1%	0.2%	3.5%	0.07%
Korea	–	3.7	12	46	91	146	3,800	3.8%	1.5%	10.4%	0.39%
Latin America	–	–	–	–	–	311	56,660	0.5%	3.1%	–	5.77%
Netherlands	181	240	255	241	275	262	1,600	16.4%	2.6%	0.8%	0.16%
Oman	0	0	0	6	–	84	840	10.0%	0.8%	9.9%	0.09%
Poland	–	–	–	–	–	143	2,590	5.5%	1.4%	–	0.26%
Portugal	–	–	–	–	–	9	6,290	0.1%	0.1%	–	0.64%
Qatar	<1	<1	<1	<1	37	44	260	16.9%	0.4%	8.4%	0.03%
Romania	–	–	–	–	–	7.4	630	1.2%	0.1%	–	0.06%
Serbia & Montenegro	–	–	–	–	–	9.5	580	1.6%	0.1%	–	0.06%
Slovakia	–	–	–	–	–	176	360	48.9%	1.8%	–	0.04%
Slovenia	–	–	–	–	–	9.5	190	5.0%	0.1%	–	0.02%
South East Asia	<1	<1	<1	<1	<1	5	29,270	0.0%	0.0%	–	2.98%
Southern Africa	1	2	6	6	7	10	4,500	0.2%	0.1%	5.1%	0.46%
Spain	3	8	12	60	350	380	5,700	6.7%	3.8%	10.9%	0.58%
Sweden	–	–	–	–	–	44	350	12.6%	0.4%	–	0.04%
Switzerland	–	–	–	–	–	100	790	12.7%	1.0%	–	0.08%
UK	0	0	0	5	5	5	2,160	0.2%	0.1%	–	0.22%

Table 2 (continued)

Country/region	Average annual MAR volume in the decade centred on date (Mm³/year)						Annual gw use (Mm³/year)[b]	MAR as % of global reported capacity[c]	MAR as % gw use	MAR vol growth (%/year)[d]	gw use as % global use
	1965	1975	1985	1995	2005	2015[a]	2010	2015	2015	to 2015	2010
USA	302	494	768	1,218	2,026	2,569	112,000	25.8%	2.3%	4.7%	11.41%
Total	1,029	2,656	3,272	4,334	6,296	9,945	414,110	100%	2.4%	4.9%	42.2%
Global gw use	–	–	–	–	–	–	982,000	NA	1.0%	NA	100%
Total (km³/year)	1.0	2.7	3.3	4.3	6.3	9.9	414.1	100%	2.4%	4.9%	42.2%

gw groundwater; *–* no reported data; *NA* not applicable

[a] Average annual MAR volume at 2015, based on 2011–2015

[b] Margat and van der Gun (2013)

[c] Global reported capacity is the sum of the volumes shown in this table for 2015

[d] Calculated compound growth rate from 1965 to 2015. If no data were available in 1965, data in the first year reported were used with the corresponding time duration to 2015 to calculate compound growth rate

inconsistently among European countries as reported by Sprenger et al. (2017), but where two estimates were available, the better supported estimate was used. Hence, the table is regarded as the best available conservative estimate of current national and global MAR, and its publication is intended to stimulate more rigorous reporting of MAR in future.

In the 50 years from 1965 to 2015, MAR capacity has grown from 1 to 10 km³/year. The average annual growth rate was 4.5% for the 15 countries with reliable data in 1965 (Fig. 2) and those countries account for 34% of global groundwater use in 2010. Over the same period there was a 2.7% annual rate of growth in groundwater use for nine countries that account for two thirds of global groundwater use in 2010 (Margat and van der Gun 2013). Table 2 suggests that MAR increased by 0.5 km³/year between 2005 and 2015 compared with the increase in groundwater use of 53 km³/year between 2000 and 2010 for the nine reference countries reported by Margat and van der Gun (2013). Even relatively high growth rates in MAR are far from being an adequate solution to over-abstraction of groundwater. However, MAR is a management tool to consider with and complement new efficiency measures in irrigation, switching to low water use crops, conjunctive use of surface water and groundwater resources including substituting use of recycled water for groundwater, and foregoing extraction. Managed aquifer recharge viability depends on ranking the economics of these various options (Dillon et al. 2012) and social acceptance.

India (only estimated for five states) with 31% of reported global MAR capacity in 2015 and the USA (26%) account for the majority of the reported global MAR capacity. Germany ranked third with 9%, most of which is bank filtration for city water supplies that have been in use since before the 1870s (Sprenger et al. 2017). Other European countries and Australia also make modest contributions to the global total, with European contributions largely through bank filtration. Figure 3a summarizes the reported MAR volumetric capacity in 2015 by region, while Fig. 3b shows MAR capacity by region as a percentage of groundwater use in 2010 in only those countries or areas reporting MAR, as per Table 2. Although Asia, Europe and North America have the highest reported volumes of MAR there is enormous variability in MAR uptake within regions that is not explained by groundwater use alone.

Although the five states in India and the USA have high groundwater use, so do China, Latin America and South East Asia where MAR is not yet well established (Fig. 4), suggesting major opportunities for MAR in these regions. Preliminary investigations in heavily developed Chao Phraya basin of Thailand (Pavelic et al. 2012) and in the Ganges Basin of India (Pavelic et al. 2015) suggest that widespread MAR at basin scale could have a vital role in managing water variability and reducing water-related disasters (floods and droughts).

Countries with high MAR capacities in relation to groundwater use such as Germany, Italy, Hungary, and Netherlands, reflect significant long-term reliance on bank filtration in drinking water

Fig. 2 International evolution of MAR capacity by decade from 1960s to 2000s and 2011–2015. This figure only includes the countries or regions where historical estimates from 1965 were available. These 15 countries/areas account for 76% of reported installed MAR capacity in 2015 and for 34% of global groundwater use in 2010. Bar stacks from bottom up follow the alphabetical order of countries as per the legend
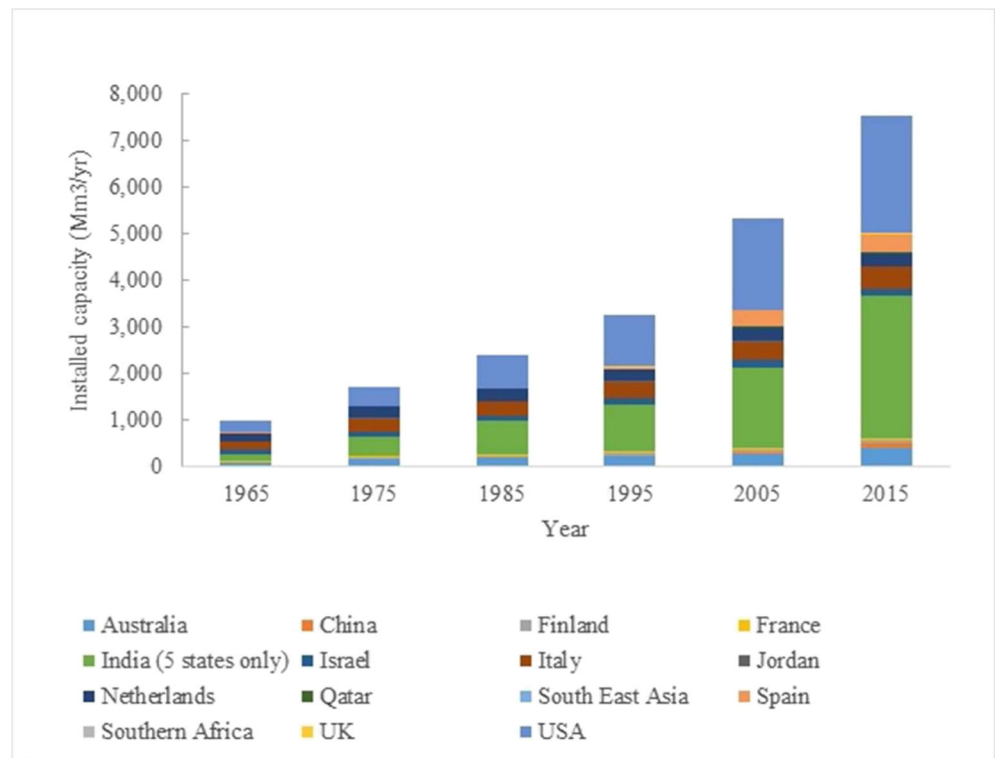
supplies. Australia, with large MAR contributions from urban stormwater and mine water reuse, and Spain and Israel with large infiltration systems for agricultural irrigation, demonstrate growth in systems with diverse objectives. High recent growth rates (>8%/year) for MAR in Finland, Korea, Oman, Qatar and Spain show that MAR is relevant to a wide range of water-resource-management issues. The large variations in commencement and rate of uptake are thought to be more related to information exchange and capability development than to divergence of opportunities at national scales—Sprenger et al. (2017) reported considerable opportunities for uptake in Europe.

The proportion of national or regional groundwater use in 2015 that is contributed by MAR also has a wide range of values, for various reasons. Hungary and Slovakia (91 and 49%) were highest due to the historical dominance of bank filtration for urban water supplies. Other countries where bank filtration constitutes a significant proportion of groundwater use are Germany (28%) and Sweden (13%). Reporting of bank filtration varied among sites. In some cases the total annual groundwater abstraction adjacent to a stream was counted and in others this was scaled by the proportion of extraction that originated from the stream; for future consistency, the latter is recommended for reporting. In several other countries where MAR is commonly used for water quality improvement, there is also a sizable proportion of MAR to groundwater use: Finland (23%), Netherlands (16%), and Switzerland (13%). Among semi-arid to arid areas where recharge of natural water and/or recycled water is
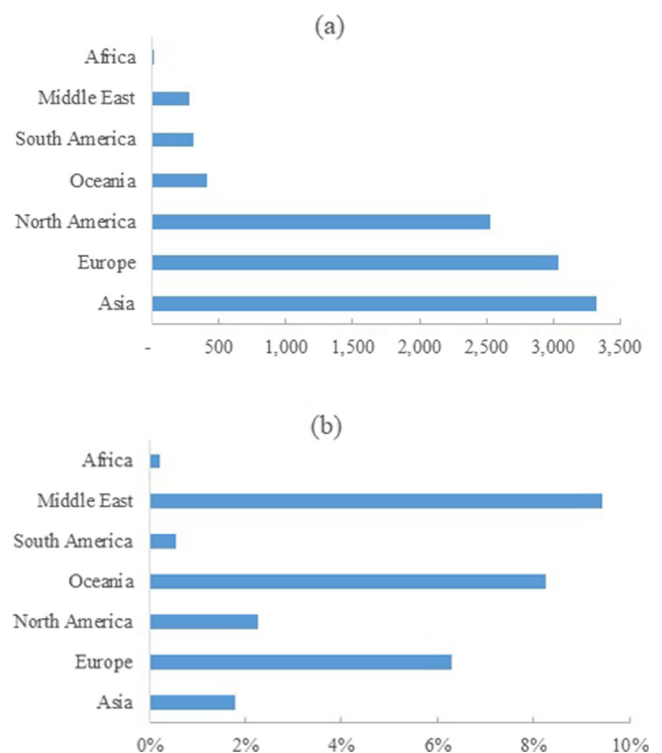

Fig. 3 Reported MAR capacity in 2015 by region expressed **a** volumetrically and **b** as a percentage of groundwater use (from Margat and van der Gun 2013) for reporting countries (or states) of each region: **a** by region (Mm³/year); **b** by region as a percentage of groundwater use in 2010
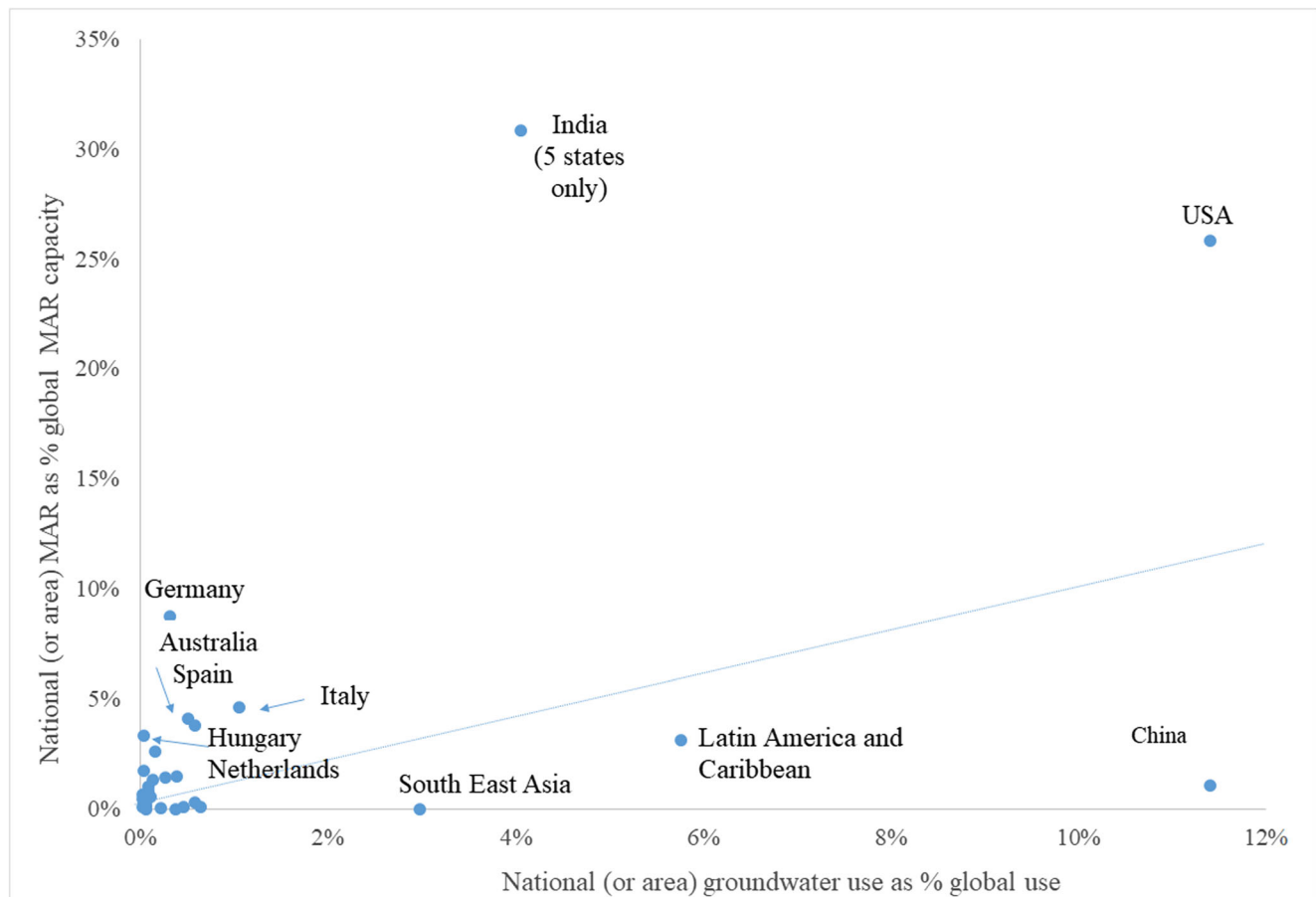
Fig. 4 National or areal MAR capacity as a percentage of reported global MAR capacity versus national or areal groundwater use as a percentage of global groundwater use

practiced, the ratio of MAR to groundwater draft varies: Qatar (17%), Israel (11%), Oman (10%), Australia (8%), 5 Indian states (8%), Spain (7%), Italy (4%) and USA (2.3%).

Notable for the very minor reported contribution by MAR and, where known, low growth rate, are Latin America and the Caribbean (0.5%), Southern Africa (0.2%), China (0.1%), and SE Asia (<0.1%). These regions cover a wide variety of climate, lithologies, and demand for drinking and irrigation water and groundwater stress (Stefan and Ansems 2018; Bonilla Valverde et al. 2018); thus, there is clearly significant potential for uptake of MAR.

Based on current application of MAR it is likely that the demand for MAR where groundwater systems are under stress would be of the order of 10% of water demand; hence the current status of MAR development (~10 km^3/year) is likely to expand to the order of 100 km^3/year. The rate of expansion will depend on having a sound understanding of the capabilities and constraints of the suite of techniques, effective risk management and knowledge of the economics of MAR in comparison with alternatives (Ross and Hasnain 2018).

Development of specific MAR techniques

A wide variety of methods are used for managing aquifer recharge, and they are addressed here in four broad categories—streambed channel modifications, bank filtration, water spreading and recharge wells; while a fifth category, runoff harvesting, used in the IGRAC MAR Portal, refers to any of these methods. The sequence followed here reflects the level of maturity of these approaches from oldest to newest, and the ramping up of research that has enabled these techniques to be refined or developed. Descriptions of the different recharge methods are given in Dillon (2005) and in Stefan and Ansems (2018) as used in the IGRAC MAR Portal (IGRAC 2018), through which all MAR projects may be reported. Figure 5 illustrates the way that the choice of MAR technique is influenced by the local hydrology, hydrogeology and ambient groundwater quality. A gallery of photographs and diagrams of various recharge techniques can be found in ESM2.
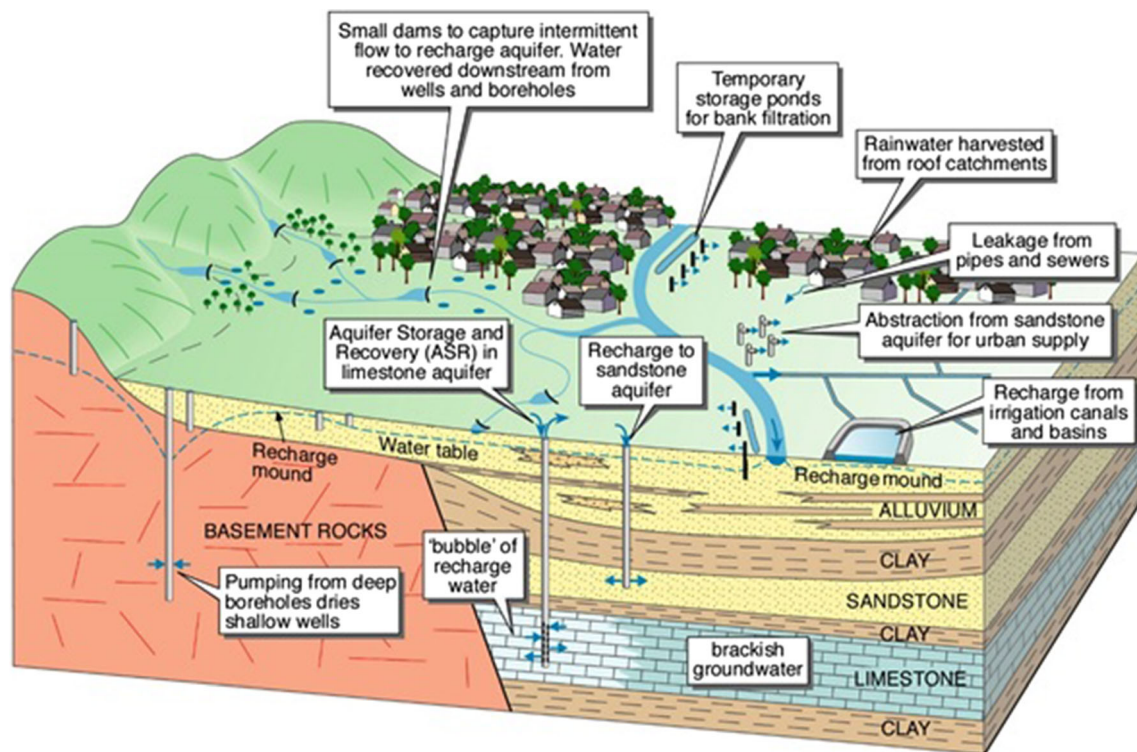
Fig. 5 Managed aquifer recharge is adapted to the local hydrology and hydrogeology, and is usually governed by the type of aquifer, topography, land use, ambient groundwater quality and intended uses of the recovered water. This diagram shows a variety of recharge methods and water sources making use of several different aquifers for storage and treatment with recovery for a variety of uses. An understanding of the hydrogeology of the locale is fundamental to determining options available and the technical feasibility of MAR projects. Recharge shown here occurs via streambed structures, riverbank filtration, infiltration basins and recharge wells. (Adapted from Gale 2005, with permission in Dillon et al. 2009b)

Streambed channel modifications

The information on this earliest form of recharge enhancement is focussed on India, but no doubt also occurred in other semi-arid regions. Sakthivadivel (2007) reports that more than 500,000 tanks and ponds dispersed throughout India have been constructed and some are several thousand years old, as also reported for China (Wang et al. 2014). These have been used to detain surface runoff to supply water for drinking water and irrigation both directly and by infiltration to replenish aquifers. This focus is only on the infiltrated component. Gale et al. (2006) studied three streambed structures and recently Dashora et al. (2018) studied four and reviewed studies of 20 more revealing that infiltration rates from in-stream water detention are one to two orders of magnitude less than that reported for off-stream infiltration basins where flow and quality of water can be controlled. Structures need to be located in such a way that the streambed is scoured naturally by high flow, or else desilting will be required to conserve detention capacity and maintain infiltration rates. They also need to be located taking account of potential hydraulic

connection with groundwater that reduces and even negates recharge, which can complicate assessment of recharge suggesting that several types of measurement methods and calculations be performed such as applied at an extensive drainage depression in southern India (Boisson et al. 2014) and Perrin et al (2012). The Indian government and NGO investment in percolation tanks to infiltrate detained monsoon runoff in drought prone areas has been enormous, and is projected to expand under an ambitious master plan for "artificial recharge" in India by a further 11 million structures in urban and rural areas at an estimated cost of US$10 billion (CGWB 2005, 2013).

The design of MAR structures has changed little since the 1960s when concrete check dams and spillways for percolation tanks were introduced and standardized through guidelines issued by state irrigation departments and the Central Ground Water Board (CGWB 2000, 2007). While there are many papers that conceptually evaluate positioning of streambed recharge structures in relation to geomorphic variables, there is a lack of field measurement and monitoring that would inform policies on MAR density within the context of catchment scale water sharing plans. Figure 6 shows a recent large-

Fig. 6 Ahin recharge dam on the Batinah Plain, Oman, constructed in 1994, is a large dam (crest length 5,640 m, crest height 8 m) with a detention capacity of 6.8 Mm^3. This is one of the 43 recharge dams, with an aggregated capacity of 95 Mm^3 constructed in Oman during the period 1985–2011 (Oman MRMWR 2012). Their purpose is to enhance aquifer recharge primarily to support irrigation; and also to protect the villages and agricultural land in the coastal zone against (previously) devastating flash floods. The dams intercept floods from a catchment with a mean annual rainfall less than 140 mm and potential evaporation around 2,000 mm/year. The detained water is released in a controlled way to recharge the aquifer zone downstream (Photo: Jac van der Gun, taken in 1995)

scale streambed structure, one of many in use in Oman, and more examples are given in ESM2.

Bank filtration

Bank filtration (BF) describes a natural process where surface-water infiltration is induced through nearby groundwater extraction. Bank filtrate can be extracted from dug, vertical or horizontal wells, drains or using other techniques. The raw water abstracted, e.g. from a production well, typically consists of a mixture of infiltrated surface water and groundwater recharged on the landside catchment. Statistics on use of bank filtrate are often not reliable because (1) there is no clear definition of the minimum travel time after which infiltrated surface water could be termed groundwater, as many water companies prefer to deliver seemingly safer groundwater to consumers, resulting in very modest reporting of numbers for BF; (2) the contribution of landside groundwater is often not known or not taken into account, resulting in the reporting of exaggerated numbers for BF. Furthermore, the term river "bank" filtration is often replaced by authors by the term river "bed" filtration to describe it more specifically (Milczarek et al. 2010) or not used at all if the abstraction scheme (e.g. drain pipe) is embedded in the riverbed. As bank filtration at most sites was and is a combination of bank and bed filtration, the term BF should be seen as a general term, which could be further subdivided into river (RBF), lake (LBF) and canal (CBF) bank (and/or bed) filtration, with RBF currently being the most commonly practiced method.

In Europe, BF systems have been in place at a large scale since 1870 (Jülich and Schubert 2001), providing about 50% of the public water supply of Slovakia and Hungary, 9% in Germany (Hiscock and Grischek 2002), 7% in Netherlands (Stuyfzand et al. 2006) and 25% in Switzerland (von Rohr et al. 2014). The city of Budapest (Hungary) is fully supplied with bank filtrate from the Danube River (Laszlo 2003) from 762 wells with a total maximum capacity of 1 million m^3/day. In the US, bank filtration systems have been in use for more than 60 years (Ray et al. 2002), including the world's largest horizontal collector wells with single capacities of more than 150,000 m^3/day (Ray et al. 2003). Today in Europe, BF is mainly used for pre-treatment, the focus lying on attenuation of water quality variations and removal of turbidity, pathogens and organic compounds. In the US, India and Egypt, BF is mainly used to remove particles and pathogens. In some countries, including China and Italy, BF is used to prevent overexploitation of aquifers.

In the two decades following the founding of IAH, only a few publications on BF appeared in Europe, focusing on technical issues and removal of bacteria. Intensive investigations in Germany and Netherlands started after the pollution of the Rhine River by the Sandoz accident in 1986 (Sontheimer 1991) and with further development of analytical techniques for identifying trace organic compounds. In the US, RBF

came into focus between 1990 and 2010 with the Environmental Protection Agency (EPA)'s Groundwater Under Direct Influence of Surface Water (GWUDISW) rules to ensure removal of pathogens (protozoa, viruses; e.g. Tufenkji et al. 2002; Weiss et al. 2005; Ray et al. 2003). Meanwhile, numerous studies have shown bank filtration to be effective in the removal and/or degradation of microorganisms, turbidity, pesticides, dissolved and total organic carbon, and organic micropollutants (e. g. Stuyfzand 1998b; Kuehn and Mueller 2000; Hiscock and Grischek 2002; Jekel and Grünheid 2005; Eckert and Irmscher 2006; Ray et al. 2003; Maeng et al. 2008; Hoppe-Jones et al. 2010; Lorenzen et al. 2010; Henzler et al. 2014; Hamann et al. 2016 and references therein).

A series of conferences and workshops on RBF was organized between 2000 and 2006 with significant support from IAH members. As a result, interest has been growing outside of Europe and the USA to implement RBF as an alternative to surface water abstraction, which faces the problems of turbidity, pathogens and increasing pollution (Ray 2008), especially in Asia. In India, a large potential for RBF was identified for the alluvial deposits along the Ganga River and various tributaries (Dash et al. 2010; Sandhu et al. 2011). Consequently, one EU-Indian and one German-Indian RBF project were started in 2005 and 2008, respectively, and the Cooperation Centre for Riverbank Filtration was established in 2007 at the RBF site Haridwar (India), which was recognized by the IAH Commission on MAR as a demonstration site in 2009. In 2011, the Indo-German Competence Centre on RBF was founded under the guidance of the National Institute of Hydrology, Roorkee, following the approval by the Indian Ministry of Water Resources. The EU-Indian project "Saph Pani" (2013–2015) included a work package on RBF (Wintgens et al. 2016). In parallel, South Korea became a leading country in Asia in constructing horizontal collector wells for RBF (e.g. Lee et al. 2009). In Thailand, a master plan for RBF was developed between 2011 and 2013 and potential areas were selected from the existing 25 river basins in the country (DGR 2013). In Vietnam, existing sites are under investigation as further use of RBF has to take into account disadvantages such as dissolution of arsenic (Postma et al. 2017) and advantages in combination with flood protection (Feistel et al. 2014). The Wakaf Bunut water treatment plant in the state of Kelantan is Malaysia's largest RBF scheme and it operates via a combination of RBF and ultrafiltration systems. The plant was commissioned in March 2013 and is capable of producing a maximum of up to 14,000 m³/day (Chew et al. 2015). Othman et al. (2015) report on investigations at a new RBF pilot site in Sungai Perak, Malaysia, and Mauro and Utari (2011) on a pilot site on the Kurkut River in Indonesia.

Only a few BF sites are known from South America, probably as a result of sufficient (surface) water resources and information sharing limited to national journals. In English-language publications, the main emphasis was given to the removal of turbidity and bacteria (Garnica 2003; Blavier et al. 2014) and cyanobacteria (Freitas et al. 2012; Romero et al. 2014). In Australia, the potential for BF in semi-arid areas is limited, with major aspects reported including algae and brackish aquifers (Dillon et al. 2002).

In Egypt, a core group was formed in the major state water company to promote RBF along the Nile River according to the potential identified (Shamrukh and Abdel-Wahab 2008; Ghodeif et al. 2016)—an example is shown in Fig. 7. Beach wells are also used in Egypt to pretreat seawater before desalination (Bartak et al. 2012). Beach sand filtration is the abstraction of seawater via beach wells or infiltration galleries that are located along a seashore (Voutchkov 2005). Large seawater reverse osmosis plants are in operation at the Bay of Palma plant in Mallorca, equipped with vertical wells having a total capacity of 46,000 m³/day (Ray et al. 2002), in Malta with a combined capacity of 190,000 m³/day, and the Pemex Salina Cruz plant in North America, which uses three Ranney-type collector wells with a capacity of 15,000 m³/day each (Voutchkov and Semiat 2008). Missimer et al. (2013) demonstrated the water quality improvements and economic efficiency of subsurface intakes for seawater reverse osmosis systems.

In countries where new BF schemes are planned, innovative methods for site assessment are needed to address major issues (e. g. Wang et al. 2016) such as induced clogging of river/lake beds (Hubbs 2006; Soares et al. 2010; Pholkern et al. 2015), prediction of attenuation rates and bank filtrate quality as well as further treatment requirements (Wintgens et al. 2016; AquaNES 2016; Sharma et al. 2012b). New technical developments are reported mainly from the US: drilling of angle wells for RBF at the Missouri River, use of an inflatable dam to enhance RBF at the Russian River (Ray et al. 2011) and construction of a tunnel with laterals beneath the Ohio River bed by the Louisville Water Company (Hubbs et al. 2003), finalized in 2011 and exceeding all known abstraction rates per km river length, leading to a high risk of riverbed clogging.

In countries with long-term BF scheme operation, recent issues and developments include: river hydrology and clogging (Martin 2013; Grischek and Bartak 2016), economic and/or technical optimization, modeling redox processes responsible for iron and manganese release and attenuation of micropollutants (Sharma et al. 2012a, b; Henzler et al. 2016), innovative sensing and management schemes (Rossetto et al. 2015), adaptation to changing conditions such as water demand and climate change (e.g. Gross-Wittke et al. 2010, Sprenger et al. 2011, Schoenheinz and Grischek 2011), measures to protect against flooding (Sandhu et al. 2018), and combination with sophisticated post-treatment techniques (AquaNES 2016)—more examples are shown in ESM2.

Fig. 7 Drilling of riverbank filtration wells at the Nile River, Luxor, Egypt, March 2018. Seasonal low river water level, frequent spills of oil and other pollutants and high turbidity during high flow cause problems in surface-water abstraction and subsequent treatment. A short distance between the abstraction wells and the river bank is sufficient to remove particles, to buffer spills and to ensure a high portion of bank filtrate and a low portion of manganese-rich land-side groundwater (Photograph courtesy of T. Grischek, University of Applied Sciences Dresden, HTWD)

Water spreading

Spate irrigation, where floodwater is spread to increase soil moisture for food production on otherwise dry cropping land, has been a widely practiced custom in semi-arid countries (Steenbergen et al. 2010), also unintentionally causing groundwater recharge. However, not until irrigation with groundwater became common in the twentieth century did the spreading of water intentionally in recharge basins begin to be used at scale. This scale-up was founded on two main strands of pioneering research initiated in Arizona (USA) with experimental tests and pilot projects in the 1960s and 1970s, and in Europe centred in the Netherlands.

In Arizona two research organizations carried out most of this work; the United States Water Conservation Laboratory (USWCL), a division of the United States Department of Agriculture, located in Phoenix, and the Water Resources Research Center (WRRC) at the University of Arizona in

Tucson. In the mid-1960s, pilot recharge basins were constructed and operated by Dr. Sol Resnick (WRRC) at the foot of McMicken Dam in Phoenix. In 1967, the USWCL, under the direction of Dr. Herman Bouwer and with some assistance from the Salt River Project (SRP), constructed and operated the Flushing Meadows project, a pilot project that consisted of six long and narrow infiltration/recharge basins excavated in the bed of the Salt River. This project was followed in 1975 by the 23rd Avenue Recharge Project located adjacent to one of the city of Phoenix wastewater treatment plants. It had six recharge basins located on the north bank of the Salt River. The two USWCL projects' source water was treated municipal wastewater which was intermittently infiltrated via basins. These were operated principally to study and develop this form of water treatment and storage which became known as soil aquifer treatment (SAT).

Concurrently, the WRRC carried out research in MAR using both basins and wells. The passing of the 1980 Groundwater Act (Arizona) and the approaching completion of the Central Arizona Project Aqueduct to Phoenix in the early to mid-1980s contributed to the planning for the use of MAR to store the Colorado River (CAP) water. In 1978, the Salt River Project sponsored the first MAR symposium in Arizona. This symposium was followed by another in 1985 and from then on every 2 years. Recurring research themes were hydraulics, solute transport and modelling of MAR operations, causes and management of clogging, geochemistry of aquifer recharge, fate of pathogens and organics, and subsurface water-quality changes. There were also many case studies describing MAR projects, their role in integrated water management, economics, and progress in regulations and governance arrangements. In 1986, the Groundwater Recharge and Underground Storage and Recovery Act was passed by the Arizona Legislature (1994). This law defined the ownership of the surface water stored in the aquifer by managed recharge, and it also defined many other regulatory issues of MAR operations opening the way for the development of underground water storage facilities, mainly those storing CAP water. One of these was the city of Phoenix Cave Creek Recharge Project that would convert many of its production wells to dual-purpose injection and recovery wells to store part of its CAP water allocation. Injection and recovery of water using the same well is known as aquifer storage and recovery (ASR; Pyne 2005) and will be discussed later.

Commencing in 1986, the SRP working closely with several Phoenix area municipalities, and many members of the Arizona Municipal Water Users Association (AMWUA), planned for a large aquifer storage facility. This facility would store surplus CAP water—a site located in the dry bed of the Salt River downstream of SRP's Granite Reef Diversion Dam was selected. After several years of hydrologic, hydrogeologic, engineering and environmental studies at this site, the Granite Reef Underground Storage Project (GRUSP) obtained the

necessary federal and state permits and started operating in 1994. Parallel to the efforts in Phoenix, the city of Tucson developed the Sweetwater Recharge Project to store a portion of its reclaimed water and tested one of their well fields to store treated CAP water. They followed by developing a large surface-water-spreading facility, the Central Avra Valley Recharge Project. Pima County and other water entities started planning, constructing and operating pilot recharge projects.

In the early 2000s, the Central Arizona Water Conservation District (CAWCD), now known as the Central Arizona Project (CAP), started constructing water-spreading recharge facilities in the Phoenix and Tucson areas and became the entity with the largest aquifer storage capacity. They presently own and operate four storage facilities in or near Phoenix and two in Tucson. The stored water in these projects is CAP water. To store their surplus reclaimed water and obtain credits for future reclaimed water uses, many municipalities in the Phoenix area developed their own MAR facilities. These are usually of small capacity using basins, with more entities introducing the use of vadose zone recharge wells because of land constraints. The SRP constructed and operates the New River Agua Fria Recharge Project (NAUSP) at the terminus of their canal system. This basin recharge facility commenced operation in 2008 and presently stores mostly reclaimed water from two municipalities and is also permitted for CAP water storage. The quantities of water derived from CAP and municipal wastewater was quantified in Arizona and resultant increases in groundwater levels recorded in these active management areas (Scanlon et al. 2016).

The GRUSP facility obtained a permit to store reclaimed water from the city of Mesa in 2007 and became a two-water-source-MAR operation. Aquifer storage and recovery, used by

very few water utilities in Arizona, is employed when there are land limitations and also when there are unused production wells that can be retrofitted for recharge. The source water for the ASR wells is predominantly reclaimed water although some store treated and untreated CAP water. Most of the MAR facilities in Arizona are owned and operated by public utilities but there are a few with private ownership like the large MBT Ranch basin recharge facility located west of Phoenix. The increase in the direct use of CAP water has stopped the development of large capacity MAR projects in Arizona. Those municipalities which are fortunate to obtain a water right from a surface storage facility, like the town of Payson, will develop their own nonreclaimed water MAR projects; however, these will be very infrequent in Arizona's semi-arid climate. The majority of new MAR recharge facilities will be for the storage and recovery of reclaimed water. Figure 8 shows an example from Mexico that has been operating since 2007 (Humberto et al. 2018). A substantial research project on intermittent infiltration of treated wastewater (soil aquifer treatment) was undertaken by Fox (2006) and further work has progressed in Israel and Australia and is reported in Stuyfzand and Hartog (2017).

Infiltration basins were also in early use in California, commencing with dam diversions in 1928 to Saticoy spreading grounds north west of Los Angeles and used since 1954 in Orange County south of Los Angeles to assist recharge of water from the Santa Ana River and from tertiary treated wastewater (Mills 2002). Spreading basins were also developed in the Central Valley beginning in the 1960s to support irrigated agriculture (Scanlon et al. 2016). Research on clogging of basins and on water quality changes and water treatment requirements was undertaken in both Arizona and

Fig. 8 At San Luis Rıo Colorado, Sonora, Mexico, oxidation lagoons (at a wastewater treatment plant in the background), have annually discharged 8.2 Mm^3 treated water to intermittent infiltration basins (located at a distance in the middle of the photo) for more than 10 years, and in the foreground some water is starting to be used to establish constructed wetlands (Humberto et al. 2018) (Photo, April 2018, courtesy of Hernández Humberto, Organismo Operador Municipal de Agua Potable, Alcantarillado y Saneamiento de San Luis Rıo Colorado, Sonora, Mexico)

California and reported through several conference series and subsequently in the scientific literature, with summaries of various aspects given by Bouwer (1978, 2002) and Bouwer et al. (2008). In Namibia, recharge basins were constructed downstream of the OMDEL dam in 1997 to recharge an alluvial aquifer in a very arid area. The dam detains floodwater from the normally dry Omaruru River and allows settling of sediment before water is released to recharge basins to replenish the aquifer (Zeelie 2002). A similar approach is being investigated in Saudi Arabia except that ASR wells are to be used instead of recharge basins (Missimer et al. 2014).

In the Netherlands, dune infiltration was also practiced to improve the quality of river water for drinking water supplies and to buffer water supplies. Intensive research there led to improved understanding of the geochemical processes associated with infiltration systems and the consequent fate of organic material, nutrients and pathogens. The introduction of MAR systems in the Netherlands in the mid-1900s raised and continues to nurture many technical and scientific questions. In the period 1940–1975, research mainly focused on the engineering aspects of MAR systems, regarding the minimum travel time needed to remove pathogens, the attenuation of salinity and temperature fluctuations in the infiltration waters, the clogging of basins and wells, and the effects of aquifer passage on main constituents. This knowledge informed much of the handbook on artificial recharge by Huisman and Olsthoorn (1983).

In the period 1965–1985, the worsening quality of the Rhine and Meuse rivers provoked research into the behavior of macroparameters, nutrients, heavy metals and some classical organic micropollutants during detention in spreading basins and aquifer passage (Piet and Zoeteman 1980; Stuyfzand 1989, 1998a). It also stimulated research into the effects of eutrophication on algae blooms in recharge basins and on oligotrophic phreatophytic plant communities in dune valleys around them (Van Dijk 1984). It was discovered in the 1980s that rainwater lenses can form in between infiltration ponds and remote recovery systems, and that flow-through (seepage) lakes in between can disrupt these lenses and stimulate local eutrophication (Stuyfzand 1993). This research was based on multi-tracing to discern infiltrated river water from autochthonous dune groundwater (locally infiltrated rainwater). Later hydrochemical studies yielded further insight in the performance of various (potential) tracers (Stuyfzand 2010), the behavior of trace elements (Stuyfzand 2015), the behavior of organic micropollutants (Noordsij et al. 1985; Hrubec et al. 1986, 1995; Stuyfzand 1998b; Greskowiak et al. 2006; Stuyfzand et al. 2007; Eschauzier et al. 2010) and pathogens (Schijven and Hassanizadeh 2000; Schijven 2001; Medema and Stuyfzand 2002). In Israel, at the Shafdan wastewater treatment plant, soil aquifer treatment of recycled water has contributed significantly to groundwater development over many years (Schwarz et al. 2016). In Italy, since the 2016 release of a regulation for permitting MAR, the first two

infiltration basins have been authorized (one of these is included in a series of photographs of infiltration basins in ESM2).

Various modeling approaches were pursued to simulate and predict the behavior of pollutants, radionuclides, bacteria and viruses, and main constituents during detention in recharge basins and during aquifer passage. One of the first such models was Easy-Leacher (Stuyfzand 1998c), which is a two-dimensional (2D) reactive transport code set in an Excel spreadsheet, combining chemical reactions (volatilization, filtration, dissolution-precipitation, sorption, (bio)degradation), with empirical rules regarding the reaction sequence. It assumes a constant input quality, flow and clogging layer conditions, but takes account of the leaching of reactive aquifer constituents. More sophisticated models were built using the MODFLOW/ MT3DMS and PHREEQ-C based reactive multicomponent transport model PHT3D, including reaction kinetics (Prommer and Stuyfzand 2005; Wallis et al. 2010; Antoniou 2015; Seibert et al. 2016). On the other hand, simpler models set in an Excel spreadsheet were developed such as Reactions+, a mass balance (inverse) model to identify and quantify the inorganic mass transfer between, for instance, the infiltrating surface water and a well downgradient (Stuyfzand 2011), and INFOMI, an analytical model to predict the behavior of trace metals and organic micropollutants (Stuyfzand 1998c).

Recharge wells

Recharge wells were used as early as 600 AD in Tamil Nadu, India, to recharge rainwater collected in ponds to replenish shallow aquifers used as drinking water supplies (Sakthivadivel 2007). It is reported that thousands of these wells still exist in southern coastal areas where aquifers are brackish and are used for a variety of purposes. In northern India, step wells called *baolis*, which are impressive architectural monuments, harvested rainwater from public paved surfaces and increased groundwater supplies, at a likely risk to drinking water quality. In Turkmenistan, Central Asia, Pyne (2005) reports that for several hundred years recharge has been enhanced in an area with 100-mm annual rainfall and silty-clay soils between dunes, by construction of trenches leading to pits within the dunes and recovered from adjacent dug wells. In India, at a smaller scale, traditional household rainwater harvesting schemes have diverted rooftop rainwater into dug wells to freshen and augment water supplies in water short areas. Until the 1960s, such wells spread widely based on local knowledge and hundreds of thousands of these were implemented without government involvement. Over the last 50 years, governments have assisted the spread through provision of scientific information to improve the management of recharge.

The following account of development of recharge well systems focuses on several main areas: Israel, USA, northern

Europe and Australia. In Israel recharge wells started to be used in about 1955 (Harpaz 1971) and by 1967 there were 135 wells recharging 10 Mm^3/year, with scientific advances being recorded concurrently (e.g. Bear and Jacobs 1965). In the US, the first injection wells were established in the 1950s in California to create barriers to seawater intrusion in Orange and Los Angeles counties; ESM2 contains a diagram and photo of the current groundwater replenishment program at Orange County. Subsequent development in recharge technology led to aquifer storage and recovery (ASR), which opened opportunities in confined and brackish aquifers as well as the aquifers for which other techniques may also be used. The first ASR wellfield in the USA that is still in operation is in Wildwood, New Jersey. The wellfield began operation in 1969 and is utilized to meet seasonal peak water demands during summer months. Prior to 1969, the US Geological Survey conducted research investigations at several different sites nationwide, none of which continued in operation after the initial research program was completed, but provided the basis for further development (Asano 1985; Johnson and Finlayson 1989; Johnson and Pyne 1995; Aiken and Kuniansky 2002; Pyne 2005; Maliva and Missimer 2010). Subsequent operational projects were mostly implemented by local government agencies having a need for expansion of water supply capacity or reliability. By 1983, three ASR projects were operational in USA, including two in New Jersey and one in California. The Lake Manatee ASR project in Florida began operation in 1983 and won a major national award in 1984. Publicity from that award galvanized ASR interest and activity nationwide so that by 1995 about 25 ASR projects were operational or in development in several states.

In the late 1990s, the city of Scottsdale, Arizona, started the operation of the Water Campus Facility. This innovative project uses vadose zone recharge (VZR) wells, also called "dry wells" to store advanced-treated-municipal wastewater or treated stormwater in the aquifer. It has now operated very successfully for more than 15 years and is now widely used by many municipalities.

ASR activity accelerated during the late 1990s in, e.g. the United Kingdom, while in Florida it encountered a major setback in 2001. If arsenic is present in specific minerals in the aquifer comprising an ASR storage zone such as arseniferous pyrite, and the recharge water contains oxygen, nitrate and, e.g. chlorine or ozone, the arsenic will mobilize and may occur at concentrations exceeding drinking water standards in the water recovered from an ASR well (Stuyfzand 1998a; NRMMC, EPHC, and NHMRC 2009). Pretreatment of the recharge water to remove oxidants is effective at controlling arsenic mobilization; however, their removal tends to be complex and expensive, while post-treatment to remove arsenic is also expensive. A simple solution, which was demonstrated to be effective, e.g. in Florida, at many drinking water ASR wellfields since 1985, is to initially form and maintain an oxidized zone around the ASR well (Pyne 2005). Mobilized arsenic remains dissolved within the generally anoxic buffer zone, situated between the oxidized zone and the outer anoxic mixing zone. In the oxidized zone, most arsenic is normally precipitated or adsorbed to the aquifer matrix during storage and recovery by subsurface geochemical processes, but can be released if mixed zone water reduces either the oxidation state of the storage zone or the sorptive capacity of amorphous iron oxides (Wallis et al. 2010, 2011). Hence, ASR operations need to monitor volumes and quality stored and recovered, ensuring that none of the buffer zone volume is recovered.

By 2016 over 500 ASR wells in 175 ASR wellfields were operating in USA, spread among at least 25 of the 52 states. Most are storing drinking water; however, others are storing partially treated surface water, groundwater from different aquifers or from the same aquifer at a different location, or highly treated, purified water from wastewater reclamation projects. Aquifer storage and recovery wells are from 50 to 900 m deep in a wide variety of geologic settings, while storage is in confined, semi-confined and unconfined aquifers containing freshwater to brackish groundwater with total dissolved solids concentration up to ~20,000 mg/L. Individual ASR well yields range from ~2,000 to ~30,000 m^3/day. To date, 28 different objectives for ASR projects have been identified, the most common of which are to meet seasonal peak demands, long-term water storage (water banking) and emergency storage. Other common applications of ASR are to maintain flows and pressures at distal locations in water distribution systems, and to reduce disinfection by-products and seasonal elevated water temperatures. Most ASR wellfields meet multiple water demand and water quality objectives—an example from Florida is shown in ESM2.

In Europe, a more research-oriented approach to recharge-well development was underway during the period 1973–1982, when extensive research on the clogging mechanisms of infiltration wells was carried out by Kiwa (renamed KWR in 2006). This yielded the new clogging potential indicators Membrane Filter Index (MFI; Schippers and Verdouw 1980) and Assimilable Organic Carbon (AOC; Hijnen et al. 1998). Also, the insight was born that cumbersome clogging can only be prevented by thorough pretreatment (that included at least a coagulation step and rapid sand filtration), leading to MFI <2 and AOC <10 μg C/L, combined with frequent back-pumpings of short duration (Olsthoorn 1982; Peters et al. 1989). By the 1990s, a large-scale ASR project was operating in northern London, England, UK (Pyne 2005).

The clogging of wells or drains has always been a hot topic in MAR systems because of their extreme vulnerability. Studies by van Beek (2010) among others, revealed that infiltration basins and recovery wells in aquifer storage transfer and recovery (ASTR) systems (that is, separate injection and recovery wells) are more vulnerable to (bio)chemical clogging by hydrous ferrihydrite, whereas bank filtration wells in the anoxic fluvial plain

are prone to clog by aquifer particles that are retained by the borehole wall if damaged by residual drilling muds. This growing understanding of the aquifer biogeochemical processes provided a platform to enable intelligent design to avoid these issues in well recharge systems. Much of the research methodology developed in the Netherlands on water quality, and described in the preceding section 'Water spreading', was also applied to geochemical, microbiological and organic chemical changes near recharge wells. Their use has grown in the Netherlands for drinking water supplies, in part due to tensions between use of dunes for wildlife habitat in nature reserves and for natural water filtration in public water supplies. Aquifer storage and recovery is being applied for drinking water supply only on a very small scale (Stuyfzand et al. 2012) however, it is rapidly expanding in the supply of (1) rainwater from roofs for crop irrigation in greenhouses, and (2) freshwater for irrigation of orchards (Zuurbier 2016). Other work in Europe includes evaluation and prediction, based on water quality, of the timescale for clogging around injection wells that form a barrier to seawater intrusion (Masciopinto 2013).

In Australia, ASR had captured the imagination of water managers and users particularly in urban areas, and in the early 1990s the method was in use in South Australia for harvesting winter stormwater, storing in limestone or hard rock aquifers that originally contained brackish groundwater, with effective recovery of freshwater for irrigation of parks and gardens in summer. An urban stormwater ASR research site was established in 1993 at Andrews Farm, to evaluate the effectiveness of injection and to understand subsurface processes affecting mixing and water quality. Subsequently in 1996, the City of Salisbury established at The Paddocks its first ASR project, as described in ESM2. Then in 1996–2005, the first recycled water ASR trial began at Bolivar, South Australia, and resulted in substantial advances in measurement methods, modelling and process understanding (Dillon et al. 2003; Greskowiak et al. 2005; Pavelic et al. 2006a, b, 2007; Ward et al. 2009; Vanderzalm et al. 2009; Page et al. 2010a). Both sites subsequently led to 49 ongoing ASR projects of 0.01 to 1 Mm^3/year in Adelaide (Kretschmer 2017), recharging 20 Mm^3/year and enabled ground-truthing of the Australian Guidelines for MAR (NRMMC, EPHC and NHMRC 2009) as well as their use as examples of applying the guidelines (Page et al. 2010b), combining complementary natural and engineered treatments for water recycling (Dillon et al. 2008) and expansion of MAR in Australia (Parsons et al. 2012). In 2006, a research project to evaluate the effectiveness of stormwater ASTR in brackish aquifers commenced at Parafield in the nearby city of Salisbury and by 2011 this was the hub site of major research project to evaluate the risk management requirements for use for stormwater ASR to
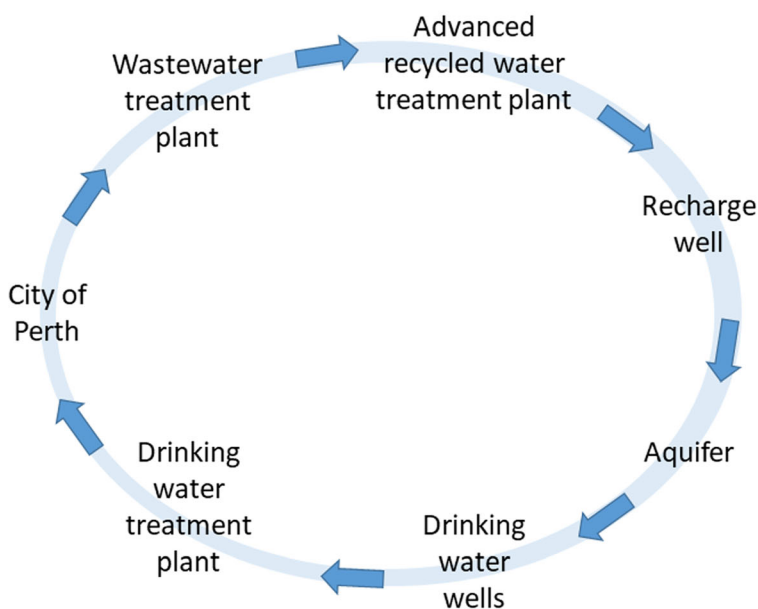

Fig. 9 Perth Groundwater Replenishment Project, Western Australia which commenced operations in 2017 at 14 Mm^3/year using advanced treated recycled water to recharge a confined aquifer that is an important contributor to Perth's drinking water supply. It will double its annual recharge by 2019 when treatment plant and a total of four wells will store water in the Leederville and Yarragadee aquifers. The project will prevent saline intrusion and allow expansion of use of the groundwater system to meet water supplies in an area experiencing a drying climate where surface water supplies have reduced over the last 40 years and population has steadily increased. (Photo and diagram courtesy of Water Corporation, Western Australia)

produce drinking water in Adelaide, and was found to have a lower cost and had higher public acceptance than seawater desalination (Dillon et al. 2014b).

Following on from the Bolivar research, Scatena and Williamson (1999) showed the potential for ASR in Perth, Western Australia, and Toze and Bekele (2009) led a study on MAR pilot projects in Perth. Subsequently the water utility undertook extensive water treatment and injection trials having deep engagement with health and environmental regulators and the public on groundwater replenishment with advanced treated recycled water. Injection wells were separate from the drinking-water-supply wells in the same aquifer. Intensive water quality monitoring was undertaken and aquifer geochemical interactions studied (e.g. Patterson et al. 2011; Seibert et al. 2016). The trials were successful on all dimensions and groundwater replenishment with recycled water was approved in 2013 as the next water supply for Perth. In 2017, the 14 Mm^3/year stage 1 of the groundwater replenishment system was commissioned (Water Corporation 2017) and its capacity will be doubled in 2019. This project won a Global Water Award in 2017 (Global Water Awards 2017) and is the first step of a plan to replenish via wells more than 100 Mm^3/year, enough to source 20% of Perth's water by 2050. Figure 9 contains a diagram and a photo of the first recharge well.

Lawrie et al. (2012), in seeking groundwater resources in a semi-arid western New South Wales (NSW, Australia), undertook extensive airborne electromagnetic studies, drilling, geomorphic, geochemical, hydrogeological and clogging studies and with an innovative integrating analysis identified several compelling opportunities for recharge enhancement via wells adjacent the Darling River near Menindee, NSW. This 10 Mm^3/year water supply for the city of Broken Hill using ASR which has been priced at less than half the projected cost

of a surface-water supply, during drought would provide higher security and reduce competition for water.

Research and communications to support MAR

Considerable research in recent years has helped advance the understanding of natural processes involved in MAR and the design of any complementary engineered processes, and how to better manage such systems in a widening array of hydrogeological settings. This summary paper demonstrates the progress made in a number of areas; however, the objective of this section is not to be an exhaustive literature review and the authors recognise that many high quality and important papers are not cited. Much research is encapsulated here by reference to anthologies rather than the numerous individual specific contributions these contain.

Two significant symposia series initiated in USA have helped to bring scientific focus to the practices of MAR and help advance from trial and error approaches, and local traditional knowledge, to a scientific footing giving greater assurance of technical viability, water quality protection and improvement, environmental restoration, economic feasibility, community acceptance and resilience of systems. The Salt River Project convened the First Symposium on Artificial Recharge in 1978 in Phoenix, Arizona. This has now extended to 16 biennial symposia subsequently organized by Arizona Hydrological Society and now run jointly with Groundwater Resources Association of California and known as the Biennial Symposia on Managed Aquifer Recharge (BSMAR). In 1988, the American Society of Civil Engineers (ASCE) conducted the First International Symposium on Artificial Recharge of

Table 3 Papers, presented in the International Symposia on Managed Aquifer Recharge (ISMAR) series over the period 1988–2016, that have been published

Date	ISMAR	Location	No. of papers	Proceedings or special issues [a]	Reference
1988	ARG1	Anaheim	63	B	Johnson and Finlayson (1989)
1994	ARG2	Orlando	84	B	Johnson and Pyne (1995)
1998	TISAR	Amsterdam	83	B	Peters et al. (1998a)
2002	ISAR4	Adelaide	91	B	Dillon (2002)
2005	ISMAR5	Berlin	133	eB	Fritz et al. (2005)
2007	ISMAR6	Phoenix	124	B	Fox (2007)
2010	ISMAR7	Abu Dhabi	115	eB	Herrman (2010)
2013	ISMAR8	Beijing	122	SIJ-17	Zhao and Wang (2015)
				SIJ-12	Sheng and Zhao (2015)
				SIJ-14	Megdal and Dillon (2015)
2016	ISMAR9	Mexico City	88	SIJ-18	Stuyfzand and Hartog (2017)
				SIJ-18	Dillon et al. (2018)
All	–	–	903	B/eB-7, SIJ-79	–

[a] *B* book, *eB* e-book, *SIJ-18* special issue of a journal with 18 papers

Groundwater at Anaheim, California, that commenced what is now known as the International Symposia on Managed Aquifer Recharge (ISMAR), since IAH and UNESCO joined with ASCE in organizing these in Amsterdam in 1998. On two occasions when timing and location has been favorable, the national and international conference series have merged (in Phoenix 2007 and in Mexico City 2016). The number of papers at each symposium is shown in Table 3.

An evaluation of the topics under which papers were presented showed some perennial themes. These include the description of design, operation, management and impacts of MAR systems. Also, clogging of recharge systems and hydraulic evaluation of fate of recharged water and the ability to recover it. For clogging, in spite of huge progress in understanding mechanisms (e.g. Olsthoorn 1982; Baveye et al. 1998; Rinck-Pfeiffer 2000; Perez-Paricio 2001; Pavelic et al. 2006a, b; Wang et al. 2012; Pedretti et al. 2012; Martin 2013; Newcomer et al. 2016; Xia et al. 2018 among many others), lack of standardized predictive instruments (Dillon et al. 2016), and the previous lack of adequate water quality monitoring and geochemical, mineralogical and biological evaluations at operational sites has inhibited the formation of better predictive tools and more efficient management. A Working Group of the IAH Commission on MAR has produced one monograph on clogging (Martin 2013), and a subsequent monograph on management of clogging is in preparation to help address this.

In general, water quality is better reported in recent symposia, with geochemical evaluations now quite common, particularly for well injection systems, and there is better information on water quality improvements in aquifers particularly for organic chemicals. This new knowledge is also of value more widely in hydrogeology—for example in contaminated site remediation where introduced volumes of water and masses of constituents are normally unknown. In MAR, the stoichiometry can be explicitly defined; similarly, mixing processes and biogeochemical reactions in natural aquifer systems are generally inferred after equilibrium, whereas in MAR the kinetics of these processes are also observable at field scale. Isotopes have been used to study origin and age of ambient groundwater, mixing processes and travel times of recharged water and biogeochemical processes such as denitrification, sulfate reduction, fate of organic carbon and dissolution of minerals due to disequilibrium. The IAEA (2013) provides an anthology of methods and their numerous applications to MAR investigations.

The rates of attenuation of pathogenic micro-organisms and toxic or carcinogenic trace organic chemicals measured at MAR sites or in relevant laboratory experiments have been assembled and discussed by Drewes et al. (2008), NRMMC, EPHC, and NHMRC (2009) and Regnery et al. (2017). Large variations in attenuation rates are partially explained by environmental variables (such as temperature and redox state) and

co-metabolites (e.g. labile organic carbon) and aquifer minerals (e.g. those containing iron); however, site-specific studies may be needed to meet the requirements for risk assessment and approval for reliance on aquifer treatment. The developed understanding of attenuation processes has led to the coupling of bank filtration and surface spreading to more effectively treat water through a sequence of contrasting environmental conditions. Sequential managed aquifer recharge technology (SMART) as it has been termed by Regnery et al. (2016) has now been applied at field scale in Colorado demonstrating improved degradation of some trace organic chemicals. Modelling of flow and water quality changes in MAR operations has also been extensive and a review of the range of models (unsaturated/ saturated flow, solute transport and reactions, geochemistry and clogging) and their uses in planning, design, and improving operations at MAR sites for all types of MAR are summarized by Ringleb et al. (2016). A recent example by Rodríguez-Escales et al. (2017) simulates improved degradation of organics by varying the flow fields beneath infiltration basins to vary redox conditions.

In recent years there has been increased reporting of economic impacts of MAR and governance arrangements (e.g. Megdal and Dillon 2015). Ross and Hasnain (2018) have recently proposed a systematic methodology to calculate the costs of MAR schemes and inform future investment in MAR including water-banking systems where benefits accrue in future droughts of unknown timing and magnitude.

Significant publications on MAR in the Spanish language are also available, and de la Orden and Murrillo (eds) (2009) and Escolero Fuentes et al. (2017) have highlighted advances in MAR developed and relevant to Spain and Latin America, respectively, but are also broadly applicable.

There is evidence in these papers and elsewhere of repetition of past problems of similar sites suggesting some proponents are unaware of experience previously documented. This also partially explains the prolific number of projects that are reported as "world firsts". Clearly, these symposia could play a more valuable role in facilitating information exchange and giving opportunity for more reliable and efficient MAR, particularly to those attempting their first projects. Until ISMAR7 in 2010, all papers presented at these symposia were published in a hardbound proceedings or were available to download from the website (ISMAR5, ISMAR7). However, for ISMAR8 and ISMAR9, only abstracts and posters were published on the web and selected

Table 4 Indicative number of peer-reviewed journal papers published in the field of MAR by decade

Years	1960s	1970s	1980s	1990s	2000s	2011–2017
No. of papers	7	69	95	47	115	275

papers were extended, reviewed, revised and published as special issues of three journals and two, respectively, in an effort to document noteworthy research and investigations more comprehensively.

In the past, when literature searches were painstaking tasks, the US Geological Survey provided the very helpful service of publishing annotated bibliographies on artificial recharge including Todd (1959), Signor et al. (1970) and Weeks (2002). A SCOPUS search (May 2018) for journal papers (articles and reviews) on "managed aquifer recharge", "artificial recharge" or "water banking" in the title of the paper has shown that the number of such papers has grown considerably (Table 4); this narrow search would not have detected most papers cited in this current paper. While it is likely that papers in earlier years are under-represented by electronic bibliographic services, there has been a substantial growth in research and information sharing over the last two decades that is showing no sign of abating.

Considerable headway has been made through concerted efforts around the world, and including multinational collaborations in projects financed by the European Commission— Artificial Recharge of Groundwater (EC 2001), ArtDemo, AquaRec, Reclaim Water, Saph Pani, GABARDINE, DEMEAU, DEMOWARE, MARSOL, H2020 AquaNES, LIFE REWAT, IMPROWARE—and of the Water Research Foundation and Water Reuse Foundation of USA. There are still however knowledge gaps due to the intersection of different hydrogeology, groundwater quality and surface-water quality at each new site, although with decreasing predictive uncertainty as the number of documented sites expand. In Europe, the Action Group MAR Solutions - Managed Aquifer Recharge Strategies and Actions (AG128) was started within the European Innovation Partnership on Water, aimed at involving the principal stakeholders and small and medium enterprises (SMEs) and transferring project results into guidelines and policy to facilitate uptake of MAR.

Far more can be done with better documentation of existing operating sites, transparent reporting of problems and effectiveness of solutions. There are few sites where effects of different treatments or different aquifer properties can be compared unconfoundedly. Systematic evaluation, validation and comparison of methods to predict clogging and efficient means to manage it are still awaited. Aquifer microbiological ecosystems evaluation methods are warranted to provide a health check on sustained aquifer attenuation capacity for contaminants particularly in changing geochemical conditions. The gap in knowledge of water treatment requirements for MAR systems is closing, but could do so at a faster rate with improved risk assessments and probabilistic approaches applied to mixing processes in aquifers, and thus on recovered groundwater. The water quality and mixing aspects where MAR is used for long-term water banking and as saline intrusion barriers would be helped by improved aquifer and aquitard characterization, accounting for parameter uncertainty and density-dependent flow would also help build confidence for investment.

Methods for mapping of MAR opportunities are still diverse, and remain poorly founded in the absence of comparative information among methods and in relation to practical experience. Several areas that have been mapped are currently reported in IGRAC MAR Portal, but too often the huge value of aquifers is overlooked due to lack of awareness of their potential.

Operational performance of ASR systems is much better known than the far more abundant and longer-standing streambed modifications, essentially due to lack of basic monitoring of the latter. This warrants comparative evaluations with multiple methods across multiple sites and then investment in appropriate training of local custodians, and sharing of data to enable synthesis and feedback.

Evolution of governance of MAR

Clearly, MAR implementation is proceeding at pace, fuelled by need and with the management aspects supported by research that improves risk assessment on resource sustainability and water quality. To ensure MAR continues to generate its intended benefits and avoids excessive piezometric pressures or waterlogging, failure during drought, and pollution of aquifers, water resources management and environment protection authorities need to be familiar with the opportunities and constraints of MAR. This is most efficiently controlled by setting soundly based policies and guidelines to ensure that MAR is undertaken in a way which protects the status of groundwater and the requirements of its receptors, including the wider environment.

State policies such as in Arizona, California and Florida, have also been developed for the specific types and purposes of MAR in those states—for example in Arizona, state-supported aquifer recharge was permitted under the Underground Storage and Recovery Act, 1986, and the Underground Water Storage, Savings and Replenishment Program, 1994, in the most developed MAR regulatory system that involves three permits. "Underground storage facility permits" require that the proponent demonstrate: technical and financial capability, that the storage is hydrologically feasible, no unreasonable harm would be caused by water levels or water quality, and they have a right to the floodplain for building a detention basin. A "water storage permit" is needed to allow an entity with an excess renewable supply to store water at a permitted storage facility, and this gives the same entitlement of stored water as for its source. Thirdly, "recovery well permits" are issued to allow recovery of the equivalent volume of water stored, whereby recovery may be outside the area of hydrologic impact of the recharge, provided it is consistent

with a management plan that constrains the rate of drawdown and proximal impacts.

A policy framework for MAR was developed in Australia on entitlements to use a water source for recharge, entitlement to recharge (that there is available aquifer capacity) and entitlement to recover (Ward and Dillon 2011). The entitlement to recover is transferable subject to constraints on impacts, and accounting for depreciation of stored volume particularly in brackish aquifers or those with a steep hydraulic gradient) and has standard end-use conditions relating to water use efficiency and acceptable impacts on nearby groundwater users. The framework is intended to give flexibility in use of MAR in water trading and water banking and adheres to a national system of robust water entitlements. This presents a possible model for consideration although in some jurisdictions current groundwater planning and management rules do not provide a secure entitlement to recover water stored in aquifer or allow recovery after an extended time period (beyond 3–5 years; Ross 2017).

Australian national guidelines for MAR (NRMMC, EPHC and NHMRC 2009) are the only risk-management-based guidelines that conform with the World Health Organization's water-safety-planning approach and assure protection of human health and the environment. They not only apply to all types of source waters, aquifers, recharge methods and end uses of water, and account for water quality changes within the subsurface, but they also follow a staged approach starting with a desktop assessment, investigations, commissioning, and monitoring and reporting to provide a pathway to demonstrating that risks are effectively managed. Water quality hazards addressed, based on results of recent research include the following—pathogens, inorganic chemicals, salinity and sodicity, nutrients (nitrogen, phosphorus and organic carbon), organic chemicals, turbidity and particulates, and radionuclides. These guidelines also address hazards associated with pressure, flow rates, volumes and groundwater levels, contaminant migration in fractured rock and karstic aquifers, aquifer dissolution and stability of well and aquitard, aquifer and groundwater-dependent ecosystems, and energy and greenhouse gas considerations. Nine examples of applications of these guidelines applied to case study MAR projects (Page et al. 2010b) have assisted uptake. Risk-based guidelines are data intensive and so "a stepping stone" guideline for water quality for MAR in India was produced for "natural" water sources using visual observations within a water-safety-planning framework applied at the village level (Dillon et al. 2014a).

Capone and Bonfanti (2015) reviewed European legislation regarding water policy and groundwater quality protection relevant to MAR. The Water Framework Directive and the Groundwater Directive recognize MAR as a water management tool which may be used for supporting the achievement of good groundwater status, but require member states to enact their own policies in relation to the application of MAR, respecting as a minimum the "prevent and limit" requirements of the directives, which entails taking all reasonable measures to ensure the prevention of pollutants reaching groundwater (European Commission 2007). The reviewers found differences among established national legislations and a lack of a comprehensive legal framework dealing with MAR schemes in each surveyed member state. In Italy, regulation was issued in 2016 requiring compliance with the EU Water Framework Directive through two stages of project development and at least 1 year of monitoring regarding quality and quantity.

In the USA, the US Environmental Protection Agency (USEPA 1974) has explicit federal provisions encompassed in the "Underground Injection Control Regulations and Safe Drinking Water Act" that apply in each state unless the state has its own regulations that are at least as strict. These cover the requirements for design of injection wells and the quality of water that may be injected and the monitoring to be undertaken. The "Safe Drinking Water Act" also has provisions to protect drinking water sources that envelope infiltration systems, and for which some soil attenuation capacity in the unsaturated zone is considered. In general, these apply not only to the quality of recharge water but also include allowance for changes in water quality that may occur due to travel time and distance in an aquifer, including metals mobilization and attenuation. Monitoring is required so that water quality changes can be detected. ASCE Environmental and Water Resources Institute is currently revising its guidelines on MAR, intended for release in 2019, which will advise proponents on how to develop MAR projects.

In India there is a government manual on artificial recharge (CGWB 2007) which specifies how to plan, design, monitor levels and water quality and evaluate the economics of recharge augmentation by streambed recharge structures and urban rainwater harvesting. For natural water sources, a water quality guide to MAR in India was developed based on UN Water Safety Planning approach, and capable of use based on visual observations by trained villagers (Dillon et al. 2014a). China and New Zealand are both considering the development of health and environmental guidelines and policy frameworks for MAR.

It is evident that the governance frameworks need attention in many countries to ensure that MAR is sustainable and protects groundwater quality. Monitoring of existing operations and maintaining a public repository of site information, reports and data is a fundamental starting point for providing assurance of effective operations and for developing the information to assist future uptake of MAR including research and governance.

Conclusions and next steps

In a period of 60 years, there has been remarkable growth in MAR and a growing awareness of its potential to replenish over-allocated aquifers, restore brackish aquifers, and even enable energy recovery. However, the rate of growth of MAR has not kept pace with the global rate of groundwater depletion, and much more needs to be done in levering from MAR to facilitate demand management and engage communities in cooperative management of groundwater resources. Development of MAR has occurred at different rates among and within countries for various reasons, including aquifer availability for MAR, the level of awareness and confidence in MAR among water stakeholders, and having clear approval processes. While the currently reported annual volume of MAR is only 1% of global groundwater use, in some countries it is considerably higher (especially where bank filtration is practiced), suggesting that global opportunities are only just starting to be tapped.

The growth in research has enlarged the MAR repertoire, especially using wells, widened the types of source waters for recharge, reduced the costs of water treatment for sustainable operations, improved the quality and quantity of recovered water, and given greater certainty for safe and efficient operation of MAR systems. In spite of these advances, there remain a number of basic steps that would improve efficiency of investment in MAR and underpin the uptake of MAR where this is currently low. These can be expressed in the categories of extending case study information to include economic evaluations, extending research on fundamental processes to better site, design operate and monitor MAR operations, and translating scientific evidence into governance arrangements for water allocations and water quality protection.

Documenting exemplary case studies (particularly those relevant for developing countries, such as compiled by Tuinhof and Heederick 2003) and through symposia discussed earlier, should instill confidence among those who are yet to apply MAR so that, if good practices are followed for site selection, investigation and implementation, it will be reliably successful. Current efforts to form a global inventory of MAR (Stefan and Ansems 2018; IGRAC 2018) will help with identification of geographically and typologically proximal MAR sites for those considering locally pioneering projects. More work is needed to document the costs and benefits of MAR (e.g. Ross and Hasnain 2018), including work in relation to alternative water supplies or places of storage and in identifying scenarios where MAR is likely to produce the least-cost water supply and greatest benefit accounting for all objectives, including current economic externalities such as resource and environmental benefits. In particular, considering the promise

of riverbank filtration, this lacks assessment of costs and benefits. Similarly, considering the proposed magnitude of investment in streambed modification and distributed detention, evaluation is warranted at the catchment scale, accounting for maintenance and environmental flow requirements and downstream benefits and costs. National monitoring and research programs are warranted, initially sized at 2–10% of the planned investment in new recharge infrastructure, in order to steer this investment to maximise net benefits. Recharge structures have greater potential to be used to reinforce irrigation community expectations and efforts at reducing demand on groundwater, through a range of water and soil conservation measures.

While much research on subsurface physical and chemical processes in MAR has been valuable for informing solutions to local problems, more could be done to synthesize what has been learned and extend the benefits. Standardizing methods to assess and predict clogging and treatment and remediation requirements to manage it, and of methods to cost-effectively validate the fate of viruses in aquifers under a wide range of scenarios will help advance MAR. Current MAR research that warrants continuing includes—innovations to optimize ASR systems in brackish to saline aquifers (e.g. Ward et al. 2009; Zuurbier 2016), optimizing ASR systems for drinking or rainwater storage by reducing adverse water-sediment interaction (Antoniou 2015), improving water-spreading-systems capacity to cope with variable and intermittent inflows and changing redox conditions, while protecting wet dune valleys and reducing water quality problems. There will be an ongoing need to determine and predict the behavior of emerging priority pollutants such as pharmaceuticals, personal care products, new pesticides, flame retardants and nanoparticles for use in risk assessments.

Some documents now exist at the national level in only a few countries that provide guidance for health and environmental protection at MAR operations. This could be extended and made easier through use of modern sensor networks and data acquisition and control systems to facilitate decision support and risk analysis. A few jurisdictions have governance requirements that improve security of water resources entitlements generated through MAR, and documenting this experience would provide guidance on the effectiveness of alternative candidate regulatory pathways elsewhere. More regulatory effort in building water security through MAR for longer-term water banking, and in conjunctive use of dams and groundwater could create extra value out of existing dams. Furthering the knowledge of downstream impacts of MAR operations in catchments is needed. Most countries need governance frameworks strengthened to ensure that MAR is sustainable and protects groundwater quality and generates benefits for all members of groundwater-dependent communities,

particularly during drought. Fundamental steps include monitoring of existing operations and keeping a public repository of site information, to assist new MAR developments and the formation of effective evidence-based governance arrangements.

There will be a continuing need to share new knowledge on MAR widely, using seminars, training workshops, linking with planners, local governments, community groups and water users to ensure that appropriate investigations are made before construction and that operators fully understand the challenges. For micro-scale systems such as rainwater harvesting there needs to be adequate local technical support to avoid potential problems. There is much to be done and IAH will have an ongoing role with other organizations in advancing MAR. The IAH Commission's goal is to make all new MAR projects sustainable and safe, based on sound scientific evidence, thereby providing a pathway to increased confidence and wise use of MAR within the groundwater management portfolio, and ultimately maximizing its appropriate use.

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