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Research article

Developing managed aquifer recharge (MAR) to augment irrigation water resources in the sand and gravel (Crag) aquifer of coastal Suffolk, UK

K.M. Hiscock [a,*], N. Balashova [a], R.J. Cooper [a], P. Bradford [b], J. Patrick [b], M. Hullis [c]

[a] School of Environmental Sciences, University of East Anglia, Norwich Research Park, Norwich, NR4 7TJ, UK
[b] Felixstowe Hydrocycle Ltd, Unit 3b, Tuddenham Road Business Centre, Tuddenham Road, Grange Farm, Ipswich, IP4 3QN, UK
[c] Growth, Highways and Infrastructure, Suffolk County Council, Endeavour House, Russell Road, Ipswich, IP1 2BX, UK

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ABSTRACT

Managed aquifer recharge (MAR) offers a potential innovative solution for addressing groundwater resource issues, enabling excess surface water to be stored underground for later abstraction. Given its favourable hydrogeological properties, the Pliocene sand and gravel (Crag) aquifer in Suffolk, UK, was selected for a demonstration MAR scheme, with the goal of supplying additional summer irrigation water. The recharge source was a 4.6 km drainage channel that discharges to the River Deben estuary. Trialling the scheme in June 2022, 12,262 m^3 of source water were recharged to the aquifer over 12 days via a lagoon and an array of 565 m of buried slotted pipes. Groundwater levels were raised by 0.3 m at the centre of the recharge mound with an approximate radius of 250 m, with no detrimental impact on local water features observed. The source water quality remained stable during the trial with a mean chloride concentration (133 mg L^{-1}) below the regulatory requirement (165 mg L^{-1}). The fraction of recharge water mixing with the groundwater ranged from 69% close to the centre and 5% at the boundary of the recharge mound, leading to a reduction in nitrate-N concentration of 23.6 mg L^{-1} at the centre of the mound. During July–September 2022, 12,301 m^3 of recharge water were abstracted from two, 18 m boreholes to supplement surface irrigation reservoirs during drought conditions. However, the hydraulic conductivity of the Crag aquifer (~10 m day^{-1}) restricted the yield and thereby reduced the economic viability of the scheme. Construction costs for the MAR system were comparatively low but the high costs of data collection and securing regulatory permits brought the overall capital costs to within 18% of an equivalent surface storage reservoir, demonstrating that market-based mechanisms and more streamlined regulatory processes are required to incentivise similar MAR schemes.

1. Introduction

Population growth and the development of agriculture, industry and tourism worldwide have led to high demand for water resources and increased groundwater abstractions (Gleeson et al., 2012; UNESCO, 2022; Bhattarai et al., 2023; Schipanski et al., 2023). When groundwater is systematically over-exploited, withdrawn groundwater volumes cannot be easily replaced by the natural recharge of aquifers (Casanova et al., 2016). The over-exploitation of groundwater resources can have an adverse impact on groundwater quantity and quality, for example unsustainable abstraction of major aquifers (Cui et al., 2022; Ji and Senay, 2023) and saltwater intrusion in coastal aquifers (Hingst et al., 2023). The deterioration of groundwater quality from multiple chiral pollutants (for example, pesticides, polychloro-biphenyls, polyaromatic hydrocarbons and brominated flame retardants; Basheer, 2018a) and the pervasive presence of new emerging contaminants such as endocrine-disrupting pharmaceutical residues and their metabolites in the environment (Ali et al., 2009; Burri et al., 2019) are a further serious threat to groundwater resources. Faced with these pressures, innovative ways of treating and modelling contaminated water resources to control environmental contamination (Basheer, 2018b; Basheer and Ali, 2018) and of managing groundwater resources are needed to conserve aquifers to sustain groundwater use (Taylor et al., 2014; Cuthbert et al., 2023).

An example of a sustainable groundwater management technique for improving or maintaining aquifer levels and enhancing groundwater recharge is managed aquifer recharge (MAR), which is the purposeful recharge of water to aquifers with subsequent recovery (re-abstraction) or environmental benefit (Ross and Hasnain, 2018; Stefan and Ansems,

* Corresponding author.
E-mail address: k.hiscock@uea.ac.uk (K.M. Hiscock).

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2018). MAR encompasses a range of recharge methods with the four main categories being in-channel modification, bank filtration, water spreading and recharge wells (Dillon et al., 2019). A fifth category of runoff harvesting is recognised by IGRAC (2023). MAR can restore over-used or brackish aquifers, enhance water quality, protect groundwater-dependent ecosystems and improve water security (Dillon et al., 2009, 2019; Zheng et al., 2021). Although MAR storage volumes are typically low relative to surface reservoirs, MAR can be an important local-scale strategy to help alleviate regional water stress (Scanlon et al., 2023). Since the 1960s, the implementation of MAR has accelerated at a rate of ~5% a^{-1}. Currently, MAR has reached an estimated 10 $km^3\ a^{-1}$, about 2.4% of groundwater abstraction in countries reporting MAR (or about 1% of global groundwater abstraction) (Dillon et al., 2019).

According to Sprenger et al. (2017), MAR schemes are widely distributed and applied at various scales and for various purposes in European countries. The most widespread MAR type is induced bank filtration (57% of active sites), with surface-spreading methods ranked second (34%) and well injection third (5%). Most MAR sites (67% of active sites) are situated in unconsolidated geological formations given the predominance of bank filtration and surface-spreading schemes. Geological formations such as fluviatile and glacial sediments, as well as aeolian deposits, are commonly utilised in Europe, while MAR sites situated in consolidated geological strata are comparatively rare (Sprenger et al., 2017).

MAR schemes provide environmentally sustainable groundwater storage able to support drought and emergency supplies. Together with surface reservoirs, MAR slows the movement of water through catchments and basins, giving greater resilience to water supplies by augmenting groundwater reserves during wet seasons that can be drawn upon during a drought, for example when water supplies are in demand for agricultural use (Rawluk et al., 2013). Such conjunctive water resources management to expand local storage options supports a climate change adaptation strategy where water users utilise both surface water and groundwater depending on the instantaneous level of abundance and cost (Evans and Dillon, 2017; Dillon et al., 2022; Scanlon et al., 2023).

Sustainable aquifer recharge and recovery rates and periods are likely to depend on multiple variables that are established during a trial period. These factors include aquifer characteristics, recharge field and borehole performance, quality of the source and receiving waters, environmental characteristics and protected rights of receptors, source water availability and abstractor demand profiles (Maliva, 2015; Song et al., 2019; Fuentes and Vervoort, 2020; Zheng et al., 2023). The aquifer hydraulic conductivity is an important factor for MAR in terms of infiltration and later abstraction. Generally, high hydraulic conductivity and low specific yield of the aquifer, natural boundaries to stop groundwater escaping horizontally and vertically, and low salinity of existing groundwater are favourable characteristics (Knapton et al., 2019). A further important consideration is the risk of aquifer contamination, especially where reclaimed water is used for recharge, with pre-treatment potentially needed. Even though natural filtration removes many contaminants, pathogens and trace chemicals can remain (Yuan et al., 2019).

Despite advances in research that has enlarged the scope of MAR, Dillon et al. (2019) considered that there remain several basic steps that would improve efficiency of investment in MAR and underpin its uptake where this is currently low. To address this challenge, Dillon et al. (2019) advocated extending case study information to include economic evaluations, extending research on fundamental processes to better locate, design, operate and monitor MAR schemes, and to translate scientific evidence into governance arrangements for water allocations and water quality protection. Bruce et al. (2023) introduced recharge net metering (ReNeM) as a cost-effective, market-based mechanism to incentivise distributed groundwater recharge on privately owned land to overcome the hurdles often confronted by water management agencies when implementing MAR (Rohde, 2023).

MAR is not commonly practiced in the UK due to regulatory concerns about groundwater quality, the hydrogeological conditions of the principal consolidated aquifers, and the need for clear guidance to farmers and environmental groups of the benefits of MAR. An early technical review of artificial recharge practice is provided by Hunter Blair (1970) and a later review of the potential for aquifer storage and recovery in major aquifers in England and Wales is presented by Jones et al. (1998). Previous pilot schemes in the Triassic sandstone in Nottinghamshire and Lower Cretaceous Greensand in Sussex are summarised by Downing (1986). An example of a fully developed scheme is the North London Artificial Recharge Scheme involving the storage and recovery of water from the Cretaceous Chalk aquifer to support low river flow and reservoir levels in the Lea Valley (Downing, 1986; O'Shea and Sage, 1999).

As experienced elsewhere, water resources in East Anglia, eastern England, are under pressure due to population growth, demand for irrigated crops, environmental requirements and climate change. In this region, an additional $570 \times 10^3\ m^3\ day^{-1}$ for public water supply and an estimated $444 \times 10^3\ m^3\ day^{-1}$ for other users, including agriculture, power generation and industry, will be needed between 2025 and 2050 (Environment Agency, 2020; Water Resources East, 2022). Traditionally, farmers have used lined surface reservoirs to store water for summer irrigation use. This solution can require large areas of potentially productive land and involve high capital and environmental costs. Matching growth with enhanced environmental protection requires innovative solutions and MAR offers the possibility of storing excess winter flows underground for later abstraction during periods of peak demand (DEFRA, 2021).

Given the background to water resources availability in East Anglia and the need for extended case studies that combine evaluation of hydraulic, water quality, regulation and economic aspects of MAR schemes that add to global understanding (Dillon et al., 2019; Bruce et al., 2023), the aims of this study were to demonstrate the viability of the MAR approach in providing a cost-effective and sustainable solution to storing surplus surface water for later recovery. The research, which is novel for the UK, and an example of an integrated approach to the conjunctive use of surface water and groundwater resources, is based on a MAR scheme developed in the Pliocene marine sand and gravel (Crag) aquifer in an area of coastal Suffolk, a region prone to dry climatic conditions and in need of water storage to meet local irrigation demand for high-value crops.

2. MAR site selection, characterisation and investigation methods

2.1. Site selection

The Crag aquifer of coastal Suffolk has good groundwater storage properties and lies close to the surface beneath permeable topsoil, making MAR possible as a low-technology solution to supplement local irrigation water use. Stored water can be considered potentially available for recovery to avoid draining naturally into adjacent springs and rivers. In selecting a suitable site, the key elements of the MAR scheme, including identifying the source of recharge water, the recharge method, design of the recharge site, the water recovery method and ultimate uses of the recovered water, were in accordance with other MAR schemes (Yuan et al., 2016; Zhang et al., 2020; Salameh et al., 2019).

To demonstrate that the Crag aquifer is able to retain water that can then be recovered, a hydrogeological assessment was completed at the investigation stage of the project. The site chosen for the MAR trial was determined by several factors including proximity to the recharge source water, thickness of the Crag aquifer, sufficient depth of the unsaturated (vadose) zone to store recharged water, distance away from groundwater discharge zones, distance from groundwater-dependent features such as springs and ponds, land access, and within proximity of a power supply. Within these constraints, Bucklesham (latitude 52° 1′ 42″ N, longitude 1° 16′ 54″ E) was selected in an area of agricultural land on the interfluve between surface water tributaries to the west and northwest and a small tributary of the Mill River to the southeast and east (Fig. 1).

2.2. Physiographic setting

The topographic elevation across most of the study site is level at ~26 m Ordnance Datum (OD) (sea level) with gentle slopes towards the tributaries. The area experiences a temperate maritime climate, with a mean annual temperature of 10.9 °C and a mean annual precipitation total of 569 mm (1991–2020) (Meteorological Office, 2023). Land use in the region includes arable/horticultural land (40%), urban area (32%), grassland (16%), woodland (11%) and heath/bog land (1%) (UKCEH, 2023). Predominant crops in the area include cereals (28%), beans (18%), potatoes (18%) and maize (10%) (DEFRA, 2023).

The Pliocene Crag Formations, which extend across the Suffolk coastal area (Fig. 1), comprise a locally important intergranular aquifer, up to about 80 m thick (Allen et al., 1997). In the study area, the Crag Formation is up to 30 m thick with an unsaturated zone thickness of between 5 and 10 m. The geological succession at the recharge site is summarised in Table 1 and comprises the Eocene London Clay Formation, overlain by the Pliocene Red Crag and Norwich Crag Formations, which are in turn overlain by Quaternary glacial sand and gravel deposits (Mathers and Smith, 2002). The groundwater level in the Crag aquifer at the recharge site indicated an unsaturated zone thickness of about 12 m and a saturated zone thickness of approximately 5–6 m. Together with a low hydraulic gradient, the available groundwater storage and residence time was considered sufficient for the development of a MAR scheme.

The geological succession at the recharge site showed a layered heterogeneity with the Crag and glacial sand and gravel deposits in hydraulic continuity. Groundwater yields in the Crag are typically moderate to low, of the order of 10 L s^{-1} from well point systems, although better yields of between 30 and 40 L s^{-1} have been obtained from gravel packed or screened boreholes (British Geological Survey, 2022b).

2.3. Hydrogeological properties

A 5-day, constant rate pumping test of an abstraction borehole at the Bucklesham recharge site (ABH1, Fig. 2c) with an average pumping rate of 101 m^3 day^{-1} (1.17 L s^{-1}) was conducted in December 2021. Analysis of drawdown data for ABH1 and the adjacent observation borehole OBH6 at a distance of 45 m produced a calculated Crag transmissivity value of between 49 and 62 m^2 day^{-1}. The calculated transmissivity value is lower than the interquartile range of 238–722 m^2 day^{-1} compiled by Jones et al. (2000), most likely due to the nature of the medium-grained, poorly sorted Crag lithology at the recharge site. Based on the calculated transmissivity value, and for a saturated thickness of 5 m, the Crag hydraulic conductivity at the site is ~10–12 m day^{-1}. The pumping test analysis gave an estimated storage coefficient of between 0.06 and 0.07, within the interquartile range of 0.004–0.11 for the Crag aquifer (Jones et al., 2000).

The Crag aquifer at the recharge site is unconfined and overlain by a shallow, freely draining, slightly acidic sandy and loamy topsoil. Soil infiltration tests at the site gave an average vertical hydraulic conductivity value of 0.56 m day^{-1}. Free drainage of the topsoil makes it vulnerable to the leaching of agricultural fertilisers and pesticides to groundwater, while low soil moisture limits crop yields without irrigation (Cranfield University, 2022). Further tests in trial pits indicated infiltration rates of between 50 and 80 mm h^{-1} (1.20–1.92 m day^{-1}), with hydraulic conductivity increasing at depth.

Preliminary investigation of the groundwater storage and recovery potential of the Crag aquifer was modelled by the Environment Agency using the Crag component of its numerical North-East Anglia Chalk (NEAC) groundwater model (Black et al., 2012). Model results indicated that potentially up to 120,000 m^3 of water could be stored and recovered

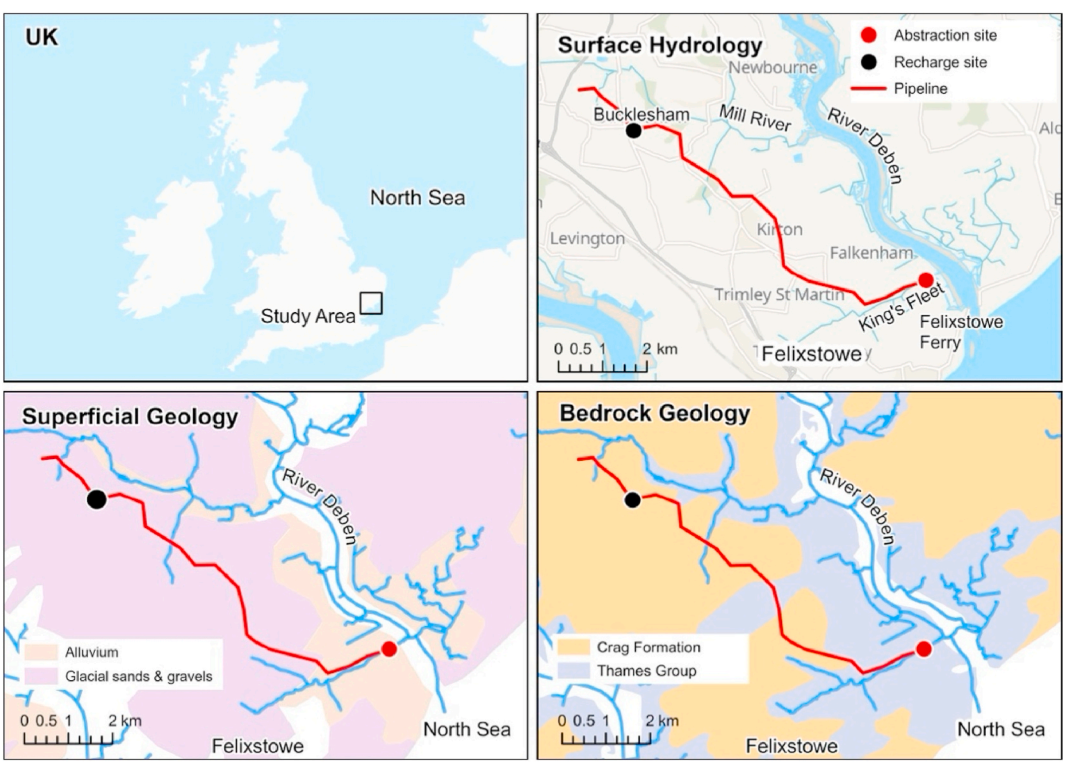

Fig. 1. Location of the Felixstowe area of Suffolk showing surface hydrology and superficial and bedrock geology (after the British Geological Survey, 2022a). Also shown are the source water abstraction site on the King's Fleet at Felixstowe Ferry and the recharge field site at Bucklesham. The red line represents the 14-km dual-pipeline to transfer water inland to surface storage irrigation reservoirs and the MAR recharge site. Contains British Geological Survey materials © UKRI [2023]. Contains OS data © Crown Copyright and database right 2023. Contains data from OS Zoomstack.

Table 1
Geological succession at the Bucklesham MAR site based on site investigation borehole logging and Mathers and Smith (2002).

Period	Epoch	Formation	Member	Lithology	Thickness (m)
Quaternary	Holocene 0–10 ka			Top soil: firm brown fine dry friable organic sand with roots	0–0.5
	Pleistocene (Anglian Stage) 0.4–0.45 Ma			Glacial sands and gravels	0.5–6.8
Neogene	Pliocene 0.9–3.6 Ma	Norwich Crag	Chillesford Clay	Silty clay, buff and grey with sand laminae and shell debris	6.8–9.0
			Chillesford Sand	Well-sorted, fine- to medium-grained, micaceous, quartz sand	
		Red Crag		Medium-grained, poorly sorted, shelly sands with an overall fining-upward trend, commonly decalcified in the upper parts	9.0–18.6
Palaeocene	Eocene 52.0–54.8 Ma	Thames Group (London Clay)		Blue-grey clay and silty clay deposits	18.6–19.6

Fig. 2. (a) Recharge source water abstraction from the King's Fleet drainage channel showing the two eel-friendly Riverscreen pumps. (b) Recharge lagoon at the Bucklesham MAR site in operation. (c) Recharge field site map showing the locations of abstraction boreholes (ABH1 and ABH2) and observation boreholes (3, 5, 6, 7, 8 and 9), including the position of the recharge lagoon and layout of backfilled infiltration trenches. (d) Recharge distribution trench with slotted pipe under construction.

over a 9-month cycle (4 months of recharge – 1 month of storage – 4 months of recovery) with negligible leakage into or from nearby watercourses.

2.4. Abstraction source

The source water for the MAR scheme is the 4.6 km King's Fleet drainage channel that discharges to the River Deben estuary and North Sea at Felixstowe Ferry (latitude 51° 59′ 44″ N, longitude 1° 22′ 41″ E) (Fig. 1). The distribution system relies on two high-volume, low-head abstraction pumps (Fig. 2a) each delivering 0.03 $m^3 s^{-1}$ operating either individually or together, with abstraction primarily during the high-flow winter months. The total capacity of the MAR scheme is consented for infiltration and re-abstraction of up to 40,000 $m^3 a^{-1}$. Water is transferred inland via two, 14-km length, 200 mm-diameter pipelines (Fig. 1) to supply nine surface storage irrigation reservoirs and the MAR recharge site.

2.5. Recharge and recovery system design

Recharge at the MAR site is induced through a recharge lagoon (Fig. 2b) connected to an array of backfilled infiltration trenches (Fig. 2c) positioned in the Crag unsaturated zone at a depth of ~2.7 m, in which slotted pipes were installed (Fig. 2d) (see Supplementary Material for further engineering design details). The infiltration system was chosen given its several advantages compared to open recharge basins. Heilweil et al. (2015) reported that a covered trench infiltration rate is an order of magnitude higher than an open basin. Advantages include reduced clogging and biofilms and minimum land take compared with open basins. Additionally, covered trench infiltration avoids low-permeability surface soils and low temperatures that increase water viscosity and so further reducing hydraulic conductivity.

An overview of vadose zone infiltration is given by Bouwer (2002) who described the technique as typically less than 1 m wide and up to 4 m deep, backfilled with coarse sand or fine gravel, with water normally applied through a perforated pipeline. An analytical equation for estimating the steady-state infiltration rate for a trench in the vadose zone, not in connection with the regional water table, is given by Heilweil et al. (2015), modified from Bouwer (2002), as follows:

$$Q_f = \frac{0.4\pi K_{sat} H^2}{ln\left(4\frac{H}{w}\right) - 1} \quad \text{Equation 1}$$

where Q_f is the final or steady-state infiltration rate per unit area of the trench floor, K_{sat} is saturated hydraulic conductivity, H is height of the water column in the trench, and w is the trench width.

For conditions at the recharge site, selecting values of K_{sat} in the range 2–12 m day^{-1} based on the trial pit and pumping test data, $H = 0.75$ m and $w = 0.5$ m produces an infiltration rate using Equation (1) of 1.79–10.7 m^3 m^{-2} day^{-1}. Therefore, a potential total recharge volume of 900 m^3 day^{-1} (10.4 L s^{-1}), yielding 108,000 m^3 for a 120-day pumping period, would require a length of infiltration trenches equal to 168–1009 m. In practice, as shown in Fig. 2c, the final length of installed, parallel covered trenches was 565 m.

An analytical solution for the development of a steady-state groundwater mound resulting from artificial recharge for long-strip basins (analogous to the infiltration trenches), in which groundwater flow away from the strip can be approximated as linear horizontal flow, is given by Bouwer et al. (1999) as:

$$H_c - H_n = \frac{iW}{2T}\left(\frac{W}{4} + L_n\right) \quad \text{Equation 2}$$

where H_c is the height of the groundwater mound at the centre of the recharge area, H_n is the height of the groundwater table in the control area at a distance, L_n, from the edge of the recharge area, i is the infiltration rate in the recharge area, W is the width of the recharge area, and T is the aquifer transmissivity, ignoring the effects of water table height on T.

Natural groundwater discharge to springs draining to the Mill River 750 m northeast of the recharge site at an elevation of about 5 m OD provides a control on the feasibility of the MAR scheme in respect of the elevation of the recharge mound that can develop under equilibrium conditions. For an infiltration rate of 1.77 m day^{-1} (the permitted daily recharge rate (1000 m^3 day^{-1}) applied over a trench length of 565 m and an assumed width of the recharge area of 1 m) and a Crag transmissivity of between 49 and 62 m^2 day^{-1} as estimated from the 5-day pumping test, then the elevation of the groundwater mound beneath the recharge site is, using Equation (2), calculated in the range 15.7–18.6 m OD. The average rest groundwater level recorded in May 2022 at the location of ABH1, near the centre of the expected recharge mound, was 14.0 m OD (Table S2). Therefore, under equilibrium conditions, the increase in groundwater level at the centre of the recharge mound is acceptable at ~2–3 m above the rest level and well below the ground elevation at ABH1 of 27 m OD.

Water was re-abstracted from the Crag aquifer using submersible pumps in two abstraction boreholes (ABH1 and ABH2, Fig. 2c) sited adjacent to the join between the main and secondary recharge trenches. The boreholes were drilled to the base of the Crag aquifer to a depth of 18 m below ground level (bgl) and lined with 165-mm diameter casing material with a slotted screen (Fig. S1) with the aim of improving yield from approximately 9 m bgl to the base of the aquifer. Water recovered from the MAR system is then pumped to surface water reservoirs via the inland pipeline where it can then be used for irrigation.

2.6. Water level monitoring

The observation borehole (OBH) network for the scheme was designed to monitor groundwater levels proximal to the recharge field and towards the edges of the recharge mound. All OBHs monitored to the base of the Crag aquifer, with details of locations, datum levels, geological strata and rest groundwater levels given in Tables S1 and S2. The site of OBH10 was chosen as a sentinel borehole for the spring-fed lakes located 500 m northeast of the recharge site. Groundwater levels were recorded with a pressure transducer and datalogger at 15-min intervals with an accuracy of ±0.1%.

2.7. Water quality monitoring

The main contaminants of concern were chloride (with possible sources in the coastal pumped drainage ditches, saline intrusion and salt washing from a major road) and nitrate and herbicides from agricultural sources. Grab samples of the recharge source water were collected periodically and analysed for a list of over 570 substances between July 2020–June 2022 ($n = 13$). Monitored parameters included: (i) parameters indicative of saline or other intrusions, including electrical conductivity; (ii) trichloroethylene and tetrachloroethylene; and (iii) substances, ions or indicators which may occur both naturally and/or because of human activities, including arsenic, cadmium, lead, mercury, chloride, sulphate, nitrate and ammonium. Of less concern were polycyclic aromatic hydrocarbons (PAHs) and metals from road runoff. Fluorosurfactants (PFOS, perfluorooctane sulfonic acid and PFOA, perfluorooctanoic acid) were added to the list of analyses for the recharge source water from October 2021 ($n = 8$) as emerging contaminants of concern.

Pumped samples from the Crag aquifer were collected between October 2021–September 2022 ($n = 46$). Groundwater samples from abstraction borehole ABH1 were analysed for the full suite of inorganic species and organic compounds between October–December 2021 ($n = 3$) prior to and during a preliminary trial of the recharge and re-abstraction system (6352 m^3 of source water were recharged between 15 and 21 November 2021 and 525 m^3 of groundwater were re-abstracted between 8 and 14 December 2021). Further samples were collected from abstraction borehole ABH1 in November–December 2021 ($n = 3$) and June–September 2022 ($n = 5$) and analysed for a more restricted suite consisting of major ions and electrical conductivity, with the summer 2022 samples collected during the main trial of the recharge and re-abstraction system. Pumped samples from observation boreholes OBH3, OBH5, OBH6 and OBH7 collected in November–December 2021 ($n = 5$ each) were analysed for major ions during the preliminary recharge and re-abstraction trial. In addition, pumped samples from abstraction borehole ABH2 and observation boreholes OBH5 and OBH8 were collected in June–September 2022 ($n = 5$ each) and analysed for major ions and electrical conductivity during the main recharge and re-abstraction trial.

In total, chemical analysis of 59 water samples was conducted by a certified commercial laboratory (ALS Laboratories (UK) Ltd). Laboratory methods included spectrophotometric analysis of chloride, sulphate, nitrate and nitrite with limits of detection (LoDs) of, respectively, <2, <2, <0.3 and < 0.05 mg L^{-1}. Ammonium was analysed by colorimetry with a LoD of <0.3 mg L^{-1}. Dissolved metals were analysed by ICP-MS with LoDs for arsenic, cadmium, lead and mercury of, respectively, <0.5, <0.08, <0.2 and < 0.01 μg L^{-1}. Phenols, PAHs, volatile organic compounds, acid herbicides and suites of pesticides were analysed by mass spectrometry with LoDs of, respectively, <0.5, <0.005, <1, <0.1 and < 0.01 μg L^{-1}. PFOS and PFOA were analysed by LC-MS with a LoD of <0.65 ng L^{-1}.

Total organic carbon (TOC) concentrations were measured by high-temperature combustion with the carbon dioxide produced measured by non-dispersive infrared adsorption with a LoD of <0.7 mg L^{-1}. Dissolved oxygen was analysed by the Winkler titration method with a LoD of 0.3

mg L^{-1} and electrical conductivity (at 20 °C) with a probe with a LoD of <5 μS cm^{-1}. The concentration of total suspended solids (TSS) as a measure of both organic and inorganic solids in water was analysed by the amount of material retained by a 1.2 μm pore size filter with a LoD of <2 mg L^{-1}.

To explore the change in water quality resulting from the introduction of recharge source water into the Crag aquifer, a conservative mixing approach was adopted. For conservative mixing of two end-member solutions:

$$c_{i,mix} = (f_{sw} \times c_{i,sw}) + (1 - f_{sw})c_{i,gw} \qquad \text{Equation 3}$$

where c_i is the concentration of ion, *i*, and *mix*, *sw* and *gw* indicate the conservative mixture and end-member recharge source water and groundwater, respectively, and f_{sw} is the fraction of recharge source water.

2.8. Regulatory requirements

The groundwater investigation consent issued by the environmental regulator specified limits on the maximum infiltration rate (1200 m^3 day^{-1}) and re-abstraction rate (250 m^3 day^{-1}), and the requirement for all infiltrated water, which should not exceed 18,000 m^3, be recovered. Monitoring of the water level of the recharge lagoon, soil moisture and precipitation above the recharge field were stipulated. Water level monitoring of the observation boreholes and two abstraction boreholes at the recharge site, together with monitoring of abstraction borehole pumping rates, were also required.

To legally permit MAR infiltration, the discharge conditions were risk-assessed based on environmental quality standards (EQS) and drinking water standards (DWS) (European Commission, 1998, 2008). An additional safeguard was provided by setting an alarm to stop the recharge water entering the infiltration array at any concentration above the designated threshold. In the example of chloride and nitrate-N, an increase in concentration of the local groundwater quality was permitted if the maximum concentrations in the source water did not exceed 165 mg L^{-1} and 7.5 mg L^{-1}, respectively.

3. Results and discussion

3.1. Water quantity and water levels

Between 9 and 20 June 2022 during the recharge period of the main MAR trial, 12,262 m^3 of source water were recharged to the aquifer at an average infiltration rate of 1022 m^3 day^{-1} (Fig. 3). Of the recharged water, 12,301 m^3 were successfully abstracted from the Crag aquifer from 18 July–10 September 2022, augmenting surface storage irrigation reservoirs during drought conditions.

Abstraction borehole ABH1 delivered a maximum pumping rate of approximately 90 m^3 day^{-1} (~1 L s^{-1}). Given this modest abstraction rate, a second abstraction borehole (ABH2) adjacent to the recharge field was installed to provide a combined discharge rate of 250 m^3 day^{-1}. Hence, the results indicated that a significant factor constraining the volume of water that can be usefully stored and recovered is the rate of re-abstraction from the Crag aquifer. Without further abstraction boreholes, the hydraulic conductivity of the Crag at the recharge site (K = 2–12 m day^{-1}) and thin saturated zone (5–6 m) limit the yield of the MAR scheme, limitations that are reported for other MAR schemes (Knapton et al., 2019).

Groundwater levels monitored at the observation boreholes during the period May–September 2022 varied between 13.3 and 14.8 m OD, with less variation of levels observed at OBH10 (11.4–11.6 m OD) (Fig. 3). Levels recorded at ABH1 and ABH2 varied between 9.5 and 15.0 m OD due to the influence of groundwater pumping. Crag groundwater levels were lower at OBH5 and OBH7 to the east of the recharge lagoon by ~0.5 m, suggesting an eastward flow component from the area of the MAR site.

The application of artificial recharge in June 2022 caused an increase in groundwater levels at all sites except for OBH9 and OBH10. As shown in Fig. 4a, the relative increase in groundwater level during the recharge period ranged from 0.1 m at OBH5, OBH6 and OBH7 to 0.2 m at ABH2 and OBH8 and 0.3 m at ABH1 and OBH3. These results imply a good delivery of artificial recharge to the area of the abstraction boreholes and to the end of the pipe array near OBH8, but lesser receipt to the east of the lagoon, perhaps due to a localised decrease in aquifer hydraulic conductivity at this location.

If the extent of the recharge mound is delimited by OBH9 where no increase in groundwater level was observed, then the radius of the artificial recharge mound is approximately 250 m. Using the equation

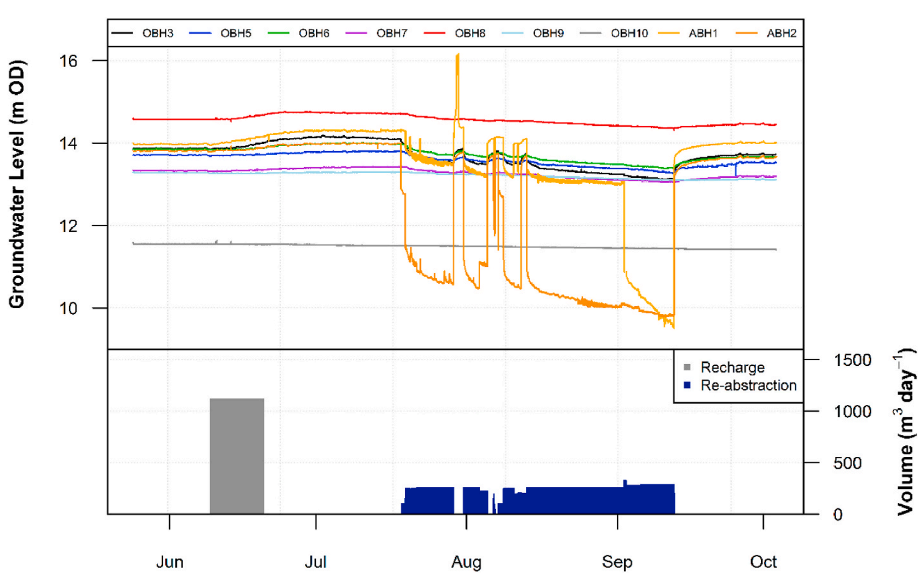

Fig. 3. Record of groundwater levels at abstraction (ABH) and observation (OBH) boreholes recorded during the main recharge and recovery trial (June–September 2022). Also shown are the daily volumes of source water recharge and re-abstraction during the trial. Spikes in water levels for ABH1 and ABH2 show temporary pump outages.

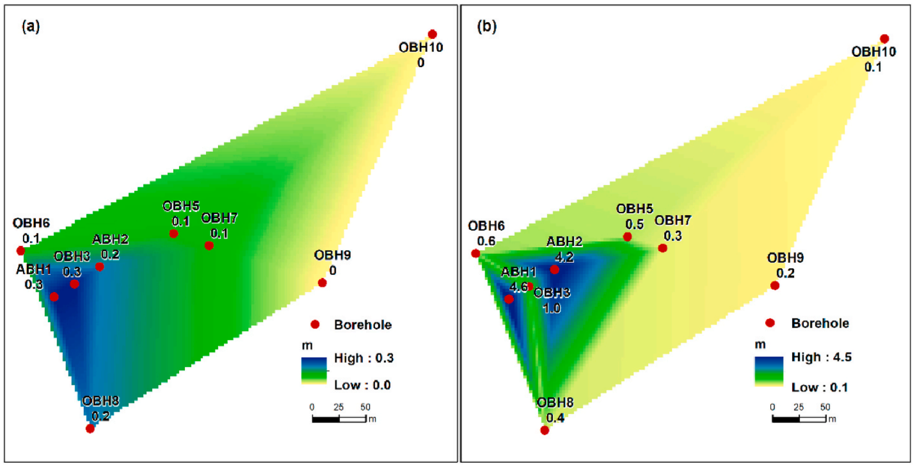

Fig. 4. Schematic diagram showing (a) the relative increase and (b) decrease (drawdown) in groundwater levels (in metres) during the recharge and recovery periods of the summer 2022 MAR trial. Water table surfaces drawn using triangulated irregular network (TIN) analysis to raster format in ArcGIS Pro.

for the volume of a spherical cap, and assuming a Crag aquifer porosity in the range 25–40% (Jones et al., 2000), then an approximation of the height of the recharge mound with a radius of 250 m to accommodate an injected pore volume of 12,262 m^3 is between 0.3 and 0.5 m, corresponding to the actual observed value of 0.3 m at the assumed centre of the recharge mound adjacent to ABH1 and OBH3.

The relative decrease (drawdown) in groundwater level during the following recovery (re-abstraction) stage of the trial was greatest at the abstraction boreholes, ABH1 and ABH2 (4.6 and 4.2 m, respectively), with a 1.0 m decrease in groundwater level observed at OBH3 equidistant between the two abstraction boreholes (Fig. 4b). The decrease in groundwater levels varied between 0.4 and 0.6 m in OBH5, OBH6 and OBH8. The smallest decreases in groundwater level of between 0.1 and 0.3 m were observed in OBH7, OBH9 and OBH10. The water level response at the boreholes varied with changes in pump rate/outages, except for the more distant OBH8 where the response to these events was muted. As shown in Fig. 3, water level recovery after every pump outage was rapid such that it is reasonable to assume that a good recovery in water level occurred after re-abstraction had ceased.

Regarding the impact of artificial recharge on local water features, groundwater level data from the trial suggested that the recharge mound remained within 250 m of the recharge site, based on the lack of response from the more distant OBH9 and OBH10. During the recovery period, the results indicated that the re-abstraction did not impact on spring-fed lakes northeast of the recharge site.

Table 2
Summary results of water quality monitoring of the recharge source water abstracted from the King's Fleet drainage channel between July 2020–June 2022.

	Minimum	Maximum	Mean	1σ	*n*
Recharge source water					
Conductivity at 20 °C (μS cm^{-1})	770	1160	890	0.13	13
Dissolved oxygen (mg L^{-1})	5.50	11.4	9.26	2.11	13
TSS (mg L^{-1})	2.00	9.50	3.02	2.14	13
TOC (mg L^{-1})	5.41	9.98	7.42	1.44	13
Chloride (mg L^{-1})	106	178	134	18.3	13
Sulphate (mg L^{-1})	75.6	138	102	20.8	13
Nitrate-N (mg L^{-1})	0.07	7.59	3.00	2.87	13
Nitrite-N (mg L^{-1})	0.02	0.10	0.04	0.02	13
Ammoniacal-N (mg L^{-1})	0.30	0.39	0.31	0.02	13
Mercury (μg L^{-1})	0.01	0.01	0.01	0	13
Arsenic (μg L^{-1})	0.58	1.35	0.94	0.21	13
Cadmium (μg L^{-1})	0.08	0.08	0.08	0	13
Lead (μg L^{-1})	0.20	0.44	0.22	0.06	13
Propyzamide (μg L^{-1})	0.01	0.02	0.01	0	2
Simazine (μg L^{-1})	0.02	0.03	0.02	0	2
Triallate (μg L^{-1})	0.02	0.05	0.03	0.01	6
Desphenyl-chloridazon (μg L^{-1})	0.85	1.62	1.21	0.30	8
2,6-dichlorobenzamide (μg L^{-1})	0.05	0.21	0.12	0.04	11
Fluoranthene (μg L^{-1})	0.01	0.22	0.09	0.07	6
Phenanthrene (μg L^{-1})	0.01	0.13	0.05	0.05	7
Pyrene (μg L^{-1})	0.02	0.11	0.06	0.04	5
Total PAH (μg L^{-1})	0.12	0.45	0.31	0.14	4
PFOA (ng L^{-1})	1.01	3.20	2.25	0.73	8
Total PFOS (ng L^{-1})	1.78	5.23	2.54	1.09	8

3.2. Water quality

3.2.1. Recharge source water

A summary of monitoring results for the recharge source water abstracted from the King's Fleet drainage channel is given in Table 2 for the monitoring period July 2020–June 2022, and time series data for selected determinands are presented in Fig. 5. Elevated values of electrical conductivity (1160 μS cm^{-1} in January 2022), chloride (178 mg L^{-1} in February 2022) and sulphate (138 mg L^{-1} in March 2022) (Fig. 5b) were caused by abstraction from the King's Fleet. The lowering of the surface water level relative to sea level during abstraction stimulated saline water intrusion. During the recharge period of the main MAR trial (9–20 June 2022), the source water quality was stable with a mean electrical conductivity value of 834 μS cm^{-1} and mean chloride concentration of 133 mg L^{-1}, below the regulated threshold concentration of 165 mg L^{-1}.

The high nitrate-N concentrations observed in April 2021 and February 2022 (7.6 and 7.2 mg L^{-1}, respectively; Fig. 5c) resulted from winter flushing of residual nitrogen from the soil profile and the timing of spring applications of fertilisers to arable crops in the King's Fleet catchment, responses that are observed in other catchments in East Anglia (Outram et al., 2016; Garrard et al., 2023). At other times of the year, nitrate-N concentrations are between 0.1 and 3.1 mg L^{-1} and demonstrated the removal of nitrate by biological uptake and denitrification within the riparian zone of the King's Fleet, as observed for other surface water systems and wetlands (Walton et al., 2020; Steiness et al., 2021; Raulerson et al., 2023).

Ammoniacal-N and nitrite-N concentrations remained within the ranges 0.2–0.3 mg L^{-1} and 0.02–0.1 mg L^{-1}, respectively (Table 2). Elevated nitrite-N concentrations of 0.05, 0.1 and 0.07 mg L^{-1} recorded in October 2020, October 2021 and May 2022, respectively (Fig. 5c), coincided with low nitrate-N concentrations (0.3, 2.3 and 3.1 mg L^{-1}, respectively) and likely represent nitrite-N produced as an intermediate product of denitrification in the riparian zone (Rivett et al., 2008).

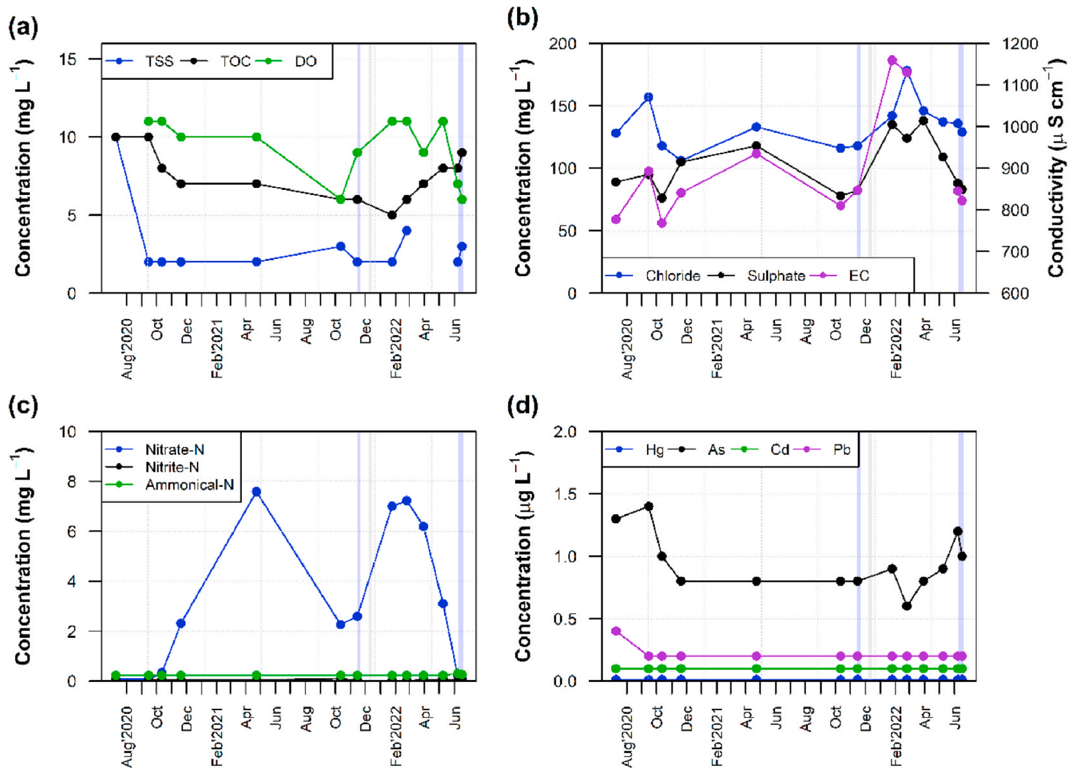

Fig. 5. Time series monitoring of recharge source water quality abstracted from the King's Fleet drainage channel from July 2020–June 2022. The two light blue lines represent the periods of source water recharge during the preliminary and main trials of the MAR scheme in November 2021 and June 2022, respectively. The single light grey line represents the re-abstraction of groundwater in December 2021 during the preliminary trial of the MAR scheme.

Dissolved oxygen concentrations in the recharge source water were generally saturated, with peaks occurring in October 2020 (11.0 mg L^{-1}) and in February 2022 (11.4 mg L^{-1}) (Fig. 5a). The lowest concentrations of dissolved oxygen of 5.5 mg L^{-1} were recorded in October 2021 and June 2022 most likely because of warmer surface water temperatures promoting reducing conditions, similar to conditions observed by Itoh et al. (2007) in riparian wetlands in a temperate forest catchment and Cooper et al. (2020) in shallow headwater streams in an arable catchment.

TSS ranged between 2.0 and 9.5 mg L^{-1} (Table 2), with most of the samples recording <4 mg L^{-1} (Fig. 5a). TOC concentrations ranged between 5.4 and 10.0 mg L^{-1} throughout the whole monitoring period (Fig. 5a). Hence, the loading of suspended inorganic and organic material from the recharge source water to the Crag aquifer was low and so limiting the risk of clogging and decline in permeability of the sands and gravels following infiltration. The formation of clogging layers on the bottom of recharge basins or other infiltration surfaces is well documented in the literature (Bouwer, 2002; Schubert, 2002; Song et al., 2019). Different factors determine the reduction in infiltration rates, including effluent water quality, hydraulic loading rate and recharge cycles (Lippera et al., 2023). However, the rate and degree of clogging are controlled mainly by the rate of suspended solids deposition, the size distribution of the fines and the size distribution of the receiving sediments (Hutchinson et al., 2013). For the current MAR scheme, should clogging become a problem in the longer term, then solutions include desilting or other pre-treatment. For example, the installation of sand filters and periodic backwashing of injection wells were required at an aquifer storage transfer and recovery (ASTR) site in The Netherlands where tile drainage water is stored in a sandy brackish coastal aquifer (Kruisdijk et al., 2023).

Mercury and cadmium concentrations remained stable at 0.01 and 0.08 μg L^{-1}, respectively (Table 2 and Fig. 5d). Ranges in arsenic and lead concentrations were within 0.58–1.35 μg L^{-1} and 0.20–0.44 μg L^{-1}, respectively. Occasional detections of chromium (mean concentration = 6.6 μg L^{-1}, $n = 2$), copper (1.8 μg L^{-1}, $n = 1$) and manganese (126 μg L^{-1}, $n = 1$) were observed. Apart from manganese, the trace metal results were within the DWS and indicated a source in urban and road runoff at the head of the King's Fleet drainage channel.

Very few organic compounds were detected (Table 2), with most concentrations below the DWS. The pesticide metabolites desphenyl-chloridazon (mean concentration = 1.21 μg L^{-1}, $n = 8$) and 2,6-dichlorobenzamide (mean concentration = 0.12 μg L^{-1}, $n = 11$) were detected. The herbicides simazine, triallate and propyzamide were detected on fewer occasions at concentrations of <0.05 μg L^{-1}. The herbicide pendimethalin and molluscicide metaldehyde were each detected on one occasion with concentrations of 0.03 and 0.02 μg L^{-1}, respectively.

A mean concentration of total PAH of 1.21 μg L^{-1} ($n = 4$ detections) was recorded, with fluoranthene (mean = 0.09 μg L^{-1}, $n = 6$), phenanthrene (mean = 0.05 μg L^{-1}, $n = 7$) and pyrene (mean = 0.06 μg L^{-1}, $n = 5$) the most frequently detected (Table 2). Mean concentrations of total PFOS and PFOA of 2.54 and 2.25 ng L^{-1}, respectively, were detected for all sampling occasions ($n = 8$) (Table 2), comparable to median concentrations of 3.2 ng L^{-1} for PFOS and 3.1 ng L^{-1} for PFOA reported in a review of surface waters by Zareitalabad et al. (2013). In the case of PFOS, the mean concentration is above the European annual average EQS for inland waters of 0.65 ng L^{-1} (Lindim et al., 2016). The most probable sources of PAH, PFOS and PFOA compounds are urban, municipal and road runoff at the head of the King's Fleet drainage channel.

3.2.2. Groundwater

The range in Crag groundwater composition monitored across the Bucklesham recharge site during the period November 2021–September 2022 (Table 3) showed that no exceedances of regulatory limits were observed, except for nitrate-N that exceeded the DWS of 11.3 mg N L^{-1}. Concentrations of chloride, sulphate and nitrate-N showed similar

Table 3
Summary results of water quality monitoring of Crag groundwater at the Bucklesham MAR site between November 2021–September 2022.

	Minimum	Maximum	Mean	1σ	n
Crag groundwater					
Conductivity at 20 °C (μS cm^{-1})	680	1070	900	0.1	45
TOC (mg L^{-1})	3.00	3.00	3.00	0	2
Chloride (mg L^{-1})	82.9	198	117	28.7	45
Sulphate (mg L^{-1})	70.6	141	114	19.5	29
Nitrate-N (mg L^{-1})	4.83	46.5	29.5	13.9	43
Nitrite-N (mg L^{-1})	0.02	0.04	0.02	0.02	3
Ammoniacal-N (mg L^{-1})	0.30	0.30	0.30	0	2
Mercury (μg L^{-1})	0.01	0.01	0.01	0	2
Arsenic (μg L^{-1})	0.50	0.50	0.50	0	2
Cadmium (μg L^{-1})	0.08	0.08	0.08	0	2
Lead (μg L^{-1})	0.20	0.32	0.26	0.1	2
Desphenyl-chloridazon (μg L^{-1})	0.96	1.62	1.34	0.34	3
2,6-dichlorobenzamide (μg L^{-1})	0.11	0.21	0.17	0.05	3
Oxadixyl (μg L^{-1})	0.16	0.27	0.22	0.06	3

trends to those observed in the recharge source water, with peaks in concentrations (198, 141 and 46.5 mg L^{-1}, respectively) recorded during the winter recharge period November–December 2021 (Fig. 6).

A measured mean TOC concentration of 3.0 mg L^{-1} compares with baseline concentrations in UK groundwaters, which, as observed here, are generally lower than for surface waters (Gooddy and Hinsby, 2009). Mean concentrations of nitrite-N and ammoniacal-N were 0.02 and 0.30 mg L^{-1}, respectively, throughout the monitoring period. Low concentrations of metals of <0.5 μg L^{-1} were detected in groundwater samples, within the ranges of baseline concentrations for the Crag aquifer reported by Ander et al. (2006).

Concentrations of organic compounds analysed from October–December 2021 ($n = 3$) were below the limits of detection except for the pesticide metabolites desphenyl-chloridazon (mean concentration = 1.34 μg L^{-1}, $n = 3$ detections) and 2,6-dichlorobenzamide (mean concentration = 0.17 μg L^{-1}, $n = 3$ detections). In addition, the fungicide oxadixyl was detected on three occasions with a mean concentration of 0.22 μg L^{-1}. The herbicide dinitro-*o*-cresol and the PAH compound benzo[ghi]perylene were each detected on one occasion with concentrations of 0.49 and 0.02 μg L^{-1}, respectively.

While average concentrations of chloride, ammoniacal-N and metals in groundwater were comparable to the average concentrations of these determinands in surface water, the average concentration of nitrate-N (29.5 mg L^{-1}) and, to a lesser extent, sulphate (114 mg L^{-1}) were noticeably higher than the recharge source water (Table 3). Elevated concentrations of these compounds in groundwater are likely due to agricultural land use in the vicinity of the recharge site and are similar to the ranges of values reported for the Crag aquifer in northern East Anglia (nitrate-N, 0–49 mg L^{-1} and sulphate, 8–1480 mg L^{-1}; Ander et al., 2006).

Fig. 6 shows the effect of recharging the King's Fleet source water on the water quality of the Crag aquifer. For both the preliminary and main MAR trials, the chloride concentration was observed to increase. The effect was most pronounced for abstraction borehole ABH1 during the preliminary trial in November 2021 with a 52% increase in chloride concentration (equivalent to 51 mg L^{-1}) before declining to a background concentration of 113 mg L^{-1} following the re-abstraction period in December 2021 (Fig. 6a). The effect was smaller during the main trial

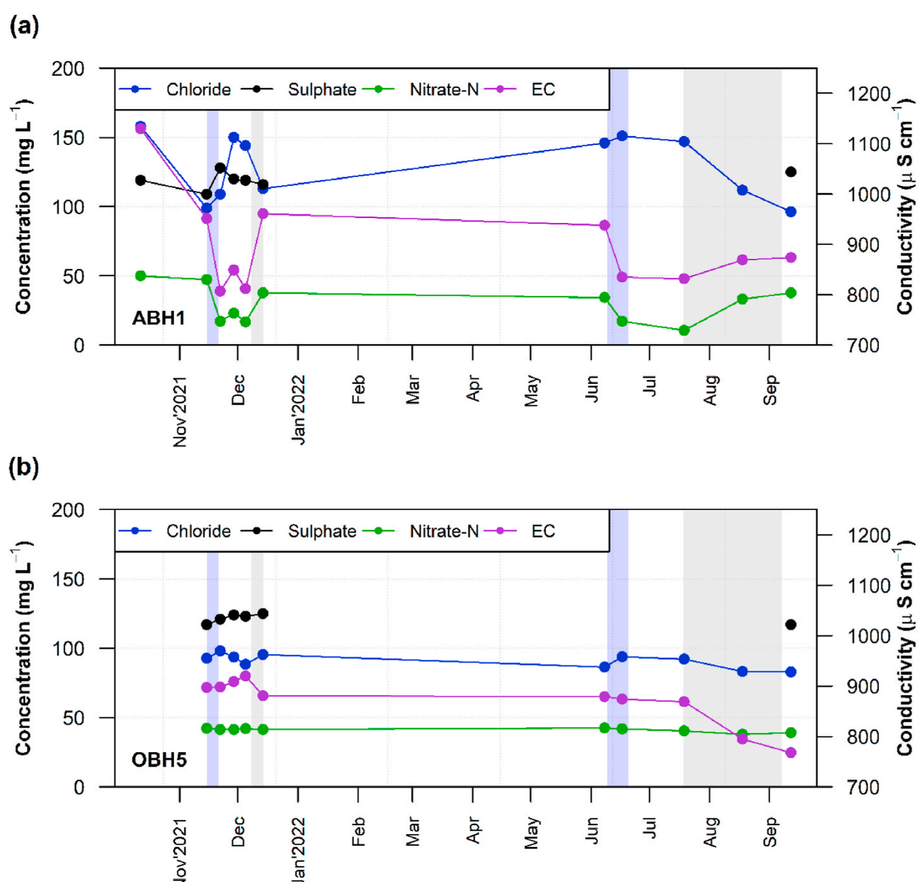

Fig. 6. Time series monitoring of Crag groundwater quality for (a) abstraction borehole ABH1 and (b) observation borehole OBH5 at the Bucklesham MAR site from October 2021–September 2022. The two light blue lines represent the times of source water infiltration during the preliminary and main trials of the MAR scheme in November 2021 and June 2022, respectively. The two light grey lines represent the re-abstraction of groundwater during the preliminary and main trials of the MAR scheme in December 2021 and July–September 2022, respectively.

in June 2022 with an increase of only 5 mg L^{-1} before declining to a background concentration of 96 mg L^{-1} in September 2022 following the re-abstraction period. The more limited sulphate data showed an increase in concentration of 19 mg L^{-1} at ABH1 in November 2021 (Fig. 6a) but with a less discernible increase of 7 mg L^{-1} at the location of OBH5 at a greater distance (~110 m) from the recharge mound (Fig. 6b).

The groundwater nitrate-N and electrical conductivity data showed an inverse response to source water infiltration compared with chloride, particularly at ABH1 during the preliminary trial in November 2021 with decreases of 65% in nitrate-N concentration (equivalent to 30.6 mg L^{-1}) and 15% in electrical conductivity (equivalent to 139 μS cm^{-1}) before recovering to background levels following the short re-abstraction period (Fig. 6a). Similar responses were observed during the main MAR trial in June 2022 with decreases of 69% in nitrate-N concentration (equivalent to 23.6 mg L^{-1}) and 11% in electrical conductivity (equivalent to 106 μS cm^{-1}) during the recharge period. The effect of the recharge is to improve the groundwater quality with minimum nitrate-N concentrations of 16.8 and 10.7 mg L^{-1} recorded in ABH1 after the recharge periods of the preliminary and main MAR trials, respectively (Fig. 6a). Similar improvements in water quality due to dilution are reported for other MAR studies. For example, Garciá-Menéndez et al. (2021) reported a decrease in electrical conductivity of 80–90% at a distance of 80 m from recharge wells in a brackish alluvial plain aquifer in coastal eastern Spain. Although the recovered groundwater did not meet drinking water standards, Rafiq et al. (2022) observed the mixing of 90% fresh recharge water with native, arsenic-rich groundwater within 4 m of an array of infiltration wells in a confined sand aquifer in a coastal area of southwest Bangladesh.

Given the greater distance from the recharge mound, little effect on nitrate-N concentrations is observed at OBH5 for either the preliminary or main MAR trials (Fig. 6b). Similarly, only a small effect was observed in groundwater electrical conductivity values at OBH5, decreasing by 101 μS cm^{-1} during the re-abstraction period of the main MAR trial towards a background value of 768 μS cm^{-1} for the final sample taken on September 12, 2022.

The degree of mixing between the infiltrated source water and background Crag groundwater would be expected to be greatest close to the centre of the recharge mound, at ABH1, becoming less with distance, for example at OBH5. Table 4 presents chloride and nitrate-N concentrations for the recharge source water and Crag groundwater at the MAR site during the recharge and recovery periods of the main MAR trial. The strongest evidence for mixing was represented by nitrate-N given the two orders of magnitude difference in concentrations between the source water (0.20 mg L^{-1}) and Crag groundwater (range from 34.3 to 42.5 mg L^{-1}) on June 8, 2022, the day before recharge commenced. At the end of the recharge period on July 19, 2022 (the day after re-abstraction had commenced) nitrate-N concentrations at ABH1, ABH2 and OBH5 were 10.7, 22.4 and 40.4 mg L^{-1}, respectively.

Based on conservative mixing of source water and groundwater end-member solutions (Equation (3)), and assuming no denitrification for an infiltration rate above 0.7 ± 0.2 m day^{-1} (Schmidt et al., 2011), as achieved in this study (1.81 m day^{-1}), the nitrate-N concentrations at ABH1, ABH2 and OBH5 represent mixed fractions of recharge source water to Crag groundwater of 69%, 39% and 5%, respectively. Therefore, and as expected, the fraction of recharge water mixed with Crag groundwater decreased with distance from the centre of the recharge mound. These results are consistent with those presented by Kattan et al. (2010) for aquifer storage and recovery in a shallow alluvial Quaternary aquifer in the Damascus basin, Syria, in which a hydrochemical mass balance showed that the stored water remains within close proximity (<100 m) to the injection well with the proportion of recharged water (volume = 24,160 m^3) in the range 50–90% at a distance of 80 m.

A similar mixing trend was not observed for chloride given the similarity of values between the recharge source water (136 mg L^{-1}) and Crag groundwater (123 mg L^{-1} at ABH1) prior to the start of recharge on June 8, 2022. Following the end of the recovery period on September 12, 2022, chloride values recorded at the two abstraction boreholes, ABH1 and ABH2 (96.3 and 90.4 mg L^{-1}, respectively), were similar to the groundwater at OBH5 (82.9 mg L^{-1}) suggesting that re-abstraction of the mixed groundwater was largely completed with background Crag groundwater with a lower chloride concentration now being drawn towards the abstraction boreholes.

3.3. Regulatory considerations

Regulatory considerations, which are important in the implementation of a MAR scheme, include issues related to water quantity and aquifer storage aspects, the protection of human health and the environment in relation to water quality, and the general regulatory framework for MAR schemes including cross-cutting aspects such as economic costs and benefits (Zhang et al., 2020).

In the example of the MAR scheme presented here, data collection followed by discussion with the environmental regulator indicated that the primary regulatory concern was the potential impact of the MAR scheme on chloride concentrations within the receiving Crag aquifer. The regulator indicated that for licensing purposes it would use an 'absolute threshold' for chloride of 250 mg L^{-1} based on the DWS, and a 'relative threshold' limiting chloride concentrations in the aquifer to an increase of no more than 10% above baseline concentrations.

The results of the MAR trial demonstrated that the 'absolute threshold' was not challenged. Chloride concentrations within the source water averaged 134 mg L^{-1}, significantly below the 250 mg L^{-1} DWS limit. However, the 'relative threshold' may be more problematic. Background chloride concentrations in Crag groundwater below the recharge field averaged 117 mg L^{-1}, similar to the source water, but across the Crag aquifer, chloride concentrations vary from 13 to 2680 mg L^{-1} (Ander et al., 2006). The MAR trial further showed that the risk can be minimised by: (i) setting a limit to the chloride concentration in the source water; and (ii) introducing relatively small volumes of water into the aquifer and ensuring that a high proportion of this infiltration is re-abstracted.

Additional regulatory concerns related to other potential contaminants and water resource impacts on nearby water features (streams, licensed abstractions and protected rights). The risks of other contaminants were considered low in the case of the current MAR scheme. The source water was monitored 13 times over 24 months for over 570 compounds of potential concern and none were found to be problematic.

Table 4
Concentrations of chloride and nitrate-N in the recharge water and Crag groundwater at ABH1, ABH2 and OBH5 during the summer 2022 MAR trial (recharge from 9 to 20 June and recovery (re-abstraction) from 18 July–10 September). Concentrations are given in mg L^{-1}.

Period	Date	Recharge water		ABH1		ABH2		OBH5	
		Cl^-	NO_3^--N	Cl^-	NO_3^--N	Cl^-	NO_3^--N	Cl^-	NO_3^--N
Pre-recharge	08/06/22	136	0.20	146	34.3	123	36.8	86.4	42.5
Recharge	17/06/22	129	0.09	151	17.2	131	21.4	93.9	41.8
Recovery	19/07/22	–	–	147	10.7	121	22.4	92.2	40.4
Recovery	18/08/22	–	–	112	33.2	99.4	38.4	83.3	37.9
Post-recovery	12/09/22	–	–	96.3	37.7	90.4	42.2	82.9	39.1

Water resource impacts were also considered a low risk with any effects on the spring-fed lakes to the northeast of the recharge site likely to be below detection thresholds. Overall, the MAR scheme demonstrated that water quality and water resource requirements were successfully met, and that the MAR system can be managed to augment water resources in addition to providing an alternative means of storing high surface flows.

3.4. Economic assessment

For MAR to be a viable solution to alleviate water resource pressures, it must prove to be a cost-effective alternative to, for example, lined surface storage reservoirs. Compared to alternative resource options, the cost of MAR schemes can be favourably low (Hasan et al., 2019). The largest cost component of the MAR scheme presented here (Table 5) was for monitoring, data collection and licensing, which together accounted for approximately 57% of the total budget of GBP 178,815 at 2022 prices. The costliest elements were the installation of observation boreholes and loggers followed by water quality testing and analysis. Construction costs including the recharge field, abstraction boreholes and pumping systems accounted for the remaining 43% of the capital costs.

The breakdown of costs of the MAR scheme differed significantly from that of a surface storage reservoir of equivalent cost where construction work accounts for up to 86% of the overall budget of GBP 217,008 at 2022 prices. Design, reservoir licensing, environmental assessment, fencing and landscaping costs typically come to only 20% of the total budget (Weatherhead et al., 2014), although planning and archaeological investigation costs vary widely and can add significantly to the capital costs.

Construction costs for the MAR system were comparatively low but the high costs of data collection and securing regulatory permits brought the overall capital costs to within 18% of an equivalent surface storage reservoir. From an abstractor's perspective, unless a larger volume of water is available from a MAR scheme, this marginal difference is unlikely to outweigh the perceived difficulties and uncertainties involved in constructing and operating a MAR system.

The annual operational costs of the MAR scheme were estimated at 12% (GBP 21,458) of the total capital costs, the greatest proportion of which is the additional costs of pumping the recovered water into a balancing facility so that it can be re-abstracted at higher rates sufficient to run an irrigation system. In comparison, the annual operating costs for an equivalent reservoir are about 7% (GBP 15,191) of the capital costs and are comprised of, in diminishing order of size, power supply, maintenance costs and loss of income due to land-take (Weatherhead et al., 2014).

The higher data collection and licensing costs for the MAR scheme reflected the greater regulatory challenges. Aquifer recharge is a relatively novel technique in the UK and the licensing regime is required to address potential impacts on both groundwater quality and water resources. In contrast, agricultural surface storage reservoirs have a long history of development and licence considerations are limited to water resources impacts only. Water resources mitigation measures, such as protected minimum flows, are well understood and the licensing regime has matured to accommodate reservoir abstraction permits. It is likely, however, that if more MAR schemes are developed, the regulatory process will become more streamlined, reducing licensing overheads and making MAR more attractive to agricultural irrigators.

Table 5
Capital and annual operating costs for the MAR scheme presented in this study compared with a typical lined surface storage reservoir of equivalent cost.

Component		Cost (GBP)	Cost (GBP)
		Managed aquifer recharge	Surface storage reservoir
Capital costs	Construction	76,996	187,095
	Monitoring and testing	77,157	12,900
	Permits and licences	24,662	17,013
	Total	178,815	217,008
Annual operating costs		21,458	15,191

4. Conclusions

The key findings of this research are summarised as follows:

1. Overall, the MAR scheme developed in this study demonstrated that water quality and water resource requirements were successfully met, and that MAR can be managed to augment irrigation water resources. The recharge source water contained concentrations of hazardous substances below the relevant minimum reporting values with only the non-hazardous substances (chloride, sulphate and nitrate) considered to be slightly above the permitted limits on several occasions.
2. The MAR scheme proved technically feasible, with the recharge mound restricted to within 250 m of the recharge site for an injected volume of 12,262 m^3 during a period of 12 days. During this time, hydrochemical modelling showed that the fraction of recharge water mixing with the Crag groundwater ranged from 69% to 5% close to the centre and boundary of the recharge mound, respectively, leading to an improvement in background groundwater quality with a reduction in nitrate-N concentration of 23.6 mg L^{-1} at the centre of the recharge mound. Following infiltration, a volume of 12,301 m^3 of recharged water was successfully recovered over a period of 55 days for irrigation water use, during which the groundwater quality returned to background concentrations.
3. As applied in this study, the risk of groundwater quality deterioration can be minimised by setting thresholds on the source water and by introducing relatively small volumes of water into the aquifer and ensuring that a high proportion of this infiltration is re-abstracted.
4. As in other MAR studies (Knapton et al., 2019), the aquifer permeability was an important factor, with the Crag aquifer hydraulic conductivity ($\sim$10 m day^{-1}) and thin saturated zone ($\sim$5 m) at the recharge site found to be limiting factors constraining the volume of water that could be usefully stored and recovered.
5. Importantly, the loading of suspended inorganic and organic material from the recharge source water to the Crag aquifer was low and so limiting the risk, at least in the short term, of clogging and decline in permeability of the sands and gravels in the covered infiltration trenches, a problem that is reported for other MAR schemes (Kruisdijk et al., 2023).
6. In comparison to surface storage reservoirs that can require large areas of potentially productive land and involve high capital and environmental costs, MAR offers the opportunity of matching growth with enhanced environmental protection. Additionally, MAR provides an alternative means of storing high surface flows to ensure water supplies during periods of peak demand.
7. Construction costs for the MAR system designed in this study were comparatively low but the high costs of data collection and securing regulatory permits brought the overall capital costs to within 18% of an equivalent surface storage reservoir.
8. The marginal difference in costs in comparison to surface storage reservoirs is unlikely to outweigh the perceived difficulties and uncertainties involved in constructing and operating a MAR system without cost-effective, market-based mechanisms to incentivise groundwater recharge schemes on privately owned land (Bruce et al., 2023). Also, more streamlined regulatory processes to reduce licensing overheads are required.

9. This study showed that this and similar MAR schemes should ideally operate in conjunction with surface reservoir storage facilities to balance the demand, although the incorporation of reservoir storage has further economic implications. Alternatively, MAR could be used directly for low-volume irrigation systems such as trickle tape or mini sprinklers.
10. The wider benefits of the MAR scheme presented here are reflected in the contribution made to the sustainable use of water resources as an important strategy in helping alleviate regional water stress (Scanlon et al., 2023), while at the same time building partnerships that adhere to the international Sustainable Development Goals (United Nations, 2015).

Author contribution statement

Kevin Hiscock: Writing- original draft, Conceptualization, Formal analysis, Methodology, Investigation. **Natalia Balashova**: Writing-original draft, Formal analysis, Data curation. **Richard Cooper**: Writing-review and editing, Formal analysis. **Paul Bradford**: Writing-review and editing, Conceptualization, Investigation, Funding acquisition. **John Patrick**: Writing-review and editing, Conceptualization, Investigation, Funding acquisition. **Matt Hullis**: Writing-review and editing, Conceptualization, Funding acquisition.

Declaration of competing interest

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

Data availability

Data will be made available on request.

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Appendix A. Supplementary data

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References

Ali, I., Singh, P., Aboul-Enein, H.Y., Sharma, B., 2009. Chiral analysis of Ibuprofen residues in water and sediment. Anal. Lett. 42, 1747–1760.
Allen, D.J., Brewerton, L.J., Coleby, L.M., Gibbs, B.R., Lewis, M.A., MacDonald, A.M., Wagstaff, S.J., Williams, A.T., 1997. The physical properties of major aquifers in England and Wales. Brit. Geol. Surv. Tech. Rep. WD/97/34 8, 312. Environment Agency R&D Publication.
Ander, E.L., Shand, P., Wood, S., 2006. Baseline Report Series: 21. The Chalk and Crag of North Norfolk and the Waveney Catchment. British Geological Survey Commissioned Report No. CR/06/043N, p. 75.
Basheer, A.A., 2018a. Chemical chiral pollution: impact on the society and science and need of the regulations in the 21st century. Chirality 30, 402–406.
Basheer, A.A., 2018b. New generation nano-adsorbents for the removal of emerging contaminants in water. J. Mol. Liq. 261, 583–593.
Basheer, A.A., Ali, I., 2018. Stereoselective uptake and degradation of (±)-*o*,*p*-DDD pesticide stereomers in water-sediment system. Chirality 30, 1088–1095.
Bhattarai, N., Lobell, D.B., Balwinder-Singh, Fishman, R., Kustas, W.P., Pokhrel, Y., Jain, M., 2023. Warming temperatures exacerbate groundwater depletion rates in India. Sci. Adv. 9, eadi1401 https://doi.org/10.1126/sciadv.adi1401.
Black, A.D., Lewis, R.T., Grout, M.W., Witterick, W.R., 2012. Crossing boundaries, the influence of groundwater model boundaries and a method to join and split MODFLOW models. In: Shepley, M.G., Whiteman, M.I., Hulme, P.J., Grout, M.W. (Eds.), Groundwater Resources Modelling: A Case Study from the UK, vol. 364. Geological Society, London, Special Publications, pp. 155–172.
Bouwer, H., 2002. Artificial recharge of groundwater: hydrogeology and engineering. Hydrogeol. J. 10, 121–142.
Bouwer, H., Back, J.T., Oliver, J.M., 1999. Predicting infiltration and ground-water mounds for artificial recharge. J. Hydrol. Eng. 4, 350–357.
British Geological Survey, 2022a. 1:50,000 bedrock geology. Available at: https://mapapps2.bgs.ac.uk/geoindex/home.html.
British Geological Survey, 2022b. Principal aquifers in England and Wales: the Crag. Available at: https://www2.bgs.ac.uk/groundwater/shaleGas/aquifersAndShales/maps/aquifers/Crag.html.
Bruce, M., Sherman, L., Bruno, E., Fisher, A.T., Kiparsky, M., 2023. Recharge net metering (ReNeM) is a novel, cost-effective management strategy to incentivize groundwater recharge. Nat. Water 1, 855–863.
Burri, N.M., Weatherl, R., Moeck, C., Schirmer, M., 2019. A review of threats to groundwater quality in the anthropocene. Sci. Total Environ. 684, 136–154.
Casanova, J., Devau, N., Pettenati, M., 2016. Managed aquifer recharge: an overview of issues and options. In: Jakeman, A.J., Barreteau, O., Hunt, R.J., Rinaudo, J.-D., Ross, A. (Eds.), Integrated Groundwater Management: Concepts, Approaches and Challenges. Springer Open, pp. 413–434. https://doi.org/10.1007/978-3-319-23576-9.
Cooper, R.J., Hiscock, K.M., Lovett, A.A., Dugdale, S.J., Sünnenberg, G., Vrain, E., 2020. Temporal Hydrochemical Dynamics of the River Wensum, UK: Observations from Long-Term High-Resolution Monitoring (2011–2018). Sci. Total Environ., p. 724. https://doi.org/10.1016/j.scitotenv.2020.138253
Cranfield University, 2022. The Soils Guide. Cranfield University, UK. Available at: landis.org.uk.
Cui, W., Hao, Q., Xiao, Y., Zhu, Y., Li, J., Zhang, Y., 2022. Combining river replenishment and restrictions on groundwater pumping to achieve groundwater balance in the Juma River Plain, North China Plain. Front. Earth Sci. 10, 902034 https://doi.org/10.3389/feart.2022.902034.
Cuthbert, M.O., Gleeson, T., Bierkens, M.F.P., Ferguson, G., Taylor, R.G., 2023. Defining renewable groundwater use and its relevance to sustainable groundwater management. Water Resour. Res. 59, e2022WR032831 https://doi.org/10.1029/2022WR032831.
DEFRA, 2021. Water Abstraction Plan: Catchment Focus. Department for Environment, Food and Rural Affairs, London. Available at: https://www.gov.uk/government/publications/water-abstraction-plan-2017/water-abstraction-plan-catchment-focus.
DEFRA, 2023. Crop Map of England (CROME) 2020. Department for Environment, Food and Rural Affairs, London. Available at: https://environment.data.gov.uk/DefraDataDownload/?mapService=RPA/CropMapOfEngland2020SUF&mode=spatial.
Dillon, P., Pavelic, P., Page, D., Beringen, H., Ward, J., 2009. Managed aquifer recharge: an introduction. Waterlines Report Series No 13.. National Water Commission, Canberra, p. 64.
Dillon, P., Stuyfzand, P., Grischek, T., Lluria, M., Pyne, R.D.G., Jain, R.C., Bear, J., Schwarz, J., Wang, W., Fernandez, E., Stefan, C., Pettenati, M., van der Gun, J., Sprenger, C., Massmann, G., Scanlon, B.R., Xanke, J., Jokela, P., Zheng, Y., Rossetto, R., Shamrukh, M., Pavelic, P., Murray, E., Ross, A., Bonilla Valverde, J.P., Palma Nava, A., Ansems, N., Posavec, K., Ha, K., Martin, R., Sapiano, M., 2019. Sixty years of global progress in managed aquifer recharge. Hydrogeol. J. 27, 1–30.
Dillon, P., Alley, W., Zheng, Y., Vanderzalm, J. (Eds.), 2022. Managed Aquifer Recharge: Overview and Governance. IAH Special Publication, p. 90.
Downing, R.A., 1986. Development of groundwater. In: Brandon, T.W. (Ed.), Groundwater: Occurrence, Development and Protection. The Institution of Water Engineers and Scientists, London, pp. 485–542.
Environment Agency, 2020. Meeting Our Future Water Needs: a National Framework for Water Resources. Environment Agency, Bristol, p. 13.
European Commission, 1998. Council Directive 98/83/EC of 3 November 1998 on the quality of water intended for human consumption. Off. J. Eur. Communities - Legislation 330, 32–54.
European Commission, 2008. Council Directive 2008/105/EC of 16 December 2008 on environmental quality standards in the field of water policy. Off. J. Eur. Union L 348/84, 1–14.
Evans, R.S., Dillon, P., 2017. Linking groundwater and surface water: conjunctive water management. In: Villholth, K.G., Lopez-Gunn, E., Conti, K., Garrido, A., van der Gun, J. (Eds.), Advances in Groundwater Governance. CRC Press, London, pp. 329–352.
Fuentes, I., Vervoort, R.W., 2020. Site suitability and water availability for a managed aquifer recharge project in the Namoi basin, Australia. J. Hydrol.: Reg. Stud. 27, 100657 https://doi.org/10.1016/j.ejrh.2019.100657.
García-Menéndez, O., Renau-Pruñonosa, A., Morell, I., Ballesteros, B.J., Esteller, M.V., 2021. Hydrogeological changes during managed aquifer recharge (MAR) in a salinised coastal aquifer. Appl. Geochem. 126, 104866 https://doi.org/10.1016/j.apgeochem.2020.104866.
Garrard, N.L., Hiscock, K.M., Cooper, R.J., Marca, A.D., Sünnenberg, G., 2023. Hydrochemical and dual-isotope approach to the identification of denitrification in

arable field drainage in the Wensum catchment, eastern England. Appl. Geochem. 158, 105803 https://doi.org/10.1016/j.apgeochem.2023.105803.
Gleeson, T., Wada, Y., Bierkens, M.F.P., van Beek, L.P.H., 2012. Water balance of global aquifers revealed by groundwater footprint. Nature 488, 197–200.
Gooddy, D.C., Hinsby, K., 2009. Organic quality of groundwaters. In: Edmunds, W.M., Shand, P. (Eds.), Natural Groundwater Quality. Blackwell, Oxford, pp. 59–70.
Hasan, M.B., Driessen, P.P.J., Majumder, S., Majumber, S., Zoomers, A., van Laerhoven, F., 2019. Factors affecting consumption of water from a newly introduced safe drinking water system: the case of managed aquifer recharge (MAR) systems in Bangladesh. Water 11, 2459. https://doi.org/10.3390/w11122459.
Heilweil, V.M., Benoit, J., Healy, R.W., 2015. Variably saturated groundwater modelling for optimizing managed aquifer recharge using trench infiltration. Hydrol. Process. 29, 3010–3019.
Hingst, M.C., McQuiggan, R.W., Peters, C.N., He, C., Andres, A.S., Michael, H.A., 2023. Surface water-groundwater connections as pathways for inland salinization of coastal aquifers. Groundwater 61, 626–638.
Hunter Blair, A., 1970. Artificial recharge of groundwater. In: Technical Paper 75, the Water Research Association, Medmenham. Marlow, p. 64.
Hutchinson, A., Milczarek, M., Banerjee, M., 2013. Clogging phenomena related to surface water recharge facilities. In: Martin R, R. (Ed.), Clogging Issues Associated with Managed Aquifer Recharge Methods. IAH Commission on Managing Aquifer Recharge, Australia, pp. 95–106.
IGRAC, 2023. MAR Portal. International Groundwater Resources Assessment Centre. https://www.un-igrac.org/ggis/mar-portal.
Itoh, M., Ohte, N., Koba, K., Katsuyama, M., Hayamizu, K., Tani, M., 2007. Hydrologic effects on methane dynamics in riparian wetlands in a temperate forest catchment. J. Geophys. Res. 112, G01019 https://doi.org/10.1029/2006JG000240.
Ji, L., Senay, G.B., 2023. Temporal trends in agricultural water use and the relationships to hydroclimatic factors in the High Plains aquifer region. J. Am. Water Resour. Assoc. 59, 950–969.
Jones, H.K., Macdonald, D.M.J., Gale, I.N., 1998. The Potential for Aquifer Storage and Recovery in England and Wales. British Geological Survey Report WD/98/26, p. 36.
Jones, H.K., Morris, B.L., Cheney, C.S., Brewerton, L.J., Merrin, P.D., Lewis, M.A., MacDonald, A.M., Coleby, L.M., Talbot, J.C., McKenzie, A.A., Bird, M.J., Cunningham, J., Robinson, V.K., 2000. The Physical Properties of Minor Aquifers in England and Wales, vol. 68. British Geological Survey Technical Report, WD/00/4. Environment Agency R&D Publication, p. 234.
Kattan, Z., Kadkoy, N., Nasser, S., Safadi, M., Hamed, A., 2010. Isotopes and geochemistry in a managed aquifer recharge scheme: a case study of fresh water injection at the Damascus University Campus, Syria. Hydrol. Process. 24, 1791–1805.
Knapton, A., Page, D., Vanderzalm, J., Gonzalez, D., Barry, K., Taylor, A., Horner, N., Chilcott, C., Petheram, C., 2019. Managed aquifer recharge as a strategic storage and urban water management tool in Darwin, Northern Territory, Australia. Water 11, 1869. https://doi.org/10.3390/w11091869.
Kruisdijk, E., Ros, J.F., Ghosh, D., Brehme, M., Stuyfzand, P.J., van Breukelen, B.M., 2023. Prevention of well clogging during aquifer storage of turbid tile drainage water rich in dissolved organic carbon and nutrients. Hydrogeol. J. 31, 827–842.
Lindim, C., van Gils, J., Cousins, I.T., 2016. Europe-wide estuarine export and surface water concentrations of PFOS and PFOA. Water Res. 103, 124–132.
Lippera, M.C., Werban, U., Vienken, T., 2023. Improving clogging predictions at managed aquifer recharge sites: a quantitative assessment on the vertical distribution of intrusive fines. Hydrogeol. J. 31, 71–86.
Maliva, R.G., 2015. Managed aquifer recharge: state-of-the-art and opportunities. Water Sci. Technol. Water Supply 15, 578–588.
Mathers, S.J., Smith, N.J.P., 2002. *Geology of the Woodbridge and Felixstowe District: a Brief Explanation of the Geological Map Sheets 208 and 225 Woodbridge and Felixstowe*. British Geological Survey, Nottingham, p. 34.
Meteorological Office, 2023. *UK climate averages: Levington 1991–2020*. Meteorological Office, Exeter. Available at: https://www.metoffice.gov.uk/research/climate/maps-and-data/uk-climate-averages/u10zuu3ye.
O'Shea, M.J., Sage, R., 1999. Aquifer recharge: an operational drought-management strategy in North London. J. Inst. Water Environ. Manag. 13, 400–405.
Outram, F.N., Cooper, R.J., Sünnenberg, G., Hiscock, K.M., Lovett, A.A., 2016. Antecedent conditions, hydrological connectivity and anthropogenic inputs: factors affecting nitrate and phosphorus transfers to agricultural headwater streams. Sci. Total Environ. 545–546, 184–199.
Rafiq, M.R., Ahmed, K.M., Rietveld, L.C., van Breukelen, B.M., 2022. Monitoring and inverse modelling of hydrogeochemical processes during managed aquifer recharge in Southwestern Bangladesh. Appl. Geochem. 146, 105472 https://doi.org/10.1016/j.apgeochem.2022.105472.
Raulerson, S., Jeffers, J.B., Griffiths, N.A., Rau, B.M., Matteson, C., Jackson, C.R., 2023. Rapid denitrification of nitrate-contaminated groundwater in a low-gradient blackwater stream valley. Biogeochemistry. https://doi.org/10.1007/s10533-023-01077-0.
Rawluk, A., Curtis, A., Sharp, E., Kelly, B.F.J., Jakeman, A.J., Ross, A., Arshad, M., Brodie, R., Pollino, C.A., Sinclair, D., Croke, B., Qureshi, M.E., 2013. Managed aquifer recharge in farming landscapes using large floods: an opportunity to improve outcomes for the Murray-Darling Basin? Australas. J. Environ. Manag. 20, 34–48.
Rivett, M.O., Buss, S.R., Morgan, P., Smith, J.W.N., Bemment, C.D., 2008. Nitrate attenuation in groundwater: a review of biogeochemical controlling processes. Water Res. 42, 4215–4232.
Rohde, M.M., 2023. Financial incentives can leverage existing infrastructure to replenish groundwater. Nat. Water 1, 831–832.
Ross, A., Hasnain, S., 2018. Factors affecting the cost of managed aquifer recharge (MAR) schemes. Sustain. Water Resourc. Manag. 4, 179–190.
Salameh, E., Abdallat, G., van der Valk, M., 2019. Planning considerations of managed aquifer recharge (MAR) projects in Jordan. Water (Switzerland) 11, 182. https://doi.org/10.3390/w11020182.
Scanlon, B.R., Fakhreddine, S., Rateb, A., de Graaf, I., Famiglietti, J., Gleeson, T., Grafton, R.Q., Jobbagy, E., Kebede, S., Kolusu, S.R., Konikow, L.F., Long, D., Mekonnen, M., Schmied, H.M., Mukherjee, A., MacDonald, A., Reedy, R.C., Shamsudduha, M., Simmons, C.T., Sun, A., Taylor, R.G., Villholth, K.G., Vörösmarty, C.J., Zheng, C., 2023. Global water resources and the role of groundwater in a resilient water future. Nat. Rev. Earth Environ. 4, 87–101.
Schipanski, M.E., Sanderson, M.R., Méndez-Barrientos, L.E., Kremen, A., Gowda, P., Porter, D., Wagner, K., West, C., Rice, C.W., Marsalis, M., Guerrero, B., Haacker, E., Dobrowolski, J., Ray, C., Auvermann, B., 2023. Moving from measurement to governance of shared groundwater resources. Nat. Water 1, 30–36.
Schmidt, C.M., Fisher, A.T., Racz, A.J., Lockwood, B.S., Los Huertos, M., 2011. Linking denitrification and infiltration rates during managed groundwater recharge. Environ. Sci. Technol. 45, 9634–9640.
Schubert, J., 2002. Hydraulic aspects of riverbank filtration – field studies. J. Hydrol. 266, 145–161.
Song, Y., Du, X., Ye, X., 2019. Analysis of potential risks associated with urban stormwater quality for managed aquifer recharge. Int. J. Environ. Res. Publ. Health 16, 3121. https://doi.org/10.3390/ijerph16173121.
Sprenger, C., Hartog, N., Hernández, M., Vilanova, E., Grützmacher, G., Scheibler, F., Hannappel, S., 2017. Inventory of managed aquifer recharge sites in Europe: historical development, current situation and perspectives. Hydrogeol. J. 25, 1909–1922.
Stefan, C., Ansems, N., 2018. Web-based global inventory of managed aquifer recharge applications. Sustain. Water Resourc. Manag. 4, 153–162.
Steiness, M., Jessen, S., van't Veen, S.G.M., Kofod, T., Højberg, A.L., Engesgaard, P., 2021. Nitrogen-loads to streams: importance of bypass flow and nitrate removal processes. J. Geophys. Res.: Biogeosciences 126, e2020JG006111. https://doi.org/10.1029/2020JG006111.
Taylor, R.G., Scanlon, B., Döll, P., Rodell, M., Beek, R.V., Wada, Y., Longuevergne, L., Leblanc, M., Famiglietti, J.S., Edmunds, M., Konikow, L., Green, T.R., Chen, J., Taniguchi, M., Bierkens, M.F.P., MacDonald, A., Fan, Y., Maxwell, R.M., Yechieli, Y., Gurdak, J.J., Allen, D.M., Shamsudduha, M., Hiscock, K., Yeh, P.J.F., Holman, I., Treidel, H., 2014. Groundwater and climate change. Nat. Clim. Change 3, 322–329.
UKCEH, 2023. *National River Flow Archive: 35014 – Mill River at Newbourne*. UK Centre for Ecology and Hydrology. Available at: https://nrfa.ceh.ac.uk/data/station/spatial/35014.
UNESCO, 2022. Groundwater: Making the Invisible Visible. The United Nations World Water Development Report 2022. UNESCO, Paris, p. 225.
United Nations, 2015. Transforming our world: the 2030 agenda for sustainable development. Available at: https://sdgs.un.org/goals.
Walton, C.R., Zak, D., Audet, J., Petersen, R.J., Lange, J., Oehmke, C., Wichtmann, W., Kreyling, J., Grygoruk, M., Jabłońska, E., Kotowski, W., Wiśniewska, M.M., Ziegler, R., Hoffmann, C.C., 2020. Wetland buffer zones for nitrogen and phosphorus retention: impacts of soil type, hydrology and vegetation. Sci. Total Environ. 727, 138709 https://doi.org/10.1016/j.scitotenv.2020.138709.
Water Resources East, 2022. Draft Regional Water Resources Plan for Eastern England. Water Resources East (WRE) Ltd, Norwich, p. 91.
Weatherhead, K., Knox, J., Daccache, A., Morris, J., Groves, S., Hulin, A., Kay, M., 2014. Water for Agriculture: Collaborative Approaches and On-Farm Storage. Water for Agriculture (FFG1112) Final Report, March 2014. Cranfield University, Cranfield, p. 59.
Yuan, J., Van Dyke, M.I., Huck, P.M., 2016. Water reuse through managed aquifer recharge (MAR): assessment of regulations/guidelines and case studies. Water Qual. Res. J. Can. 51, 357–376.
Yuan, J., Van Dyke, M.I., Huck, P.M., 2019. Selection and evaluation of water pretreatment technologies for managed aquifer recharge (MAR) with reclaimed water. Chemosphere 236, 124886. https://doi.org/10.1016/j.chemosphere.2019.124886.
Zareitalabad, P., Siemens, J., Hamer, M., Amelung, W., 2013. Perfluorooctanoic acid (PFOA) and perflorooctanesulfonic acid (PFOS) in surface waters, sediments, soils and wastewater – a review on concentrations and distribution coefficients. Chemosphere 91, 725–732.
Zhang, H., Xu, Y., Kanyerere, T., 2020. A review of the managed aquifer recharge: historical development, current situation and perspectives. Phys. Chem. Earth 118–119, 102887. https://doi.org/10.1016/j.pce.2020.102887.
Zheng, Y., Ross, A., Villhoth, K., Dillon, P., 2021. Managing Aquifer Recharge: A Showcase for Resilience and Sustainability. UNESCO, Paris, p. 379.
Zheng, Y., Vanderzalm, J., Hartog, N., Fernández Escalante, E., Stefan, C., 2023. The 21st century water quality challenges for managed aquifer recharge: towards a risk-based regulatory approach. Hydrogeol. J. 31, 31–34.