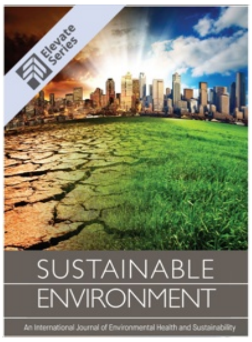

Taylor & Francis
Taylor & Francis Group

Sustainable Environment

An international journal of environmental health and sustainability

ISSN: 2765-8511 (Online) Journal homepage: www.tandfonline.com/journals/oaes21

Water quality dynamics following re-wetting of agriculturally drained peatlands: A mesocosm experiment

Teresa Silverthorn, Kate Blake , Elya Monsen-Elvik , Tom Burton , Terhi Riutta, Ian Holman, Caroline Gurd , Angus McEwen , Chris Bell , Francesca Southon , Jasmine Hughes , Chris D. Evans, Mark Simpson , Ross Morrison, Graham Howell, Josh Oakley & Mike Peacock

To cite this article: Teresa Silverthorn, Kate Blake , Elya Monsen-Elvik , Tom Burton , Terhi Riutta, Ian Holman, Caroline Gurd , Angus McEwen , Chris Bell , Francesca Southon , Jasmine Hughes , Chris D. Evans, Mark Simpson , Ross Morrison, Graham Howell, Josh Oakley & Mike Peacock (2026) Water quality dynamics following re-wetting of agriculturally drained peatlands: A mesocosm experiment, Sustainable Environment, 12:1, 2730667, DOI: 10.1080/27658511.2026.2730667

To link to this article: https://doi.org/10.1080/27658511.2026.2730667

SUSTAINABLE ENVIRONMENT
2026, VOL. 12, NO. 1, 2730667
https://doi.org/10.1080/27658511.2026.2730667

ENVIRONMENTAL SOLUTIONS | RESEARCH ARTICLE

OPEN ACCESS Check for updates

Water quality dynamics following re-wetting of agriculturally drained peatlands: A mesocosm experiment

Teresa Silverthorn[a], Kate Blake[b], Elya Monsen-Elvik[b], Tom Burton[b], Terhi Riutta[b], Ian Holman[c], Caroline Gurd[d], Angus McEwen[d], Chris Bell[e], Francesca Southon[a], Jasmine Hughes[b], Chris D. Evans[e], Mark Simpson[b], Ross Morrison[b], Graham Howell[d], Josh Oakley[d] and Mike Peacock[a,f]

[a]Department of Geography and Planning, School of Environmental Sciences, University of Liverpool, Liverpool, UK; [b]UK Centre for Ecology & Hydrology, Wallingford, UK; [c]Cranfield Water Science Institute, Cranfield University, Cranfield, UK; [d]School of Environment, Earth and Ecosystem Sciences, The Open University, Milton Keynes, UK; [e]UK Centre for Ecology & Hydrology, Bangor, UK; [f]Department of Aquatic Sciences and Assessment, Swedish University of Agricultural Sciences, Uppsala, Sweden

ABSTRACT

Peatland rewetting is an effective restoration strategy to reduce carbon losses, halt soil subsidence and restore ecosystem functioning. However, rewetting drained and degraded peatlands may release stored nutrients and potentially toxic elements into surface waters, raising concerns about eutrophication and metal mobilisation. Conversely, rewetting might have positive effects, whereby contaminants in source waters are retained within the peat itself. To assess these effects, we conducted a mesocosm experiment using peat cores from five agricultural lowland peatlands (raised bogs and fens) across England. The cores were subjected to three hydrological regimes: drained (control), raised (rewetted) and raised but fluctuating water table conditions. We monitored water quality determinands, including organic and inorganic nitrogen and phosphorus, potassium, sulphate, calcium, magnesium, sodium, chloride and fluoride, in the mesocosm surface water prior to treatment initiation and in mesocosm pore water (as an indicator of mobility) over nine weeks. Overall, pore water chemistry remained relatively consistent over the duration of the experiment. There were no site-wide effects of the water table treatments, except on nitrate concentrations, which were highest under drained conditions. Responses for individual determinands varied among sites, likely due to differences in peat type (bog vs fen), peat chemistry and land use history. Our results suggest that rewetting is unlikely to trigger widespread downstream water quality deterioration, at least not for the determinands we measured, though the effects vary by site and solute. These findings highlight the importance of site-specific assessments, particularly if nitrogen, prior to rewetting.

ARTICLE HISTORY
Received 25 March 2026
Accepted 3 September 2026

KEYWORDS
Peatland; rewetting; drainage; agriculture; water quality

1. Introduction

Anthropogenic pressures on peatland ecosystems are high, with about 12% of peatlands degraded to the extent that peat is no longer formed and carbon is being lost (United Nations Environment Programme [UNEP], 2022). Peatland drainage involves lowering the water table which introduces oxygen to the peat, accelerating decomposition and leading to increased carbon losses (Massop and Hessel, 2024). In response, peatland rewetting through raising water tables, either partially or fully, is proposed as a measure to reduce carbon losses by restoring anoxic conditions that slow decomposition; although the water quality consequences of this remain uncertain (Freeman et al., 2022).

Whether peatland rewetting mobilises or retains pollutants depends on site-specific factors that are poorly understood. Agricultural activities on drained peatlands can expose them to excess nutrients and heavy metals.

CONTACT Teresa Silverthorn teresa.silverthorn@liverpool.ac.uk; Mike Peacock m.peacock@liverpool.ac.uk Department of Geography and Planning, School of Environmental Sciences, University of Liverpool, Roxby Building, Liverpool L69 7ZT, UK
Supplemental data for this article can be accessed at https://doi.org/10.1080/27658511.2026.2730667.

These substances may be retained in the system while under drained conditions, but may become mobile, transported into surrounding surface waters following the raising of water tables (Kaila et al., 2016; Zak & Gelbrecht, 2007). This mobilisation could include various substances such as dissolved organic carbon (DOC), phosphorus (P) and ammonium (NH_4^+) (Kaila et al., 2016; van der Laan et al., 2024; van Dijk et al., 2004; Zak & Gelbrecht, 2007). For example, the mobilisation of phosphate (PO_4^{3-}) and NH_4^+, which can accumulate in the upper peat layers can be a particular concern when rewetting agriculturally drained peatlands (Zak & Gelbrecht, 2007). However, most studies focus on a handful of determinands (e.g. DOC, inorganic P and the different forms of nitrogen [N]) (Kaila et al., 2016; Zak & Gelbrecht, 2007). Less is known about the retention or leaching of other less-frequently measured nutrients, metals and trace elements, which can originate from agricultural inputs, air pollution and the direct dumping of waste (Breward, 2003; Garcés-Pastor et al., 2023; Tiemeyer et al., 2007). The potential mobilisation of these substances from peat to downstream surface waters would have implications for aquatic ecology, drinking water quality, greenhouse gas (GHG) emissions and recreational water uses (Breen et al., 2018; Jones, 1992; Vuori, 1995; Xu et al., 2018). Conversely, given the capacity of intact peat to retain nutrients, rewetting may be an effective strategy to bind substances that might otherwise leach to surface waters under drained conditions, effectively remediating the downstream impact of peatlands that have been subject to long-term agricultural intensification (Vroom et al., 2020). In addition, where external surface waters are used for rewetting, the incorporation of some pollutants such as nutrients and organic contaminants into the saturated peat matrix could enable either their retention (e.g. through adsorption or incorporation into newly forming peat) or long-term breakdown into less harmful substances (e.g. mineral N to N_2) (Temmink et al., 2023; Urbanová et al., 2011). These processes, analogous to those that occur in constructed wetlands, would provide an ecosystem service by improving downstream water quality (Berset et al., 2001; Martin-Ortega et al., 2014).

The uncertainty regarding rewetting water quality impacts is particularly critical in the UK, where industrial and agricultural legacies have created unique contamination profiles that may respond differently to rewetting than less-impacted sites studied elsewhere. Many rewetting studies have been conducted in North America (Chimner et al., 2017) or in continental North-West Europe (e.g. the Netherlands and Germany) (Andersen et al., 2017). As the birthplace of the industrial revolution, the UK has a distinct land use context; UK peatlands have been heavily impacted by industrialisation (Garcés-Pastor et al., 2023) and a large proportion of its drained peatlands are used for agriculture, rather than forestry or peat extraction (Evans et al., 2017). These legacies have contributed to heavy metal pollution (Rothwell et al., 2010) and resulted in some of the most highly degraded peatlands globally (Fluet-Chouinard and Stocker, 2023). Therefore, some UK peatlands may have relatively high concentrations of accumulated nutrients and heavy metals (Comber et al., 2023; Smith et al., 2005) that could potentially be released to downstream waters if rewetting takes place.

Early rewetting efforts in the UK were concentrated on upland blanket bogs, and the majority of the UK rewetting evidence base is derived from these systems (Parry et al., 2014; Williamson et al., 2023). However, it is estimated that about 90% of UK lowland peatlands, originally fens and raised bogs, have been drained for agriculture (Evans et al., 2017). These peatlands represent some of the highest-value farmland in the UK, accounting for a large proportion of UK horticultural crop production, with an economic value of several billion pounds per year (National Farmers' Union, 2019). Despite their importance for UK food production and economies, their agricultural use results in the emission of 2.5% of all reported UK greenhouse gas emissions (Bennett et al., 2025; Evans et al., 2017), as well as leading to other environmental problems such as land subsidence and structural damage to infrastructure, and increased flood risk (Page et al., 2020). As such, increasing attention is being given to partial or full rewetting of the UK's lowland peatlands. Whilst full rewetting (i.e. raising water tables to or near the peat surface) is favoured for conservation management and maximum delivery of most ecosystem services, other than food production, partial rewetting (i.e. raising water tables higher within the peat, but not to the peat surface) is proposed as a possible middle ground between reducing GHG emissions and maintaining agricultural output and economic productivity. There remain clear knowledge gaps about how lowland peatland rewetting will affect biogeochemical cycles and ecosystem services.

In this mesocosm experiment, we obtained peat cores from five agriculturally drained lowland peatlands across England and exposed them to three simulated water table conditions: drained, rewetted and fluctuating. Consistently high water levels are likely to minimise GHG emissions (Evans et al., 2021) but are difficult to maintain, and unlikely to be compatible with continued high-value agricultural production. Fluctuating water tables most closely mimic conditions in many rewetted and managed peatlands (Ahmad et al., 2021; Blodau & Moore, 2003; Peacock et al., 2019) and, in a real-world rewetting scenario, serve as a middle ground to balance crop production and the GHG mitigation potential of wetter soils. Conversely, wet-dry cycles may promote the release of nutrients and metals (Broder & Biester, 2015; Laine et al., 2013; Pinto et al., 2021). Thus, the objective of our study was to quantify how drained, rewetted and fluctuating water table treatments impact the mobilisation or retention of nutrients, metals and trace elements from five agriculturally drained lowland peatlands. We further aimed to determine whether treatment effects would vary across sites with differing geographies and land use histories. Given that the contamination history of UK lowland peatlands is heterogeneous across the UK, we also examined whether baseline surface water chemistry varied substantially between sites, which could subsequently influence treatment responses. We focused on lowland agricultural peatlands, because these are the systems where peatland emissions are highest (Evans et al., 2016) and where rewetting efforts for GHG mitigation are increasingly focused in the UK (Department for Environment, Food & Rural Affairs [DEFRA], 2021) and around the world (Lestari et al., 2024; Regulation (EU) 2024/1991, 2024). Such mesocosm experiments have proven insightful previously for examining GHG exchange in UK lowland peatlands (Matysek et al., 2019; Wen et al., 2020), while here we focus on water quality.

2. Methods

2.1. Experimental setup

We collected peat cores from five lowland peatlands in England (two from the northwest and three from the southeast) that are, or have been, actively cultivated using conventional drainage-based farming methods (Figure 1, Table 1). Railway View is a grassland, Pymoor is arable with cereals and Wrights, Rosedene1 and Rosedene2 are arable with horticulture crops (e.g. lettuce, celery, radish). The two Rosedene sites are ~2.5 km apart, have different specific cropping histories, and Rosedene1 was subject to an experimental water table raise in June 2024. Depending on which definition of peat is used, e.g. >30% organic material to a depth of 30 cm (Joosten & Clarke, 2002), Pymoor may not be considered a true peatland, but rather 'wasted peat' (JNCC, 2011). These soils form part of the extensive area of former thick peat in the lowlands of Eastern England that has been heavily depleted by oxidation following several centuries of agricultural drainage (Evans et al., 2017). Wasted peat soils retain high topsoil carbon content and are biogeochemically distinct from mineral soils within the same region (D'Acunha et al., 2026). As such, wasted peat soils are an important component of peatland management examined in this study. For more detailed information about peat chemistry at all sites (except Pymoor), refer to our complementary publication (Silverthorn et al., 2026). All sites have temperate climates: mean annual temperatures and precipitation are ~11 °C and 570 mm for the southeastern sites and ~10 °C and 830 mm for the northwestern sites (UK Met Office, 2025). Between 4 July and 14 August 2024, 12 intact peat cores were collected adjacent to one another at each site by carefully driving PVC drainpipes (length: 0.8 m, diameter: 0.23 m) into the soil using a mechanical excavator, discarding cores with inadequate soil volume or excessive soil compaction (Figure S1) and after first removing surface vegetation. Cores in the PVC pipes were packed in sealed plastic sacks and maintained vertically during transport to the UK Centre for Ecology & Hydrology (UKCEH) in Wallingford, Oxfordshire, UK where the mesocosm array was established in an outside area (Figure S2).

Three water level treatments were randomly assigned to the cores ($n = 4$ cores per site, per treatment): rewetted (water table maintained 10 cm below the peat surface), conventionally drained (water table maintained 10 cm above the bottom of the peat core, approximately 60 cm below the peat surface) and fluctuating (water table alternating between the rewetted and conventionally drained scenarios, every 2 weeks) (Figure 2a; Table S1). The cores were kept within their PVC drainpipes (which had 0.6 cm holes every 0.1 m at 180 ° along their lengths to allow water exchange), which were individually placed into 80-L black plastic containers designed with overflow holes and taps to allow precise manipulation of water levels through water additions directly to the container (Figure 2b). Rainfall was permitted to fall directly onto the cores, but the surrounding container was covered with several layers of blue plastic wrap to exclude rainwater inputs and block sunlight to prevent algal growth and temperature artefacts (Figure S3). This set-up was chosen for its resemblance to field conditions, where water levels are controlled by land managers using porous pipes running below the field, terminating in ditches at each side. After extensive consideration, tap water was chosen as the best and

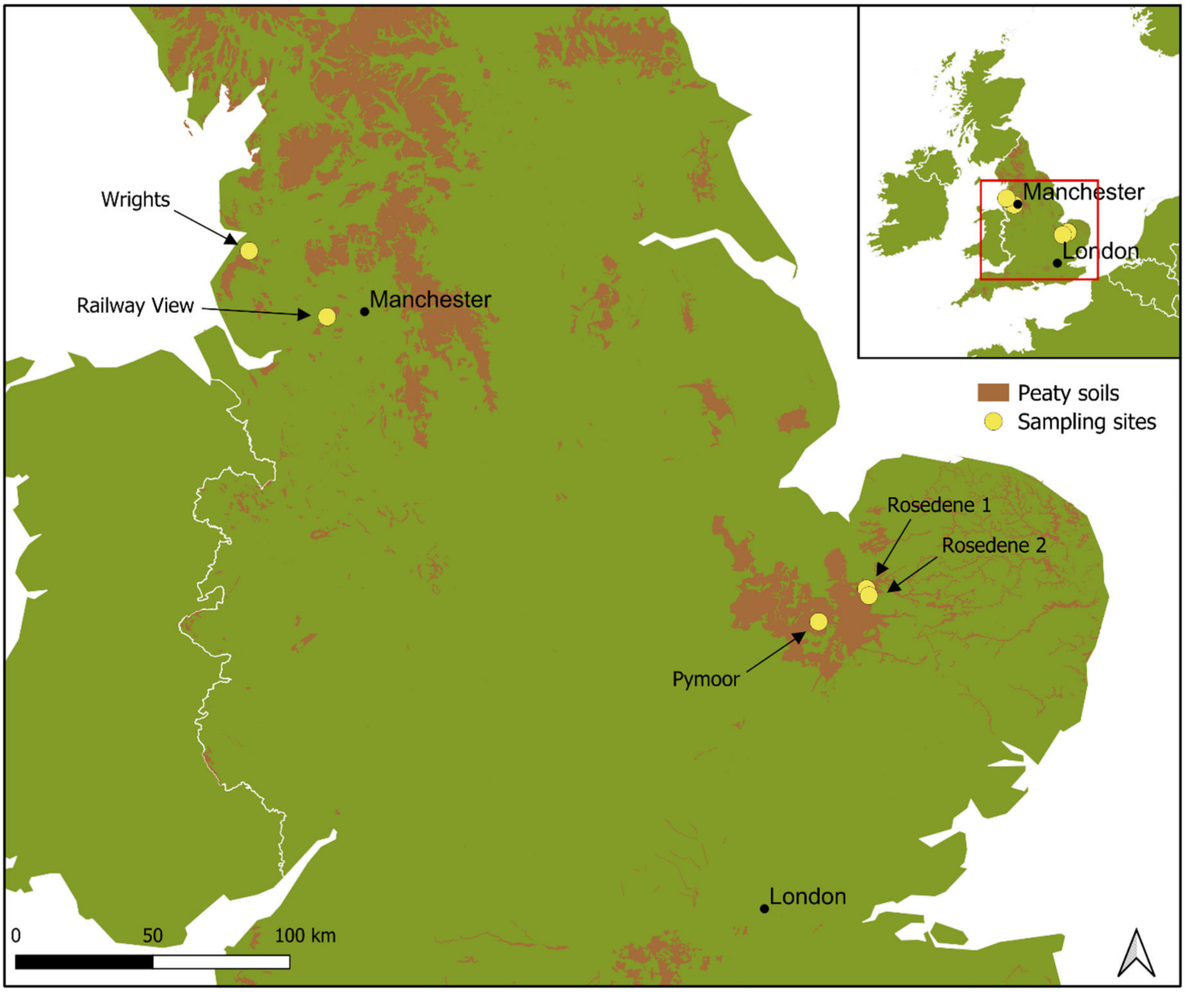

Figure 1. Mesocosm peat core sampling locations. England peaty soils extent from Natural England (2025).

Table 1. Name, region, peat type, land cover type, peat core sampling date, and water table depth details of the lowland peatlands used for the mesocosm experiment. Water table depths (reported in m below ground level) are on the sampling date at dipwells in the same field, as close as possible to the mesocosm peat core sampling points.

Site	Region	Latitude, longitude	Peat type	Land cover	Peat thickness (m)	Sampling date	Water table depth (m)
Pymoor	East Anglia	52.44328, 0.2023	Fen	Arable (cereals)	<1	4th July	0.7
Railway View	Manchester Mosses	53.46342, −2.44975	Raised bog	Grassland	1.5	14th August	0.8
Rosedene1	East Anglia	52.54603, 0.46469	Fen	Arable (horticulture)	>1	10th July	0.3
Rosedene2	East Anglia	52.52369, 0.47666	Fen	Arable (horticulture)	>1	11th July	0.8
Wrights	Lancashire Mosses	53.67711, −2.88331	Raised bog	Arable (horticulture)	0.5	24th July	0.7

most feasible water source for the water table manipulations (see Supplementary Text 1 for more details). Briefly, although site-specific river or ditch water would have allowed for a more realistic experiment, this presented a major logistical challenge, as well as limited replicability due to the spatiotemporal variability in water chemistry.

All cores were initially kept under conventional drainage (water table 0.1 m above the bottom of the peat core), mimicking their original state. To assess leaching of contaminants into surface waters under fully drained conditions, we first collected 125 mL pre-treatment surface water samples (n = 60) directly from the mesocosm containers after peat cores had been under these pre-treatment conditions for 1 to 6 weeks (due to extraction occurring on different dates—mean daily temperature during this period was 17.9 °C). Subsequently, mesh-covered dipwells for pore water sampling were installed using a narrow-gauge auger centrally in all mesocosms to allow entry of pore water 0.1 m from the bottom of the mesocosm through a ring of 0.6 cm diameter holes. Pore water samples were extracted from the dipwells using either a small petrol

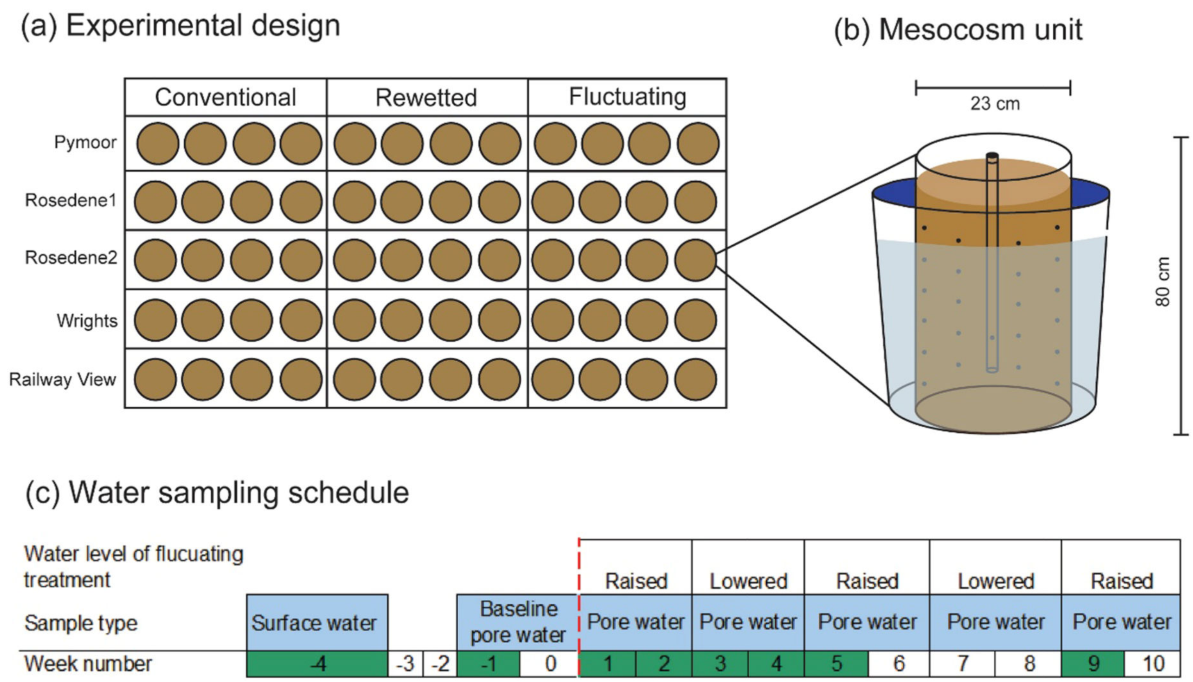

Figure 2. (a) Experimental design of the mesocosm experiment depicting cores collected from five sites, and split into replicated rewetted, conventional drainage and fluctuating water table treatments ($n = 4$ per site). (b) A simplification of the mesocosm unit consisting of a dipwell within the peat core within a PVC pipe, placed in an 80-L container. (c) The water sampling and water level treatment schedule, which denotes: the sampling weeks (green background); initiation of dividing the mesocosms into conventional drainage, rewetted and fluctuating treatments (after the red vertical line); and time periods when the fluctuating treatment was lowered to conventional levels (0.6 m below peat surface) and raised to rewetted levels (0.1 m below peat surface).

syphon or a point source bailer (Solinst, Canada), representing a cumulative 125 mL sample since the previous collection. Baseline pore water samples ($n = 60$) from the dipwells were collected on the week of 9 September 2024 (assigned as water table treatment week −1), prior to the experimental phase of the project commencing on the week of 23 September 2024 (meaning that all cores had a stabilisation period in the mesocosm array under drained conditions for at least nine weeks). (Figure 2c). Water table treatments were then imposed on the cores, and mesocosm pore water was sampled from the dipwells at 1, 2, 3, 4, 5 and 9 weeks after the start, for a total of $n = 360$ samples over an experimental duration of 9 weeks (excluding the baseline period).

2.2. Laboratory analyses

Laboratory analyses were conducted in the Ecosystems Laboratories at the Open University, England. Electrical conductivity (EC) and pH were measured on unfiltered samples using an Orion VERSA STAR multiparameter metre (Thermo Scientific, USA). Dissolved organic carbon (DOC) was measured as non-purgeable organic carbon on a total organic carbon analyser (Shimadzu TOC-L, Shimadzu, Japan). Inductively coupled plasma optical emission spectroscopy (ICP-OES) and SPS 4 autosampler (Agilent Technologies, USA) were used to measure concentrations of: total phosphorous (TP), iron (Fe), manganese (Mn), aluminium (Al), copper (Cu), lead (Pb), arsenic (As), cadmium (Cd), chromium (Cr), nickel (Ni), silicon (Si) and zinc (Zn). Samples were run undiluted after at least 30 minutes settling time to obtain a clear solution. Over-range samples of clear solution were re-run after dilution with 2% nitric acid (HNO_3). Anions fluoride (F^-), chloride (Cl^-), nitrate (NO_3^-), nitrite (NO_2^-), phosphate (PO_4^{3-}) and sulphate (SO_4^{2-}) as well as cations ammonium (NH_4^+), sodium (Na), potassium (K), calcium (Ca), magnesium (Mg) and lithium (Li) were analysed using ion chromatography on filtered samples (0.45 µm) (930 Compact IC Flex and 889 IC Sample centre, Metrohm, Switzerland). Over-range samples were re-run after dilution with 18.2 MΩ·cm water. Due to logistical limitations, several determinands were only measured in surface water, and not pore water samples: pH, EC, DOC, *P*, Si, Al, As, Cd, Cr, Cu, Fe, Mn, Ni, Pb and Zn. For more details on these analyses, including the determination of the lower limit of detection (LLD) and limit of quantification (LoQ) for each determinand, see Supplementary Text 2 and Table S2 for a list of the LLD and LoQ values for each determinand.

2.3. Statistical analysis

We used the non-parametric Kruskal–Wallis test to determine significant differences between sites for the pre-treatment determinands. To summarise the pre-treatment water chemistry data, we used principal components analysis (PCA) on scaled and centred variables to visualise differences and similarities between the sites. For the PCA, we kept the first two principal components, based on a visual inspection of the scree plot using the broken-stick method (Figure S4, Table S3), which together explained 55% of the variance in the data. For As and Pb all concentrations (and the majority of Li and Cd) were <LLD, and therefore results are not plotted or statistically analysed. Otherwise, concentrations below the LLD were treated as LLD*0.5, in line with best practices used in other similar studies (Finckh et al., 2022; Holman et al., 2010). To evaluate the individual and (two- and three-way) interactive effects of site, water table treatment and week on the pore water quality variables (excluding the baseline period), we used linear mixed-effects models (LMMs), modelling each variable separately using the *lmer* function from the lme4 package (Bates et al., 2015). We included mesocosm ID as the random effect to account for repeated measures, for the model structure: response variable ~ site * treatment * week + (1| mesocosm ID). We examined model residuals and Q-Q plots to assess homoscedasticity and normality; applying log-transformation to response variables to meet model assumptions, which was necessary in all cases except Cl^- and F^-. Statistical reporting of the LMM results in text includes the beta coefficient (β), representing the expected difference in the response relative to a reference level for categorical predictors (site and treatment) or for a one-unit change in numerical predictors (week), and the significance (p-value) of this effect. The fixed effects were tested using Type III analysis of variance (ANOVA) using Satterthwaite's approximation for degrees of freedom using *anova* from the lmerTest package (Kuznetsova et al., 2017). The ANOVA results are reported in text with their F-value and p-value. Pairwise comparisons were performed using estimated marginal means using the emmeans package (Lenth, 2025). We used Spearman correlations (R_S) to test for significant monotonic relationships between water quality variables, visualised as a correlation matrix using the corrplot package (Wei & Simko, 2024). To account for multiple comparisons in the correlation matrix, p-values were adjusted using a Benjamini–Hochberg correction. Means are reported ± one standard deviation (SD) throughout. All differences were considered significant when $p < 0.05$.

3. Results

In 3.1 we present and discuss the pre-treatment mesocosm surface water samples, which were collected before the start of the experiment. We use this pre-treatment data simulating drained conditions to (1) infer how effectively peat retains chemicals from the added source water or releases chemicals sequestered within the peat, and (2) compare biogeochemistry between sites. From 3.2 onwards we present and discuss the pore water data during the baseline and experimental phase of the project.

3.1. Pre-treatment surface water chemistry

Pre-treatment surface water Cl^-, F^-, K, Mg, NO_2^- and SO_4^{2-} concentrations were generally higher than tap water concentrations (Figure 3). Ca concentrations at both bog sites (grassland Railway View and arable Wrights Farm) were lower than tap water concentrations, while the two arable Rosedene Farm fen sites were approximately equivalent to tap water, and Pymoor Farm Ca concentrations were three times higher than the tap water (Figure 3a). For NH_4^+, surface water concentrations for all sites except for railway view were below tap water concentrations (Figure 3g). Variations in water concentrations of NO_3^- generally spanned the tap water concentration (Figure 3i). PO_4^{3-} concentrations were lower in surface water than tap water for all sites (Figure 3j).

There were differences in water quality in the pre-treatment surface water samples between the five field sites (Figure 4). Most water quality parameters had significant differences between sites (Kruskal–Wallis test,

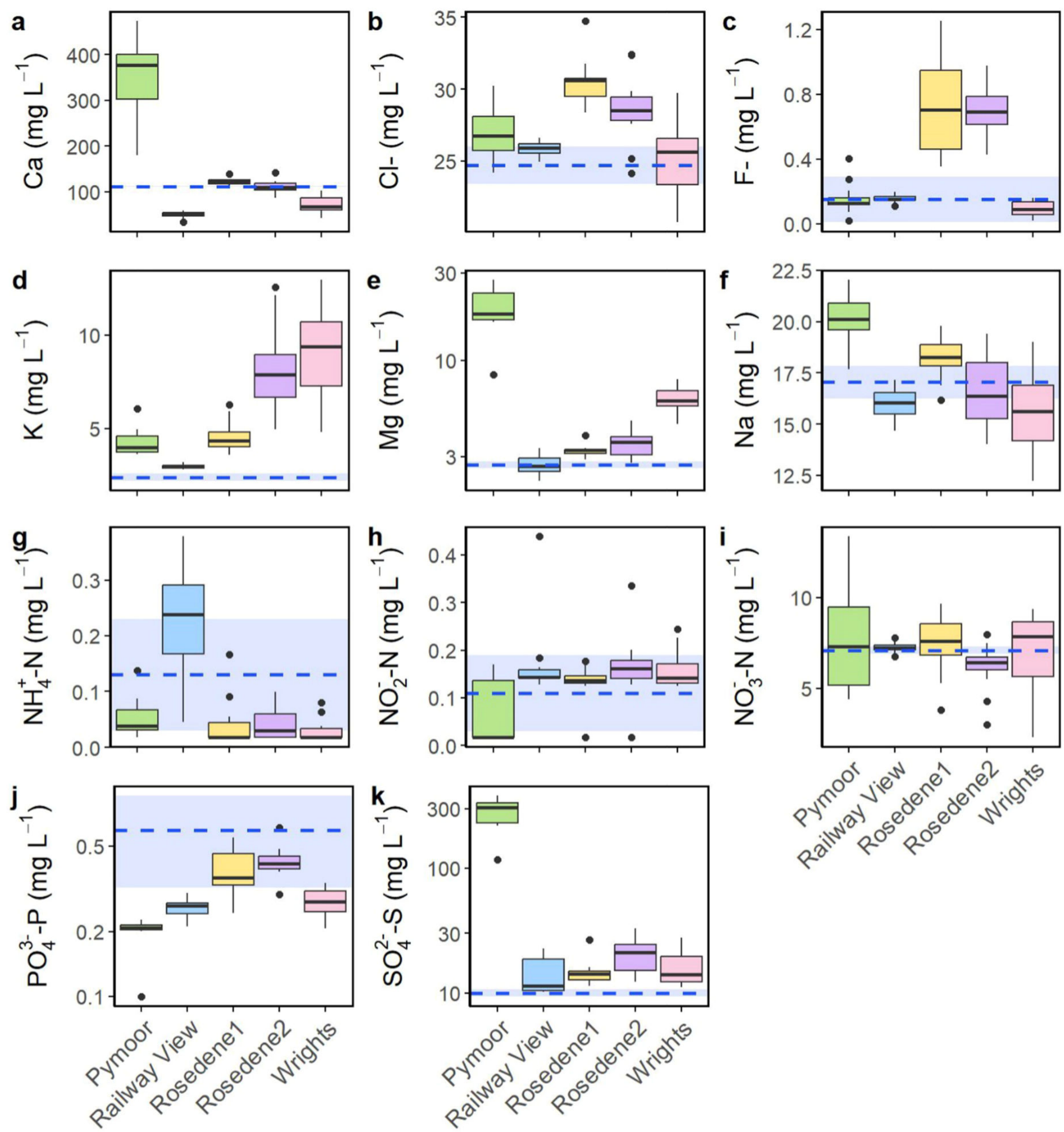

Figure 3. Mesocosm pre-treatment surface water determinands grouped by site with mean tap water concentrations for reference (horizontal dashed line, with standard deviation as pale blue band). Each box represents 12 samples. Note that some variables are plotted on a logarithmic scale for clarity. Kruskal–Wallis tests show significant differences between sites at $p < 0.001$, with the exceptions of NO_3^- ($p = 0.17$) and NO_2^- ($p = 0.02$). Box plots display the median, 25th and 75th percentiles, and whiskers display values at the 1.5× interquartile range. See Table S4 for tap water values.

$p < 0.05$), with the exception of NO_3^- ($p = 0.17$). The first two PCA axes together explained just over half (53.6%) of the total variance in the data (Figure 4). The first PCA axis (35.2%) shows a gradient from high concentrations of weathering-source ions (SO_4^{2-}, Mg, Ca, etc.) to high concentrations of biologically cycled ions (PO_4^{3-}, NO_2^-). The second axis (18.4%) shows a gradient from waters high in DOC and metals (Ni, Cr, Fe, Al, etc.) to high salt and nutrient concentrations (Na, Cl^-, NO_3^-).

Pymoor farm surface water had a unique water quality profile when compared to the other sites, driven by higher concentrations of SO_4^{2-}, EC, Ca, Mg and Mn than the other sites (Figure 3, Figure 4, Figure S5). There were significant positive correlations between SO_4^{2-}, EC, Ca and Mg when all sites were considered together (Figure S6).

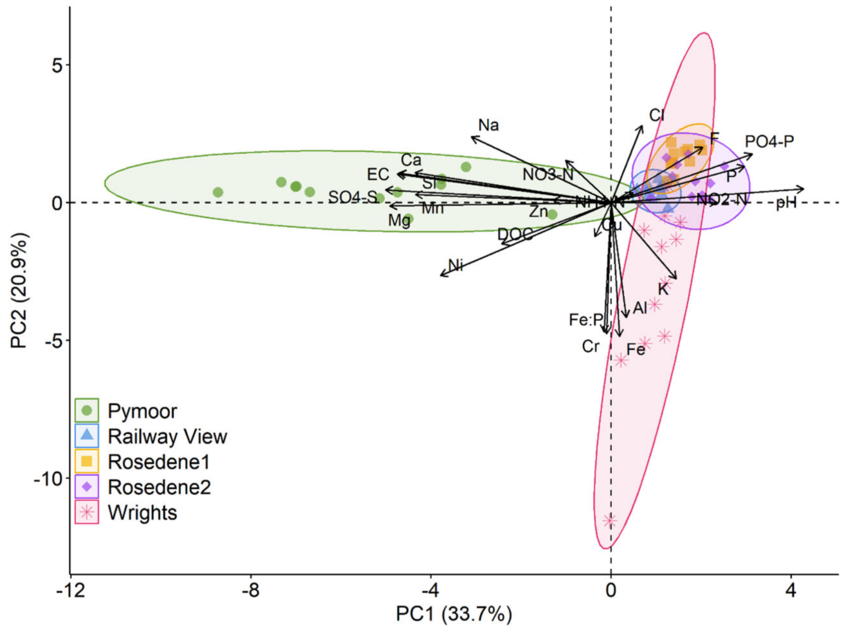

Figure 4. Principal components analysis of water quality variability across pre-treatment surface water samples, grouped by site.

The two Rosedene farm sites had similar water quality profiles, with the Rosedene1 cluster falling inside the Rosedene2 cluster (Figure 3). They were characterised by higher PO_4^{3-} and F^- when compared to the other sites (Figures 2 and 3). There were some differences between the Rosedene farm sites, notably, K and Cu concentrations were higher at Rosedene2 than Rosedene1 (Figure 3, Figure S5).

Railway view mesocosm surface water chemistry showed a distinct and tight cluster at the centre of the PCA plot, indicative of little variation and chemically intermediate conditions (Figure 4). Railway View, a bog grassland, had lowest concentrations of Ca (50 mg L^{-1}) and Mg (2.7 mg L^{-1}), and lowest EC (389 µs cm^{-1}), as well as the highest NH_4^+ concentrations (0.23 *N* mg L^{-1}) (Figure 3, Figure S5).

Wrights Farm (bog, arable) surface water had the highest concentrations of metals Al (1426 µg L^{-1} vs a mean of 88 µg L^{-1} for the other sites), Fe (1230 vs 55 µg L^{-1} for the other sites) and Cr (6.2 vs 1.4 µg L^{-1} for the other sites) as well as the highest Fe:P (4.25 vs. 0.33 for the other sites) (Figure S5).

3.2. Effects of rewetting on pore water chemistry

3.2.1. Nitrogen

NO_3^- concentrations declined significantly over time across all treatments ($\beta = -0.21$, $p < 0.001$). Mean NO_3^- concentrations declined from 8.5 ± 7.5 *N* mg L^{-1} in the baseline period converging to low values with low variability in week 9 (1.4 ± 1.5 N mg L^{-1}) (Figure 5a). NO_3^- concentrations significantly differed between sites ($F = 10.8$, $p < 0.0001$) with the highest concentrations measured at Pymoor Farm. NO_3^- concentrations also significantly differed between treatments ($F = 6.5$, $p = 0.002$). Estimated marginal mean comparisons revealed that conventionally drained cores had higher NO_3^- concentrations than both fluctuating ($p = 0.002$) and rewetted ($p = 0.001$) cores. There was a significant interaction between site and the rewetted treatment at Pymoor ($\beta = 9.8$, $p = 0.03$), indicating that rewetted treatment NO_3^- concentrations were higher at Pymoor compared to other sites. Additionally, a significant three-way interaction between site, treatment (rewetted) and week at Pymoor ($\beta = -0.24$, $p = 0.02$) indicates that NO_3^- concentrations declined more steeply over time at Pymoor than for the rewetted treatment at other sites.

Overall, NH_4^+ concentrations slightly increased over time after the baseline period ($\beta = 0.21$, $p = 0.02$) (Figure 5b). NH_4^+ concentrations did not significantly differ between sites ($F = 1.1$, $p = 0.37$). NH_4^+ concentrations also did not significantly differ between treatments ($F = 2.7$, $p = 0.07$).

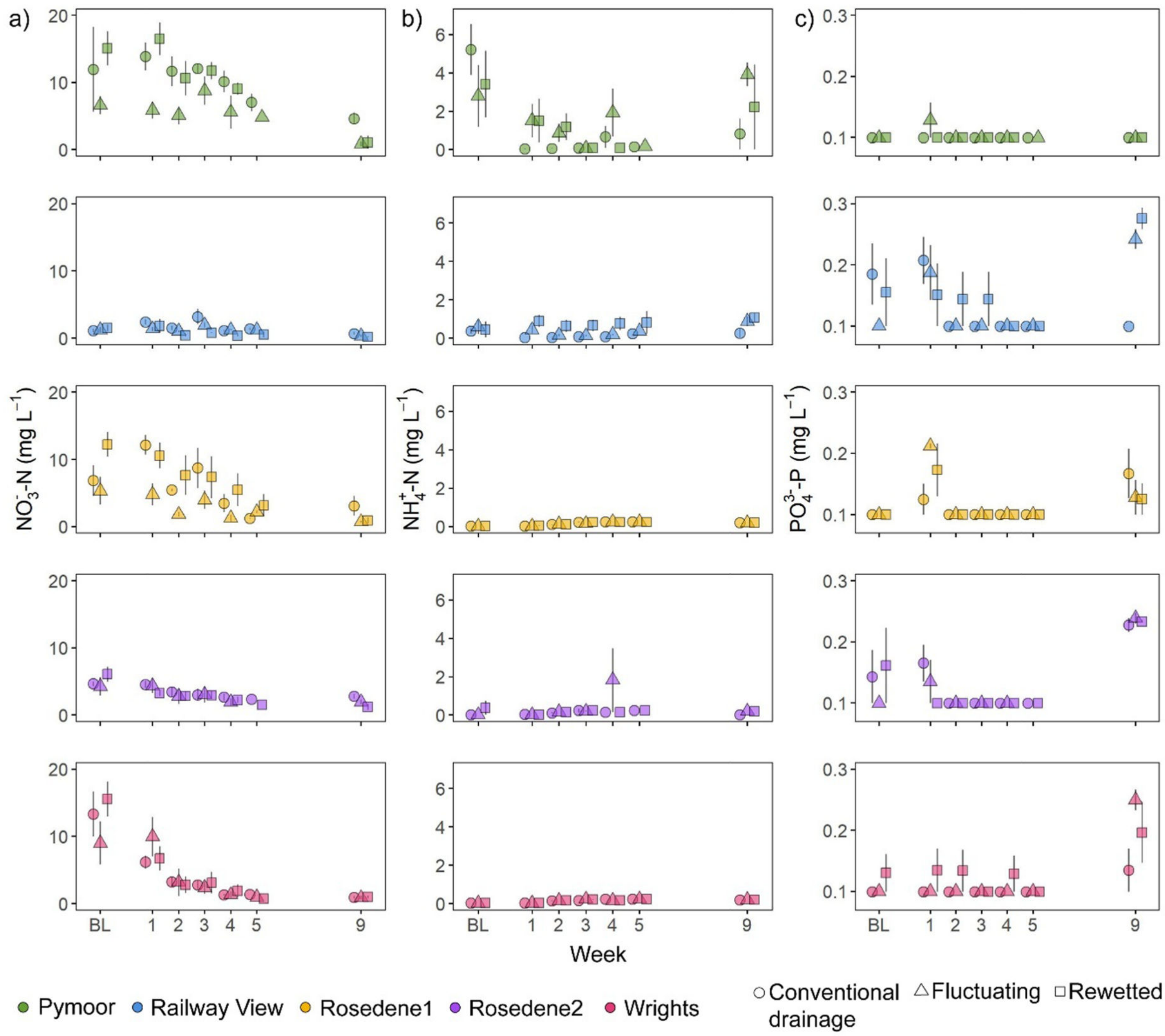

Figure 5. Pore water a) nitrate (NO_3^-), b) ammonium (NH_4^+) and c) phosphate (PO_4^{3-}) concentrations over the duration of the experiment, by week from the baseline period ('BL') in conventional drainage, rewetted and fluctuating water table treatments. See Figure 2 for the water table fluctuation schedule.

3.2.2. Phosphate

PO_4^{3-} concentrations did not have a significant temporal trend ($\beta = 0.04$, $p = 0.08$). Although concentrations across most sites (except Pymoor) and treatments were elevated at week 9. PO_4^{3-} concentrations significantly differed between sites ($F = 13.2$, $p < 0.0001$) and treatments ($F = 3.3$, $p = 0.04$). However, the effect of treatment was not strong enough to be observed in pairwise comparisons. The highest PO_4^{3-} concentrations were measured in cores from Railway View (bog, grassland) (Figure 5c).

3.2.3. Other ions (K, SO_4^{2-}, Ca, Mg, Na, Cl^-, F^-)

For the remaining determinands, concentrations varied widely within treatments (Figure S7), obscuring any consistent treatment-level patterns. Their concentrations all differed significantly between sites ($p < 0.001$) but not treatments, with low within-site variation (Figure 6). The only exception to this trend was F^-, which had a significant overall effect of treatment ($p = 0.03$), although not strong enough to be observed in the pairwise comparisons. Concentrations of the other ions were generally consistent over time with no significant temporal effects. Wrights Farm (bog, arable) had the highest concentrations of K (Figure 6a) which declined slightly after the baseline period. Rosedene2 had the second-highest mean K concentrations after Wrights Farm, and these dropped at week 3 in the conventionally drained mesocosms but then increased again so that differences with rewetted and fluctuating treatments were minor (Figure S8a). The remaining sites had K concentrations < 10 mg L^{-1}, with minimal variation, for the duration of the experiment. Pymoor Farm had

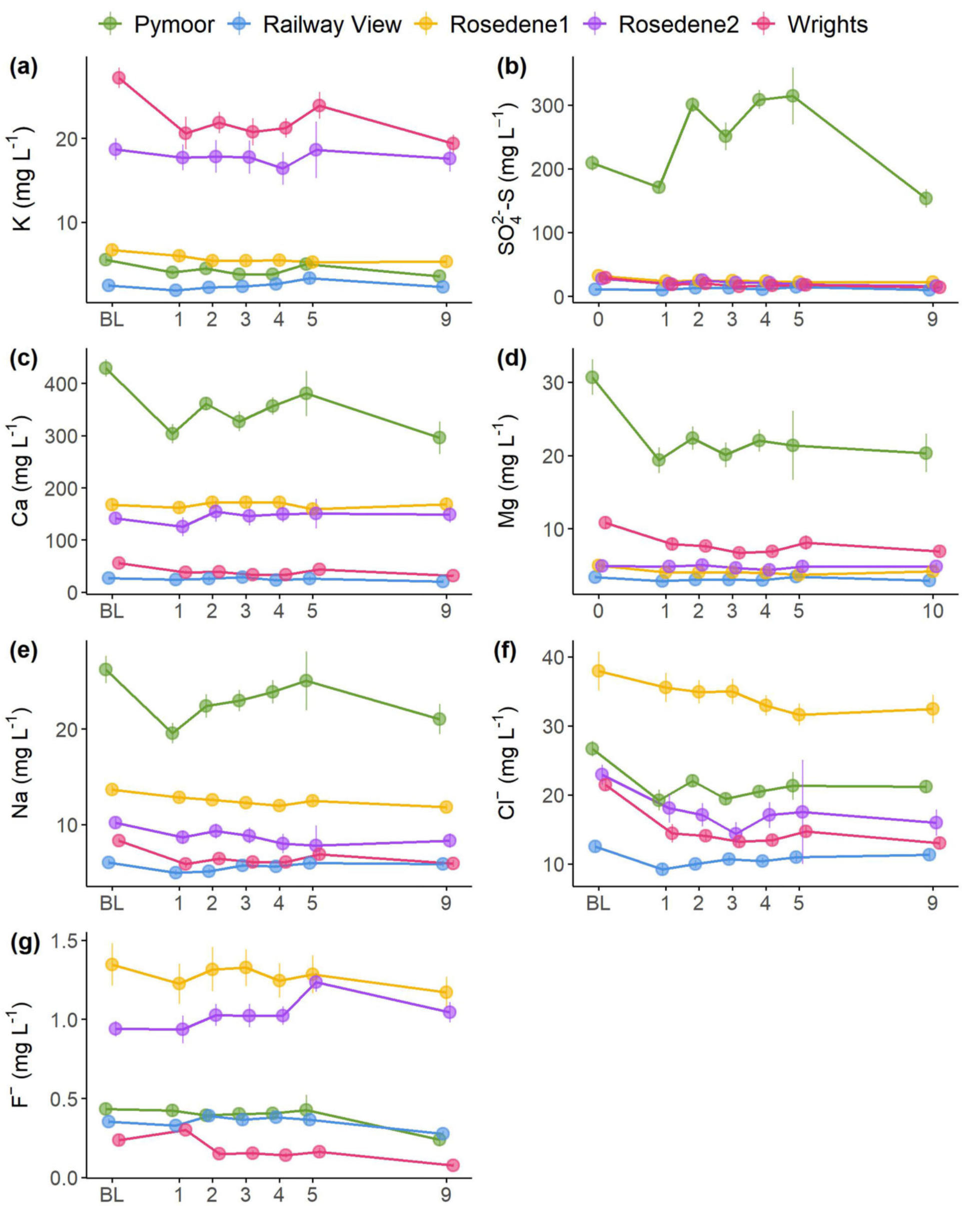

Figure 6. Mesocosm water quality parameters over the duration of the experiment, grouped by site for all treatments, by week from the baseline period (week 0). The lines connecting the points (mean ± SE of each site) are to aid in visual interpretation and do not indicate continuous measurements. Note that high values of SO_4^{2-} (>20 S mg L^{-1}) were above the calibration range and estimated using linear extrapolation.

the highest concentrations of SO_4^{2-}, Ca, Mg and Na, and these followed a small decline after the baseline period (Figure 6). Rosedene1 had the highest concentrations of Cl^- and F^-, and these were relatively consistent over time (Figure 6, Figure S8).

4. Discussion

4.1. Pre-treatment surface water chemistry

4.1.1. Pre-treatment leaching vs retention

During the pre-treatment phase, all mesocosms were kept under conventional drainage, with the water table 0.1 m above the bottom of the peat core (approximately 0.6 m below the peat surface). These pre-treatment surface water samples thus represent a product of exchange processes between the surface water and the lower layers of peat. If concentrations of determinands were higher in the pre-treatment surface water than in the source water that was added to the mesocosm, we presumed that net release from the lower peat into the surface water had occurred—this was the case for Cl^-, F^-, K, Mg, NO_2^- and SO_4^{2-}. Conversely, if surface water concentrations were lower than tap water, we presumed that retention within the peat had occurred—this was the case for PO_4^{3-}. These patterns were generally consistent across sites, except for some cases. For example, the Pymoor Farm cores leached notable amounts of Ca compared to other sites. This Ca could be of groundwater origin, or perhaps more likely from the mineral soil fraction as this site has thin, mineral-mixed 'wasted' peat overlying calcareous geology (British Geological Survey, 2019). Several of the substances that appeared to be mobilised from the cores during the pre-treatment phase (e.g. SO_4^{2-}, Ca, Mg) were positively correlated. Furthermore, some are derived from mineral sources. Therefore, we assumed they were retained in the *in situ* mineral fraction of peat under field conditions, and then subsequently released into the surface water in the mesocosm array when the comparatively low-Ca water, was added (Hill & Siegel, 1991).

For all pre-treatment surface water samples, concentrations of metals As and Pb were below the LLD, and only some Li and Cd samples were slightly above their LLD. A wide body of research shows that Pb, as well as other metals including Cu and Zn are largely immobile in ombrotrophic peat (Novak et al., 2011), though metal dynamics in minerotrophic fens differ due to groundwater as an additional metal source. Nevertheless, generally low metal concentrations observed across all pre-treatment surface water samples suggest limited mobility regardless of peat type.

4.1.2. Pre-treatment site differences

Pre-treatment surface water chemistry revealed notable differences between sites based on their peatland type, site histories and current management. The two Rosedene Farm sites (both arable, ~2.5 km apart) had similar water quality profiles, characterised by higher PO_4^{3-} and F^- when compared to the other sites. PO_4^{3-} and F^- were positively correlated, and this is feasibly due to fertiliser inputs at these sites, given that PO_4^{3-} fertilisers are important sources of anthropogenic F^-, sourced from impurities in phosphorite rock, the raw material used in PO_4^{3-} fertiliser production (Li and Ma, 2023). There were, however, some differences between the Rosedene Farm sites, notably, K and Cu concentrations were higher at Rosedene2 than Rosedene1. Although K can be derived from groundwater, it was not correlated to Ca across our full dataset, suggesting site-specific inputs, e.g. differing fertiliser or cropping regimes between the two sites. Likewise, the higher Cu could be from fertiliser applications (Nicholson et al., 2006).

Railway view, mesocosm surface water had the lowest concentrations of Ca, Mg and EC, as would be expected for an ombrotrophic peatland with limited groundwater inputs (Bleuten et al., 2006). In contrast, Pymoor Farm surface water had a unique water quality profile high in weathering-sourced ions (e.g. SO_4^{2-}, Mg, Ca). This may be reflective of the thin, mineral-mixed 'wasted' peat compared to deeper peat at most other sites. Railway View also had the highest NH_4^+ concentrations. This could arise because a higher in-field water table and limited disturbance compared to the arable sites (i.e. ploughing) could promote anaerobic conditions, thus suppressing nitrification and promoting NH_4^+ accumulation (van der Laan et al., 2024). Alternatively, historical land use, such as direct deposits of sewage sludge and urban wastes, which occurred directly onto the Manchester Mosses during the 19th century, could play a role (Breward, 2003).

Wrights Farm surface water had the highest concentrations of metals Al, Fe and Cr. These metals are possibly from fertiliser inputs (Nicholson et al., 2006), or due to agricultural drainage which will have exposed deeper peat and mineral soil layers, resulting in the release of Fe and Al from organic matter, or clays and iron oxides (Jansen et al., 2004; Zak et al., 2010). Along with the railway view, Wrights Farm is situated in the north-west of England near Greater Manchester, which was a major hub of manufacturing during the Industrial Revolution (Douglas et al., 2002). Therefore, elevated metal concentrations may have also come from regional historic coal combustion and metal smelting (Fletcher & Ryan, 2018; Osborne et al., 2024).

4.2. Effects of water table manipulations on pore water chemistry

4.2.1. Pore water nitrogen

Peatland drainage, rewetting and fluctuating water tables will all affect N cycling. Lower water tables may accelerate N cycling because N mineralisation in peatlands is more effective under aerobic than anaerobic conditions (Laine et al., 2013; Olde Venterink et al., 2002). We observed an overall decline in NO_3^- and an increase in NH_4^+ over the duration of the experiment. NO_2^- concentrations are not frequently measured in peatland rewetting studies, and average values are generally close to zero (Howson et al., 2021), as we also observed in our study. Declines in NO_3^- following rewetting have been observed elsewhere and attributed to saturated conditions suppressing nitrification in both long-term field and mesocosm studies (Frank et al., 2014; Kaila et al., 2016; Nachtigall & Giani, 2022; van der Laan et al., 2024). Although in our study, the observed NO_3^- decline occurred across all treatments, not just rewetted ones, suggesting a time-dependent process rather than just water table effects. However, the observed decline in NO_3^- concentrations over time significantly steeper at the rewetted treatments at Pymoor, as indicated by the significant three-way effect. Initial NO_3^- concentrations were also the highest at Pymoor, thus, this trend likely indicates a rapid depletion of a relatively large NO_3^- pool in response to rewetting. Saturated conditions favour denitrification, which is a facultative anaerobic process that converts NO_3^- and NO_2^- to nitrous oxide and di-nitrogen (Pinto et al., 2021). Therefore, we might expect to see not only the lowest production of NO_3^- (due to limited mineralisation) but also the highest removal of NO_3^- (through denitrification) in the rewetted and fluctuating mesocosms. Our results aligned with this expectation only at the Pymoor site, where we observed lower NO_3^- in the fluctuating treatment compared to the conventional drainage treatment. It is unclear why the fully rewetted treatment did not also have lower NO_3^-compared to the drained treatment. It is possible that the alternating wet/dry conditions in the fluctuating treatment may have promoted denitrification or limited mineralisation more intensely compared to the persistently rewetted mesocosms. This has been observed in streambed sediments, where wet-dry cycles enhanced *N* losses due to a coupling of denitrification and nitrification (Pinto et al., 2021).

We observed a small increase in NH_4^+ over the duration of the experiment, and no significant effects of treatments. In other studies, peatland rewetting has resulted in increased pore water NH_4^+, attributed to the low levels of nitrification in saturated peat (van der Laan et al., 2024; Zak & Gelbrecht, 2007; Zak et al., 2010). In a rewetting mesocosm experiment using peat cores from a Dutch grassland fen, NH_4^+ concentrations increased following rewetting, attributed to anaerobic conditions suppressing nitrification and promoting NH_4^+ accumulation (van der Laan et al., 2024). In a mesocosm experiment simulating water table fluctuations using cores from a peatland in southern Finland, NH_4^+ concentrations increased following rewetting in pristine peat mesocosms but not in the mesocosms using drained peat (Laine et al., 2013). This was attributed to the pristine peat retaining a larger pool of undecomposed organic *N* that becomes rapidly mineralised after temporary aeration, while long-term drainage has already depleted labile *N* substrates in the drained peat. The *N* was then released into the pore water under a fluctuating water table as mineralisation of organic matter increased under a low water table and was flushed out under a high water table. Surprisingly, we found that Pymoor Farm had the highest NH_4^+ concentrations, notably in the rewetted and fluctuating mesocosms, despite having very shallow, degraded and mineral-mixed peat. Therefore, these elevated pore water NH_4^+ concentrations may be due to field fertiliser inputs, although this is speculative without additional evidence.

4.2.2. Pore water phosphate

PO_4^{3-} mobilisation is a recognised problem resulting from rewetting formerly drained agricultural peatlands (Van De Riet et al., 2013; van der Laan et al., 2024; Zak et al., 2010). This is because *P* from fertilisers accumulates in the surface peat layers where it binds to redox-sensitive compounds, particularly Fe(III) hydroxides (as well as non-redox-sensitive Al). As redox potential decreases upon rewetting, this *P* becomes mobilised via reduction of Fe(III) (van der Laan et al., 2024). We observed a general increase in PO_4^{3-} in week 9, although there was no significant effect of treatment. The drainage/rewetting responses of PO_4^{3-} are not straightforward, because they are further modulated by Fe and Al, as well as pH, (Forsmann & Kjaergaard, 2014; Zak et al., 2010), and thus very site-specific. The tendency of high PO_4^{3-} in rewetting and fluctuating railway views could be driven by low Fe:P molar ratios. Fe:P molar ratios have been shown to negatively correlate with *P* release in rewetted fens, and a ratio > 3 in pore water indicates strong *P* retention capacity (Zak et al., 2010). Thus, more *P* is retained in the Wrights Farm (and Pymoor Farm to a lesser extent) cores via sorption to Fe when compared to the Rosedene Farm and Railway View sites, as indicated in our surface water samples (Zak et al., 2010).

4.2.3. Other pore water chemistry

The remaining determinands (K, SO_4^{2-}, Ca, Mg, Na, Cl^-, F^-) did not have any significant temporal trends (besides a small decline in F^-), nor were there any significant differences between treatments. The principal differences were between the sites. As such, we can conclude that the drainage vs. rewetting methods generally had less of an effect on most pore water chemistry determinands—outside of *N* and P—than peat conditions, at least in the short term. There was only a marginally significant effect of treatment on F^- concentrations, which were relatively low (0.52 mg L^{-1}) across all sites, reflecting the low natural abundance of F^- in peatlands. There is very little, if any, research examining F^- concentrations following peatland rewetting. Rewetting of extracted bogs has led to post-rewetting declines (over 15 years) in Cl^- at some, but not all, sites (Lundin et al., 2017). If surface water high in SO_4^{2-} (and NO_3^-) is used for rewetting, it can promote eutrophication, increase decomposition rates and trigger *P* mobilisation (van der Laan et al., 2024). However, in a mesocosm experiment comparing surface water with 10× more SO_4^{2-} and NO_3^- than groundwater, there was not an effect of water source on pore water SO_4 concentrations (van der Laan et al., 2024). K concentrations in pore water have been observed to increase following rewetting due to leaching and inputs of K-rich source waters (Koerselman et al., 1993; van der Laan et al., 2024).

5. Limitations

Mesocosm experiments allow for precise manipulation in a controlled environment, but present challenges for faithfully replicating field conditions. Given the nature of a nine-week rewetting simulation mesocosm experiment, there were certain concessions we had to accept as study limitations. Firstly, we acknowledge that the use of tap water additions, raises certain concerns due to its unique water quality profile compared to typical rewetting sources (ditches, rivers, groundwater), namely with high concentrations of inorganic *N* and *P* (Table S1). Although tap water PO_4^- concentrations were within the range of mean concentrations in water samples taken from ditches draining some of fields that mesocosm cores were extracted from (Supplementary Text 1). Furthermore, before application to the mesocosms the tap water was allowed to sit in containers to allow Cl (used during the water treatment process) to degas.

The lack of an observable overall treatment effect on most of our determinands could be an artefact of the pore water sampling depth. Sampling pore water from a fixed point 0.1 m above the mesocosm base allowed for pore water sampling across all treatment mesocosms and was preferable to aggregating multiple depths with uncertain relative contributions. However, water chemistry responses to rewetting can differ between shallow and deeper pore water (Gaffney et al., 2018), with shallower pore water chemistry often experiencing a more pronounced response to raised water tables than deeper pore water. Future mesocosm studies could consider multi-port samplers to characterise the vertical variability in pore water chemistry, albeit at an increased sampling cost.

A further limitation of our study that may have introduced variation is that all cores were not collected on the same day due to logistical constraints and therefore spent differing amount of time ex situ prior to the experiment. To minimise the potential effects of this lag time, after collection, all cores were maintained under a low water table that closely matched their in situ hydrological conditions.

Finally, our experimental phase was limited to nine weeks (due to logistical and budgetary constraints), therefore, our conclusions should be considered as short-term rewetting impacts, and long-term field studies (e.g. Frank et al., 2014; Gaffney et al., 2018; Lundin et al., 2017; Nachtigall & Giani, 2022) should be consulted in parallel.

6. Conclusions and implications

Overall, our findings have implications for peatland management and environmental regulation. In our study context, rewetting posed a relatively limited, short-term leaching risk for most measured nutrients and ions. However, between-site variation as a function of peat type and land-use history was high (e.g. site-specific peaks in metal concentrations), and pre-restoration assessment of both soil and water chemistry therefore remains important to develop effective site-specific rewetting strategies to minimise post-rewetting nutrient loading. Several determinands (Cl^-, F^-, K, Mg, NO_2^- and SO_4^{2-}) were presumed to be released into surface waters upon tap water addition. Our findings that NO_3^- consistently declined after re-wetting while NH_4^+ increased suggest that pre-restoration soil *N* concentrations should be taken into account, as well as *N* concentrations within the rewetting source water, which could lead to internal eutrophication. At Pymoor, the fluctuating and fully rewetted treatments had unique NO_3^- responses that merit further investigation regarding the impacts of wet-dry cycles on denitrification. PO_4^{3-} generally peaked during the last week of the experiment, with Fe:P dynamics potentially playing a role in the site-specific responses.

This information will help develop effective site-specific rewetting strategies to minimise nutrient loading following restoration. Overall, our results do not suggest that peatland re-wetting will lead to significant downstream biogeochemical impacts, and therefore that concerns about water quality should not act as a barrier to peatland re-wetting for climate mitigation or biodiversity.

Acknowledgements

We thank all the landowners who provided site access, permissions and site information, including Martin Hammond and Danny Sherlock (G S Shropshire & Sons), Kathryn Wright (Wright Farm Produce Limited), Sarah Taylor and Craig Taylor (Pymoor) and Steve Ormerod (Natural England). For the laboratory analyses, we acknowledge the Open University Ecosystems Lab Team led by Sonia Newman, including Aline Cecchetti and Sneha Ravji. We thank Rachael Armitage at UKCEH for providing Wallingford weather station data. We thank the Environment Agency for funding this research, and acknowledge Sian Davies, Douglas Bray and Iain Detrey for their input and guidance.

All authors. Author order, besides core authors TS, KB and MP, was assigned randomly.

Author contributions

CRediT: **Teresa Silverthorn:** Visualization, Writing – original draft, Writing – review & editing; **Kate Blake:** Investigation, Writing – review & editing; **Elya Monsen-Elvik:** Conceptualization, Funding acquisition, Writing – review & editing; **Tom Burton:** Investigation, Writing – review & editing; **Terhi Riutta:** Investigation, Writing – review & editing; **Ian Holman:** Conceptualization, Funding acquisition, Writing – review & editing; **Caroline Gurd:** Investigation, Writing – review & editing; **Angus McEwen:** Investigation, Writing – review & editing; **Chris Bell:** Conceptualization, Funding acquisition, Writing – review & editing; **Francesca Southon:** Investigation, Writing – review & editing; **Jasmine Hughes:** Investigation, Writing – review & editing; **Chris D. Evans:** Conceptualization, Funding acquisition, Writing – review & editing; **Mark Simpson:** Investigation, Writing – review & editing; **Ross Morrison:** Conceptualization, Funding acquisition, Writing – review & editing; **Graham Howell:** Investigation, Writing – review & editing; **Josh Oakley:** Investigation, Writing – review & editing; **Mike Peacock:** Conceptualization, Funding acquisition, Investigation, Writing – original draft, Writing – review & editing.

Disclosure statement

No potential conflict of interest was reported by the author(s).

Funding

This study was supported by UK Environment Agency.

ORCID

Teresa Silverthorn https://orcid.org/0000-0001-6152-6573
Terhi Riutta https://orcid.org/0000-0002-8308-5307
Ian Holman https://orcid.org/0000-0002-5263-7746
Chris D. Evans https://orcid.org/0000-0002-7052-354X
Ross Morrison https://orcid.org/0000-0002-1847-3127
Graham Howell https://orcid.org/0000-0002-5158-1891
Mike Peacock https://orcid.org/0000-0002-3086-2854

Data availability statement

The data that support the findings of this study are openly available in figshare at http://doi.org/10.6084/m9.figshare.31836970.

References

Ahmad, S., Liu, H., Alam, S., Günther, A., Jurasinski, G., & Lennartz, B. (2021). Meteorological controls on water table dynamics in fen peatlands depend on management regimes. *Frontiers in Earth Science*, *9*, 630469. https://doi.org/10.3389/feart.2021.630469

Andersen, R., Farrell, C., Graf, M., Muller, F., Calvar, E., Frankard, P., Caporn, S., & Anderson, P. (2017). An overview of the progress and challenges of peatland restoration in Western Europe. *Restoration Ecology*, *25*(2), 271–282. https://doi.org/10.1111/rec.12415

Bates, D., Mächler, M., Bolker, B., & Walker, S. (2015). Fitting linear mixed-effects models using lme4. *Journal of Statistical Software*, *67*(1), 1–48. https://doi.org/10.18637/jss.v067.i01

Bennett, L., Blannin, L., Cottey, R., Miles, J., Mitchell, J., Putman, S., Richmond, B., & Willis, D. (2025). Greenhouse gas inventories for England, Scotland, Wales and Northern Ireland: 1990–2023. Report for the Department for Energy Security and Net Zero, The Scottish Government, The Welsh Government and The Northern Ireland Department for Agriculture, Environment and Rural Affairs. https://naei.energysecurity.gov.uk/sites/default/files/2025-06/DA_GHG_Inventories_1990-2023.pdf

Berset, J. D., Kuehne, P., & Shotyk, W. (2001). Concentrations and distribution of some polychlorinated biphenyls (PCBs) and polycyclic aromatic hydrocarbons (PAHs) in an ombrotrophic peat bog profile of Switzerland. *Science of the Total Environment*, *267*(1–3), 67–85. https://doi.org/10.1016/S0048-9697(00)00763-4

Bleuten, W., Borren, W., Glaser, P. H., Tsuchihara, T., Lapshina, E. D., Mäkilä, M., Siegel, D., Joosten, H., & Wassen, M. J. (2006). Hydrological processes, nutrient flows and patterns of fens and bogs. In J. T. A. Verhoeven, B. Beltman, R. Bobbink, & D. F. Whigham (Eds.), *Wetlands and Natural Resource Management* (Vol. *190*, pp. 183–204). Berlin Heidelberg: Springer. https://doi.org/10.1007/978-3-540-33187-2_9

Blodau, C., & Moore, T. R. (2003). Experimental response of peatland carbon dynamics to a water table fluctuation. *Aquatic Sciences*, *65*(1), 47–62. https://doi.org/10.1007/s000270300004

Breen, B., Curtis, J., & Hynes, S. (2018). Water quality and recreational use of public waterways. *Journal of Environmental Economics and Policy*, *7*(1), 1–15. https://doi.org/10.1080/21606544.2017.1335241

Breward, N. (2003). Heavy-metal contaminated soils associated with drained fenland in lancashire, england, UK, revealed by BGS soil geochemical survey. *Applied Geochemistry*, *18*(11), 1663–1670. https://doi.org/10.1016/S0883-2927(03)00081-7

British Geological Survey. (2019). Geology Viewer (Version 1.9.0) [Data set]. https://geologyviewer.bgs.ac.uk

Broder, T., & Biester, H. (2015). Hydrologic controls on DOC, as and pb export from a polluted peatland – the importance of heavy rain events, antecedent moisture conditions and hydrological connectivity. *Biogeosciences*, *12*(15), 4651–4664. https://doi.org/10.5194/bg-12-4651-2015

Chimner, R. A., Cooper, D. J., Wurster, F. C., & Rochefort, L. (2017). An overview of peatland restoration in North America: Where are we after 25 years? *Restoration Ecology*, *25*(2), 283–292. https://doi.org/10.1111/rec.12434

Comber, S., Lunt, P., Taylor, M., Underwood, N., Crocker, R., & Schindler, R. (2023). Restoration management of phosphorus pollution on lowland fen peatlands: A data evidence review from the somerset levels and moors. *Agricultural Water Management*, *287*, 108419. https://doi.org/10.1016/j.agwat.2023.108419

D'Acunha, B., Evans, C. D., Bodo, A., Cooper, H., Crabtree, D. E., Cumming, A., Rhymes, J. M., Rylett, D., Artz, R. R. E., & Morrison, R. (2026). Drained agricultural peatlands as persistent carbon sources: Implications for carbon and water use intensity in food production. *Global Change Biology*, *32*(3). e70796. https://doi.org/10.1111/gcb.70796

Department for Environment, Food & Rural Affairs [DEFRA]. (2021). England Peat Action Plan. https://assets.publishing.service.gov.uk/media/6116353fe90e07054eb85d8b/england-peat-action-plan.pdf

Douglas, I., Hodgson, R., & Lawson, N. (2002). Industry, environment and health through 200 years in manchester. *Ecological Economics*, *41*(2), 235–255. https://doi.org/10.1016/S0921-8009(02)00029-0

Evans, C. D., Morrison, R., Burden, A., Williamson, J., Baird, A., Brown, E., Callaghan, N., Chapman, P., Cumming, A., & Dean, H. (2016). Lowland peatland systems in England and Wales–evaluating greenhouse gas fluxes and carbon balances (Final report to Defra on Project SP1210). Centre for Ecology and Hydrology. https://oro.open.ac.uk/50635/

Evans, C. D., Artz, R., Moxley, J., Smyth, M.-A., Taylor, E., Archer, N., Burden, A., Williamson, J., Donnelly, D., Thomson, A., Buys, G., Malcolm, H., Wilson, D., Renou-Wilson, F., & Potts, J. (2017). Implementation of an emissions inventory for UK peatlands (p. 88) [Report to the Department for Business, Energy and Industrial Strategy]. Centre for Ecology and Hydrology. https://lowlandpeat.ceh.ac.uk/sites/default/files/2024-04/1904111135_UK_peatland_GHG_emissions.pdf

Evans, C. D., Peacock, M., Baird, A. J., Artz, R. R. E., Burden, A., Callaghan, N., Chapman, P. J., Cooper, H. M., Coyle, M., Craig, E., Cumming, A., Dixon, S., Gauci, V., Grayson, R. P., Helfter, C., Heppell, C. M., Holden, J., Jones, D. L., Kaduk, J., … Morrison, R. (2021). Overriding water table control on managed peatland greenhouse gas emissions. Nature, *593*(7860), 548–552. https://doi.org/10.1038/s41586-021-03523-1

Finckh, S., Beckers, L.-M., Busch, W., Carmona, E., Dulio, V., Kramer, L., Krauss, M., Posthuma, L., Schulze, T., & Slootweg, J. (2022). A risk based assessment approach for chemical mixtures from wastewater treatment plant effluents. *Environment International*, *164*, 107234. https://doi.org/10.1016/j.envint.2022.107234

Fletcher, W., & Ryan, P. (2018). Radiocarbon constraints on historical peat accumulation rates and atmospheric deposition of heavy metals at Holcroft Moss, Warrington. *North West Geography*, *18*(2), 18–28.

Fluet-Chouinard, E., Stocker, B. D., Zhang, Z., Malhotra, A., Melton, J. R., Poulter, B., Kaplan, J. O., Goldewijk, K. K., Siebert, S., & Minayeva, T. (2023). Extensive global wetland loss over the past three centuries. *Nature*, *614*(7947), 281–286. https://doi.org/10.1038/s41586-022-05572-6

Forsmann, D. M., & Kjaergaard, C. (2014). Phosphorus release from anaerobic peat soils during convective discharge—Effect of soil fe: P molar ratio and preferential flow. *Geoderma*, *223*, 21–32. https://doi.org/10.1016/j.geoderma.2014.01.025

Frank, S., Tiemeyer, B., Gelbrecht, J., & Freibauer, A. (2014). High soil solution carbon and nitrogen concentrations in a drained atlantic bog are reduced to natural levels by 10 years of rewetting. *Biogeosciences*, *11*(8), 2309–2324. https://doi.org/10.5194/bg-11-2309-2014

Freeman, B. W. J., Evans, C. D., Musarika, S., Morrison, R., Newman, T. R., Page, S. E., Wiggs, G. F. S., Bell, N. G. A., Styles, D., Wen, Y., Chadwick, D. R., & Jones, D. L. (2022). Responsible agriculture must adapt to the wetland character of mid-latitude peatlands. *Global Change Biology*, *28*(12), 3795–3811. https://doi.org/10.1111/gcb.16152

Gaffney, P. P. J., Hancock, M. H., Taggart, M. A., & Andersen, R. (2018). Measuring restoration progress using pore- and surface-water chemistry across a chronosequence of formerly afforested blanket bogs. *Journal of Environmental Management*, *219*, 239–251. https://doi.org/10.1016/j.jenvman.2018.04.106

Garcés-Pastor, S., Fletcher, W. J., & Ryan, P. A. (2023). Ecological impacts of the industrial revolution in a lowland raised peat bog near Manchester, NW England. *Ecology and Evolution*, *13*(2), 1–22. https://doi.org/10.1002/ece3.9807

Hill, B. M., & Siegel, D. I. (1991). Groundwater flow and the metal content of peat. *Journal of Hydrology*, *123*(3–4), 211–224. https://doi.org/10.1016/0022-1694(91)90091-U

Holman, I. P., Howden, N. J., Bellamy, P., Willby, N., Whelan, M. J., & Rivas-Casado, M. (2010). An assessment of the risk to surface water ecosystems of groundwater P in the UK and Ireland. *Science of the Total Environment*, *408*(8), 1847–1857. https://doi.org/10.1016/j.scitotenv.2009.11.026

Howson, T., Chapman, P. J., Shah, N., Anderson, R., & Holden, J. (2021). A comparison of porewater chemistry between intact, afforested and restored raised and blanket bogs. *Science of the Total Environment*, *766*, 144496. https://doi.org/10.1016/j.scitotenv.2020.144496

Jansen, B., Nierop, K. G. J., & Verstraten, J. M. (2004). Mobilization of dissolved organic matter, aluminium and iron in podzol eluvial horizons as affected by formation of metal-organic complexes and interactions with solid soil material. *European Journal of Soil Science*, *55*(2), 287–297. https://doi.org/10.1111/j.1365-2389.2004.00598.x

Jones, R. I. (1992). The influence of humic substances on lacustrine planktonic food chains. *Hydrobiologia*, *229*(1), 73–91. https://doi.org/10.1007/BF00006992

Joosten, H., & Clarke, D. (2002). Wise use of mires and peatlands. *International Mire Conservation Group and International Peat Society*, *304*, 239–250.

Kaila, A., Asam, Z., Koskinen, M., Uusitalo, R., Smolander, A., Kiikkilä, O., Sarkkola, S., O'Driscoll, C., Kitunen, V., Fritze, H., Nousiainen, H., Tervahauta, A., Xiao, L., & Nieminen, M. (2016). Impact of re-wetting of forestry-drained peatlands on water Quality—A laboratory approach assessing the release of P, N, fe, and dissolved organic carbon. *Water, Air, & Soil Pollution*, *227*(8), 292. https://doi.org/10.1007/s11270-016-2994-9

Koerselman, W., Van Kerkhoven, M. B., & Verhoeven, J. T. (1993). Release of inorganic N, P and K in peat soils; effect of temperature, water chemistry and water level. *Biogeochemistry*, *20*(2), 63–81. https://doi.org/10.1007/BF00004135

Kuznetsova, A., Brockhoff, P. B., & Christensen, R. H. B. (2017). lmerTest package: Tests in linear mixed effects models. *Journal of Statistical Software*, *82*(13), 1–26. https://doi.org/10.18637/jss.v082.i13

Laine, M. P. P., Strömmer, R., & Arvola, L. (2013). Nitrogen release in pristine and drained peat profiles in response to water table fluctuations: A mesocosm experiment. *Applied and Environmental Soil Science*, *2013*, 1–7. https://doi.org/10.1155/2013/694368

Lenth, R. V. (2025). emmeans: Estimated marginal means, aka least-squares means [Manual]. https://doi.org/10.32614/CRAN.package.emmeans

Lestari, N. S., Rochmayanto, Y., Salminah, M., Novita, N., Asyhari, A., Gangga, A., Ritonga, R., Yeo, S., & Albar, I. (2024). Opportunities and risk management of peat restoration in Indonesia: Lessons learned from peat restoration actors. *Restoration Ecology*, *32*(1), 1–14, e14054. https://doi.org/10.1111/rec.14054

Li, H., Ma, X., Huang, X., Ji, W., Chen, K., Xu, S., & Gao, P. (2023). Fluoride contents in commonly used commercial phosphate fertilizers and their potential risks in China. *Environmental Monitoring and Assessment*, *195*(9), 1–12. https://doi.org/10.1007/s10661-023-11623-w

Lundin, L., Nilsson, T., Jordan, S., Lode, E., & Strömgren, M. (2017). Impacts of rewetting on peat, hydrology and water chemical composition over 15 years in two finished peat extraction areas in Sweden. *Wetlands Ecology and Management*, *25*(4), 405–419. https://doi.org/10.1007/s11273-016-9524-9

Martin-Ortega, J., Allott, T. E. H., Glenk, K., & Schaafsma, M. (2014). Valuing water quality improvements from peatland restoration: Evidence and challenges. *Ecosystem Services*, *9*, 34–43. https://doi.org/10.1016/j.ecoser.2014.06.007

Massop, H. T. L., Hessel, R., van den Akker, J. J. H., van Asselen, S., Erkens, G., Gerritsen, P. A., & Gerritsen, F. H. G. A. (2024). Monitoring long-term peat subsidence with subsidence platens in Zegveld, the Netherlands. *Geoderma*, *450*, 1–16, 117039. https://doi.org/10.1016/j.geoderma.2024.117039

Matysek, M., Leake, J., Banwart, S., Johnson, I., Page, S., Kaduk, J., Smalley, A., Cumming, A., & Zona, D. (2019). Impact of fertiliser, water table, and warming on celery yield and CO2 and CH4 emissions from fenland agricultural peat. *Science of The Total Environment*, *667*, 179–190. https://doi.org/10.1016/j.scitotenv.2019.02.360

Nachtigall, S., & Giani, L. (2022). Almost 40 years after raised bog restoration on black peat: How did nutrient levels in soil and water change? *Mires and Peat*, *28*, 03–17. https://doi.org/10.19189/MaP.2021.OMB.StA.2367

National Farmers' Union. (2019). Delivering for Britain: Food and Farming in the Fens. https://www.nfuonline.com/media/uvhhtjio/delivering-for-britain-food-and-farming-in-the-fens.pdf

Natural England. (2025). England Peat Map—Peaty Soil Extent [Shapefile] [Map]. https://naturalengland-defra.opendata.arcgis.com/datasets/Defra:england-peat-map-peaty-soil-extent-1/about

Nicholson, F. A., Smith, S. R., Alloway, B. J., Carlton-Smith, C., & Chambers, B. J. (2006). Quantifying heavy metal inputs to agricultural soils in England and Wales. *Water and Environment Journal*, *20*(2), 87–95. https://doi.org/10.1111/j.1747-6593.2006.00026.x

Novak, M., Zemanova, L., Voldrichova, P., Stepanova, M., Adamova, M., Pacherova, P., Komarek, A., Krachler, M., & Prechova, E. (2011). Experimental evidence for Mobility/Immobility of metals in peat. *Environmental Science & Technology*, *45*(17), 7180–7187. https://doi.org/10.1021/es201086v

Olde Venterink, H., Davidsson, T. E., Kiehl, K., & Leonardson, L. (2002). Impact of drying and re-wetting on N, P and K dynamics in a wetland soil. *Plant and Soil*, *243*(1), 119–130. https://doi.org/10.1023/A:1019993510737

Osborne, A. W., Marsden, S., Caporn, S. J., & Coulthard, E. (2024). Cutting peat: The historical ecology and dissection of the Chat Moss ecosystem. *Mires and Peat*, *31*, 21. https://doi.org/10.19189/MAP.2024.OMB.SC.2468173

Page, S. E., Baird, A. J., Cumming, A., High, K. E., Kaduk, J., & Evans, C. (2020). An assessment of the societal impacts of water level management on lowland peatlands in England and Wales (Report to Defra for Project SP1218). York https://eprints.whiterose.ac.uk/170769/1/Societal_Impacts_Report_March_2020_2_.pdf

Parry, L. E., Holden, J., & Chapman, P. J. (2014). Restoration of blanket peatlands. *Journal of Environmental Management*, *133*, 193–205. https://doi.org/10.1016/j.jenvman.2013.11.033

Peacock, M., Gauci, V., Baird, A. J., Burden, A., Chapman, P. J., Cumming, A., Evans, J. G., Grayson, R. P., Holden, J., Kaduk, J., Morrison, R., Page, S., Pan, G., Ridley, L. M., Williamson, J., Worrall, F., & Evans, C. D. (2019). The full carbon balance of a rewetted cropland fen and a conservation-managed fen. *Agriculture, Ecosystems & Environment*, *269*, 1–12. https://doi.org/10.1016/j.agee.2018.09.020

Pinto, R., Weigelhofer, G., Brito, A. G., & Hein, T. (2021). Effects of dry-wet cycles on nitrous oxide emissions in freshwater sediments: A synthesis. *PeerJ*, *9*, e10767. https://doi.org/10.7717/peerj.10767

Regulation (EU) 2024/1991. (2024). 2024/1991 of the European Parliament and of the Council of 24 June 2024 on Nature Restoration and Amending Regulation (EU) 2022/869.

Rothwell, J. J., Taylor, K. G., Chenery, S. R. N., Cundy, A. B., Evans, M. G., & Allott, T. E. H. (2010). Storage and behavior of as, sb, pb, and cu in ombrotrophic peat bogs under contrasting water table conditions. *Environmental Science & Technology*, *44*(22), 8497–8502. https://doi.org/10.1021/es101150w

Silverthorn, T., Southon, F., Chiverrell, R., Evans, C. D., Andrews, L., Baugh, L., Bell, C., Boyle, J., Baker, F., & Peacock, M. (2026). Pollution legacy and land use influence peat chemistry in temperate lowland peatlands. *JGR: Biogeosciences*, *131*(7), 1–14. https://doi.org/10.1029/2026JG010035

Smith, E. J., Hughes, S., Lawlor, A. J., Lofts, S., Simon, B. M., Stevens, P. A., Stidson, R. T., Tipping, E., & Vincent, C. D. (2005). Potentially toxic metals in ombrotrophic peat along a 400 km English–Scottish transect. *Environmental Pollution*, *136*(1), 11–18. https://doi.org/10.1016/j.envpol.2004.12.015

Temmink, R. J. M., Robroek, B. J. M., Van Dijk, G., Koks, A. H. W., Käärmelahti, S. A., Barthelmes, A., Wassen, M. J., Ziegler, R., Steele, M. N., Giesen, W., Joosten, H., Fritz, C., Lamers, L. P. M., & Smolders, A. J. P. (2023). Wetscapes: Restoring and maintaining peatland landscapes for sustainable futures. *Ambio*, *52*(9), 1519–1528. https://doi.org/10.1007/s13280-023-01875-8

Tiemeyer, B., Frings, J., Kahle, P., Köhne, S., & Lennartz, B. (2007). A comprehensive study of nutrient losses, soil properties and groundwater concentrations in a degraded peatland used as an intensive meadow–implications for re-wetting. *Journal of Hydrology*, *345*(1–2), 80–101. https://doi.org/10.1016/j.jhydrol.2007.08.002

UK Met Office. (2025). Historic station data [Data set]. https://www.metoffice.gov.uk/research/climate/maps-and-data/historic-station-data

United Nations Environment Programme [UNEP]. (2022). Global Peatlands Assessment: The State of the World's Peatlands. https://www.UNEP.org/resources/global-peatlands-assessment-2022

Urbanová, Z., Picek, T., & Bárta, J. (2011). Effect of peat re-wetting on carbon and nutrient fluxes, greenhouse gas production and diversity of methanogenic archaeal community. *Ecological Engineering*, *37*(7), 1017–1026. https://doi.org/10.1016/j.ecoleng.2010.07.012

Van De Riet, B. P., Hefting, M. M., & Verhoeven, J. T. A. (2013). Rewetting drained peat meadows: Risks and benefits in terms of nutrient release and greenhouse gas exchange. *Water, Air, & Soil Pollution*, *224*(4), 1440. https://doi.org/10.1007/s11270-013-1440-5

van der Laan, A., van Dijk, J., Rebel, K. T., & Wassen, M. J. (2024). Rewet without regret? Nutrient dynamics in fen peat exposed to different rewetting degrees. *Biogeochemistry*, *167*(5), 705–721. https://doi.org/10.1007/s10533-024-01139-x

van Dijk, J., Stroetenga, M., Bos, L., Van Bodegom, P. M., Verhoef, H. A., & Aerts, R. (2004). Restoring natural seepage conditions on former agricultural grasslands does not lead to reduction of organic matter decomposition and soil nutrient dynamics. *Biogeochemistry*, *71*(3), 317–337. https://doi.org/10.1007/s10533-004-0079-0

Vroom, R. J., Temmink, R. J., van Dijk, G., Joosten, H., Lamers, L. P., Smolders, A. J., Krebs, M., Gaudig, G., & Fritz, C. (2020). Nutrient dynamics of sphagnum farming on rewetted bog grassland in NW Germany. *Science of the Total Environment*, *726*, 138470. https://doi.org/10.1016/j.scitotenv.2020.138470

Vuori, K.-M. (1995). Direct and indirect effects of iron on river ecosystems. *Annales Zoologici Fennici*, *32*(3), 317–329.

Wei, T., & Simko, V. (2024). R package 'corrplot': Visualization of a correlation matrix [Manual]. https://github.com/taiyun/corrplot

Wen, Y., Zang, H., Ma, Q., Freeman, B., Chadwick, D. R., Evans, C. D., & Jones, D. L. (2020). Impact of water table levels and winter cover crops on greenhouse gas emissions from cultivated peat soils. *Science of the Total Environment*, *719*, 135130. https://doi.org/10.1016/j.scitotenv.2019.135130

Williamson, J., Evans, C., Spears, B., Pickard, A., Chapman, P. J., Feuchtmayr, H., Leith, F., Waldron, S., & Monteith, D. (2023). Reviews and syntheses: Understanding the impacts of peatland catchment management on dissolved organic matter concentration and treatability. *Biogeosciences*, *20*(18), 3751–3766. https://doi.org/10.5194/bg-20-3751-2023

Xu, J., Morris, P. J., Liu, J., & Holden, J. (2018). Hotspots of peatland-derived potable water use identified by global analysis. *Nature Sustainability*, *1*(5), 246–253. https://doi.org/10.1038/s41893-018-0064-6

Zak, D., & Gelbrecht, J. (2007). The mobilisation of phosphorus, organic carbon and ammonium in the initial stage of fen rewetting (a case study from NE Germany). *Biogeochemistry*, *85*(2), 141–151. https://doi.org/10.1007/s10533-007-9122-2

Zak, D., Wagner, C., Payer, B., Augustin, J., & Gelbrecht, J. (2010). Phosphorus mobilization in rewetted fens: The effect of altered peat properties and implications for their restoration. *Ecological Applications*, *20*(5), 1336–1349. https://doi.org/10.1890/08-2053.1