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Impacts of Irrigation with Reclaimed Water on Soil and Groundwater: Vegetation- and Season-Dependent Patterns from a Lysimeter Study

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Abstract The use of reclaimed water in agriculture is a potential solution to regional water shortages. A controlled lysimeter study investigated the impact of irrigation with reclaimed water on the soil's water and nutrient balance, and the potential impact on drainage water quality for groundwater recharge. Planted and unplanted lysimeters with defined soil profiles were irrigated with reclaimed or tap water for 12 months (August 2024 to August 2025). The reclaimed water was obtained from a membrane bioreactor. Water balance (precipitation, irrigation and potential evapotranspiration) and drainage water formation were recorded and sampled weekly. The analyses included physico-chemical parameters (pH, electrical conductivity, total bound nitrogen, nitrate, nitrite, and ammonium nitrogen), anions (chloride, fluoride, sulfate), and 21 organic micropollutants. In the planted lysimeters, electrical conductivity increased notably, especially during dry summer periods, suggesting evapotranspiration effects and altered flow paths within the root zone. Conversely, nitrogen concentrations in the drainage water from the planted lysimeters were substantially lower than in the unplanted lysimeters, which is consistent with plant uptake and potentially enhanced microbial conversion in the rhizosphere. In summary, both vegetation and water type markedly affected drainage water quality, with clear differences between reclaimed and tap water as well as strong seasonal effects. The results suggest that when assessing irrigation with reclaimed water, site- and management-specific conditions and vegetation-related differences in drainage water quality and groundwater recharge rates must be considered.

Keywords Groundwater recharge · Membrane bioreactor · Nitrogen concentrations · Reclaimed water · Small-scale lysimeters · Water reuse

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1 Introduction

Freshwater resources, such as groundwater, are increasingly under pressure worldwide due to climatic changes, prolonged droughts, population growth and rising water demands, all of which reduce their long-term availability (Mitchell et al., 2012; Allan et al., 2013; Salehi, 2022). Global assessments have demonstrated widespread groundwater depletion caused by declining recharge and increasing abstraction, particularly in water-limited regions (Wada et al., 2010; Döll et al., 2018). This trend is also emerging in Germany. This is particularly evident in Brandenburg, one of the driest regions in the country, with a negative annual climatic water balance of −52 mm at a mean temperature of 9.7 °C (DWD, 2019; CDC, 2023). Climate projections for the Brandenburg–Berlin region under a high-emission climate pathway (RCP8.5) for 2031–2060 indicate that groundwater recharge might decrease by up to 25%, highlighting the vulnerability of regional water resources (MLUK, 2022).

As a response to increasing water scarcity, the use of treated wastewater for agricultural irrigation is increasingly discussed and applied worldwide (FAO, 2020; EU, 2020).

Potential risks to human health, soil quality and groundwater resources have been documented, particularly in relation to salinity, nutrient accumulation and contaminant transport (EPA, 2012; Becerra-Castro et al., 2015; Helmecke et al., 2020; Hashem & Qi, 2021). Excessive nitrogen inputs increase the risk of nitrate leaching into groundwater, depending on plant uptake, irrigation regime, and soil properties (Basse & Ritchie, 2005; Syswerda et al., 2012). Long-term studies also showed that repeated inputs of organic matter can alter soil nutrient stores and leaching behaviour over time (Diacono & Montemurro, 2010). In addition to nutrients, physico-chemical water quality parameters and organic micropollutants (OMPs) may affect soil function and pose risks to groundwater quality (Qadir & Scott, 2009; Becerra-Castro et al., 2015; Dittmann et al., 2025). Understanding nutrient and contaminant dynamics under water reuse is therefore particularly important.

Lysimeter experiments are applied to investigate the water balance and solute transport in the soil and in groundwater under controlled yet field-relevant conditions. Studies addressing wastewater or reclaimed water (RW) irrigation using lysimeters have primarily focused on individual parameter groups, such as nutrient dynamics or OMPs, rather than on integrated assessments of overall water quality (Pedrero et al., 2010; Grossberger et al., 2014). Comprehensive multi-parameter evaluations of water quality impacts under irrigation with RW are therefore still limited, particularly in water-scarce regions.

This study addresses this knowledge gap by simultaneously monitoring nitrogen species (total bound nitrogen (TN_b), ammonium nitrogen (NH_4^+-N), nitrate (NO_3^--N), nitrite (NO_2^--N)), and physico-chemical parameters (pH, electrical conductivity (EC)), other anions (chloride, fluoride, sulfate), and OMPs over a full annual cycle. A 12-month small-scale lysimeter study was conducted using agricultural soil from Brandenburg, irrigated with wastewater after treatment with a membrane bioreactor (MBR) or with tap water (TW) as a reference. The primary focus of this study is the comparison between planted and unplanted lysimeters under RW irrigation, to assess the effect of vegetation on drainage water quality. Soil from the same site has previously been analysed for the transport of selected 1OMPs (Thalmann et al., 2025; Dittmann et al., 2025).

By integrating multiple water quality parameters under demand-oriented irrigation in phase I and managed groundwater recharge in phase II, this study provides a comprehensive assessment of nutrient dynamics, contaminant transport and potential risks to groundwater, thereby contributing to the development of safe and sustainable water reuse strategies in water-limited regions.

2 Materials and Methods

2.1 Lysimeter System and Soil Characteristics

The lysimeter study was conducted in Berlin-Marienfelde, Germany (52°23′51.7"N, 13°22′06.9"E). The facility houses 12 lysimeters in total. Of these, eight were operated over the entire study period and are evaluated in this study. A second planted lysimeter (TW-P2), irrigated with tap water, provided reliable data during phase II. Selected data are presented in the Supplementary Information (Fig. S1 to S2) for comparison with TW-P1. The remaining three were not available for continuous operation over the full period. The eight small-scale lysimeters were filled equally with compacted soil material from an agricultural field in Fresdorf, Brandenburg (52° 16′ 14″ N, 13° 4′ 36″ E). Each lysimeter had an inner diameter of 15 cm, a surface area of 183 cm^2, a depth of 55 cm and a lysimeter volume of 9.17 L. The total dry weight of one lysimeter is 21 kg. Drainage water was collected in 1-L glass bottles for volume quantification and further analyses.

Each lysimeter consisted of three distinct layers: 30 cm of a topsoil affected by plowing (Ap horizon), overlying 20 cm of a subsoil (B horizon), and 5 cm of a gravel drainage layer (Fig. 1, left). The soil is classified as a Stagnic Cambisol (world reference base), which is characteristic for agricultural sites in Brandenburg. It has iron and manganese concretions in the subsoil (Thalmann et al., 2025). Physico-chemical soil characteristics for the Ap and B horizons are summarised in Table 1.

Table 1 Physico-chemical characteristics of the soil layers of Ap and B horizon. (C_{org}: organic carbon)

Layer	Textual fractions [%][a]			C_{org} [%]	pH_{CaCl2}
	sand	silt	clay		
topsoil Ap	83.3	11.5	5.3	0.73	4.9
subsoil B	87.4	7.9	4.7	0.16	5.4

[a]Analysed by the PARIO method (Durner & Iden, 2021)

2.2 Setup of the Experiment

The lysimeter experiment was conducted over a 12-month period from August 12 in 2024 to August 28 in 2025. The lysimeters were installed in a specially designed shaft and integrated into the ground so that they were level with the surface of the terrain (Fig. 1, right). For weekly weighing and drainage water sampling, the lysimeters were lifted onto a precision scale (model QS64B, Sartorius AG, Germany) using a rack and a pulley system.

The sample volumes attained a maximum of 100 mL per lysimeter. Depending on the prevailing weather conditions, there were weeks without sampling due to missing drainage water. Six lysimeters received RW (RW-P1, RW-P2, RW-P3, RW-U1, RW-U2, RW-U3) and two lysimeters were irrigated with TW (TW-P1, TW-U1), serving

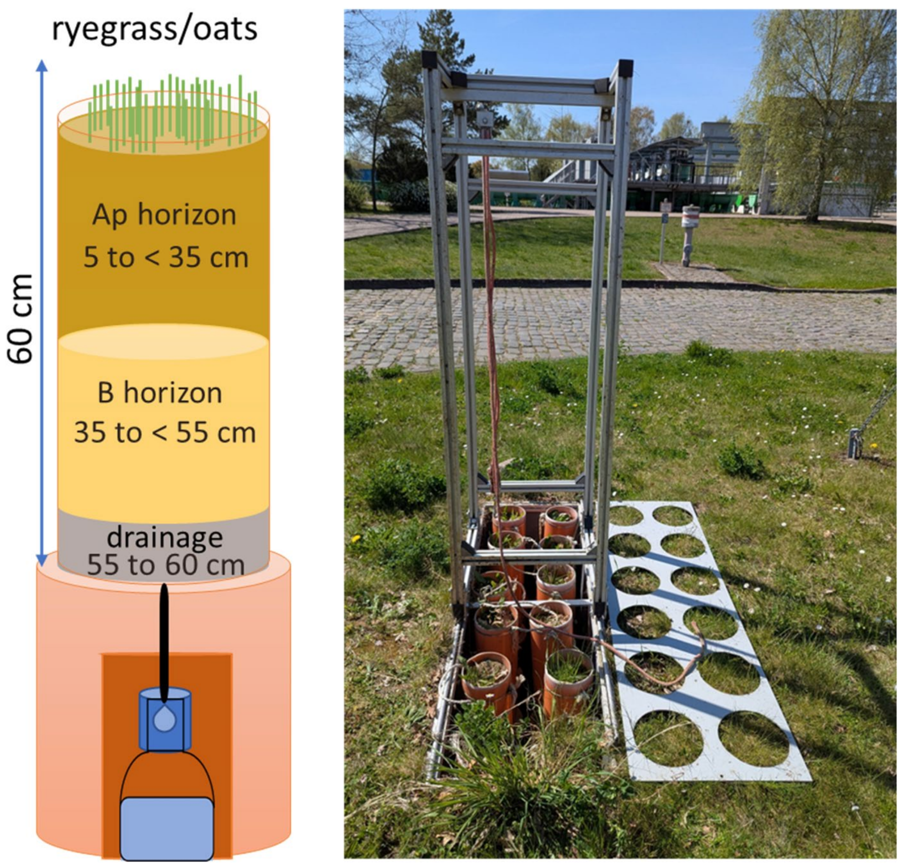

Fig. 1 Design and construction of the small-scale lysimeters (left) and photograph of the 12 lysimeters in the shaft (right), of which eight were evaluated in this study. The rack is only temporarily attached and employed for weighing the individual lysimeters once a week

as control variants. Four of the lysimeters were planted (RW-P1, RW-P2, RW-P3, TW-P1), while the other four were left unplanted (RW-U1, RW-U2, RW-U3, TW-U1). Initially, the lysimeters were planted with ryegrass (*Lolium perenne*), which was replaced by oats (*Avena sativa*) in March 2025. This change was made for two reasons. First, parts of the ryegrass stand had deteriorated over winter, likely due to a combination of overwintering conditions and the irrigation regime. Second, oats were selected to represent a common arable cereal crop relevant to irrigated agriculture and require sowing between March and April to ensure adequate crop establishment.

The study was divided into two phases (Table 2). The amount of irrigation water was calculated in advance of the experiment based on the field capacity, which is the amount of water remaining in the soil after downward percolation has ceased and surplus water has drained away (Rai et al., 2017). In phase I, irrigation was applied at 10% above field capacity to ensure that sufficient drainage water was available for weekly sampling. In phase II, the irrigation volume was increased with the aim of simulating managed aquifer recharge.

2.3 Climate Data

The Berlin and Brandenburg region is located in the temperate climate zone of the middle latitudes and represents a transition between the maritime climate of Western Europe and the continental climate of Eastern Europe, resulting in pronounced seasonal variability (DWD, 2019). The region's climate is further influenced by the alternation of humid, temperate Atlantic air masses and dry, continental air masses (DWD, 2019). Over the 30-year reference period from 1991 to 2020, the mean annual air temperature was 9.7 °C, and the mean annual precipitation amounted to 578.9 mm (DWD, 2023).

Meteorological data (temperature, potential evapotranspiration (ET_{pot}) and precipitation) were recorded by the on-site weather station close to the lysimeters throughout the entire experimental period (Fig. 2). Precipitation was measured using a tipping bucket and the ET_{pot} was calculated according to Wendling (1991). During the 12-month study period, the average air temperature was 11.2 °C, and the total precipitation amounted to 473.2 mm.

2.4 Irrigation Water

RW was produced by a pilot-scale MBR with 61 L reactor volume and 0.72 m^2 membrane area (polyethersulphone (PES), 0.04 µm pore size, MANN + HUMMEL, Germany) described in more detail by Bauer et al. (2025). The MBR treated the effluent from the primary clarifier of a wastewater treatment plant in Brandenburg, Germany (52,000 m^3/d dry weather inflow).

The MBR was operated with a target total suspended solids (TSS) concentration between 5 and 10 g/L and net flows between 11.8 and 21.7 L/h without nitrification by achieving a food-to-microorganism ratio above 0.5 kg chemical oxygen demand (COD) per kg TSS per day and a sludge retention time below 5 d in 2024. In 2025, the combination of high net flow of 21.7 L/h and a dosage of powdered activated carbon (PAC; SAE Super, Norit, Netherlands) for the removal of

Table 2 Structure of the experimental phases, classified by duration, planting and irrigation

Phase	Duration	System	Planting	Irrigation water	Irrigation volume
I	August 12 2024 to January 15 2025	RW-P1, RW-P2, RW-P3	Ryegrass (*Lolium perenne*)	RW	10% above field capacity Total weight of each lysimeter: 23.52 kg
		TW-P1		TW	
		RW-U1, RW-U2, RW-U3	-	RW	
		TW-U1		TW	
II	January 16 to August 28 2025	RW-P1, RW-P2, RW-P3	Ryegrass (*Lolium perenne*) and common oats (*Avena sativa*)	RW	53% above field capacity (volume was increased by 1 L) Total weight of each lysimeter: 24.52 kg
		TW-P1		TW	
		RW-U1, RW-U2, RW-U3	-	RW	
		TW-U1		TW	

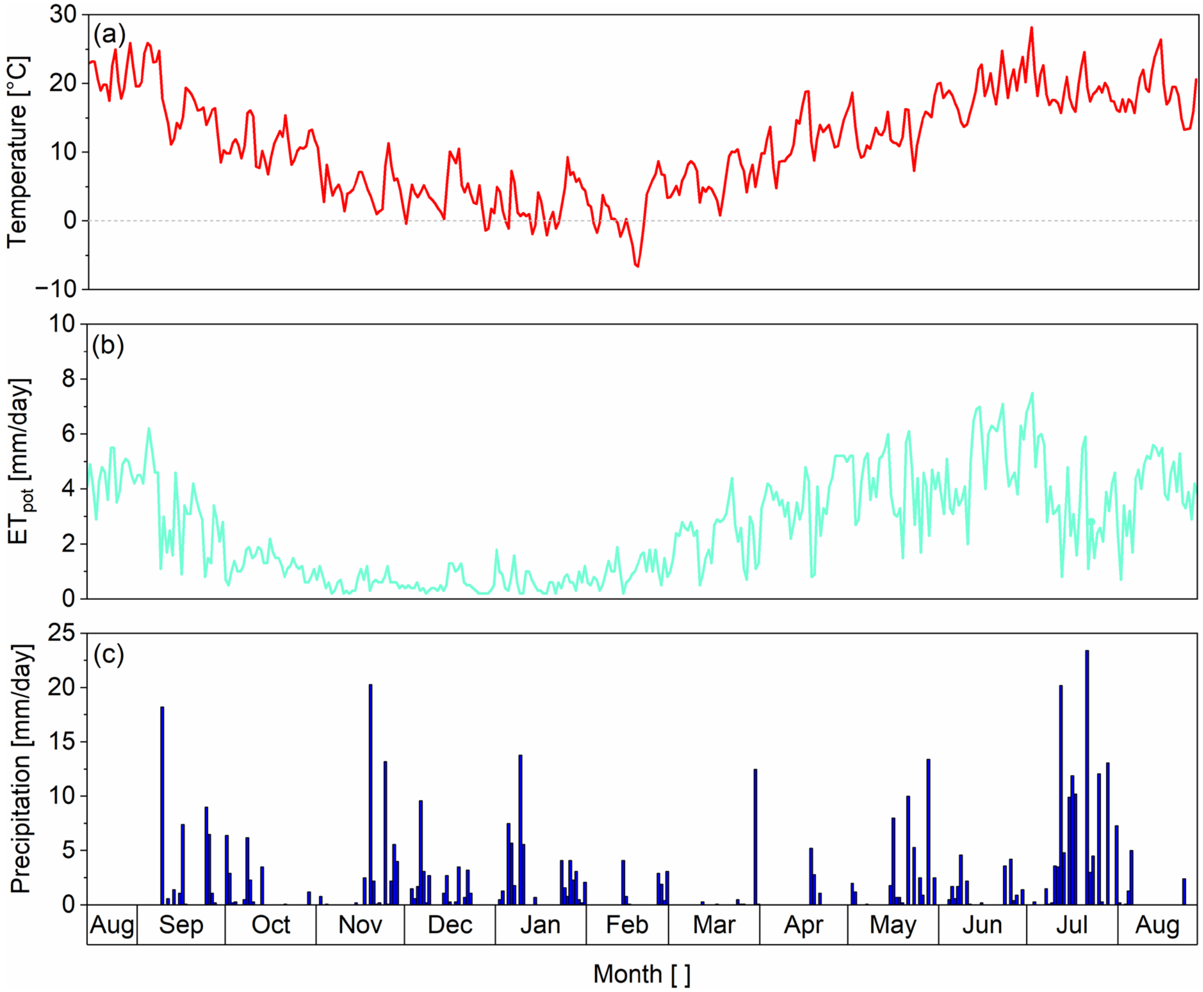

Fig. 2 Climate data from August 2024 to August 2025 showing (**a**) temperature, (**b**) potential evapotranspiration (ET_{pot}) according to Wendling (1991), and (**c**) precipitation

OMPs was investigated. The irrigation water was directly collected from the permeate pipe in 30 L canisters. The MBR operation settings from August 2024 to August 2025 are summarised in Table S1, and RW originated from different operational phases. The MBR was not in operation between late November 2024 and early May 2025. During this period, irrigation continued using RW that had been produced by the MBR during the previous operational phase and stored for later use. Each RW batch used for irrigation was analysed for the parameters listed in Table S2.

TW was collected from the same tap throughout the entire study period, and parameters were analysed on three dates. As there were only minor fluctuations, it was assumed that the water quality remained constant.

2.5 Physico-Chemical and Chemical Analyses of Irrigation and Drainage Water

The pH value and the EC were determined in unfiltered drainage water and irrigation water samples using electrodes (WTW SenTix and TetraCon, USA).

TN_b was determined in unfiltered samples via catalytic combustion and light absorption detection using a multi N/C 3300 ChD duo analyser (Analytik Jena, Germany). Concentrations of NH_4^+-N were determined using cuvette tests and photometry following manufacturer instructions (test kit LCK 304, and spectrophotometer DR1900, Hach Lange, USA).

Concentrations of anions (chloride, fluoride, nitrate, nitrite, sulfate) were determined by ion chromatography (930 Compact IC Flex with conductivity detector LF Detector 1, Metrohm, Switzerland).

A total of 21 OMPs (Table 3) were quantified using high-performance liquid chromatography (HPLC; Agilent 1290 Infinity, Agilent, Waldbronn, Germany) coupled with a triple-quadrupole mass spectrometer (MS/MS) with Turbo V ion source (Sciex, QTRAP 5500, Darmstadt, Germany), as described in detail by Zeeshan et al. (2023). Prior to measurement, all samples were filtered through 0.45 µm syringe filters (Chromafil Xtra, PET, Macherey–Nagel GmbH & Co. KG, Germany). For sample preparation, 1 mL of each undiluted sample was supplemented with 10 µL of an internal standard micropollutant stock solution. The samples were stored at 4 °C until analysis. Some compounds exhibited high concentrations and were therefore re-measured at a dilution of 1:10.

2.6 Data Analysis

The water balance was calculated based on the lysimeter surface area using the equation:

$$\Delta S = P + I - ET_{pot} - D \qquad (1)$$

where ΔS is the change in water storage (mm), P is precipitation (mm), I is irrigation (mm), ET_{pot} is potential evapotranspiration (mm), and D represents drainage (mm).

Table 3 Overview of analysed organic micropollutants (OMPs) with abbreviations, CAS numbers, and limits of quantification (LoQ)

Abbreviation	Name	CAS No	LoQ [µg/L]
AAMPS	Sodium 2-acrylamido-2-methylpropane sulfonate	5165-97-9	0.005
ACE	Acesulfame	55589-62-3	0.005
ATA	Adamantan-1-amine	768-94-5	0.005
BDMA	Benzyldimethylamine	103-83-3	0.05
BETMAC	Benzyltrimethylammonium	56-93-9	0.06
BTA	Benzotriazole	95-14-7	0.01
CBZ	Carbamazepine	298-46-4	0.0015
CG	Cyanoguanidine	461-58-5	0.05
DZA	Diatrizoate	117-96-4	0.01
DCF	Diclofenac	15307-86-5	0.001
DMBSA	Dimethylbenzenesulfonate	25321-41-9	0.01
DIOTOG	1,3-Di-*o*-tolylguanidine	97-39-2	0.005
DPG	1,3-Diphenylguanidine	102-06–7	0.005
MAPMA	N-(3-(dimethylamino)-propyl)methacrylamide	5205-93-6	0.01
MEL	Melamine	108-78-1	0.01
MPSA	2-Methyl-2-propene-1-sulfonate	1561-92-8	0.005
OXI	Oxipurinol	2465-59-0	0.02
PRI	Primidone	125-33-7	0.005
PTSS	*p*-Toluenesulfonate	104-15-4	0.005
SAC	Saccharine	81-07-2	0.01
SMX	Sulfamethoxazole	723-46-6	0.007
TFMSA	Trifluoromethanesulfonate	1493-13-6	0.001
VSA	Valsartanate (2 ‘-(2H-tetrazol-5-yl)-[1,1’-biphenyl]−4-carboxylate	164265-78-5	0.005

Mass balances for various parameters such as anions and OMPs were determined based on the cumulative mass influx and outflux. During each irrigation event, the applied volume, date and time, as well as concentrations in the corresponding RW batch were considered. For the mass balances, all concentrations below the limit of quantification (LoQ) were considered as zero.

The measured parameters in the drainage water are summarised descriptively. Weekly medians, means, standard deviations (SD) and ranges are provided for the RW replicates in Table S3 of the Supplementary Information.

Data processing and visualization were performed using Origin 2023b (OriginLab).

3 Results and Discussion

3.1 Water Balance and Formation of Drainage Water

The effect of irrigation and vegetation on the water balance and drainage water formation revealed marked differences between planted and unplanted lysimeters over the one-year period (Fig. 3). The corresponding water balance for the tap-water systems is provided in the Supplementary Information (Fig. S3).

Similar amounts of RW were applied to planted lysimeters (3,089 to 3,353 mm) and unplanted lysimeters (2,871 to 3,129 mm). Over the study period, cumulative precipitation amounted to 473 mm and ET_{pot} to 972 mm. Clear differences in drainage were observed between the different systems. On average, drainage volumes were 31% lower in planted

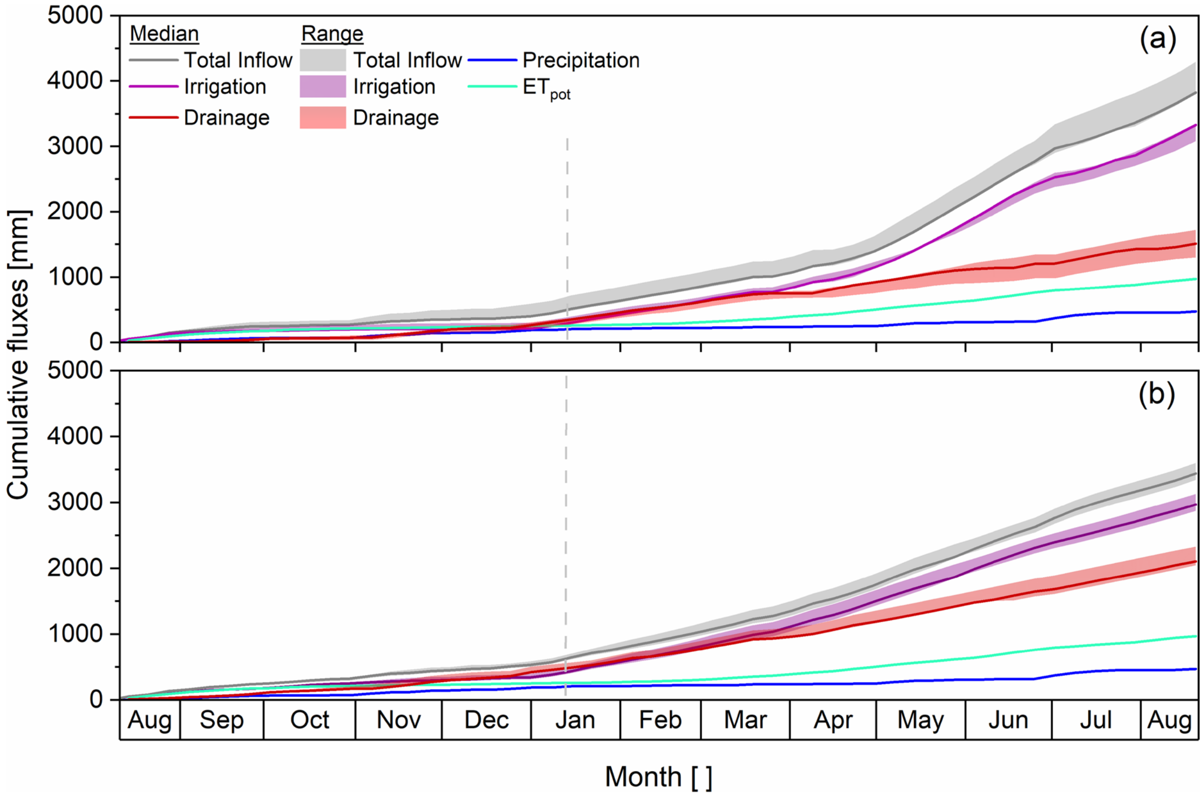

Fig. 3 Cumulative fluxes over one year in (**a**) planted and (**b**) unplanted lysimeters with reclaimed water (RW) irrigation. Fluxes are shown for total inflow (irrigation plus precipitation), drainage, and potential evapotranspiration (ET_{pot}). For total inflow, irrigation and drainage, lines show the median and shaded bands the range (minimum to maximum) across the three replicate lysimeters. The grey dotted line marks the start of increased irrigation volumes (boundary between phase I (10% above field capacity) and phase II (53% above field capacity))

lysimeters (1,302 to 1,721 mm) than in unplanted lysimeters (2,049 to 2,332 mm). The lower drainage in planted lysimeters coincided with increased plant water uptake and higher ET_{pot}, resulting in reduced percolation. This effect was particularly pronounced from March to August, corresponding to the main growing season. The results are consistent with findings of Benettin et al. (2021) and Zhao et al. (2021), who found that evaporation in temperate latitudes and semi-arid regions, respectively, is more than twice as high in vegetated sites as in sites without vegetation.

In the unplanted lysimeters, drainage increased almost proportionally to the total inflow, consistent with the absence of plant water uptake (Fig. 3). The highest irrigation demand (3,353 mm) and lowest drainage (1,302 mm) were observed in lysimeter RW-P1. As vegetation biomass decreased across the RW systems (RW-P1, RW-P2 to RW-P3), drainage volumes increased while irrigation demand decreased. TW-P1 showed the lowest irrigation demand (2,442 mm) and among the highest drainage volumes (1,724 mm). Until late March, drainage and irrigation followed similar temporal patterns. Following oat sowing in March, the curves diverged, indicating the onset of vegetation effects. Vegetation is a key factor in controlling soil water dynamics and the water balance in terrestrial ecosystems (Zhao et al., 2021). This is mainly driven by increased evapotranspiration and decreased drainage, resulting in reduced groundwater recharge compared to unplanted soil (Zhang & Schilling, 2006; Mohan et al., 2018). This pattern is also reflected in the observations of the present study.

The overall water balance was primarily governed by vegetation. In the TW systems, drainage was higher in the unplanted lysimeter (TW-U1: 2,118 mm) than in the planted one (TW-P1: 1,724 mm). This may reflect differences in infiltration and soil water retention between the individual lysimeters.

3.2 Nitrogen Dynamics

Concentrations of TN_b, NO_3^--N, NO_2^--N, and NH_4^+-N in drainage water were strongly affected by vegetation, irrigation water quality and seasonal dynamics (Fig. 4). Elevated nitrogen concentrations were deliberately maintained in the irrigation water. Consequently, the observed values do not reflect typical conditions in wastewater treatment plant effluent. To allow a comparison between TN_b and NH_4^+-N, NO_3^- and NO_2^- concentrations originally reported as ions (mg/L) were converted to an N-equivalent basis (mg/L N).

Lysimeters irrigated with RW consistently exhibited higher nitrogen concentrations in drainage water than those irrigated with TW, reflecting the additional nitrogen inputs associated with RW. Across all systems, NO_3^--N accounted for most of the measured TN_b, while NO_2^--N and NH_4^+-N occurred only in minor concentrations, typically in the range of 0.0–1.0 mg/L N and 0.0–0.2 mg/L N (Fig. 4, Fig. S4).

The unplanted lysimeters (RW-U1 to RW-U3) reached maximum NO_3^--N concentrations of 90.5 to 105.7 mg/L N from mid to end of July. The planted lysimeters showed peak concentrations of 65.8 to 73.6 mg/L N at the beginning of July. In contrast, NO_3^--N concentrations in drainage water from the TW systems were substantially lower throughout the entire study period (planted: 0.1 to 1.5 mg/L N; unplanted: 0.1 to 14.3 mg/L N).

Nitrogen concentrations were generally low during phase I and increased during spring in phase II, consistent with higher nitrogen inputs to the system (Fig. 4). Rising temperatures can accelerate microbial nitrification and mineralization, leading to higher nitrogen conversion rates (Hu et al., 2021). This seasonal increase coincides with the start of the growing season in spring. Higher soil temperatures and increased soil moisture stimulate microbial nitrogen mineralization, increasing the availability of inorganic nitrogen species for plant uptake (Robertson & Groffman, 2015).

The crop transition from ryegrass to oats may also affect nitrogen dynamics. Balkcom et al. (2019) reported that ryegrass and oats exhibit species-specific differences in nitrogen uptake (oats 54%, ryegrass 21%), which can cause short-term fluctuations in system nitrogen levels during crop rotation. After harvesting the ryegrass and sowing the oats, the concentrations of TN_b and NO_3^--N in the drainage declined rapidly in all lysimeters (Fig. 4). Since this decline was also observed in the unplanted lysimeters while irrigation remained comparable, the change is unlikely solely attributed to plant uptake or crop-specific nitrogen utilization.

Based on TN_b loads from phase II, the cumulative nitrogen supply from irrigation water in the planted

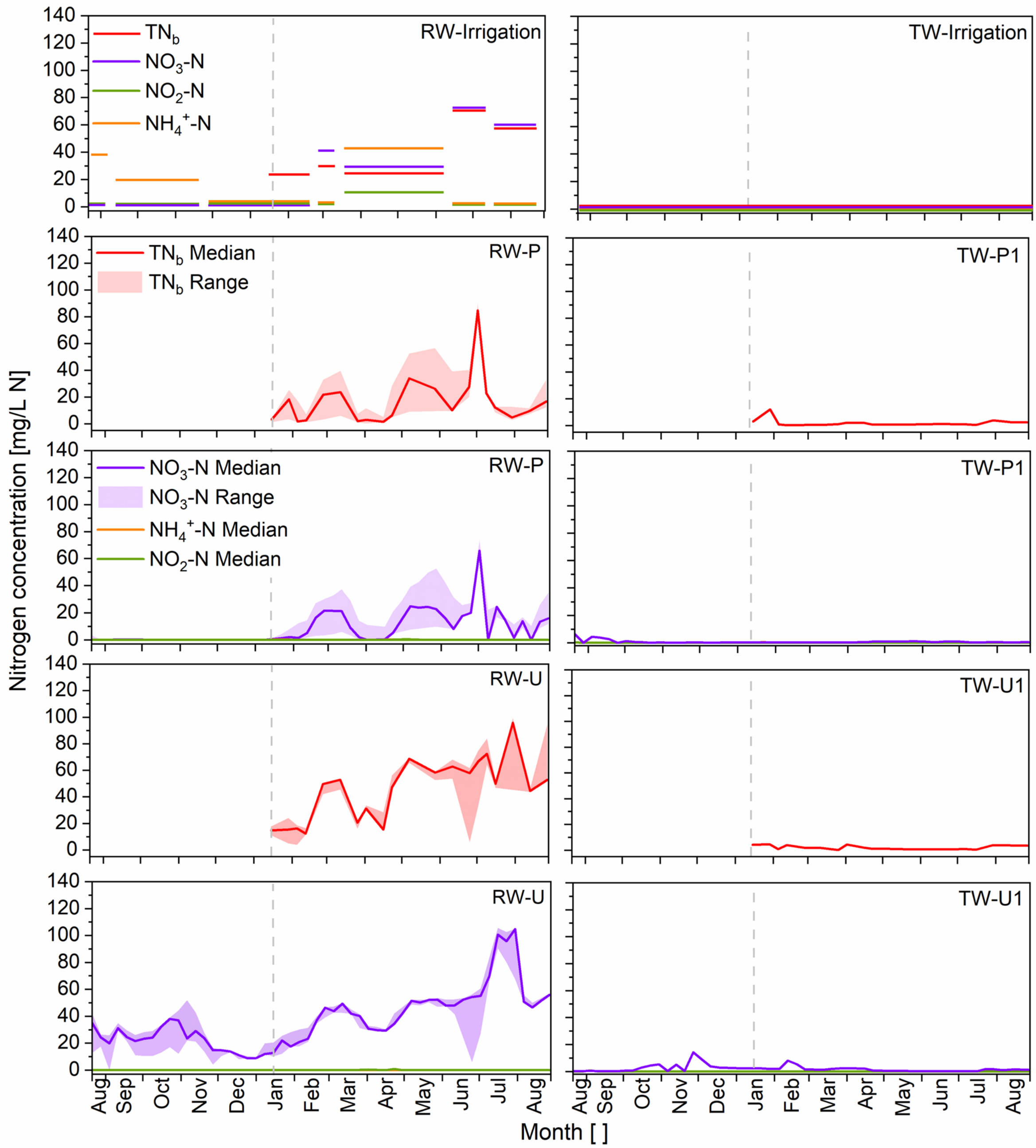

Fig. 4 Time series of total bound nitrogen (TN_b), nitrate (NO_3^--N), nitrite (NO_2^--N) and ammonium nitrogen (NH_4^+-N) in the irrigation water (top row) and in the drainage water of planted (RW-P and TW-P1) and unplanted (RW-U and TW-U1) lysimeters irrigated with reclaimed water (RW) and tap water (TW)

RW treatments amounts to 2,136–2,430 mg N, and in the unplanted lysimeters to 1,764–1,915 mg N. The cumulative nitrogen export with drainage was 140–412 mg N in the RW planted lysimeters and 764–1,033 mg N in the RW unplanted lysimeters. This corresponded to 6–19% and 40–54% of the applied nitrogen. The exported TN_b was dominated by NO_3^--N, while NO_2^--N and NH_4^+-N contributed only minor fractions (Fig. 4). The remaining fraction of the applied nitrogen was not recovered in the drainage water and therefore reflects a combination of retention and loss processes within the lysimeters. These processes may include temporary storage in the soil matrix (including organic nitrogen in soil and microbial biomass), plant uptake, and gaseous losses through denitrification and ammonia volatilisation. As none of these pathways were quantified in this study, their individual contributions cannot be determined. Denitrification may nonetheless have been relevant, given the high nitrate availability, although both its extent and the ratio of its gaseous products (N_2 and N_2O) depend on carbon and nitrate availability (Senbayram et al., 2012; Robertson and Groffman, 2015). Ammonia volatilisation was probably minor, because the initial soil substrate was acidic (pH_{CaCl2} 4.9–5.4; Table 1), and the drainage pH value remained near-neutral throughout the study. Under these conditions, nitrogen is retained as non-volatile ammonium (NH_4^+) rather than converted to volatile ammonia (NH_3) (Avnimelech and Laher, 1977).

To assess regulatory relevance, measured concentrations were compared with threshold values of the German Groundwater Ordinance (Federal Republic of Germany, 2017). In all RW systems, NO_3^--N concentrations in drainage water during phase II exceeded the specified limit of 50 mg/L NO_3^- (equivalent to 11.3 mg/L NO_3^--N) for most of the time. In contrast, the planted RW lysimeters consistently remained below this limit during phase I. NO_2^--N occasionally exceeded the threshold of 0.5 mg/L NO_2^- (equivalent to 0.15 mg/L NO_2^--N) in both planted (up to 0.9 mg/L N) and unplanted (up to 1.0 mg/L N) RW lysimeters, as well as in one TW lysimeter (0.7 mg/L N). NH_4^+-N remained below the threshold of 0.5 mg/L NH_4^+ (equivalent to 0.39 mg/L NH_4^+-N) across all lysimeters.

3.3 pH Value

The RW used for irrigation had a median pH value of 8.4 (7.4–8.4) in phase I, and a median of 6.7 (6.5–6.9) in phase II. The pH values of the drainage water from the planted lysimeters ranged from 6.9 to 8.0 in phase I, and from 6.7 to 8.0 in phase II (Fig. 5a).

These results indicate relatively stable conditions, with only moderate and temporary variations between the different lysimeters. The pH of the drainage water in unplanted lysimeters showed similar ranges (6.3 to 8.0 in phase I, 6.2 to 8.0 in phase II), but greater short-term fluctuations with more pronounced and frequent peaks (Fig. 5b). These differences between planted and unplanted systems may be linked to their nitrogen dynamics and to root-induced processes in the rhizosphere. Nitrification contributes to soil acidification and can thereby lower the pH (Ni et al., 2023). It occurs in both systems, although to a lesser extent in the unplanted lysimeters. In the planted systems, further processes occur in addition to nitrification: the exudation of root exudates (organic acids) to increase the availability of metals, the release of CO_2, which further lowers the pH in the rhizosphere, and the uptake of nitrate, which leads to an increase in pH (Hinsinger et al., 2003). These processes act together to provide greater buffering in the planted variant. As nitrification and denitrification were not measured in this study, the observed pH differences are reported as such rather than as evidence of a specific mechanism.

Overall, the pH values of all systems remained within the typical range for uncontaminated groundwater (6.0 to 8.5 according to EEA (2000)). Throughout the one-year study, the pH value of the drainage water under TW irrigation remained within a similar range to that observed under RW irrigation.

3.4 Electrical Conductivity

The EC of RW amounted to a median of 1.5 mS/cm (1.4–1.5 mS/cm) in phase I and to the same median of 1.5 mS/cm (1.2–1.5 mS/cm) in phase II. In planted lysimeters, EC exhibited distinct seasonal patterns, with higher values during summer and lower values in winter (Fig. 6a).

EC was consistently higher in RW-irrigated lysimeters (0.3 to 5.0 mS/cm) than in those receiving TW (0.2 to 0.9 mS/cm). The pronounced EC summer maxima in the planted lysimeters, especially in June and July, suggest an additional plant-mediated concentration effect during periods of high ET_{pot} and reduced percolation. In unplanted lysimeters, temporal fluctuations were less pronounced. EC in drainage water was also higher under RW irrigation (0.3 to 1.8 mS/cm) than under TW irrigation (0.2

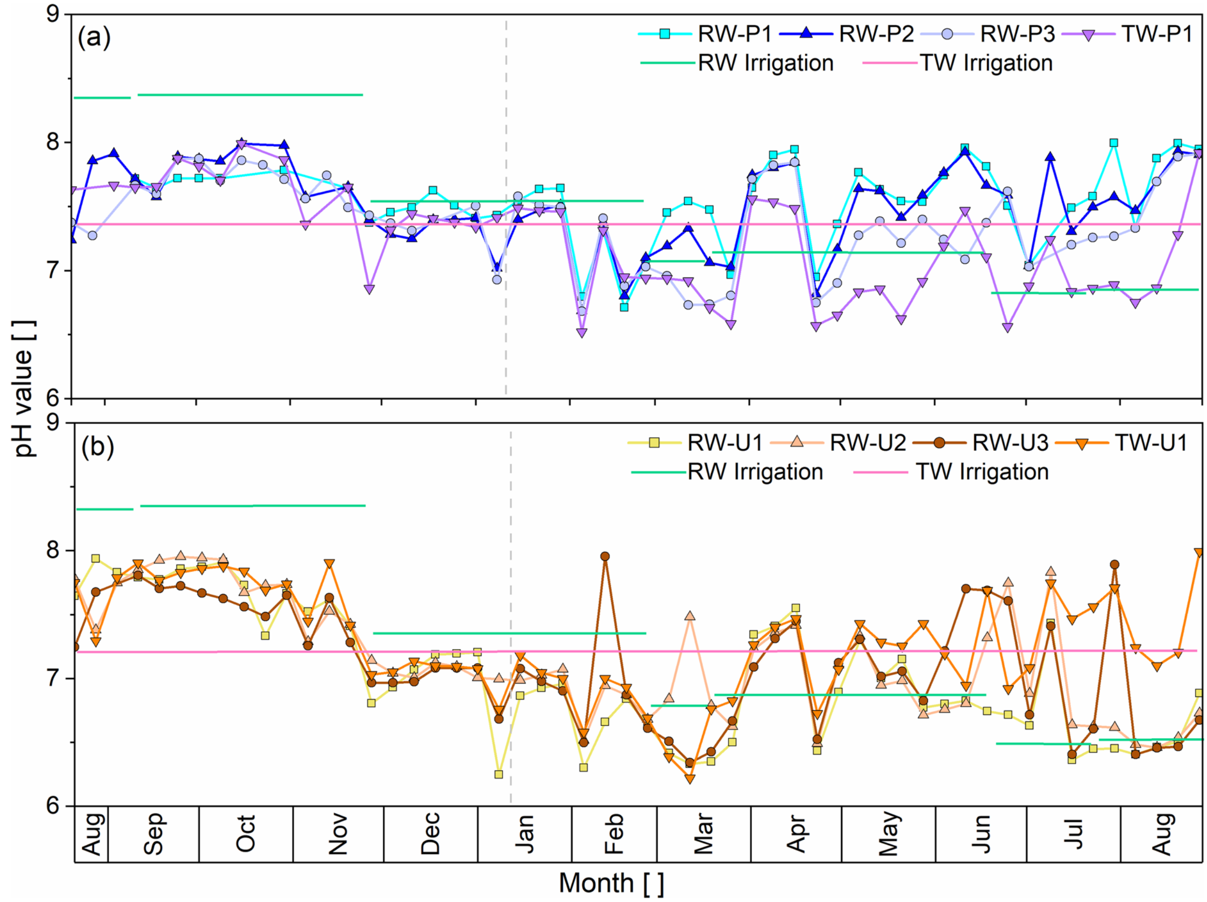

Fig. 5 Time series of pH values in drainage water of (**a**) planted and (**b**) unplanted lysimeters irrigated with reclaimed water (RW) and tap water (TW)

to 1.0 mS/cm). The higher EC under RW irrigation reflects its higher ionic content compared to TW.

3.5 Other Anions

The mass balances of the monitored anions (chloride, fluoride and sulfate) revealed distinct mobilisation patterns affected by irrigation water quality (RW and TW) and the presence of vegetation. Across all analytes, temporal variability was more pronounced in planted than in unplanted lysimeters.

In RW, median anion concentrations were higher in phase II than in phase I (sulfate: 90.0 vs. 101.2 mg/L, chloride: 91.4 vs. 178.8 mg/L, fluoride: 0.2 vs. 0.3 mg/L). In phase II, anion concentrations in drainage water increased towards late spring/summer across all systems (Fig. S5). During the same period, the EC of the drainage water from the planted lysimeters increased markedly (Fig. 6a). These patterns suggest that seasonal changes, particularly increased plant activity and shifts in the water balance, substantially affected ion concentrations (Conklin, 2014). Rising temperatures during the growing season likely enhanced evapotranspiration, promoting salt accumulation in the upper soil layers and consequently elevating concentrations in the drainage water (Ayers & Westcot, 1985; Rhoades et al., 1992).

Chloride and sulfate exhibited similar temporal patterns to each other in the drainage water, with pronounced seasonal variability in concentrations and loads. During winter and early spring, sulfate outputs exceeded the inputs from December to February and chloride outputs exceeded inputs from November to

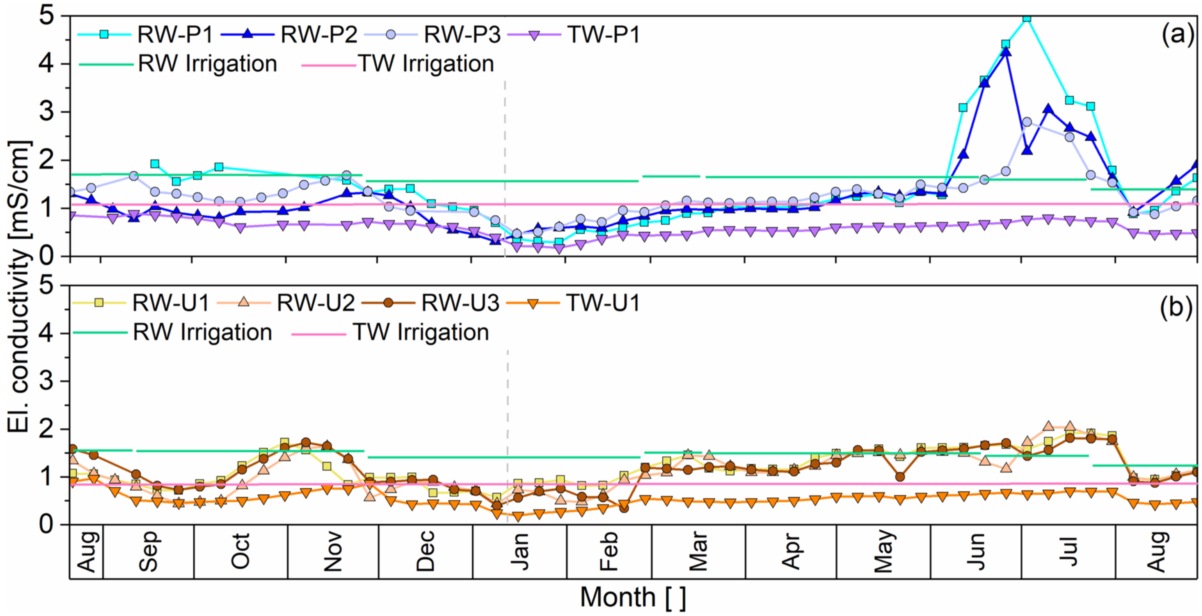

Fig. 6 Time series of electrical conductivity in the irrigation water and drainage waters of (**a**) planted and (**b**) unplanted lysimeter irrigated with reclaimed water (RW) and tap water (TW)

April (Fig. S5). This can be attributed to increased percolation rates (Fig. 3), and the mobilisation of ions retained in the soil. Sulfate is generally mobile and readily soluble, although its transport can be affected by its sorption to the soil matrix. Chloride is only weakly sorbed and therefore behaves conservatively in soils (Reuss, 1975; Conklin, 2014; Geilfus, 2018).

Concentrations of chloride and sulfate in the drainage water of unplanted RW systems occasionally approach the limit values of 250 mg/L for groundwater specified in the German Groundwater Ordinance (2017): Cl^-: Median 152.7 mg/L, min. 24.8 mg/L, max. 342.0 mg/L; SO_4^{2-}: Median 110.9 mg/L, min. 14.1 mg/L, max. 209.7 mg/L. In planted RW systems, concentrations exceeded 250 mg/L during the growing season (phase II), with peak values of 1,085 mg/L (Cl^-) and 639 mg/L (SO_4^{2-}). Reduced plant activity during the winter period probably resulted in lower retention of both ions. In TW systems, inputs and outputs were considerably lower, and unplanted lysimeters showed only minimal export values.

Fluoride (F^-) exhibited the lowest concentrations of all monitored ions and showed minimal temporal variability. Drainage water concentrations were similar in planted and unplanted lysimeters (overall median: 0.2 mg/L, range: 0.1–0.8 mg/L), with no notable peaks. Differences between input and output, and between replicates, were negligible. This suggests that fluoride was largely unaffected by vegetation or other processes related to the system.

3.6 Organic Micropollutants

The 21 OMPs exhibited highly variable leaching behaviours, leading to their division into three categories (Fig. 7). DZA, MPSA and PRI were highly leached, with over 60% of the applied mass entering the drainage water in relevant proportions. Eight substances demonstrated moderate leaching of 10% to 60%, while the remaining nine exhibited only minor leaching of less than 10%. The results only refer to RW systems, because it was not expected that the TW systems contribute significant amounts of OMPs.

For ACE, the amount detected in the drainage water was higher than the amount applied with the irrigation water. This substance was therefore excluded from the detailed leaching evaluation. The measured input and output of ACE over the study period, including the cumulative loads, can be found in the Supplementary Information (Fig. S12).

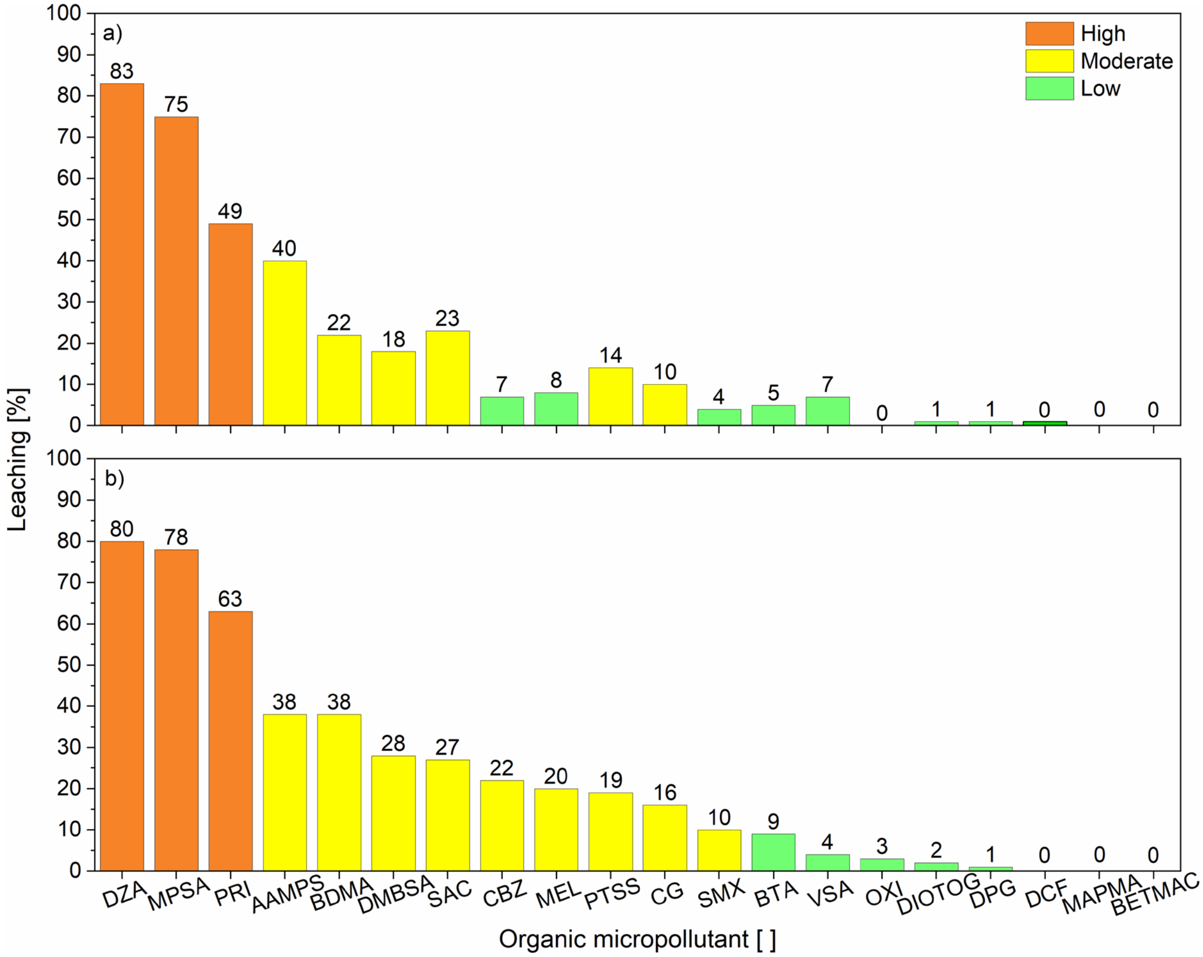

Fig. 7 Average leaching percentages of the analysed organic micropollutants, calculated from the total cumulative leached loads and aggregated across all **a**) planted and **b**) unplanted lysimeters. Abbreviations of the organic micropollutants are defined in Table 3 (Sect. 2.5)

BDMA exhibited the strongest difference between planted and unplanted lysimeters (Figs. 7 and 8). Leaching of BDMA to the drainage water was classified as moderate in both the planted (22%) and unplanted lysimeters (38%). BDMA has been detected in wastewater treatment plant effluents and wastewater impacted waters, and it can originate from a variety of sources. For instance, it is produced as a by-product of the microbiological degradation of benzalkonium chlorides (Abusallout et al., 2024; Song et al., 2024). The lower leaching observed in the planted lysimeters therefore suggests an additional, vegetation-related contribution to BDMA attenuation. This could involve plant-associated processes, such as enhanced biological activity in the rhizosphere.

CBZ showed the second strongest difference between planted and unplanted lysimeters (Fig. 7), indicating that plants reduced leaching under the conditions of this study. The leaching into the drainage water of the unplanted lysimeters was classified as moderate (22%), while the leaching of the planted lysimeters was classified as low (7%). CBZ is widely reported as a persistent pharmaceutical that is frequently detected in wastewater and is often used as an indicator compound due to its limited removal during conventional wastewater treatment processes and its recurring detection in environments impacted by water (Grossberger et al., 2014). In soils irrigated with RW, CBZ can adsorb to soil organic matter. As this sorption is partly reversible, CBZ may

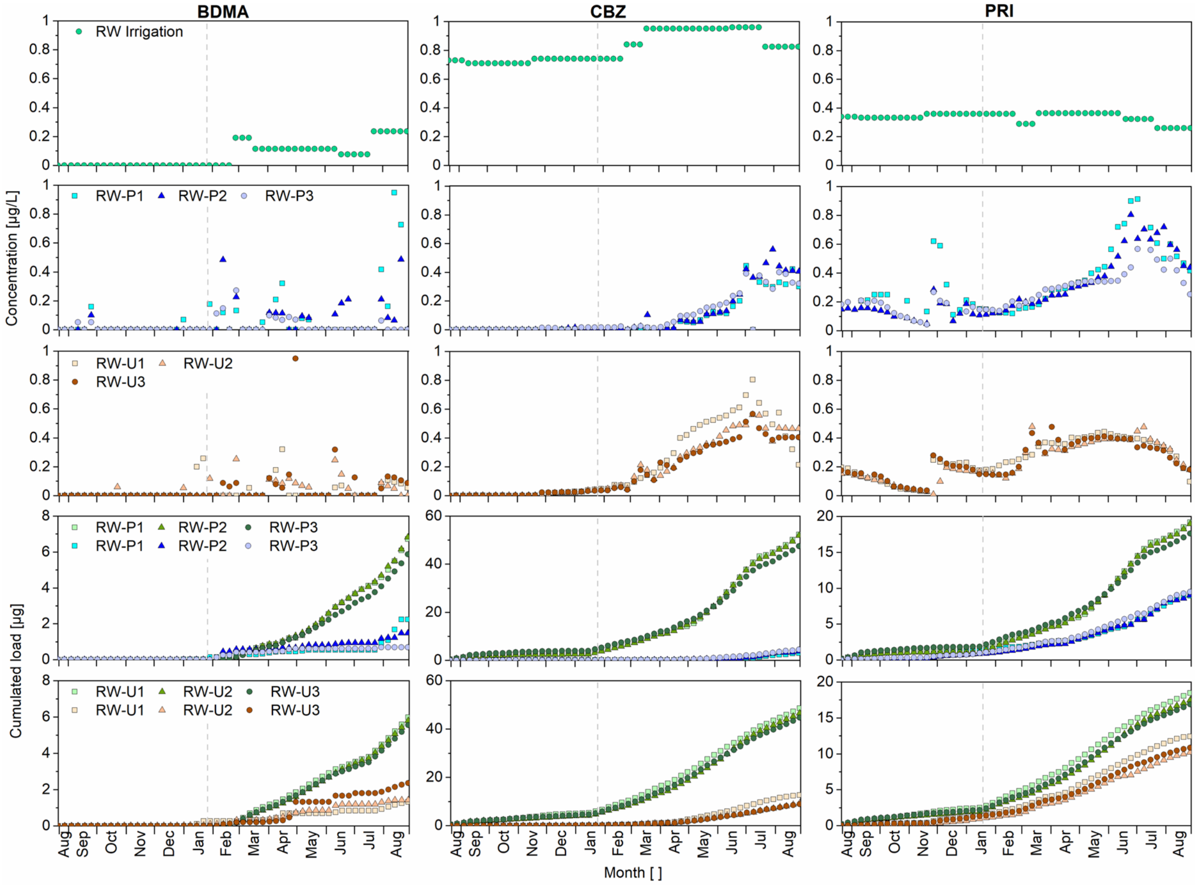

Fig. 8 Concentration and cumulative load time series of the organic micropollutants BDMA, CBZ and PRI in reclaimed water (RW) irrigation (input – green) and drainage water of planted RW-P1 to RW-P3 (output – blue) and unplanted RW-U1 to RW-U3 (output – yellow to brown) lysimeters. The upper panels show the temporal concentration profiles, while the lower panels depict the corresponding cumulative masses over the monitoring period

be retained in the upper soil layers and subsequently remobilised (Paz et al., 2016). Depending on hydrological conditions, it may therefore remain available for leaching and potentially for plant uptake (Paz et al., 2016). The lower CBZ leaching observed in planted systems compared to unplanted systems is therefore consistent with plant-associated attenuation processes. The uptake of CBZ has been observed in numerous plant species, although the extent of this uptake varies depending on the species and the experimental conditions (Mordechay et al., 2021; Manasfi et al., 2021; Riemenschneider et al., 2016).

PRI showed the third strongest difference between planted and unplanted lysimeters (Fig. 7 and Fig. 8). The leaching from the unplanted lysimeters (63%) and the planted lysimeters (49%) were classified as moderate. PRI has repeatedly been detected in wastewater treatment plant effluents and in groundwater (Hass et al., 2012). It is considered relatively persistent and mobile and is therefore sometimes used as an indicator substance in studies on subsurface transport (Scheytt et al., 2006; Hass et al., 2012). Due to its comparatively low affinity for soil sorption, retention in the soil matrix might be limited and leaching remains likely (Yu et al., 2009). However, primidone has also been detected in edible plant tissues that have been irrigated with treated wastewater (Wu et al., 2014). This suggests that the uptake of primidone by plants may contribute to its attenuation. Against this background, the lower level of PRI leaching from the planted lysimeters suggests that vegetation may reduce PRI leaching through plant uptake and, potentially, microbial transformation (Thalmann et al., 2025), which may be associated with the generally higher microbial biomass and activity reported for vegetated soils.

Neither BETMAC nor MAPMA was detected in the irrigation or drainage water. With the exception of AAMPS, DCF and VSA, concentrations leached from unplanted lysimeters were markedly higher, indicating an effect of planting. Further analysis might clarify whether the OMPs were taken up by plants or remained in the soil. While plant uptake and distribution of pharmaceuticals have been addressed in numerous studies (e.g. Wu et al., 2014; Seelig et al., 2025), information is still limited for many individual compounds. Graphical mass balances for all OMPs are shown in Fig. S6 to S11.

As no analyses of plant tissue or microbes were performed in this study, the lower leaching in planted systems was inferred from mass-balance and concentration patterns rather than demonstrated directly. The proposed plant- and rhizosphere-related mechanisms therefore remain hypotheses that are consistent with the observed data and previous reports (e.g. Mordechay et al., 2021; Riemenschneider et al., 2016). In addition, the drainage volumes of the planted RW lysimeters were, on average, 31% lower than those of the unplanted RW systems (see section Water Balance), resulting in reduced water percolation. This hydrological difference may have contributed to the lower cumulative OMP mass leached from the planted systems independently of plant uptake or rhizosphere processes. The relative contributions of hydrological and biological mechanisms cannot be determined from the presented data.

The composition of the RW changed over the course of the study, reflecting the varying operating conditions of the MBR (Table S1). As these operational changes coincided with seasonal and temporal variations, it was not possible to isolate their individual effects. All mass balances were calculated from the input concentrations measured throughout the study (Table S2) for each batch, so the reported loads and leaching values correspond to the water received by each lysimeter. As all RW lysimeters received the same water during each irrigation event, this temporal variation applied equally to the planted and unplanted systems and did not affect comparisons between them. The most significant operational change was the introduction of powdered activated carbon (PAC) in early June 2025, alongside a shift to high net flow (Table S1). Through adsorption onto the activated carbon, PAC reduced the concentrations of most OMPs in the RW. Consistent with this, the concentrations of most OMPs in the drainage water decreased from late June to early July. By contrast, the nutrient and physico-chemical parameters were deliberately maintained at elevated levels and were largely unaffected by these operational changes.

4 Conclusion

This study provides a detailed analysis of the physico-chemical composition of RW used for irrigation and of drainage water from planted and unplanted lysimeters filled with soil from Brandenburg, Germany. The findings improve the understanding of how RW irrigation affects drainage water quality and, consequently, groundwater recharge.

Overall, the study shows that the extent and composition of leaching into drainage water under RW irrigation were strongly affected by vegetation and seasonal conditions. In planted systems, vegetation altered evapotranspiration, drainage formation and attenuation, thereby modifying how irrigation water quality was reflected in drainage water quality. Vegetation can amplify salt signals through evapotranspiration, while also reducing nitrogen export. Under the conditions of this study, including deliberately elevated nitrogen concentrations in the irrigation water, the pH value of the drainage water remained stable, whereas NO_3^--N and selected OMPs represented the most relevant parameters in relation to groundwater protection.

The controlled design defines the scope of these conclusions. The findings are based on a single sandy loam soil, deliberately elevated nitrogen concentrations, and small-scale lysimeters (15 cm in diameter and 55 cm in depth) that may not fully reproduce field-scale root development. While such conditions allowed vegetation and seasonal effects to be clearly identified, they should be confirmed at a larger scale. On this basis, water reuse can be considered for irrigation and potentially also for groundwater recharge, but its environmental compatibility requires case-by-case evaluation. It depends on the constituents present in the RW (especially type and concentration of OMP), the site and system conditions affecting NO_3^--N and OMP leaching, and the extent to which vegetation affects the drainage water quality. Site-specific management strategies are therefore important when using RW for agricultural purposes.

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Data Availability The data that support the findings of this study is going to be made available upon request and after acceptance of the manuscript.

Declarations We declare that accepted principles of ethical and professional conduct have been followed.

Competing interests We declare no competing interests.

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