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Recharge performance of pond-based managed aquifer recharge in the Ramganga basin

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Abstract

Groundwater underpins India's water and food security but faces growing stress from overexploitation and climate change. This study evaluates the hydrologic performance of small-scale, pond-based managed aquifer recharge (MAR) interventions implemented under government development programs, in the context of India's plan to construct 14 million additional recharge structures. Focusing on the groundwater-stressed Ramganga basin, the study draws on three years of participatory monitoring of 23 recharge structures. Using topographic surveys, staff gauge installations, daily water-level records maintained by trained community members, and water-balance methods, 236 recharge spells were captured and associated (potential) recharge volumes quantified. While the structures demonstrated utility in recharging groundwater, performance remained below potential with a consistent declining trend. The ratio of recharge volume to pond storage averaged 1.75 in 2023 but fell to ~ 0.8 in 2024 and 2025, far below values of 5–12 reported in a nearby MAR pilot. Substantial variability was observed across structure types and seasons. Declining recharge rates were linked to siltation, inadequate maintenance, and hydrogeological constraints. Pond water availability (~ 140 d per season) was not the primary constraint; instead, low infiltration rates mostly below 40 mm d^{-1} emerged as the dominant limiting factor. High evaporation losses ($\sim 28\%$ of pond storage) further reduced effective recharge. These findings highlight the need for performance-based MAR planning, integration with watershed measures, regular operation and maintenance to manage siltation, context-specific designs particularly for multipurpose ponds and participatory monitoring to build local ownership and capacity.

1. Introduction

Groundwater plays a critical role in global food security. Around 70% of global groundwater withdrawals go toward agriculture, underpinning close to 40% of irrigated cropland worldwide (OECD 2015, United Nations 2022). Its importance is particularly evident in India, where on the order of 20 million pumps abstract around 250 billion cubic meters (BCMs) of groundwater, and accounting for two-thirds of the total irrigated area (Sikka *et al* 2021, CGWB 2025). The rapid growth of groundwater-based irrigation was a key driver of India's Green Revolution, contributing substantially to national food security and poverty reduction (Mukherji 2022).

However, intensive groundwater use for irrigation has led to unsustainable abstraction, with roughly 25% of India's groundwater classified as overexploited, critical, or semi-critical, especially in Northwest India and the Southern Peninsula (CGWB 2025). This is especially severe across Northwestern India, where more than 80% of irrigated area is dependent on groundwater to grow water-demanding crops like paddy, wheat, and sugarcane (Alam *et al* 2025). It is estimated that almost half of dry-season irrigated crop production currently is directly supported by unsustainable groundwater use (Zaveri *et al* 2016). Ensuring sustainable and resilient groundwater systems for irrigation is therefore crucial for

maintaining food security (Jain *et al* 2021), particularly amid growing climate variability and increasing frequency of floods and droughts (Sikka *et al* 2021; Mukherji, 2022).

To address groundwater depletion and enhance groundwater security, government agencies, non-governmental organizations, and the private sector have increasingly supported the design and implementation of managed aquifer recharge (MAR) interventions (Sikka *et al* 2021, Alam *et al* 2022, 2025). MAR involves targeted recharging of aquifers during periods of surface water surplus through purposefully built interventions, helping augment groundwater storage during monsoon seasons, to sustain dry season irrigation, and mitigate long-term depletion (Dillon *et al* 2019, Alam and Pavelic 2020). India's Master Plan for Artificial Recharge envisages recharging 185 BCM of water through ~ 14 million MAR structures at an estimated cost of ~ 16 billion USD (CGWB 2021). Since the 1990s, MAR has been integrated into watershed development programs, rainwater harvesting schemes, and large-scale public investments such as National Rural Employment Guarantee Scheme (NREGS) and Integrated Watershed Development Programs (Massuel *et al* 2014, Nattorp *et al* 2016, Nafil *et al* 2025). It is estimated that already >1 million recharge structures have been built in last three decades, although precise numbers are unknown (Nattorp *et al* 2016, Nafil *et al* 2025). The structures are primarily small scale, such as water harvesting ponds, check dams, tanks and recharge wells.

Robust scientific evaluations of MAR performance in India have remained limited (Nattorp *et al* 2016; Richard-Ferroudji *et al* 2018, Gale *et al* 2006). The number of published studies that document performance of recharge structures is miniscule compared to scale of investments. Most studies document fewer than 10 structures and overwhelmingly focus on hard rock parts of peninsular India (Glendenning and Vervoort 2010, Massuel *et al* 2014, Surinaidu *et al* 2016, Dashora *et al* 2018, Jalan *et al* 2025). Thus, published results giving systematic and quantitative evidence of recharge structures in large alluvial systems such as the Ganges basin has been sparse (Kaledhonkar *et al* 2003, Gale *et al* 2006, Alam *et al* 2020). Whilst a compendium of MAR case studies has been published by the Central Government Ground Water Board, it documents qualitative 'success stories' with limited data and analysis (Central Ground Water Board (CGWB) 2011). Thus, lack of systematic monitoring of recharge structures raises concerns about their capacity to address overexploitation effectively and sustainably (Alam *et al* 2025).

While MAR studies from other global contexts provide useful benchmarks for system performance (Gale *et al* 2006, Nattorp *et al* 2016, Alam *et al* 2025), the limited and largely qualitative evidence base from India constrains meaningful cross-comparison. Global reviews highlight that MAR performance is strongly conditioned by site-specific factors, including soil properties, water availability, and local hydrogeology (Alam *et al* 2021). The transferability of findings from intensively managed and well-monitored systems in countries such as the United States, Australia, or Israel where dedicated infrastructure, engineered basins, and advanced treatment objectives such as soil aquifer treatment (SAT) and agricultural SAT (Ag-SAT) have been developed under conditions of land and water scarcity (Grinshpan *et al* 2022) to the diverse, data-scarce alluvial settings of the Ganges basin therefore also remains uncertain. Without systematic monitoring and quantitative performance data from India's recharge structures, benchmarking against such global MAR sites remains a challenge.

Within this broader context, the paper presents the performance of small recharge ponds ($\sim <2$ ha in size) based upon a representative subset of existing 23 sites monitored over 3 years in an alluvial aquifer setting within the Ganges basin where groundwater stress is most acute. The performance is assessed by quantifying the infiltration water flux from the base of ponds to estimate potential groundwater recharge. For brevity, this flux is referred to as 'recharge' throughout the text. The aim is to compare their performance levels, generate insights on factors limiting performance and develop recommendations to support more efficient and sustainable MAR programming.

2. Study area and methods

2.1. Description of the study area

The study was carried out in the Moradabad and Rampur districts, located in the western part of the state of Uttar Pradesh in India and falling entirely within the Ramganga River Basin (figure 1(a)). Ramganga river is a major tributary of the Ganga river system. The Ramganga basin was selected as the it lies in the Western Gangetic Plain, where groundwater depletion is most acute (Alam *et al* 2025). Within the basin, two districts were purposively selected, both categorized as semi-critical (groundwater abstraction 70%–90% of annual groundwater recharge) in terms of groundwater development status by the Central Ground Water Board (Central Ground Water Board (CGWB) 2025). The selection was further informed by prior documentation of MAR pilot interventions in the region (Alam *et al* 2020) and

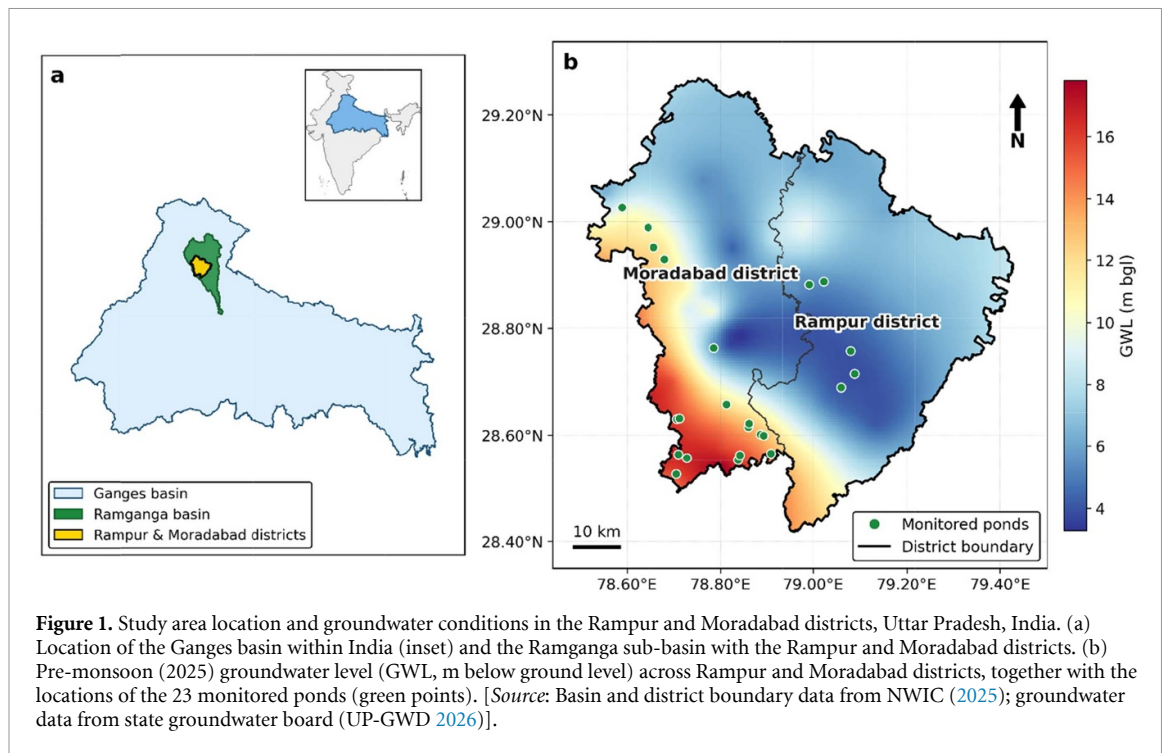


Figure 1. Study area location and groundwater conditions in the Rampur and Moradabad districts, Uttar Pradesh, India. (a) Location of the Ganges basin within India (inset) and the Ramganga sub-basin with the Rampur and Moradabad districts. (b) Pre-monsoon (2025) groundwater level (GWL, m below ground level) across Rampur and Moradabad districts, together with the locations of the 23 monitored ponds (green points). [Source: Basin and district boundary data from NWIC (2025); groundwater data from state groundwater board (UP-GWD 2026)].

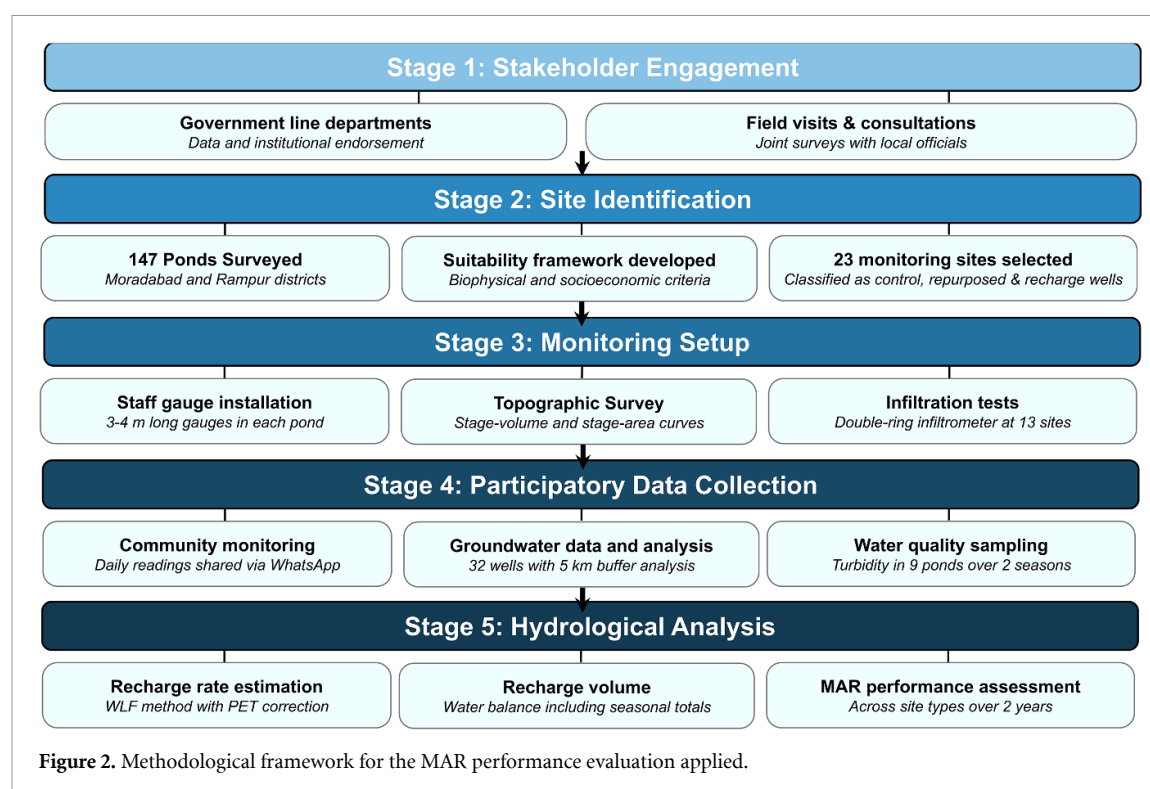
the substantial scale of recent public investment in recharge infrastructure, in rejuvenation of water bodies and recharge wells under initiatives such as Amrit Sarovar (Alam *et al* 2025).

In Moradabad and Rampur district, the mean annual rainfall is about 840 mm and 1121 mm, respectively, of which the majority (>85%) occurs during the southwest monsoon from June–September (Alam *et al* 2025; Pai *et al* 2014). The annual potential evapotranspiration (PET) in the sub-humid region is estimated at ~1400 mm, which exceeds the average rainfall, thereby highlighting the high dependence on groundwater resources (Behera 2020, Mehrotra 2020).

The study area is underlain by Quaternary alluvium deposited by the Ganga and Ramganga river systems. These deposits comprise sand, silt, and clay in varying proportions, with their relative abundance and distribution varying spatially across the district (Behera 2020, Mehrotra 2020). The alluvial aquifers reach depths of up to >200 m below ground level (mbgl), with unconfined aquifers typically found within the top 100 m (Behera 2020, Mehrotra 2020). Cropping patterns in the Rampur and Moradabad districts are primarily dominated by three major water-intensive crops, rice, wheat, and sugarcane, which together occupy over 85% of the cultivated area with >90% area being irrigated and 90% of this irrigation being dependent on groundwater sources (Alam *et al* 2025). Groundwater resources in the area are tapped through shallow tube wells (up to 30–50 m in depth) (Behera 2020, Mehrotra 2020).

The regional groundwater gradient in the study area trends from the north-eastern parts (shallow water levels ~3–5 mbgl in post monsoon season) towards the southern parts (deeper groundwater levels ~12–15 mbgl), reflecting the natural direction of groundwater flow (Behera 2020, Mehrotra 2020, figure 1). However, due to this extensive reliance on groundwater for water intensive crops, groundwater levels have been declining consistently, with an average post-monsoon depletion rate of 0.24–0.28 m per year and a pre-monsoon depletion rate of 0.15–0.17 m per year (Alam *et al* 2025). In Moradabad, the overall stage of groundwater extraction in the district is approximately 84.9% with 6 blocks (an administrative/revenue sub-division which is intermediate between district and village level, sometimes called ‘tehsil’/‘taluka’ equivalent) classified as overexploited or critical or semi critical whereas in Rampur over groundwater extraction is 71.5% with 4 blocks out of 6 classified as semi critical (CGWB 2025). Over the years, especially in the southern parts, this has resulted in the surface mounted diesel pumps reaching their suction limits and replacement of surface mounted pumps with submersible shallow tube wells.

To manage depletion, government has invested in range of water harvesting structures under multiple government programs such as the Mahatma Gandhi National Rural Employment Guarantee Scheme (MGNREGS), Amrit Sarovar, the Pradhan Mantri Krishi Sinchayee Yojana (PMKSY)-Watershed project, among others (Alam *et al* 2025). In the Ramganga basin, MGNREGS scheme alone constructed or rejuvenated 11 843 MAR structures between 2008 and 2023, most of these being water harvesting ponds, serving as recharge infiltration ponds (Alam *et al* 2025).



2.2. Methods

This study employed a structured, multi-stage methodology integrating government data, field surveys, participatory monitoring, and hydrological analysis to evaluate MAR performance of village ponds across Rampur and Moradabad districts, Uttar Pradesh (figure 2).

Existing pond datasets were compiled with input from district line departments following initial stakeholder consultations and based on that, 147 ponds were surveyed across the two districts (Alam *et al* 2026). A suitability framework was applied to select 23 representative monitoring sites spanning three pond typologies: control, repurposed, and repurposed with recharge structures (section 2.2.1). At each site, staff gauges and topographical surveys were deployed to underpin stage–volume and stage–area relationships, with water level data collected through participatory monitoring by trained local volunteers using mobile applications (sections 2.2.2 and 2.2.3). A combined water-balance and water-level-fluctuation approach, incorporating evapotranspiration losses throughout, was used to derive recharge rates and volumes (section 2.2.4). Groundwater level data from 72 regional monitoring wells across the districts were analyzed and double-ring infiltrometer tests at 13 sites was conducted to contextualize recharge performance (sections 2.2.5 and 2.2.6).

This study focused on quantifying the infiltration water flux from the base of the ponds as a measure of potential recharge contribution. It did not incorporate dedicated groundwater level monitoring in the immediate vicinity of ponds as previous studies (Alam *et al* 2020) have shown a detailed network is needed to ascertain any kind of groundwater response in this region as the hydraulic mounding associated with recharge from individual ponds is limited by the high aquifer transmissivity conditions (Alam *et al* 2020).

2.2.1. Selection of monitoring sites

Site selection began with compiling existing pond records from district line departments and local officials. Based on that, 147 ponds concentrated in the south-western part of the study area (figure 1(b)), where groundwater levels are comparatively deeper, were surveyed. This inventory covered both recently rejuvenated ponds and ponds without recent intervention (Alam *et al* 2026). A structured field questionnaire was then administered at each surveyed pond to record biophysical and socio-economic attributes—pond area, depth, ownership, use, accessibility, groundwater depth, and community interest (table A1). These indicators were used to select representative sites for detailed monitoring ensuring a mix of site types that included: (1) control (ponds which have not been rejuvenated); (2) repurposed (ponds which have been rejuvenated with de-siltation and embankment); (3) repurposed with recharge wells installed and (4) repurposed with recharge pits installed (figure 3).



Figure 3. From top left (a) Control ponds (Sanai Roza), (b) Repurposed ponds (Sanai Roza (Amrit Sarovar)), (c) Repurposed ponds with recharge wells (Segankendra). Water enters recharge wells from the top to ensure minimum pond water for other purposes and (d) Repurposed ponds with recharge pits (Changeri).

Repurposed ponds with recharge wells or recharge pits are repurposed ponds where, in addition to de-siltation and embankment repair, a recharge well or recharge pit has been installed. The two structures differ fundamentally in how water reaches the aquifer. A recharge well is a vertical shaft/borehole, similar to the one piloted by Alam *et al* (2020) in Rampur, with a well diameter of 100–200 mm drilled to a depth of 20–30 m in the area, ending in the saturated aquifer below the vadose zone. Recharge pits, by contrast, are shallow pits (2.5–3 m in depth, 3 m × 3 m in surface area) filled with graded filter media consisting of coarse boulders (≈ 40 mm) at the bottom layer, followed by medium-sized gravel (≈ 20 mm) in the middle layer, and fine gravel (≈ 10 mm) in the upper layer. The top surface of the filter media is covered with a 50% garden shade net to prevent entry of debris/sediment. These recharge pits were installed prior to the monsoon in 2025 in four ponds (table A3). These sites were selected based on: (i) presence of a clay layer at the pond bottom, (ii) availability of water in the pond for >120 d in the preceding years, and (iii) low recharge rates (<30 mm d^{-1}) in preceding years. Recharge pits were preferred over recharge wells at these sites for two main reasons: (i) water entering recharge pits percolates through the vadose zone, allowing natural filtration before reaching the aquifer, unlike recharge wells, which bypass this layer; and (ii) recharge pits are easier to construct, maintain, and clean (CGWB 2007).

Following endorsement from local stakeholders, site-level monitoring began with 15 ponds in 2023; 8 additional ponds joined in 2024 (23 in total), and by 2025 monitoring had ceased at 5 ponds for operational reasons, leaving 18 actively monitored. The details of the selected sites are provided in table A2. For analysis and presentation of results, recharge pit sites are considered in the repurposed ponds category for the year 2023 and 2024 and recharge pits category in 2025.

2.2.2. Participatory pond water level monitoring

Each pond was equipped with a single staff gauge, with maximum length based on the pond depth. In most ponds, a 3–4 m staff gauge was installed, while a few ponds were equipped with a 4 m staff gauge. The gauge was installed at a location representing average water level fluctuations, free from obstructions such as vegetation or debris, and away from inlets or outlets. Community members in nearby villages willing to support monitoring activities were identified. These individuals received hands-on training, which focused primarily on (i) accurate reading and interpretation of staff gauge measurements, (ii) systematic documentation of pond water levels, and (iii) capturing and transmitting photographic records of staff gauges via WhatsApp on a daily basis. During the monsoon (recharge season when ponds have water), these community members recorded water levels by photographing the staff gauge from the pond embankment, ensuring the scale is clearly visible and legible. To ensure accuracy, a mobile application

that automatically records the location, date, and time was installed on each monitoring person's phone. This enabled verification of the exact time and place of data collection. Monitoring was carried out daily from 1 July to 31 December for each monitoring year. In addition, nine of the monitored ponds were also sampled for water quality parameters, including turbidity, over two monsoon seasons: 2024 and 2025.

2.2.3. Development of stage–volume curves

To derive stage volume curves for ponds giving pond volume corresponding to different levels in the pond, topographical surveys were carried out. A traverse survey covering both the pond edge and interior points was carried out, along with water surface profiling at different stages to determine elevations precisely. The collected data were processed for creating contour map. To derive stage volume curves for ponds, the staff gauge location was compared to a pond contour map (from topographical surveys) to get its relative elevation. Thereafter, stage–volume data for each site were compiled and analyzed using regression-based LOcally WEighted Scatter-plot Smoother method (Cleveland 1979). Similarly, a stage–area curve capturing pond water spread area for a given water level was developed individually for each site. The best-fit models were identified and applied to develop mathematical relationships between stage–volume and stage–area. The model equations, polynomial degrees, and coefficients of determination (R^2) are presented in table A4 and A5. These best-fit models were subsequently used to estimate the water volume and water spread surface area corresponding to staff gauge levels. In 2024, staff gauges were repositioned across four monitoring sites to more accessible and representative locations. Consequently, the stage–volume relationships were recalibrated, and the datasets were updated for each year (2023 and 2024) for the same sites, as presented in appendix 4.

2.2.4. Estimation of recharge rate and recharge volume

To estimate recharge rates, first recharge spells were identified from the monitored water levels by segregating periods when the pond water level shows a continuous decline with no recorded rainfall and inflow into the pond from any other sources. To ensure consistency and remove any low- or high-level errors, each recharge spell was bounded by a minimum water level threshold (0.10 m) and below overflow (different for each pond) conditions. The estimation of recharge rate has been carried out using water level fluctuation method. This method estimates final net recharge rate as the total decline in water level minus the average PET (equation (1)) using the Blaney–Criddle formula (FAO 1986). The Blaney–Criddle method was selected due to its requirement of only temperature and daylight data, although it may lack accuracy due to the absence of critical factors such as solar radiation and wind speed. The average daily recharge rate within each recharge spell was calculated and used to estimate the representative recharge rate for that spell,

$$r_{mm} = \left[\frac{(h_{t0} - h_{t1})}{n} \right] - \text{PET} \quad (1)$$

where,

r_{mm} = Recharge rate in mm d^{−1}

$h_{t0} - h_{t1}$ Actual change in water level over the interval where h_{t0} and h_{t1} is the water level at the start and end of the recharge spell. $h_{t0} - h_{t1}$ is always positive as recharge spells are defined by the recession curve following rainfall events

n = Number of days in the interval

PET = Potential evapotranspiration

For estimating PET using the Blaney–Criddle formula, monthly maximum, minimum, and mean air temperature data were collected from the nearest automatic weather station installed by the India Meteorological Department. The mean daily percentage of annual daytime hours (p) corresponding to each month was obtained from a standard table.

For each recharge spell, change in pond volume was estimated using the stage–volume curve (section 2.2.3) from which evaporative losses were subtracted (equation (2)). The daily evaporative losses were estimated by multiplying pond surface spread area (for a given level using area–level relationship section 2.2.2) with estimated PET. For days falling outside the identified recharge spells (i.e. gap days between two consecutive spells), the recharge volume was estimated assuming the recharge rate to be same as the average daily recharge of the immediately preceding spell. For example, if a recharge spell started on 19 July and ended on 4 August, and the subsequent spell began on 7 August, the recharge volumes for the intervening days (5 and 6 August) were estimated by applying the average daily recharge

calculated from the 19 July–4 August spell. This assumption is supported by the distribution of gap lengths across the dataset. Non-recharge (gap) days between consecutive spells account for only 15% of total monitored days across all site-years (1003 of 6679 d), and the majority of individual gaps are short (median = 1.5 d; 69% of gaps are ≤ 3 d, 84% are ≤ 7 d), limiting the influence of this assumption on total seasonal recharge estimates for most spells (table A6). A minority of gaps exceeded 14 d (15 of 186, 8%), together accounting for 56% of total gap-days; for these, the average recharge rate of the preceding spell may not hold, introducing additional uncertainty into seasonal recharge volumes for the affected site-years.

Finally, the seasonal groundwater recharge was obtained by summing the recharge volumes of all identified spells along with the estimated recharge for the intervening days,

$$R_{v(i)} = V_{t_0} - V_{t_1} - \sum_0^n \text{PET}_d * A_d \quad (2)$$

where,

R_v = Recharge volume for the recharge spell i

$V_{t_0} - V_{t_1}$ = Actual change in water volume over the interval where V_{t_0} and V_{t_1} is the water volume at the start and end of the recharge spell

PET_d = Potential evapotranspiration for day d

A_d = Water spread area on day d

n = Number of days in the interval.

2.2.5. Basic infiltration rates of pond bottom

To estimate basic infiltration rates of ponds that could be compared with the recharge rates monitored, infiltration tests using the double-ring infiltrometer were carried out (Jalan and Lakshmikantha 2024). In 2024, double-ring infiltrometer tests were conducted in 13 ponds. For each pond, infiltration measurements were taken at two locations on the pond bottom during the dry season. Two concentric metal rings (inner diameter: 45 cm; outer diameter: 60 cm) were inserted vertically into the soil to a depth of 10–15 cm with minimal disturbance. Both rings were simultaneously filled with water to a constant head, ensuring equal water levels to reduce lateral flow. The outer ring maintained hydraulic equilibrium, promoting vertical infiltration from the inner ring. The constant head method was adopted, with the volume of water required to maintain a fixed level recorded at regular intervals. The decline in water depth within the inner ring was measured until a steady infiltration rate was achieved (Andrieux *et al* 2025). Instantaneous infiltration rates were computed as the ratio of infiltrated depth to the corresponding time interval, with test continuing until the rate of water level fall becomes constant, or nearly negligible. The point where the infiltration rate stabilizes indicates that the soil is saturated, and this steady rate is the basic infiltration rate. The basic infiltration rate is determined from instantaneous measurements by continuing the infiltration test until the rate of water entry into the soil becomes constant (Brouwer *et al* 1985). The basic infiltration rate was used to assess its relationship with the computed recharge rate.

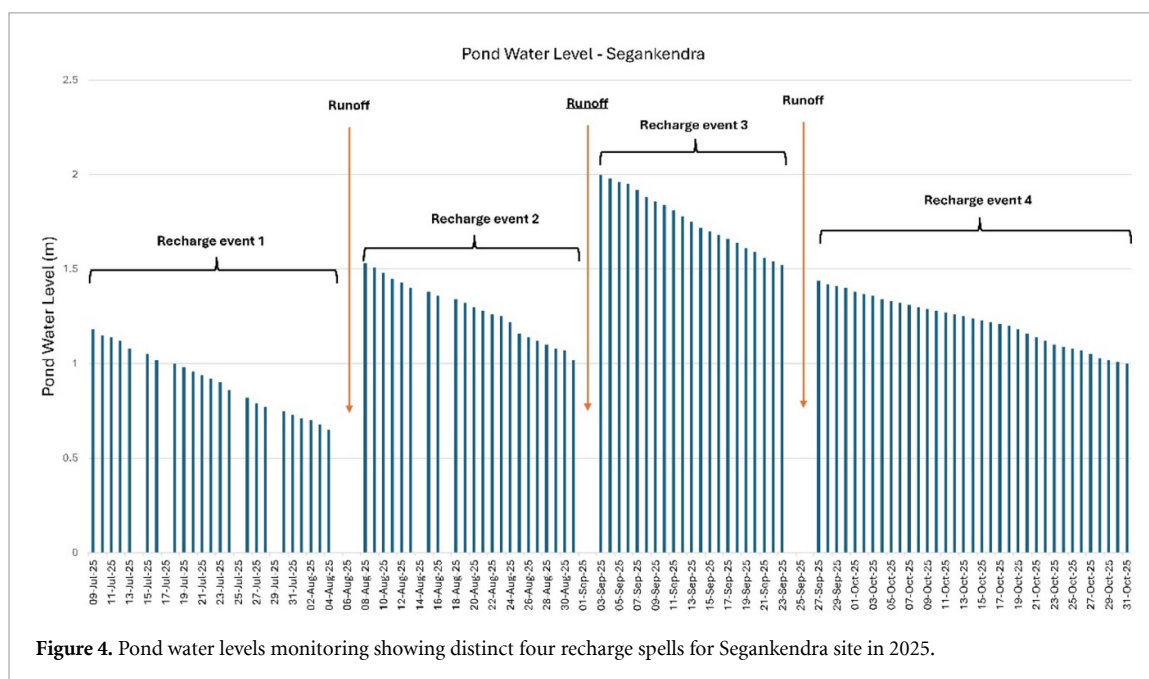
2.2.6. Groundwater monitoring wells near ponds

With groundwater levels governing available subsurface groundwater storage (and hydraulic head) for recharge, groundwater data was obtained from central and state groundwater boards. Data from 72 groundwater monitoring wells (UP-GWD, 2026) were used (figure A1), with the number of wells contributing valid pre-monsoon readings varying by year (65, 68, and 72 wells in 2023, 2024, and 2025, respectively). For each year, a continuous groundwater-level surface was interpolated across the study area using ordinary kriging (Cressie 1990) with a spherical variogram model based on geographic coordinates. The variogram was fitted separately for each year using all wells with a valid reading that year. Interpolation was implemented in Python with PyKriging v1.7.3 (Murphy *et al* 2025). Thereafter, the pre-monsoon level at each pond's exact coordinates was then extracted from that interpolated surface. Kriging weights each well's contribution according to the spatial correlation structure of the data and also provides a standard error at each interpolated point, allowing the uncertainty of each estimate to be assessed directly. Pre-monsoon groundwater level data, covering the period from 2023 to 2025, were analyzed to compare recharge rates with nearby groundwater table conditions and to assess the influence of groundwater level on recharge rate.

Table 1. Summary of recharge spells recorded for different sites.

Site type	2023	2024	2025
Control	14	35	27
Recharge wells	16	26	12
Repurposed	11	54	41 ^a

^a 17 of these recharge spells are reported from sites converted to recharge pits in 2025.

**Figure 4.** Pond water levels monitoring showing distinct four recharge spells for Segankendra site in 2025.

3. Results

3.1. Recharge performance of ponds

In total 236 recharge spells where the pond water level shows a continuous decline with no recorded rainfall and inflow into the pond from any other sources, were recorded (table 1). Figure 4 gives example of one recharge site for year 2025 which shows four distinct recharge spells during which pond water levels decline separated by inflow events. The last recharge spell in this site and in most site other sites is longest after cessation of monsoon rainfall. Table 1 gives the summary of results aggregated across site types (detailed individual site results in table A7).

Results show that mean annual recharge volumes across all monitored ponds declined from 19 425 m³ in 2023 to 11 950 m³ in 2024 and 10 612 m³ in 2025 (table 2). However, given differences in pond size across sites, the recharge ratio (volume of groundwater recharged relative to pond storage capacity) is a better indicator for comparison. The recharge ratio showed a sharp decline between 2023 and 2024, followed by stabilization at a lower level, decreasing from 1.75 in 2023 to 0.77 in 2024 and remaining broadly similar at 0.82 in 2025. Among site types, recharge well sites recorded the highest ratio in 2023 (2.34) but experienced the steepest decline thereafter (0.89 in 2024; 0.46 in 2025). Control ponds showed a less pronounced decline (1.74 in 2023, 1.08 in 2024, and 1.29 in 2025), while repurposed ponds consistently reported the lowest recharge ratios across all three years (0.77, 0.58, and 0.60, respectively). To assess whether this decline reflects a genuine trend rather than a shift in the composition of monitored ponds, a sensitivity analysis was carried out using ponds monitored consistently across years. Among 8 sites monitored in all three years, the recharge ratio declined from 1.80 in 2023 to 0.94 in 2024 and 0.93 in 2025. Similarly, among 13 sites monitored in both 2023 and 2024, the ratio declined from 1.62 to 0.84. Both comparisons reproduce the declining trend observed in the full sample, confirming that the reduction in recharge efficiency is not an artefact of the changing set of monitored ponds across years.

Mean ponded duration across all site types remained broadly stable throughout the monitoring period, averaging 147, 144, and 142 d per season in 2023, 2024, and 2025, respectively. This indicates

Table 2. Mean recharge volume per structure (in m³), recharge and evaporation ratios (recharge/evaporated volume divided by pond volume), ponded days and recharge rates (mm d⁻¹) (*in brackets minimum and maximum*) for overall and different type of ponds for the year 2023, 2024 and 2025. Note: Statistics derived from only those ponds which received water; Recharge period was taken 1st July to 31 Dec.

Year	Type	N	Recharge Volume (m ³)	Recharge ratio	Ponded days	Recharge Rate (mm d ⁻¹)	Evaporation ratio
2023	Overall	12	19 425 [1209–64 722]	1.75 [0.29–4.09]	147 [35–183]	32.8 [5.4–93.9]	0.28 [0.04–0.86]
2024	Overall	21	11 950 [133–36 968]	0.77 [0.01–2.34]	144 [14–183]	18.3 [6.8–63.5]	0.28 [0.01–0.77]
2025	Overall	16	10 612 [1568–27 149]	0.82 [0.09–2.44]	142 [37–183]	21.6 [5.2–55.5]	0.27 [0.04–0.57]
2023	Control	4	22 405 [4240–47 130]	1.74 [0.29–2.68]	148 [123–183]	27.6 [9.9–49.0]	0.19 [0.08–0.27]
2024	Control	5	12 695 [2651–25 449]	1.08 [0.15–2.34]	135 [84–178]	23.5 [10.2–45.5]	0.21 [0.07–0.34]
2025	Control	4	10 038 [1568–25 511]	1.29 [0.09–2.44]	121 [48–183]	26.7 [11.2–45.9]	0.21 [0.04–0.35]
2023	Recharge wells	5	26 540 [10 594–64 722]	2.34 [0.73–4.09]	177 [155–183]	21.4 [13.7–28.9]	0.43 [0.15–0.86]
2024	Recharge wells	5	12 072 [2818–33 346]	0.89 [0.15–1.29]	170 [134–183]	12.8 [8.0–23.9]	0.41 [0.14–0.53]
2025	Recharge wells	3	6841 [3412–11 384]	0.46 [0.18–0.79]	164 [127–183]	14.3 [5.2–25.4]	0.28 [0.13–0.43]
2023	Repurposed	3	9666 [4473–13 758]	0.77 [0.36–1.45]	94 [35–183]	58.6 [5.4–93.9]	0.14 [0.04–0.32]
2024	Repurposed	11	11 555 [133–36 968]	0.58 [0.01–1.15]	136 [14–183]	18.5 [6.8–63.5]	0.26 [0.01–0.77]
2025	Repurposed	5	9784 [3929–15 979]	0.60 [0.29–1.21]	145 [49–183]	20.3 [8.7–55.5]	0.27 [0.09–0.57]

that water availability was generally adequate and did not change substantially between years and also suggests that the observed decline in recharge volume and efficiency is not likely due to the reduced water availability. Recharge well sites recorded the longest ponded durations in all three years (177, 170, and 164 d), consistent with better-managed inlets and outlets at these sites. Control ponds showed more variability (148, 135, and 121 d across the three years), while repurposed ponds showed an apparent increase in ponded duration from 94 d in 2023–136 and 145 d in 2024 and 2025, though the low 2023 reflects the small number of monitored sites ($n = 3$) and high within-group variability (range: 35–183 d) in that year. Lower rainfall in 2024 and 2025 (999 mm and 944 mm, compared to 1424 mm in 2023) also contributed to modest reductions in ponded days at some sites, particularly control ponds, but did not translate into a large reduction in water availability across the monitoring ponds. Despite the overall adequate water availability, four out of 23 monitored ponds faced persistent constraints. Two ponds remained completely dry, one received water only through pumped groundwater, and another retained water for less than a month, with depths remaining below 0.2 m. This highlights that poor siting or inadequate inlet design can significantly limit pond performance and reduce recharge.

Recharge rate (mm d⁻¹) is identified as the key factor limiting overall recharge volumes, given that ponding durations remained relatively stable (averaging ~140–147 d per season, section 3.1) and largely unchanged. Overall mean recharge rates declined from 32.8 mm d⁻¹ in 2023 to 18.3 mm d⁻¹ in 2024 and 21.6 mm d⁻¹ in 2025. The decline was most pronounced in repurposed ponds, where mean recharge rates fell from 58.6 mm d⁻¹ in 2023 to approximately 18.5–20.3 mm d⁻¹ in subsequent years. Recharge well sites similarly declined from 21.4 mm d⁻¹ in 2023 to 12.8 mm d⁻¹ in 2024 and 14.3 mm d⁻¹ in 2025. In contrast, control ponds maintained the most consistent recharge rates across the three years (27.6, 23.5, and 26.7 mm d⁻¹). There is large intra-site variability in recharge rates across all types and years, as reflected in the minimum–maximum ranges in table 2. The progressive decline in recharge rates across most site types, even as ponding durations remained stable, points to gradual clogging of the pond bed due to siltation as the likely mechanisms driving declining recharge efficiency over time.

Evaporation losses were consistently substantial across all site types and years, with the overall evaporation ratio (the volume of evaporation relative to pond storage capacity) remaining stable at

Table 3. Measured basin infiltration rates (before 2024 recharge season) along with monitored recharge rates year 2024.

Site name	Type	Basic infiltration (mm d ⁻¹)	Average recharge rate 2024 (mm d ⁻¹)
Jamalpur	Control	56.2	27.8
Nagala Gujar	Control	43.9	45.5
Sadakpur Khichide	Control	19.4	16.9
Sanai Roza (Control)	Control	32.4	17.0
Shukla	Control	52.8	NA
Daniya Sankarpur	Recharge wells	21.6	23.9
Haryal	Recharge wells	31.7	8.5
Segankendra	Recharge wells	81.6	13.4
Tumdiya	Recharge wells	55.4	8.0
Chak Madhpuri	Repurposed	42.5	63.5
Changeri	Repurposed	11.5	6.8
Jargao	Repurposed	96.5	33.4
Sanai Roza (amrit sarovoor)	Repurposed	20.2	10.5

approximately 0.27–0.28. Recharge well sites recorded the highest evaporation ratios (0.43 in 2023 and 0.41 in 2024, declining to 0.28 in 2025), consistent with their longer ponding durations, which extend the period of open-water exposure. Control ponds consistently showed the lowest evaporation ratios (0.19–0.21), while repurposed ponds' evaporation ratio increased from 0.14 in 2023 to 0.26–0.27 in later years, broadly reflected decrease in recharge rates and increase in ponded duration.

3.2. Factors influencing recharge rates

3.2.1. Basic infiltration rates

To assess whether spatial variation in recharge rates is primarily driven by differences in soil infiltration capacity, basic infiltration tests were conducted in 13 selected ponds in 2024, following the 2023 recharge season. There was large variation in measured basic infiltration with rates ranging from 11.5 to 96.5 mm d⁻¹, with an overall mean of 43.5 mm d⁻¹ (table 3). However, mean rates were broadly similar across site types—40.9 mm d⁻¹ for control ponds, 47.6 mm d⁻¹ for recharge well sites, and 42.7 mm d⁻¹ for repurposed ponds, suggesting that inherent soil infiltration capacity alone does not explain the differences in recharge performance observed among site types, though it has an important role.

The overall correlation between basic infiltration rate and measured recharge rate is not statistically significant [Pearson $r = 0.244$, $R^2 = 0.06$, $p = 0.444$], basic infiltration rate explains only 6% of the variance in measured rates, reflecting a poor predictive relationship compounded by notable outliers (Chak Madhpuri and Nagala Gujar) in this small dataset ($N = 12$). A comparison of measured infiltration rates with mean recharge rates recorded during 2024 recharge season shows that at several sites (Segankendra, Tumidday, Haryal), actual recharge rates were substantially lower than the measured infiltration capacity. This is consistent with sedimentation observed in the ponds (section 3.2.2) as the infiltration tests were conducted after removing the surface silt layer. For example, Segankendra site measured infiltration rate of 81.6 mm d⁻¹ was much higher than the recharge rate of 13.4 mm d⁻¹. In contrast, at Chak Madhpuri (site which received the water for least number of days: 9 in 2023, 35 in 2024) the observed recharge rate (63.5 mm d⁻¹) exceeded the measured infiltration capacity (42.5 mm d⁻¹). This could be due to site-specific factors (with sampled locations not representing the site) or likely error in monitoring given the limited days water was received (with shallow water depth) with recharge spells only lasting 3–4 d. Some sites showed very low infiltration rates: Changeri (11.5 mm d⁻¹), Sana Roza repurposed (20.2 mm d⁻¹), and Daniya Sankarpura (21.6 mm d⁻¹). For these sites, observed recharge rates indicates that soil texture represents a inherent limitation on recharge potential, independently of surface sealing.

3.2.2. Siltation of sites

Impact of silting in the rejuvenated ponds, not control ponds as they were not desilted, were assessed through measurement of clogging layer depth and turbidity levels assessment in selected ponds (table 4). Water quality tests in 7 ponds showed pond-water turbidity ranging from 2–63 NTU. This was in the acceptable range for recharging water not reflecting high sedimentation. However, these turbidity measurements were not tied to specific rainfall or recharge spells, and were collected from only a subset (7 of 23) of monitored ponds, limiting their ability to characterize sediment loading during individual recharge spells. Further, these measurements likely underestimate actual turbidity during runoff events.



Figure 5. Deposited sediment in pond surface in Imratpur Seondara site (a), (b) and grassy vegetation on recharge wells in site (c) Tumdiya and (d) Daniya Sankarpur.

Table 4. Measured clogging layer depth (end of 2024 recharge season) along with monitored turbidity of water in the ponds.

S No.	Name of site	Types of site	Year of construction	Number of grids sampled	Depth of clogging layer	Turbidity (NTU)
1	Sadakpur Khichide	Control	NA	4 grid	2 cm	7.5
2	Jamalpur	Control	NA	4 grid	1 cm	—
3	Sanai Roza (Control)	Control	NA	6 grid	3 cm	6.7–13.3
4	Nagala Gujar	Control	NA	4 grid	3 cm	—
5	Bichola Dhooki	Control	NA	4 grid	2 cm	—
6	Tumdiya	Recharge wells	2020	5 grid	2 cm	—
7	Daniya Sankarpur	Recharge wells	2020	3 grid	2 cm	2.4–7.3
8	Chak Madhpuri	Repurposed	2021	4 grid	1 cm	—
9	Sanai Roza (amrit sarovoor)	Repurposed	2022	4 grid	0	—
10	Jargao	Repurposed	2022	4 grid	3 cm	63.4
11	Changeri	Repurposed	2022	4 grid	3 cm	5.3–23.8
12	Sonakpur_Nari Kund	Repurposed	2021	4 grid	3 cm	4.3
13	Imratpur Seondara	Repurposed	2022	4 grid	2 cm	—
14	Sonakpur_Jogan	Repurposed	2022	4 grid	2.5 cm	—
15	Chhiraoli	Repurposed	2021	4 grid	2 cm	6.8
16	Mahendari Sikandari	Repurposed	2022	4 grid	1 cm	—

This is because samples were collected from the upper pond water layer and several days after rainfall, by which time most suspended silt would have settled. This is reflected from the field observations across 16 sites (at the end of 2024 recharge season) confirming significant sediment and clay deposition at the pond base (figure 5, table 4). An average deposited silt layer of about 2 cm (0–3 cm) was estimated, though the thickness varied across ponds. Though, this measurement was taken only one time at the end of the 2024 season, so it captures cumulative sediment accumulation rather than the rate of deposition, and cannot itself be statistically linked to the observed decline in recharge rates.

The impact of silt deposition on recharge is reflected in table 5 which showed construction/rejuvenation year of the site along with the recharge rates for the monitoring period for recharge wells, repurposed and recharge pit sites. Data shows that all monitored sites were constructed or rejuvenated between 2020 and 2022, and the monitoring period covered one to five years after their last maintenance. Recharge rates declined steadily with years since rejuvenation, from ‘▲’ (higher rates) to ‘▼’ (lower rates), consistent with the gradual reduction in overall recharge efficiency discussed above

Table 5. Construction year and recharge rate (mm d^{-1}) of recharge-well, repurposed, and recharge-pit ponds across the 2023–2025 monitoring seasons. Recharge-pit sites ($n = 4$) are shown at the bottom; these were repurposed ponds in 2023 and 2024, with recharge pits constructed ahead of the 2025 monsoon season. Symbol show each site's recharge-rate ranking relative to its own monitored years (assigned manually per site/row): $\blacktriangle$ = highest recharge rate recorded for that site; $\bullet$ = intermediate; $\blacktriangledown$ = lowest. Where only two years of data are available, only the high ($\blacktriangle$) and low ($\blacktriangledown$) categories apply. Where two years show an identical rate, both are marked “=” (grey) to indicate no change between years.

Site	Type	Construction Year	Recharge rate (mm d^{-1})		
			2023	2024	2025
Daniya Sankarpur ^a	Recharge wells	2020	28.9 $\blacktriangle$	23.9 $\blacktriangledown$	25.4 $\bullet$
Segankedra	Recharge wells	2020	13.7 $\blacktriangle$	13.4 $\bullet$	12.3 $\blacktriangledown$
Haryal	Recharge wells	2021	23.2 $\blacktriangle$	8.5 $\blacktriangledown$	—
Ramnagar Gangpur	Repurposed	2021	—	21.4 $\blacktriangle$	10.8 $\blacktriangledown$
Signi	Recharge wells	2022	23.4 $\blacktriangle$	10.2 $\blacktriangledown$	—
Tumdiya	Recharge wells	2020	18.0 $\blacktriangle$	8.0 $\bullet$	5.2 $\blacktriangledown$
Imratpur Seondara	Repurposed	2022	—	7.3 $\blacktriangledown$	8.7 $\blacktriangle$
Mangoopura	Repurposed	2022	—	11.5 $\blacktriangledown$	13.5 $\blacktriangle$
Sana Roza (Amrit sarovoor)	Repurposed	2022	76.6 $\blacktriangle$	10.5 $\blacktriangledown$	—
Sonakpur_Jogan	Repurposed	2022	—	8.5 $\blacktriangledown$	12.8 $\blacktriangle$
Tewar Khas ^b	Repurposed	2022	—	15.1 $\blacktriangledown$	55.5 $\blacktriangle$
Changeri ^c	Recharge pits	2022	5.4 $\blacktriangledown$	6.8 $\bullet$	10.4 $\blacktriangle$
Chhiraoli ^c	Recharge pits	2021	—	12.0 $\blacktriangledown$	18.4 $\blacktriangle$
Jargao ^c	Recharge pits	2022	93.9 $\blacktriangle$	33.4 $\blacktriangledown$	51.6 $\bullet$
Sonakpur Nari Kund ^{c,d}	Recharge pits	2021	—	13.6 =	13.6 =

^a The filters in the recharge wells were cleaned before recharge season of 2025.

^b Pond was desilted before recharge season of 2025.

^c Recharge pits sites were repurposed sites for year 2023 and 2024 with recharge pits constructed for the 2025 recharge season. Recharge pit area as percentage of pond area is of 0.06% (Changeri), 0.23% (Chhiraoli), 0.16% (Jargon) and 0.26% (Sonakpur Nari kund).

(table 1). Results presented here are correlational rather than a demonstrated causal attribution of sedimentation, as the available data are limited, but it is consistent with a sedimentation-related decline in recharge efficiency.

Among sites constructed in 2022, where 2023 was therefore the first season after construction, there was a sharp decline in recharge rates between the first and second year of monitoring. For example, Jargao, and Sana Roza (Amrit sarovoor) recorded high initial rates (93.9 and 76.6 mm d^{-1} respectively in 2023), which reduced to 33.4 and 10.5 mm d^{-1} by 2024, a decline of more than 50% within a single season. This suggests rapid early clogging of the pond bed, within one year, likely driven by the deposition of fine sediment during initial inundation events, with infiltration capacity stabilizing at a lower level in subsequent years as surface sealing progresses.

Recharge well sites generally exhibited more stable trajectories, though a consistent downward trend was also evident. Sites constructed in 2020 (Daniya Sankarpura and Segankedra) showed the smallest interannual declines over the monitoring period, likely indicating that after two seasons in use any decline in performance had already stabilized.

For the monitoring period, in only 2 sites maintenance activity was observed. For example, at Daniya Sankarpur, cleaning of recharge wells (removing and cleaning filter material with water) before 2025 recharge season increased recharge from 23.9 to 25.4 mm d^{-1} and at Tewar Khas, full desilting (thorough scraping of pond bottom) of the pond before 2025 recharge season more than tripled recharge rates from 15 to 55 mm d^{-1} (reducing evaporation ratio from 0.27 to 0.09). This suggests that maintenance can arrest or partially reverse the decline in infiltration capacity.

The four sites where recharge pits were constructed prior to the 2025 season (converting previously repurposed ponds) provide a further impact of the targeted structural intervention on recharge rates. All four sites recorded higher recharge rates in 2025 than in 2024: Changeri increased from 6.8 to 10.4 mm d^{-1} , Chhiraoli from 12.0 to 18.4 mm d^{-1} , Jargao from 33.4 to 51.6 mm d^{-1} , and Sonakpur Nari Kund remained stable at 13.6 mm d^{-1} . The magnitude of improvement appeared broadly proportional to the size of the recharge pit relative to pond area, with Chhiraoli (0.23% pit area) and Jargao (0.16%) showing the largest absolute gains, while Changeri (0.06%) showed the most modest improvement. Changeri is also where the basic infiltration rates were the lowest reflecting limitations of underlying soil and need for higher intensity of pits. The results suggest that even small recharge pits can enhance infiltration by bypassing the sealed pond bed and connecting more directly to the unsaturated zone below.

Table 6. Recharge rate (mean, mm d^{-1}) for different site types at different times of the season—Early (June–July), Mid (August–September), and End (October–December) and pre-monsoon groundwater levels (mean, m below ground level), estimated via ordinary kriging interpolation (figure A1). Average kriging standard error added in brackets.

	Pre monsoon GWL (mbgl) [Kriging std. error]	Early	Mid	End
Control	15.0 [± 2.90]	36.5	24.6	18.2
Repurposed	14.7 [± 2.87]	29.0	22.2	10.5
Recharge wells	4.7 [± 2.76]	25.1	18.9	11.1

3.2.3. Seasonal effects on recharge

Table 6 shows changes in recharge rates during the seasons—from early (Jun–July) to mid (Aug–Sep) to late (Oct–Dec) period. Results show a consistent decreasing trend as the season progresses, a pattern observed in all years. This suggests that early-season recharge spells contribute more to total recharge, whereas later in the season, recharge is less and evaporation losses are higher. The decline in recharge rates during the monsoon season results from two key factors: (i) shallow groundwater levels that reduce the hydraulic gradient, especially pronounced in areas where groundwater tables are shallow, and (ii) progressive siltation of recharge structures, which accumulates over time and becomes more evident across years (as noted earlier in table 5).

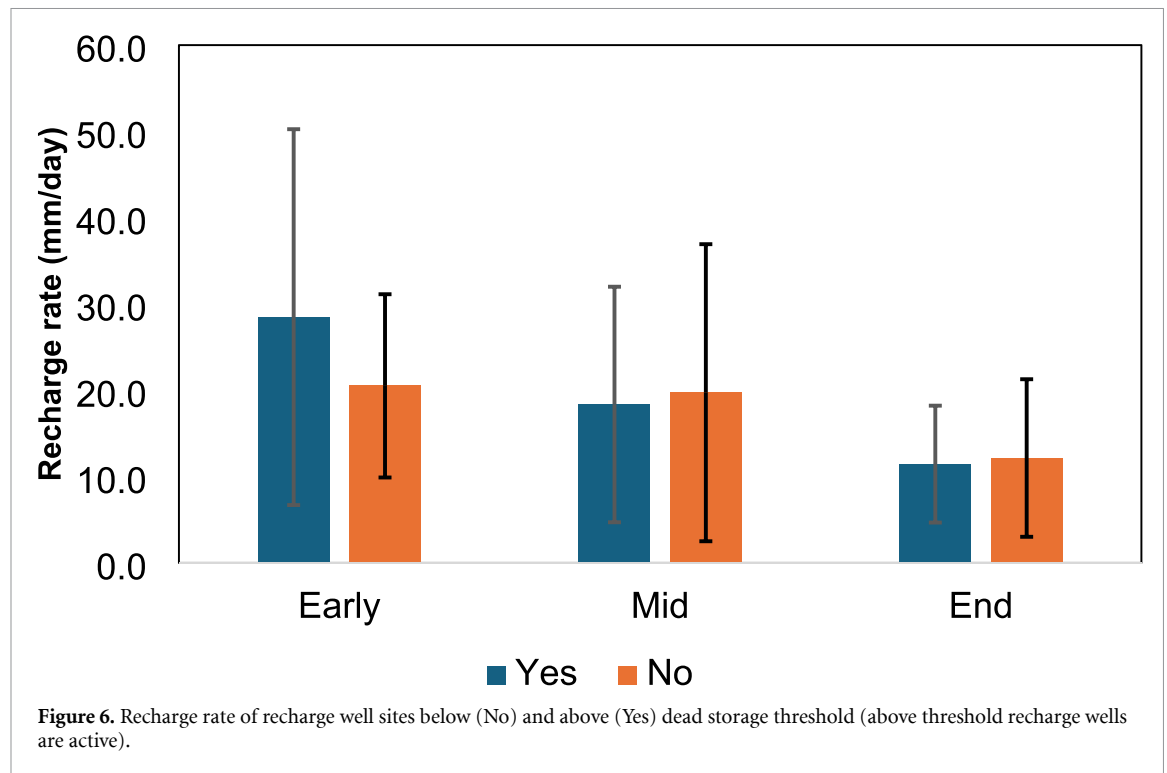
Although continuous groundwater level monitoring data was unavailable due to the absence of piezometers near most sites, groundwater levels at each pond location were estimated using ordinary kriging interpolation of pre-monsoon readings from nearby state and central monitoring wells (figure A1). This interpolation carries some uncertainty, particularly for ponds farther from any monitoring well (distance to nearest well and wells per buffer are reported in table A8) with kriging standard errors across all site-years ranged from 2.1–3.4 m, with the majority of estimates (72%) below 3 m. Pre-monsoon groundwater levels were taken which provide an indication of available storage at the start of the season, with groundwater levels increasing in the monsoon season from this base. For control and repurposed sites, groundwater levels were generally deeper (15.0 and 14.7 mbgl, respectively), whereas for recharge-well sites—mostly located in Rampur—groundwater levels were markedly shallower (4.7 mbgl) (table 6). The results suggests that recharge-well sites face greater limitations from reduced hydraulic head and available storage as the season progresses. These conditions, coupled with limited maintenance, explain why recharge rates from recharge-well sites were low relative to control sites among all site types (table 3).

3.2.4. Recharge well design and dead storage in pond

Another factor influencing the performance of recharge well sites is the designed dead storage. These sites were intentionally constructed so that the recharge well becomes active only after the pond water reaches a certain threshold level (typically 1–1.5 m above the pond bottom) (figure 3), thereby ensuring that minimum pond water levels remain available for other uses (e.g. livestock, recreational value). Consequently, during parts of the recharge season when water levels remained below this threshold, recharge through wells was absent and was limited to the base and sides of the pond.

Results show that when recharge spells are disaggregated based on whether the average water level was above or below this threshold (figure 6), early in the season, mean recharge rate was numerically higher when pond water exceeded the threshold (28.5 mm d^{-1} , $n = 11$) than when it remained below (20.5 mm d^{-1} , $n = 8$), though this difference was not statistically significant (Welch's t -test, $t = 1.05$, $p = 0.31$). This apparent gap narrowed later in the season (mid-season: 18.4 mm d^{-1} , $n = 13$, vs 19.7 mm d^{-1} , $n = 10$, $p = 0.84$; end of season: 11.4 mm d^{-1} , $n = 6$, vs 11.7 mm d^{-1} , $n = 8$, $p = 0.95$), though neither of these differences was significant either. Given that each season/threshold group contains only 6–13 recharge spells, these tests are likely underpowered to detect anything but a large effect, and the results should therefore be interpreted as a suggestive early-season pattern rather than a confirmed threshold effect.

One possible explanation for this pattern is that early in the season, when groundwater levels are deeper, the higher hydraulic gradient supports greater recharge once the threshold is exceeded. Later in the season, as groundwater levels rise and available (desaturated) aquifer storage becomes limited, the effectiveness of recharge wells declines even when water levels in the pond remain high. The results suggests that recharge wells can enhance recharge under suitable conditions, but that their effectiveness is



highly site- and time-dependent, and that many recharge-well sites in this study may not have been optimally located, as shallow groundwater conditions later in the season would have limited their contribution to recharge. Further, data is needed to probe this in detail which will require paired groundwater-level and recharge-rate monitoring at each recharge-well site, which was beyond the scope of the current dataset.

4. Discussion

Recharge performance, characterized by quantifying the potential recharge as measure of infiltration water flux from base of ponds, across the 23 monitored sites over the three-year study period varied widely, with recharge-to-storage ratios remaining below 2 throughout and declining markedly from approximately 1.75 in 2023 to around 0.8 by 2024–2025. Surface water availability was generally adequate, averaging around 140 ponded days per year, yet infiltration rates remained low and declining, mostly below 40 mm d^{-1} , particularly at recharge well and repurposed pond sites. Evaporation losses accounted for approximately 27%–28% of pond storage. Results show that recharge rates declined most sharply over time at sites that had been recently rejuvenated, whereas control ponds that had not undergone recent intervention showed comparatively stable, with low rates over time.

The infiltration rates observed in this study, most falling in the $15\text{--}40 \text{ mm d}^{-1}$ range are broadly consistent with those reported for percolation tanks and check dams across hard rock aquifer settings, although limited results are reported from alluvial aquifers in India (table 7). Dashora *et al* (2018) reported $27\text{--}42 \text{ mm d}^{-1}$ for check dams in Rajasthan; Surinaidu *et al* (2016) and Gale *et al* (2006) reported $12\text{--}35 \text{ mm d}^{-1}$ for percolation tanks in Tamil Nadu; and Ali *et al* (2012) report a mean of 24 mm d^{-1} for a percolation pond in Rajasthan. Recharge ratios in this study (0.46–1.74 for control and repurposed ponds) similarly fall within ranges reported by Central Ground Water Board (CGWB) (2011) for Maharashtra (0.11–0.48), Biswas *et al* (2017) for Bundelkhand (0.72–0.78), and NIH (1999) for Tamil Nadu (0.65–1.1). The convergence of rates across structurally different geological settings (hard rock in most of the referenced studies; alluvial in this study), likely indicates that pond bed clogging and sediment accumulation govern recharge largely independent of the underlying aquifer type. The results reflect this as rejuvenated sites showing the steepest decline in recharge rates (table 5) and similar results were reported from other studies (Gale *et al* 2006, Ali *et al* 2012). This reflects the high intensity of monsoon rainfall season amplified by the predominantly agricultural catchments leading to high runoff and sediment loads (Kim *et al* 2018a; Mishra *et al* 2007) draining into monitored ponds that accumulate in infiltration zones.

Table 7. Summary of MAR Performance Across selected Indian and global MAR sites. Rows in italic indicate global MAR studies outside India.

Study	Area Aquifer type	Structure Type (<i>n</i>) Monitored Years	Performance indicator ^a	
			Infiltration Rate (mm d ⁻¹)	Ratio of recharge to volume
Alam <i>et al</i> (2020)	Uttar Pradesh, India Alluvial	Pond with 10 recharge wells (1) 3	164–295	5.0–11.8
Dashora <i>et al</i> (2018)	Rajasthan, India Granitic gneiss Hard rock	Check dams (4) 2	27–42	1–2
Massuel <i>et al</i> (2014)	Andhra Pradesh, India Hard rock	Percolation tank (1) 3	11.3	
Surinaidu <i>et al</i> (2016)	Tamil Nadu, India Hard rock	Percolation tank (1) 3	28–35	
Gale <i>et al</i> (2006)	Tamil Nadu, India Hard rock	Tank (1) 2	12–30	0.5–1
Gale <i>et al</i> (2006)	Gujarat India Alluvial + hard rock	Check dam (1) 2	78–225	~2
Gale <i>et al</i> (2006)	Gujarat India Alluvial + hard rock	Percolation pond (1) 2	4–13	0.2
Glendenning and Vervoort (2010)	Rajasthan, India Granitic gneiss Hard rock	Percolation pond, check dams (6) 2	12–55 (27.5)	
Central Ground Water Board (CGWB) (2011)	Maharashtra, India (Jalgaon) Hard rock	Percolation pond(5) 1	—	0.11–0.48 (0.33)
Biswas <i>et al</i> (2017)	Bundelkhand, India Hard rock	Percolation pond (2) 6		0.72–0.78
NIH (1999)	Tamil Nadu, India Hard rock	Percolation pond (2) 2	5.5–55.57 5–23,3	0.65–1.1
Ali <i>et al</i> (2012)	Rajasthan, India Hard rock	Percolation pond (1) 3	19–51 (24)	
Barry <i>et al</i> (2017)	Alice Springs, Australia Alluvial	Infiltration Basins/Soil Aquifer Treatment (SAT) 6	100–1000	
Grischek <i>et al</i> (2021)	Dresden-Hosterwitz, Germany Alluvial	Infiltration Basins (part of riverbank filtration scheme) >40	2000–7000	
Lluria (2009)	Tonopah, Arizona, USA	Spreading Basins >20	600–2000	
Department of Water Affairs (2010)	Atlantis, South Africa Sedimentary coastal aquifer overlying hard-rock basement	Infiltration Ponds 30	10–160	
Racz <i>et al</i> (2012)	Pajaro valley, California Alluvial aquifer	3-ha MAR infiltration pond 1	100–1000	
Elkayam <i>et al</i> (2024)	Israel Alluvial	Infiltration/ Soil Aquifer Treatment (SAT) 50	240–2400	
Mawer <i>et al</i> (2016)	Northern Colorado, USA Alluvial	Infiltration ponds/ Riverbank filtration 1	1700	
Samanta <i>et al</i> (2020)	Belgium Alluvial	Infiltration ponds 1	200–400	
Ganot <i>et al</i> (2017)	Israeli Coastal Aquifer	Infiltration ponds 1	360–840	

^a values in parentheses denote the reported mean for the study.

The recharge rates from this study are an order of magnitude less than reported by Alam *et al* (2020) for a pilot site within the study area. Alam *et al* (2020) reported recharge rates of 164–295 mm d⁻¹ and recharge ratios of 5.0–11.8 for the case of a single pond augmented with ten recharge wells. In contrast, the recharge well ponds sites monitored in this study were equipped with only 2–4 wells and subject to minimal maintenance, achieved recharge ratios of 0.46–2.34 and rates of 12–25 mm d⁻¹, an order of magnitude lower. However, it is to be noted that Alam *et al* (2020) results were from one pilot site with intensive supervision and maintenance during the operation.

Comparison with intensively and well-managed MAR systems in other global contexts further highlight the suboptimal performance of ponds in this study. Managed infiltration ponds in California's Pajaro Valley reported whole-pond infiltration rates exceeding 1000 mm d⁻¹ during initial operations, sustained for several weeks before declining as clogging developed (Racz *et al* 2012). Similarly long term recharge rates of 100–1000 mm d⁻¹ were reported for an infiltration basin used for SAT in Alice Springs Australia (Barry *et al* 2017); spreading basins in Tonopah, Arizona, USA reported 600–2000 mm d⁻¹ (Lluria 2009); infiltration basins used for SAT in Israel achieved rates of 240–2400 mm d⁻¹ (Elkayam and Lev, 2024); MAR basin in Colorado recording mean rates approaching 1700 mm d⁻¹ (Mawer *et al* 2016), and a Belgian infiltration site operating reported rates of 200–400 mm d⁻¹ (Samanta *et al* 2020).

Direct comparison of these sites with small-scale ponds cannot be done given these sites reflect more closely managed systems operating at much larger scales (in terms of water recharged per site/system) with better technical design and siting, dedicated resources, and controlled source water. Also in many,

but not all cases, MAR is used to augment groundwater used for non-agricultural purposes specially for domestic use, which instils a need for more careful management on the quality part. However, the large contrast in recharge performance does suggest there is potential to increase recharge rates from largely unmanaged ponds in this study through better design and maintenance.

First is the importance of controlling sedimentation through maintaining source water quality. Many of the global case MAR sites with high performance (table 7) received treated wastewater with controlled turbidity, or actively managed siltation through desilting traps and regular de-siltation (Ganot *et al* 2017; Barry *et al* 2017; Elkayam and Lev 2024; Samanta *et al* 2020, Alam *et al* 2020; Mawer *et al* 2016). This is in contrast with studied pond sites with no control measures for intake water quality. Thus, the gap between realized recharge rate (table 2) and measured basic infiltration rate (table 3) visible in poor correlation among them, could be narrowed through better sediment management. This was reflected in one monitored pond where desilting tripled its recharge rate from 15 to 55 mm d⁻¹ and thus reducing evaporation ratio from 0.27 to 0.09 (tables 2, table A6). Given the large number and distributed nature of small ponds under various government programs, broader catchment-level measures (contour bunding, field bunds, vegetative barriers, and silt traps) are needed to reduce sediment mobilization at source (Pavelic *et al* 2026, Adimassu *et al* 2014, Sharma *et al* 2024 ref). This is to be complemented with more frequent desilting of pond beds. For example, NREGS program guidelines prescribed desilting intervals of 5–10 years (MoRD 2024) which is inadequate for ponds intended for groundwater recharge and results suggest desilting frequency of 2–3 years for maintaining performance in this region and in other regions should be adapted to locally observed rate of sedimentation. This should be mandated for recharge ponds with dedicated financial provisioning through local government budgets.

The maintenance aspect is critical for recharge well sites that need frequent cleaning of filters and wells (Bouwer 2002; Kim *et al* 2018b, Shahbaz Akhtar *et al* 2021). Thus, it may be prudent to concentrate recharge wells in cluster configurations, rather than dispersing them thinly across a landscape, akin to better ‘managed’ systems with dedicated resources and team. For example, Alam *et al* (2020) pilot results from the same district showed that recharge well approach holds significant unrealized potential but need active management. Also, in the recharge well sites that come at relatively higher investment, dead-storage thresholds (currently maintained at 1–1.5 m) should be reduced for structures allowing recharge to continue through a larger portion of the monsoon. This necessitates earmarking such structures exclusively for recharge rather than multiple water use systems. For ponds with multiple water-use needs (e.g. fisheries, livestock watering) that require dead storage, and where local capacity for maintenance is limited, recharge pits may offer a more compatible alternative. Shallow depth recharge pits also provide natural filtration through soil media, are easier to construct and simpler to maintain (UPGWD, Central Ground Water Board (CGWB) 2007). Results show that they can enhance recharge as also reflected in the four sites where recharge pits were installed (table 4). Increasing pit density could enhance their contribution meaningfully without impacting other pond functions.

Results also showed the importance of design and siting. The monitored recharge-well sites were placed in areas with shallow pre-monsoon groundwater, which limited available hydraulic head and storage capacity during the recharge season and resulted in the very low recharge rates observed. Alam *et al* (2020) reported a similar pattern, with recharge rates declining sharply later in the monsoon as aquifer storage was exhausted. Similarly, Jalan *et al* (2025) demonstrated explicitly how early-season aquifer saturation can render recharge interventions ineffective at a watershed scale in basaltic Maharashtra. Additionally, four of the ponds did not receive sufficient inflow to sustain ponding beyond evapotranspiration losses, with two remaining dry throughout the season (section 3.1). These results underscore the need for pre-construction hydrogeological and catchment-level assessments including estimation of runoff potential, measurement of pre-monsoon water table depth, subsurface permeability profiles and basic infiltration rates as a prerequisite for siting recharge well structures. These parameters are generally considered in desktop-based suitability analysis (Alam and Pavelic 2020, Mizan *et al* 2026) which may guide broader targeting but should be complemented with site-specific measurements (e.g. pre-monsoon and post-monsoon water-table depth at the pond location, basic infiltration testing, local soil/lithology logging, and distance/variability of nearby monitoring wells).

Lastly, participatory monitoring and performance-based evaluation should be encouraged. The three-year monitoring timeframe of this study illustrates the feasibility of community-based performance evaluation. The same has been reported in several studies on participatory monitoring of water resources, where community members have been successfully engaged as data collectors to assess the condition and performance of local water infrastructure, demonstrating that such approaches can generate reliable, longitudinal datasets while simultaneously building local ownership and stewardship capacity (Maheshwari *et al* 2014, D’Souza *et al* 2019). Simple measurements of pond water levels, combined with periodic

sediment depth assessments, are sufficient to characterize infiltration dynamics and can identify clogging trajectories, and detect the effects of maintenance interventions. Standardizing such protocols and embedding them within existing local governance structures (panchayats—village-level local government councils, water user groups, or designated water champions) would provide a low-cost mechanism for adaptive management across India's large and heterogeneous recharge endeavour. This warrants shifting evaluation frameworks across department and agencies implementing MAR from asset-creation metrics toward outcome-based indicators to better align public investment with long-term groundwater sustainability.

5. Conclusion

This study, based on three years of participatory monitoring of 23 recharge structures, shows that small-scale MAR structures in the Ramganga basin were useful in recharging groundwater. However, monitored sites consistently operated below their recharge potential with the results indicating that overall recharge was suboptimal, with recharge-to-storage ratios declining from 1.75 in 2023 to approximately 0.8 by 2025, a decline associated primarily with low infiltration rates linked to progressive siltation and inadequate maintenance. Infiltration rates, mostly below 40 mm d^{-1} , are broadly consistent with other Indian MAR studies however remain an order of magnitude below well-managed global systems. Sedimentation in the pondage area of MAR structures appears to be an important constraint, as suggested by one site where desilting tripled infiltration rates. This points to the need for more frequent desilting, upstream catchment management to reduce sediment loads, and dedicated provisioning for periodic maintenance of these small MAR structures. Recharge well sites, though expected to perform better, showed the lowest recharge rates; this may partly reflect lack of maintenance as well as siting in areas with shallow groundwater levels, where recharge space is limited consistent with the available groundwater data showing shallow water tables at several sites. The study recommends that recharge wells be implemented in actively managed cluster configurations, with prior hydrogeological assessments to avoid placement in areas with limited hydraulic head or storage capacity. In ponds with multipurpose use, recharge pits may offer a more compatible alternative. The three-year participatory monitoring demonstrated that community-based water level monitoring is both feasible and effective in generating reliable performance data. Finally, realizing India's ambition of building 14 million recharge structures to enhance groundwater recharge and build resilience will require, in addition to important asset-creation, a shift toward outcome-based performance evaluation, context-specific design, and regular maintenance embedded within local governance structures.

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Data availability statement

Restrictions apply. Data cannot be shared publicly via a URL/DOI. Part of ongoing research and work is with partners whose consent is needed before making it public.

Conflict of 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.

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