Lowland agricultural peat characteristics which influence water quality: literature review

Chief Scientist's Group research report

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Dr Robert Bradburne
Chief Scientist

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Foreword

The 'LP3+' (Lowland Peat 3 Plus) is a research project comprising a series of linked work packages and reports. Together, these further our knowledge of how peatland characteristics influence peat hydrology and water quality. The project focuses specifically on how the properties of lowland agricultural peat influence peat hydrology and water table management and how raising the water table influences the mobilisation of substances from within the peat, and therefore the likely risks to water quality.

The LP3+ project benefits from the specialist knowledge of the related Lowland Peat 3 (LP3) project and partners including the field data collection infrastructure established by the LP3 project.

LP3+ is funded by the UK Government's Nature for Climate fund through the Environment Agency's Lowland Agricultural Peat Water R&D programme and is led and managed by UK Centre for Ecology and Hydrology (UKCEH).

The LP3+ project partners are UKCEH, University of Liverpool, Cranfield University and Rigare Ltd.

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Executive summary

This report provides a rapid evidence assessment of how drainage and rewetting influence water quality in temperate lowland peatlands, focusing on the flux and concentration of carbon, nutrients, metals, and emerging contaminants. The review synthesises data from 39 field-based studies (80 sites) across the UK and mainland Europe, and from the National Soil Inventory of England and Wales.

Rewetting of lowland peatlands has the potential to alter concentrations and fluxes of substances, driven by changes in redox conditions, microbial activity and hydrology and influenced by peatland type, historical and current land-use and in some cases, location. Pore water chemistry is not always reflective of nearby surface water chemistry. The availability of published data varied, with more measurements associated with carbon and nutrients than metals and other substances.

Carbon (C): Dissolved organic carbon (DOC) concentrations and fluxes are well understood with drainage generally increasing DOC, while rewetting tends to lower concentrations, though short-term DOC flushes may occur post-rewetting. The general pattern is mediated by peat type and whether surface or pore water is considered. DOC fluxes are highest in sites with low mineral content, indicating strong links to peat organic matter. Particulate organic carbon (POC) contributes minimally to aquatic carbon export, and dissolved inorganic carbon (DIC) is primarily controlled by peatland type (bog vs fen), with limited relevance for water quality.

Nitrogen (N): Raising water tables will likely alter within-peat N cycling, increasing pore water ammonium (NH4) concentrations, but this will not automatically result in elevated concentrations in surface waters. However, nitrate and total nitrogen concentrations may decrease following rewetting although this may be mediated by water sources used for rewetting.

Phosphorus (P): Surface water phosphate (PO_4-P) and total phosphorus (TP) concentrations often exceed ecological thresholds for freshwaters in drained agricultural peatlands. Rewetting may lower phosphorus concentrations, but responses vary and are strongly influenced by iron (Fe) dynamics and peat degradation.

Sulfate: Drainage promotes sulfate (SO_4) release through sulfide oxidation, while rewetting would be expected to favour the retention of SO_4 via bacterial sulfate reduction. Fluctuating water tables may cause short-term SO_4 flushes.

Metals and Other Elements: Mobilisation risks exist for Aluminium (Al), Arsenic (As), Fe, Mercury (Hg), and Lead (Pb) under rewetting, with potential ecological impacts. Fe dynamics are critical for P retention and water colour; ochre (a substance made of ferric oxide, clay and sand) formation in acid sulfate soils is a major concern in drained fens but likely mitigated by rewetting. Data gaps exist for Copper (Cu), Manganese (Mn), and other trace metals.

Pesticides and Emerging Contaminants: Evidence on pesticide and herbicide behaviour in peatlands is limited; losses may increase under wetter conditions if agricultural use continues. Emerging contaminants (e.g., per- and polyfluoroalkyl substances (PFAS), micropollutants, microplastics) are poorly studied, though peatlands may act as sinks for some compounds.

Rewetting generally is associated with improved water quality by reductions in DOC and nitrate but may mobilise ammonium, phosphorus, and metals depending on site conditions. Future monitoring should prioritise DOC, N, P, SO_4, and metals, alongside emerging contaminants where resources allow.

1. Introduction and scope

In the UK, early studies of how peatland drainage and rewetting affect water quality were concentrated on upland peatlands (e.g. Martin-Ortega et al., 2014) due to these providing ~85% of national drinking water (Xu et al., 2018). The focus of these early studies was largely on dissolved organic carbon (DOC) due to the removal of DOC by water utilities being costly and difficult (Worrall et al., 2007). Increased attention is now being given to raising water tables in temperate lowland agricultural peatlands, either towards semi-natural states (e.g. Peacock et al., 2019) or for paludiculture and wetter farming (e.g. Mulholland et al., 2020). It is possible that drainage led to the long-term accumulation of substances, either of agricultural or industrial legacy and therefore there is a pressing need to understand if raising water tables presents a risk (or benefit) to water quality, aquatic ecology and drinking water. Growing awareness of the role that macronutrient stoichiometry (carbon, nitrogen, phosphorus, - C, N, P) plays in the health of aquatic ecosystems (Stutter et al., 2018), and the potential risks from emerging contaminants (Richardson & Ternes, 2018), means that a shift away from a carbon-only focus is required.

The aim of this review is to provide a rapid evidence assessment (i.e. not a full in-depth literature review) of water quality and substance flux within drained and rewetted peats. This will include information on macronutrients, metals, pesticides, ochre, and emerging contaminants. Additionally, the review will assess whether responses relate to peat physical or chemical properties, or to the nature of re-wetting.

2. Methods

Data were collated to understand how water table management in lowland peatlands affects water chemistry. This collation involved searching for relevant data from published papers, grey literature, repositories, and previous projects by the project team (e.g. Evans et al., 2016b).

Data were extracted for lowland fen and bog peatlands across all land covers (including cropland, grassland, semi-natural, rewetted, extraction, afforested). See a summary of the data sources in Appendix – Table 1. Although this review focuses on the UK, the collation was expanded to include sites in temperate mainland Europe due to a lack of UK studies (Table 1). This extraction resulted in 39 field-based studies with 80 individual data points. Most data points represent individual peatland sites, but some represent multiple sites of the same land cover where disaggregation to individual sites was impossible (due to how data was presented); for simplicity, data points are hereafter referred to as “sites”.

Table 1. Number of studies by country.

Country	Count
Germany	18
UK	9
Ireland	5
Netherlands	5
Sweden	3
Poland	1
France	1

If results were not provided in tables, text or repositories, then WebPlotDigitizer (https://automeris.io/) was used to extract them from figures. Extracted data included mean annual water table, peat properties (e.g. bulk density, C/N), discharge, aquatic carbon fluxes, and concentrations of macronutrients (C, N, P) metals, anions and cations, in pore or surface water. In some cases, pore water samples were taken from dipwells, meaning they would be taken below the water table level. In other cases, suction plates (of varying depths) were used to sample soil water. Depending on the depth of the suction plate and the water table levels these samples could be from above or below the water table, but from the level of available data this is not always possible to determine. Sampled surface waters were predominantly drainage ditches (n = 16), but also streams (n = 4) and lakes/ponds/pools (n = 5).

Spearman correlation coefficients are used to test for monotonic relationships (i.e. variables that change in the same relative direction, but not necessarily at a constant or linear rate) between variables, and Kruskal Wallis tests are used to test for significant differences between site types. All results are considered significant when $p < 0.05$. Unless otherwise specified, data are given as means, with minimum and maximum values in parentheses.

For Section 3.5 (Other metals, anions and cations), draws upon data (from soil extractions) from the National Soil Inventory (NSI) held by Cranfield University. The NSI covers England and Wales in a 5 km grid pattern, with individual sampling points (total 5600+) at grid intersections. Here soil data are considered from the original sampling in 1980 to determine the potential for accumulation within organic topsoils, although the associated soil series is not necessarily a peat soil.

All the NSI topsoil samples have been classified in three organic carbon classes based on their organic carbon and clay contents: mineral (n = 3429), organo-mineral (n = 1660) and peaty / peat (n = 581) according to the criteria in Hodgson (1997). The NSI additionally disaggregates lowland and upland samples.

Analytical methods for the NSI are detailed in Appendix 1 Table 2.

As well as the data collation, this review discusses the findings from other relevant research, including mesocosm studies and research from outside temperate Europe.

3. Results

Rewetting, either fully or partially, of lowland peatlands has the potential to substantially alter concentrations and fluxes of substances through various mechanisms, primarily driven by changes in redox conditions, microbial activity, and hydrology. Raising water tables will cause a change in the extent of aerobic/anaerobic zones which can alter nutrient cycling, by slowing down the mineralization of organic C, N and P. Depending on the water source used, rewetting may bring additional inputs of substances, particularly inorganic N and P, into the peatland.

For other substances (e.g. anions, cations, metals), the effect of rewetting will depend partially on peatland type: the ombrotrophic nature of bogs means they naturally have low concentrations of various substances, whilst fens will receive some elements (e.g. Fe, Ca) via groundwater inputs. Thus, rewetting a bog with external (non-rainwater) inputs would likely result in a significant change in biogeochemistry. Additionally, for bogs and fens, rewetting may lead to the release of previously bound nutrients and metals into the pore water.

The extent to which these are further mobilised into receiving waters will depend on flow paths within the peat and the hydrological connectivity to ditches. Furthermore, hydrological changes will affect mixing, and dilution of substances. The magnitude and direction of changes in substance fluxes will largely be governed by the extent of rewetting because this will determine the volume of peat in saturated and unsaturated zones (i.e. below and above the water table). Biogeochemical cycling in these zones will be vastly different.

Overall, site history and the level of rewetting will determine the balance between nutrient retention and release, with significant implications for water quality, carbon sequestration, and ecosystem functioning in lowland peatlands.

3.1. Carbon

This section considers dissolved organic carbon (DOC), particulate organic carbon (POC), and dissolved inorganic carbon (DIC). Dissolved carbon dioxide (CO2) and methane (CH4) are also lost laterally via the aquatic pathway but typically make negligible contributions in UK lowland peatlands to the total aquatic budget (respectively ~5% and ≤0.5%, Evans et al., 2016b). Furthermore, these dissolved gases do not negatively affect water quality or aquatic ecosystem functioning, and are therefore not considered here.

Dissolved organic carbon (DOC)

DOC concentrations were available for 59/80 sites, and DOC fluxes for 22. Mean pore water DOC concentration was 81.6 (range 15-196) mg l^{-1}, whilst surface water was lower, with a mean of 38.1 (9.9-120) mg l^{-1}. Mean DOC flux was 18.0 (4.1 - 67) g C m^{-2} yr^{-1}, but the median was lower (10.7 g C m^{-2} yr^{-1}) suggesting that the mean is skewed upwards by a handful of sites with high fluxes.

The effects of drainage and rewetting on DOC concentrations and fluxes are well understood for peatlands as a whole class, whereby drainage increases fluxes and concentrations compared to undrained sites, and rewetting reverses this (see syntheses by IPCC, 2014, Evans et al., 2016a). However, the IPCC Wetlands Supplement (IPCC, 2014) advised caution when extrapolating from all peatland classes to fens, due to inherent difficulties in calculating water budgets (and thus DOC fluxes) in highly managed lowland fens.

In the data synthesis, a positive relationship between mean annual water table depth and DOC concentration was found for pore water, but not for surface water (Fig. 1). Similarly, site land cover had an effect, with rewetted and near natural sites having lower pore water DOC concentrations compared to grasslands, but no effect was observed for surface water (Fig. 2). However, a lack of data for some land covers (particularly arable, extraction sites, and paludiculture) limit the power of statistical testing. Finally, a strong negative relationship between peat mineral content and DOC flux was found (Fig. 3). This relationship suggests that DOC fluxes are greatest from sites with lowest mineral content; i.e. sites with highest organic content. Such a relationship makes sense, assuming that drained sites degrade and their peat masses steadily loses organic matter, thus leading to decreased rates of DOC production (and export), as well as the fact that soils with greater organic content have more potential for DOC production and release. Additionally, there is the opportunity for DOC adsorption in soils with a high mineral content (Kothawala et al., 2012), thus decreasing concentrations and fluxes.

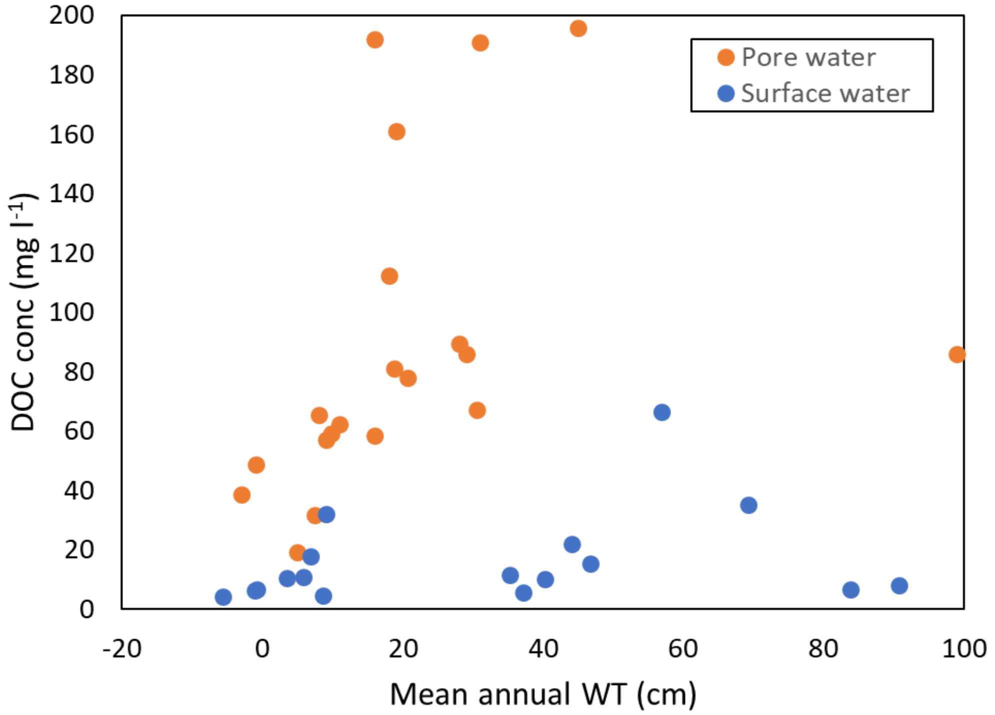

Figure 1. Scatter plot of mean annual water table (WT) and DOC concentrations in pore water and surface water. The correlation is significant for pore water (Spearman's rho = 0.62, p = 0.003) but not for surface water (rho = 0.39, p = 0.086). Water table is expressed as depth relative to the ground surface, i.e. larger positive values indicate deeper drained sites.

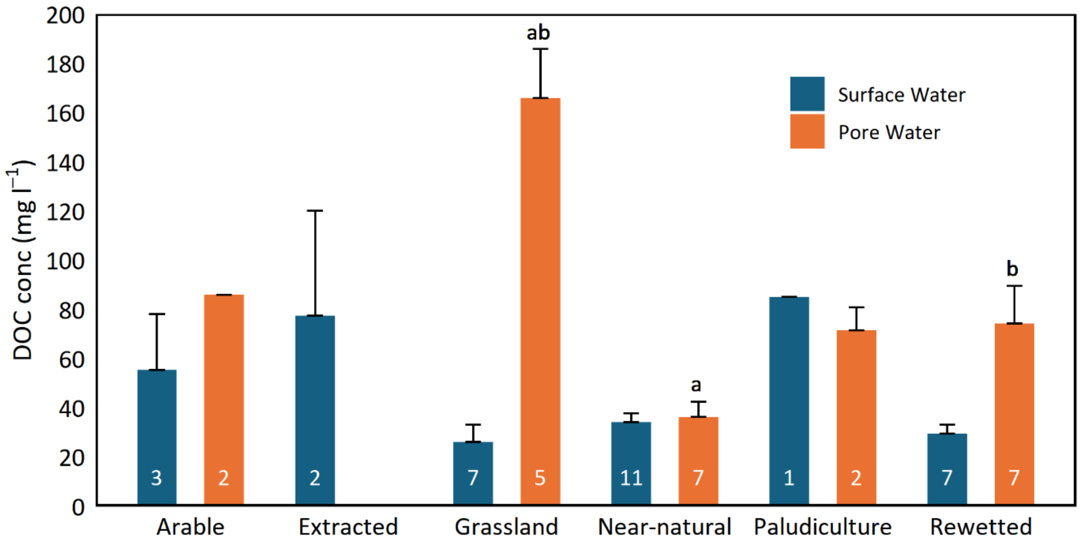

Figure 2. Mean surface and pore water DOC concentrations grouped by land cover. Kruskal Wallis tests show no significant difference between land cover types for surface water (p = 0.11) but an overall significant effect of land cover for pore water (p = 0.004); post-hoc pairwise comparisons showed significant differences between near natural vs grassland (p <0.001), and rewetted vs grassland (p = 0.023) (denoted by shared letters 'a', 'b', 'ab'). White numbers denote sample sizes for each bar. Error bars denote standard error.

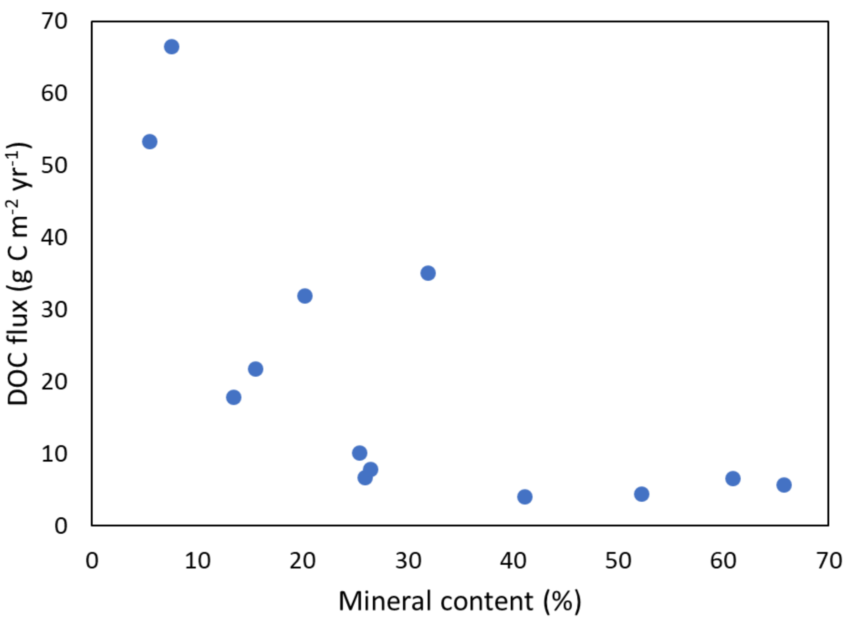

Figure 3. Scatter plot of peat mineral content and DOC flux (rho = -0.8, p < 0.001).

These results suggest that water table management and land use do play a role in DOC dynamics in lowland peat, but that other factors are also implicated which, to some extent, obscure simple empirical relationships between DOC and water table. Whilst these other factors undoubtedly include geochemical properties and land use histories, sampling protocols (e.g. frequency of sampling, pore water sampling methods) are also likely to confound site and data comparisons. When field studies use consistent methods the same drainage-induced DOC increases and post-rewetting DOC decreases are observed; e.g. in a study of 38 pristine, drained, and rewetted raised bogs in southern Finland (Haapalehto et al., 2014); in a study of small streams in lowland Irish bogs (Pschenyckyj et al., 2023); and in a comparison of a drained and rewetted fen in southern Germany (Höll et al., 2009). Thus, drainage and rewetting can generally be assumed to respectively increase and decrease long-term DOC concentrations (in surface and pore water) and fluxes, with the caveat that rewetting may be accompanied by a short-term post-rewetting flush of DOC (e.g. Strack et al., 2015). However, it cannot be assumed that drainage and rewetting will lead to change in DOC fluxes, particularly in lowland fens where hydrology can be complex, and feature groundwater and lateral inputs which can be disrupted/disconnected by changing water management regimes (Evans et al., 2016a). It is also important to consider aquatic inputs of waterborne carbon, and not just fluxes leaving the site. For example, for a semi-natural fen and rewetted grassland on fen peat in East Anglia, inputs of DOC and POC in water abstracted from nearby surface waters to maintain groundwater levels were equivalent to 27% (semi-natural) and 16% (grassland) of aquatic C losses (Peacock et al., 2019).

Particulate organic carbon (POC)

Compared to DOC, very few lowland peat studies measured POC: 11/80 sites for concentration, and 10/80 for flux. The majority (8) of these measurements were from Defra

LowlandPeat1. Overall, mean surface water POC concentration was relatively high: 4.7 (0.8-13) mg l^{-1}. In both upland and lowland UK peatlands, high POC concentrations can occur during drought or summer periods when flows are low (see Evans et al., 2018, Peacock et al., 2019). Whether these high concentrations are "real" *per se*, or arise as artefacts whereby the action of sampling in shallow ditches disturbs the sediments and releases POC is largely irrelevant; these high concentrations do not contribute meaningfully to either flow-weighted mean concentrations or fluxes (Evans et al., 2018). This is particularly true in lowland peatlands where ditch networks can contract as they dry out, and ditches become disconnected pools. Fielder et al. (2008) measured extremely high POC concentrations (≤ 125 mg l^{-1}) in the pore waters of drained and rewetted fens in southern Germany. The relevance, and interpretation, of pore water POC is an unknown, but it seems unlikely that pore water concentrations have any bearing on surface water chemistry.

Mean POC flux was 2.7 (0.3-8.6) g C m^{-2} yr^{-1}. When considering all sites where DOC and POC fluxes were measured, the mean contribution to the total aquatic flux from POC was 12.8 (3-27) %. The low sample size precludes robust statistical analysis, but in LowlandPeat1 it was noted that some of the more intensively managed arable and grassland sites had slightly higher POC fluxes (Evans et al., 2016b), but this pattern was not consistent, and some arable sites also had low POC fluxes. POC dynamics in lowland peatlands are clearly a knowledge gap. Although POC is highly relevant for studies of aquatic ecology (e.g. of macroinvertebrates, Ramchunder et al., 2012) we suggest that it is not a "must have" measurement in lowland peat studies, due to the minor contribution it makes to waterborne carbon losses when compared to DOC. However, if resources allow, more data would be welcome to enlarge the lowland peat POC knowledge base. Regardless, it seems unlikely that rewetting will have any major impact on POC concentrations or fluxes.

Dissolved inorganic carbon (DIC)

DIC concentration and flux were measured at 27/80 and 17/80 sites respectively, with means of 61 (0-249) mg l^{-1} and 11.8 (0-31) g C m^{-2} yr^{-1}. Peatland type (bog or fen) was the primary control on DIC concentration, as shown by a significant positive correlation with pH (Fig. 4) (although DIC flux did not correlate with pH; rho = 0.27, p = 0.34). This relationship simply arises because bogs receive water solely from precipitation, whilst fen peats receive inputs of base-rich groundwater, either from their own catchments or via abstraction from adjacent rivers. For the Defra LowlandPeat1 fen sites DIC fluxes were generally larger than DOC fluxes, with a mean DIC:DOC flux ratio of 1.9 (0.5-3.9). However, in these systems this DIC does not necessarily represent a loss of peatland carbon, because at least part of the DIC is generated from weathering of carbonate or siliceous minerals. As such, the extent to which DIC fluxes should be included in the peatland carbon budgets of these systems is unclear (Evans et al., 2016b). Overall, DIC concentrations and fluxes are of limited relevance when considering the effects of rewetting lowland peatlands on water quality.

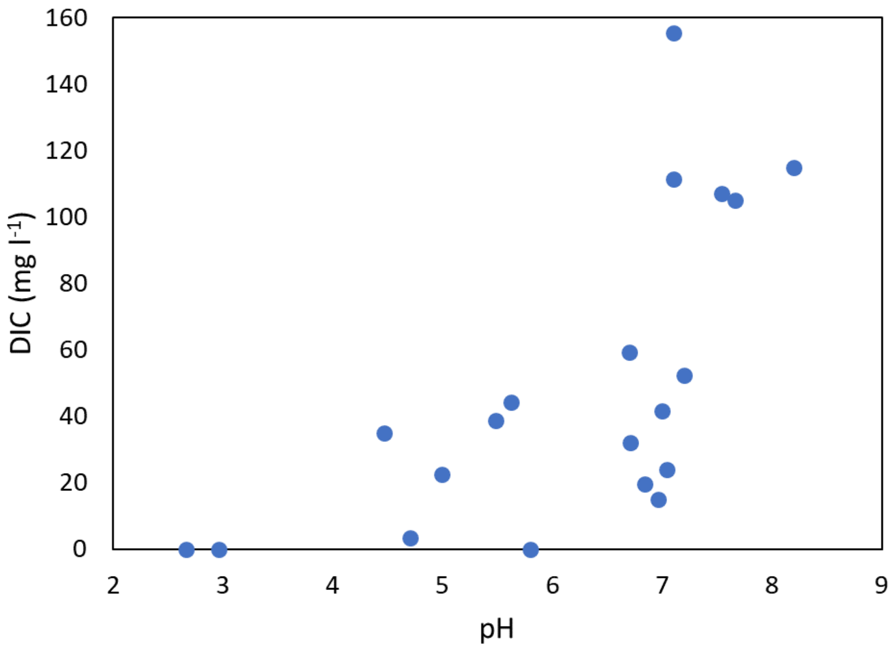

Figure 4. Scatter plot of pH and DIC concentration (rho = 0.74, p < 0.001).

3.2. Nitrogen

Nitrogen (N) concentrations were frequently reported in our collated data, with most sites having data for ammonium (NH_4, 36/80 sites), nitrate (NO_3, 33/80), total N (TN, 11/80), and/or nitrite (NO_2, 9/80). In part, this focus arises due to eutrophication/water quality concerns where lowland peatlands are used for agriculture, with NH_4, NO_2, and NO_3 all being capable of stimulating algal growth (Camargo & Alonso, 2006). The Water Framework Directive (WFD) UK Technical Advisory Group proposes TN boundaries for shallow (<3 m) lakes of 1.07 mg L^{-1} TN (clear waters) and 1.46 mg L^{-1} TN (humic waters) to achieve "good" ecological standards (WFD-UKTAG, 2020). The World Health Organisation sets drinking water limits of 50 mg l^{-1} and 3 mg l^{-1} for NO_3 and NO_2, for health reasons (WHO, 2022). Additionally, the WFD includes lowland river standards for ammonia (whereby rivers are categorised by alkalinity using $CaCO_3$ as a proxy) with "poor" quality defined as 1.1 and 2.5 mg N l^{-1} for low alkalinity (< 50 mg l $CaCO_3$) and high alkalinity (≥ 50 mg l $CaCO_3$) rivers respectively (WFD, 2015).

Ammonium (NH_4-N)

Overall mean NH_4-N concentrations were 1.51 (0.2-9.32) and 0.3 (0-1.52) mg l^{-1} for pore and surface water respectively. We were unable to detect any empirical relationships between NH_4 and other variables in our collated data. Although near-natural sites generally had the lowest pore water concentrations (0.2-1.2 mg l^{-1}) there were no clear patterns for other land covers. However, Zak et al. (2010) measured pore water NH_4 in 17 fens in northern Germany and Poland: nine rewetted fens that were previously heavily drained grasslands, four rewetted fens that were weakly drained but never used for agriculture, and four natural fens. NH_4 was similar in natural and weakly drained fens (~1.4

mg l^{-1}) but very high in the rewetted heavily drained sites (12 mg l^{-1}). Others have reported post-rewetting increases in NH_4 e.g. in pore water of a rewetted Dutch grassland fen (Dijk et al., 2004), in pore water of a rewetted extracted German bog (Nachtigall & Giani, 2022), and in the chemical composition of peat in rewetted extracted Swedish bogs (Lundin et al., 2017). The same response was observed in a rewetting mesocosm experiment using cores from a Dutch grassland fen and was attributed to anaerobic conditions suppressing nitrification and promoting NH_4 accumulation (van der Laan et al., 2024).

However, in the Swedish mires study, NH_4 concentrations in surface water actually decreased following rewetting. Similarly, no consistent difference was discernible between land cover types for surface water NH_4 concentrations (Fig. 5). As such, it can be concluded that raising water tables/rewetting will likely alter within-peat N cycling, with a resultant increase in peat pore water NH_4 concentration. However, this may not automatically result in elevated NH_4 concentrations in drainage waters. It has been suggested that partial rewetting of grassland fens (WT 20cm below the surface) may be "safer" than full rewetting as this leads to lower internal NH_4 accumulation and, therefore, perhaps less NH_4 release into drainage waters (van der Laan et al., 2024). Additionally, top soil removal may be a viable method to minimise post-rewetting NH_4 increases (see Section 3.3), although it may have severe consequences for the carbon budget; whether this is an acceptable trade-off will depend on the aims of each individual rewetting project.

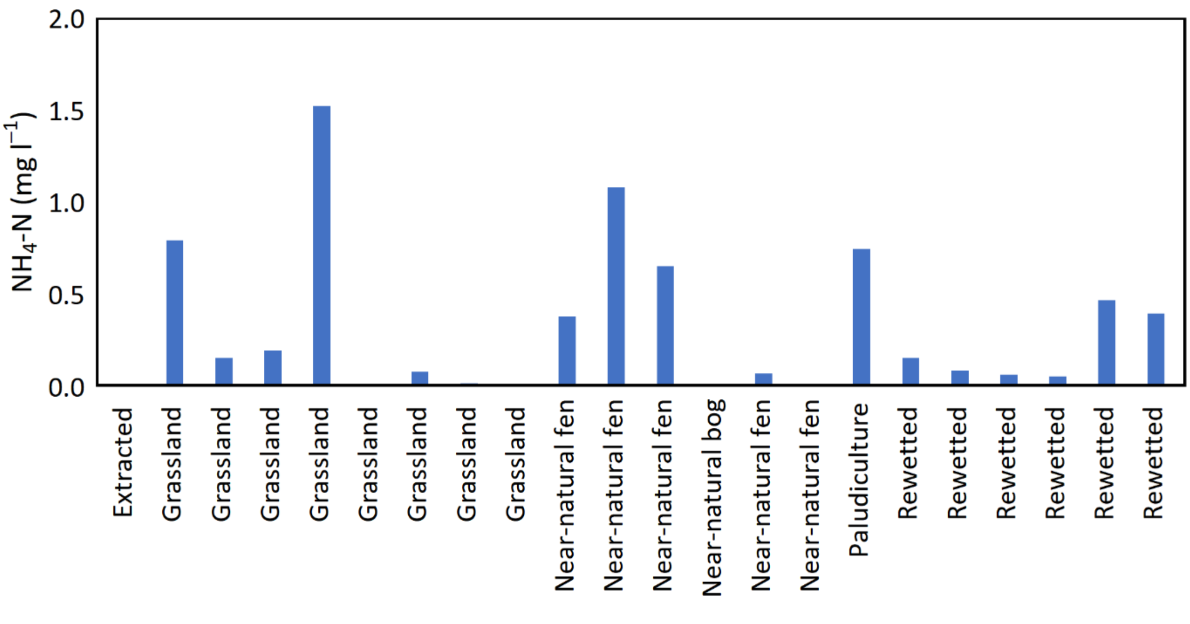

Figure 5. Mean NH_4-N concentrations in surface waters for 22 sites. Note that zero values show where NH_4 was measured but not detected.

Nitrate (NO_3-N)

Overall mean NO_3-N concentrations were 4.8 (0-37.7) and 1.9 (0-13.1) mg l^{-1} for pore and surface water respectively. As for NH_4-N, no empirical relationships were observed between NO_3-N in pore or surface water and other variables. Although land cover effects were visually obvious with mean surface water concentrations increasing from near-natural ≈ rewetted < grassland (Fig. 6), this effect was not significant (Kruskal Wallis $p = 0.31$) possibly due to the small sample size. However, such a pattern is supported by the literature, where NO_3-N concentrations are frequently higher in drained fens due to enhanced decomposition of organic matter and elevated N mineralisation (Tiemeyer et al., 2007, van der Laan et al., 2024). Additionally, drained agricultural and grassland sites may have higher NO_3-N concentrations in surface waters due to either direct leaching of fertilisers applied to fields, or via abstracted inputs of river water (see Peacock et al., 2017, 2019, for an example from fens in East Anglia, UK, where the high-level carrier used for abstraction peaked at 14 mg l^{-1} NO_3-N).

Other studies also frequently report maximum concentrations in surface waters that exceed the WHO drinking water limit of 11.3 mg l^{-1} NO_3-N (WHO, 2022); e.g. 29.8 mg l^{-1} at the outlet of a German grassland fen catchment (Tiemeyer et al., 2007), 18.1 mg l-1 in ditches draining a Danish cropland fen (Scholz & Trepel, 2004). Contrary to the response for NH_4-N, post-rewetting decreases in NO_3-N have frequently been observed in pore water and peat chemistry (Dijk et al., 2004, Lundin et al., 2017, Nachtigall & Giani, 2022).

In one German study, pore water NO_3-N concentrations decreased to approximately zero ten years after rewetting an extracted raised bog (Frank et al., 2014). Taken together, these findings suggest that rewetting, or partially raising water tables, will potentially have beneficial effects on NO_3-N dynamics, and could reduce concentrations within the peat mass and surface waters. However, presumably where water sources for rewetting have high NO_3-N concentrations, these beneficial effects may be negated.

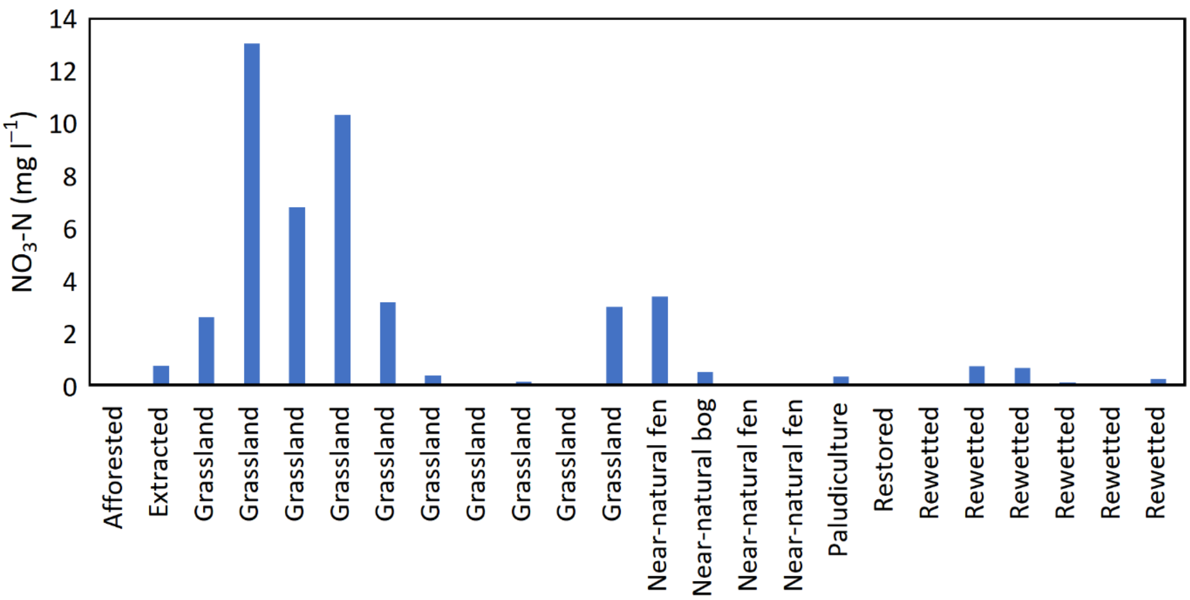

Figure 6. Mean NO_3-N concentrations in surface waters for 24 sites. Note that zero values show where NO_3-N was measured but not detected.

Total nitrogen (TN)

Overall mean TN concentrations were 5.2 (1.6-11.9) and 1.8 (1-3.7) mg l^{-1} for pore and surface water respectively. These surface water TN values are generally above the WFD-

UKTAG (2020) shallow lake threshold of 1.07-1.46 mg l^{-1} for "good" ecological status. The small sample size (n = 11) precludes any kind of statistical analysis. However, Frank et al. (2014) reported decreases in pore water TN in German bogs from intensive grassland > extensive grassland > rewetted extraction > near-natural. They suggested that rewetting successfully restored the N dynamics that would be expected in pristine ombrotrophic bogs.

Similarly, TN concentrations were significantly higher in small streams in degraded bogs when compared to natural bogs in Ireland, and this was attributed to enhanced decomposition of organic matter under drained, aerobic conditions (Pschenyckyj et al., 2023). Whatley et al. (2014) also found higher TN concentrations in drainage ditches of an extensive grassland when compared to a nature reserve, in Dutch fen peat. The evidence therefore suggests that rewetting will most likely lower TN concentrations in surface and pore waters of lowland peatlands (with the same caveat as for NO_3, that rewetting with N-rich waters may negate these effects). In many situations TN responses will be similar to those for NO_3, although this will depend on how much of the TN pool is comprised of organic N; in some lowland peatlands organic N can be the dominant N component (Frank et al., 2014).

Nitrite (NO_2-N)

NO_2-N concentrations were infrequently measured but average values were low, with an overall mean of 0.003 mg l^{-1}, well below the WHO drinking water limit of 0.9 mg l^{-1} (WHO, 2022). Studies reporting NO_2-N included one of near-natural, afforested, and restored (via tree-felling of pine and spruce) raised bogs at Flanders Moss (UK), where pore water NO_2 concentrations were below detection limits in 14% of measurements, and otherwise mostly < 0.006 mg l^{-1} (Howson et al., 2021). Mean surface water concentrations were < 0.02 mg l^{-1} for thirteen Irish fens and bogs, where sites included simple rewetting and partial or total peat removal (Lally et al., 2012). Finally, ditch water concentrations in a Dutch extensive grassland and nature reserve fen were at the limit of detection (Whatley et al., 2014).

Considering the above, and that NO_2-N concentrations in UK lowland rivers are typically low (e.g. Neal et al., 2000, Halliday et al., 2014, Homan et al., 2024), NO_2 is tentatively considered unlikely to be a determinand of major concern following rewetting of lowland peatlands. However, additional data are required to increase confidence in this assertion.

3.3. Phosphorus (P)

Phosphate (PO_4-P) and total phosphorus (TP) concentrations were frequently reported in the collated dataset, with total respective sites of 25/80 and 13/80. The WFD-UKTAG proposes TP boundaries for shallow (<3 m) lakes of 23-35 µg l^{-1} (high alkalinity) and 9 µg l^{-1} (low alkalinity) to achieve "high/good" ecological standards (WFD-UKTAG, 2016). Lakes standards were used here for comparison since slow or no-flow ditches may be more similar to lakes than rivers in their ecological response to nutrients.

As for N, this focus arises due to eutrophication and water quality concerns where lowland peatlands are used for agriculture. Mean phosphate concentrations were 443 (10-1226) and 249 (0-2452) μg l^{-1} for pore and surface water respectively, and mean surface water TP was 135 (10-440) μg l^{-1} (note that no pore water TP data was collated). These TP values were generally above the WFD-UKTAG (2016) shallow lake boundary for “good” ecological status, with the average surface water TP value falling under the “poor/bad” status. Average concentrations of PO_4–P and TP in many studies were considerably higher than 24 μg l^{-1} which is a suggested TP marker of eutrophic waters (Fig. 7, 8) (IPCC, 2019). Small within-group sample size (e.g. n = 3 for PO_4-P in near-natural sites) precludes statistical testing but concentrations were generally highest at grassland sites, demonstrating the clear effects of agricultural activities (Fig. 7, 8). A high mean surface water PO_4-P concentration of 799 μg l^{-1} (not shown in Fig. 7 for clarity) was reported from a paludiculture bog in NW Germany, likely attributed to the nutrient-rich irrigation water sourced from the surrounding agricultural landscape (Daun et al., 2023).

Near-natural sites mostly had very low P concentrations, e.g. pore waters in two Welsh fens were ~3 μg l^{-1} PO_4-P (Menichino et al., 2019); the same as concentrations in small streams in Irish bogs (Pschenyckyj et al., 2023). The high surface water PO_4 concentration (125 μg l^{-1} PO_4-P) reported for a near-natural peatland in the Netherlands is the mean from three ditches where one unexplained outlying value is skewing the mean; the median is 30 μg l^{-1} PO_4-P (Veraart et al., 2017).

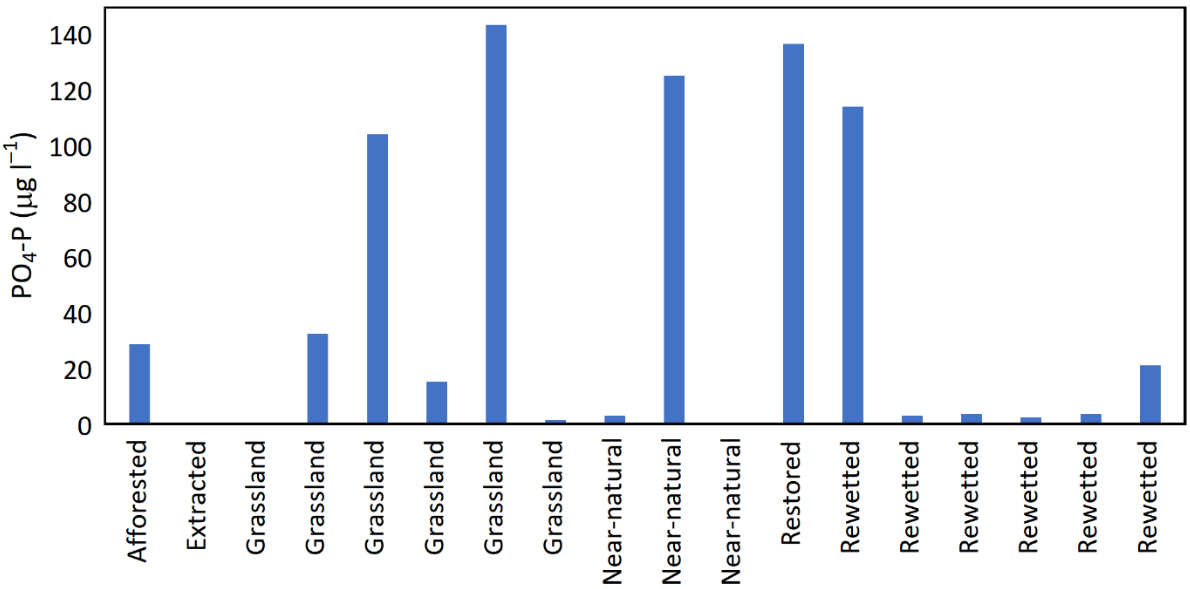

Figure 7. Mean PO_4-P concentrations in surface waters for 18 sites. Note that zero values show where PO_4-P was measured but not detected. For clarity, one paludiculture site with a concentration of 799 μg l^{-1} has been omitted.

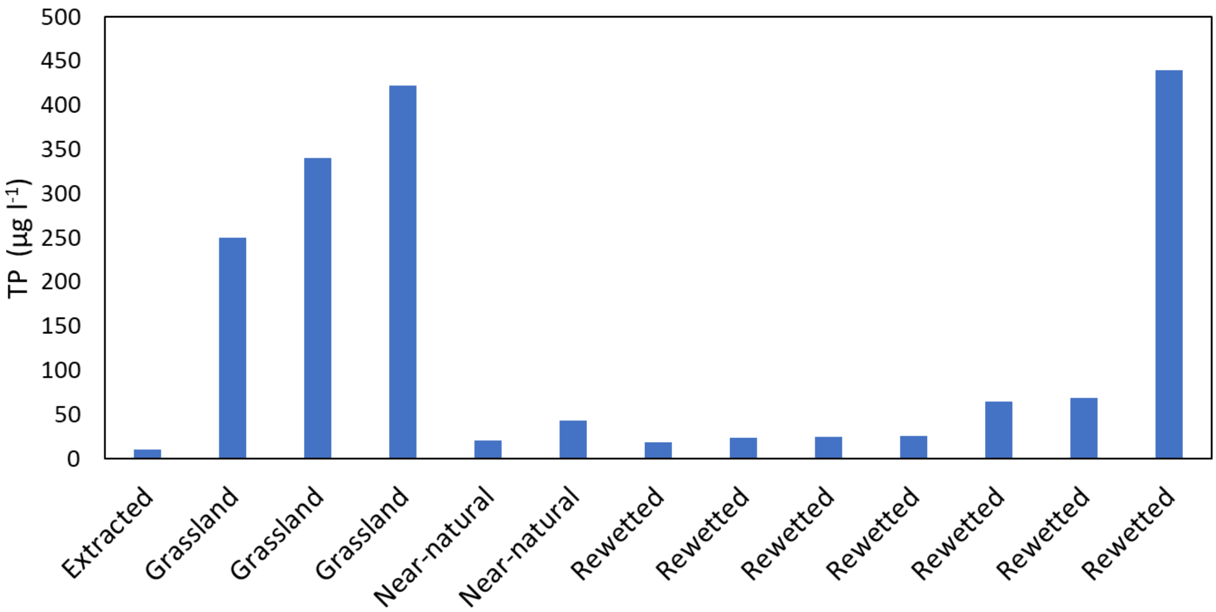

Figure 8. Mean TP concentrations in surface waters for 13 sites.

Rewetted sites generally had lower TP concentrations than grassland sites, more similar to near-natural sites (Fig. 8). A significant increase in TP concentrations was reported for a rewetted peat extraction bog with partial peat removal in Ireland, possibly due to the colonization of large open water areas by macrophytes, as seasonal die-back and decomposition can promote P release (Lally et al., 2012). However, the opposite trend of a decline in TP was observed at a rewetted site without peat removal in that same study. Similarly, rewetted fens in NE Germany with peat removal had lower surface water TP concentrations (65 µg l^{-1}) than those without peat removal (440 µg l^{-1}), due to the mobilisation of P in surface peat layers following rewetting (Zak et al., 2018).

At face value these data seem to suggest that rewetting will result in low P concentrations, but this is not necessarily the case, because many rewetting studies also feature more drastic interventions (e.g. top soil removal, TSR) alongside raising the water table. Such activities aim to specifically minimise P exports to receiving waters, because long-term drainage stimulates the transformation and mobilisation of organic P to inorganic P which, following rewetting, results in elevated pore water P concentrations (Zak et al., 2008). Although P mobilisation after rewetting is positively correlated to the extent of peat degradation, the rewetting response is not straightforward because it is further modulated by iron (Fe) dynamics. If Fe:P in peat is >10 (or pore water Fe:P is >3) then large amounts of pore water P will be retained by Fe(III) hydroxides in surface peat layers, and P concentrations in downstream waters will be lower than pore water concentrations by an order of magnitude (Zak et al., 2010, Kaila et al., 2016). However, this retention is highly dependent on redox conditions, as highly reducing conditions can dissolve Fe(III) hydroxides, releasing phosphorus (Grunth et al., 2008). If Fe:P is low, then rewetting can potentially lead to high P leaching to receiving waters and promote eutrophic conditions (e.g. Kieckbusch & Schrautzer, 2007), and this can potentially be sustained for decades (Zak et al., 2017). Together, the P response will thus be modulated by Fe and the extent to which the water table is raised (e.g. partial vs full rewetting); in the peat layers below the water table Fe(III) can be reduced to Fe(II), thus releasing P into pore water, whilst aerobic surface peat layers might act as a P sink. If there is significant water exchange between the saturated layers and ditches, then this could act as a pathway for P mobilization to downstream waters, with the net effect being dependent on the balance between the saturated and unsaturated zone, and the extent of hydrological connectivity.

The severity of this leaching can be somewhat mitigated via partial (WT at 20cm), rather than full, rewetting (van der Laan et al., 2024). Alternatively, TSR may be a viable method to minimise post-rewetting P leaching (Zak et al., 2018) as well as lowering inorganic N concentrations and DOC fluxes (Emsens et al., 2015, Harpenslager et al., 2015, Quadra et al., 2023). This involves the removal of the degraded surface layer of peat (5-10cm is enough in some cases; Quadra et al., 2023), followed by immediate rewetting so further aerobic decomposition is prevented. However, TSR may not be desirable or appropriate at all sites, and careful investigations of peat characteristics and site hydrology/topography should take place before it is carried out (Zak et al., 2018). As noted above, TSR also involves the removal of a large quantity of peatland carbon from the site which, depending on the fate of this removed material, could effectively defeat the whole purpose of re-wetting to protect the peatland carbon store.

3.4. Sulfate (SO_4)

SO_4 is commonly present, and increasing in abundance, in many freshwaters, and can be derived from natural (e.g. mineral weathering, decomposition of organic matter) and anthropogenic (e.g. agricultural fertilisers, industrial emissions, acid mine drainage) sources (Zak et al., 2021). After the retrofitting of power plants for desulfurization in the 1980s, atmospheric SO_4 deposition significantly decreased, a trend that was reflected in freshwater systems (Zak et al., 2021). This decline in atmospheric S deposition rates has been identified as a potential explanatory factor for the increased DOC export from peatlands since the 1990s (Monteith et al., 2007). The proposed mechanism behind this hypothesis is that reduced acid deposition increases soil pH, which in turn enhances soil organic matter solubility (Monteith et al., 2007). Peatland drainage has been highlighted as being a primary driver of increasing SO_4 concentrations in surface waters because aeration results in the oxidation of iron sulfides to SO_4 and Fe(III) hydroxides (Zak et al., 2009). SO_4 is of concern because it can promote eutrophication in lowland peatland waters and enhance the production of aquatic toxins such as methylmercury (Lamers et al., 2014). In our data collation there were 21 observations of SO_4, with means of 380 (0.5-2000) and 40 (0.4-176) mg SO_4-S l^{-1} for pore and surface water respectively.

The maximum pore water concentrations were measured in a coastal German fen subject to occasional flooding during storms (Wen et al., 2018), so it is assumed that this represents marine SO_4 and is not due to peatland land use. There was no consistent pattern in how land use affected SO_4 concentrations (Fig. 9). Highest concentrations were for grasslands, whilst near-natural and rewetted sites often had lower concentrations, but there were exceptions, as follows. The near natural site with higher concentrations (23 mg SO_4-S l^{-1}) was a Dutch fen nature reserve where water was pumped onto site from an adjacent agricultural river (Whatley et al., 2014). Although the water was treated to remove P it presumably still contained SO_4. The rewetted fen with ~80 mg SO_4-S l^{-1} was situated in a region with high groundwater SO_4 concentrations derived from underlying sediments (Köhn et al., 2021). Such acid sulfate soils have been documented in UK fen peatlands in East Anglia (see also Section 3.5.1, p.**Error! Bookmark not defined.**) (Evans et al., 2016b; Burton and Hodgson, 1987) and are a particular problem in the Norfolk Broads (Broads).

Lab experiments on peat samples from a Canadian swamp have shown that drought/rewetting cycles can cause the release of SO_4 via oxidation of sulfides that accumulated during wet periods (Eimers et al., 2003). Thus, it is conceivable that fluctuating water levels in partly rewetted lowland peat could lead to flushes of SO_4 (Gosling and Baker, 1980), although in permanently wet sites this is less likely. In the longer term, anaerobic conditions should favour the retention of SO_4 via bacterial sulfate reduction to sulfides and organic sulfur compounds, which, at the landscape-scale, could make meaningful reductions to riverine SO_4 loads (see Zak et al., 2009).

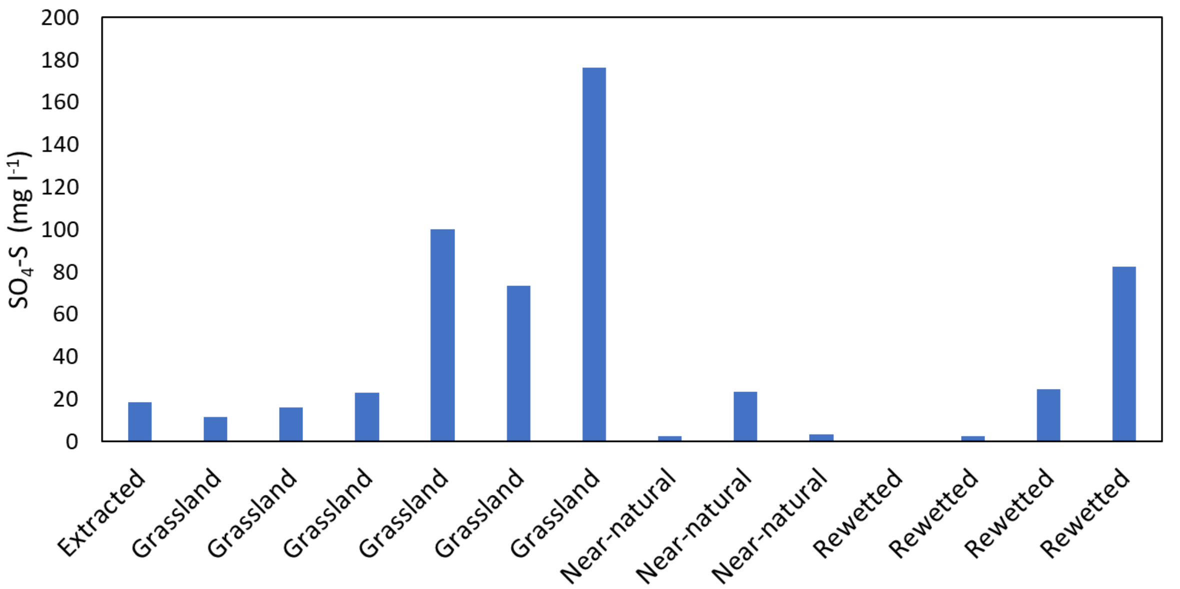

Figure 9. Mean SO_4-S concentrations in surface waters for 14 sites.

3.5. Other metals, anions and cations

This section considers the evidence for how concentrations of various metals, anions, and cations will respond to raised water tables in lowland peat.

Aluminium (Al)

There is potential for Al mobilisation following the development of acid sulfate soils (see 3.4) due to lowered pH. However, Al concentrations were scarce in our data collation. Al can have detrimental effects on aquatic organisms, with a safe exposure limit (for 95% of freshwater organisms) of 74.4 µg l^{-1} (Cardwell et al., 2018). In keeping with their Fe results, Zak et al. (2010) documented a substantial increase in peat Al content in heavily drained agricultural fens (2400 mg l^{-1}), compared to weakly drained (240 mg l^{-1}) and near-natural fens (120 mg l^{-1}). However, in a mesocosm study using cores from a drained Dutch grassland, extractable Al was approximately five times higher in cores under a flooded water table (4 cm above the surface) compared to a low water table (watered twice a week) and fluctuating water table (van de Riet, 2013).

Clearly, drainage alters internal Al biogeochemistry within the peat mass, as does rewetting. Whether this translates into elevated surface water concentrations, and whether rewetting might also cause increased Al leaching to receiving waters, remains an unknown. Lundin et al. (2017) reported concentrations 0.21-0.39 mg l^{-1} in drainage waters of two Swedish bogs used for peat extraction and found that 15 years of rewetting resulted in no concentration change, and Hendriks et al. (2024) found a mean of 0.34 mg l^{-1} in ten Dutch grassland ditches. In the NSI data aluminium concentrations were lowest in peat soils (Fig. 10) due to these soils generally containing less aluminosilicate minerals (Rawlins et al., 2012).

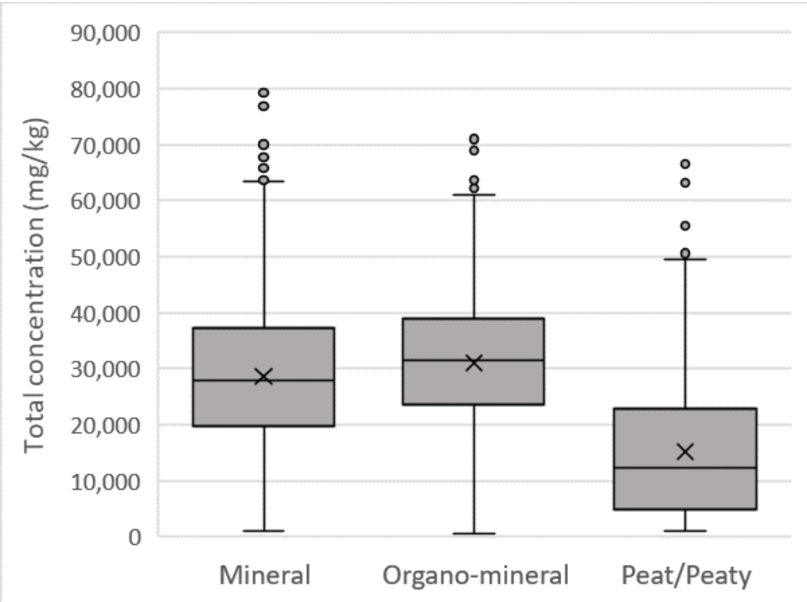

Figure 10. Boxplot showing total Al concentrations across soil classes. Medians are: mineral = 27,800 mg/kg, organo-mineral = 31,500 mg/kg, peat/peaty = 12,200 mg/kg.

Arsenic (As)

As was rarely measured or reported in lowland peat studies. As concentrations in freshwaters are typically less than 10 µg l^{-1} but can reach 1,100 µg l^{-1} in areas with high As pollution (e.g. due manufacturing) (Byeon et al., 2021). As is toxic to aquatic organisms through reduced reproduction and growth, immune disorders, cell and tissue damage, and cell death (Byeon et al., 2021). Evidence from Germany suggests that fen drainage for agricultural use, and subsequent degradation and subsidence, can lead to large As accumulation within surface soil layers (Bauer et al., 2008). This As was associated with, and could be mobilised by, Fe and organic matter. The authors suggested that this large pool of As was stable but could be slowly released if conditions became more reducing (e.g. rewetting), and they also highlighted that downstream monitoring in receiving waters should be a priority for future work. In a mesocosm study using cores from a near-natural German fen, the same group demonstrated that drought/rewetting cycles caused rapid As mobilisation and release, with concentrations up to 300 µg l^{-1} (Blodau et al., 2008). Considering the potential detrimental effects of As on aquatic ecosystems (Ghosh et al., 2022), future resources should be directed towards measuring concentrations of As in the surface waters of drained and rewetted peatlands. No difference was apparent in As concentrations in soils measured for the NSI.

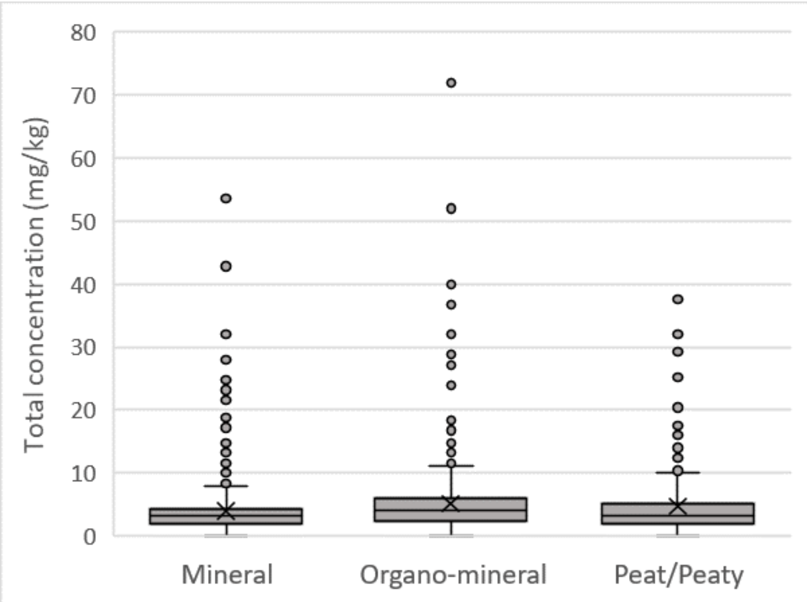

Figure 11. Boxplot showing total As concentrations across soil classes. Medians are: mineral = 3.2 mg/kg, organo-mineral = 4.0 mg/kg, peat/peaty = 3.2 mg/kg.

Calcium (Ca)

Ca concentrations were measured in 16/80 sites in the collated data, with pore and surface water means of 102 (4.5-211) and 87 (2.5-256) mg l^{-1}. There was a positive but non-significant correlation between Ca and pH (rho = 0.51, p = 16, n = 9) in surface water, hinting at the importance of groundwater connections in delivering Ca inputs to rich fens (Wassen et al., 1988).

Few studies have directly addressed the effects of changing water tables on Ca, but Zak et al. (2010) found a clear pattern in pore water Ca, whereby concentrations increased from near-natural < weakly drained < heavily drained fens. In Irish bog streams higher Ca (and Mg) concentrations were found for peat extraction sites compared to natural bogs, because peat removal and degradation lowers the bog surface and causes streams to intersect with the underlying minerotrophic fen peat (Pschenyckyj et al., 2023). Thus, depending on site type (bog vs fen) Ca concentrations can be either an indicator of degradation, or of natural functioning. Ca concentrations were, on average, lower for peat soils in the NSI (Fig. 12).

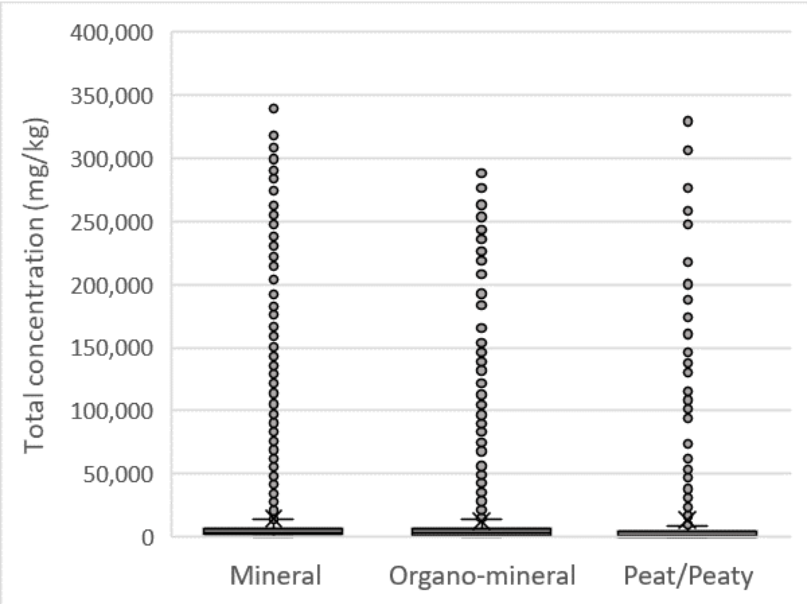

Figure 12. Boxplot showing total Ca concentrations across soil classes. Medians are: mineral = 3,469 mg/kg, organo-mineral = 3,452 mg/kg, peat/peaty = 1,108 mg/kg.

Copper (Cu)

Cu has been measured in UK lowland peats e.g. in minerotrophic and ombrotrophic areas of Cors Fochno, due to local Cu mining (Mighall et al., 2009), and at the raised bog Holcroft Moss near Manchester due to industrial Cu smelting (Garcés-Pastor et al., 2023). However, it has been suggested that it may be relatively immobile, at least in bog peat (Mighall et al., 2002), and there were no large differences in total or extractable Cu concentrations between NSI soil classes, although medians were slightly lower for peat soils (Figs. 13, 14).

Cu can also be present in agricultural additives (fertilisers, manures, effluents) (Nicholson et al., 2006) which could be one reason for higher extractable Cu concentrations in lowland peat in the NSI data (Fig. 15), especially because Cu can bind with organic matter in lowland peatlands (Rawlins et al., 2012). Despite this, measurements of Cu concentration in surface waters are almost entirely absent from the lowland peat evidence base. One study that measured water chemistry in a ditch used for rewetting a German grassland/arable fen reported a zero concentration of Cu (Bockermann et al., 2024). More data are clearly needed to understand Cu dynamics in lowland peat waterways.

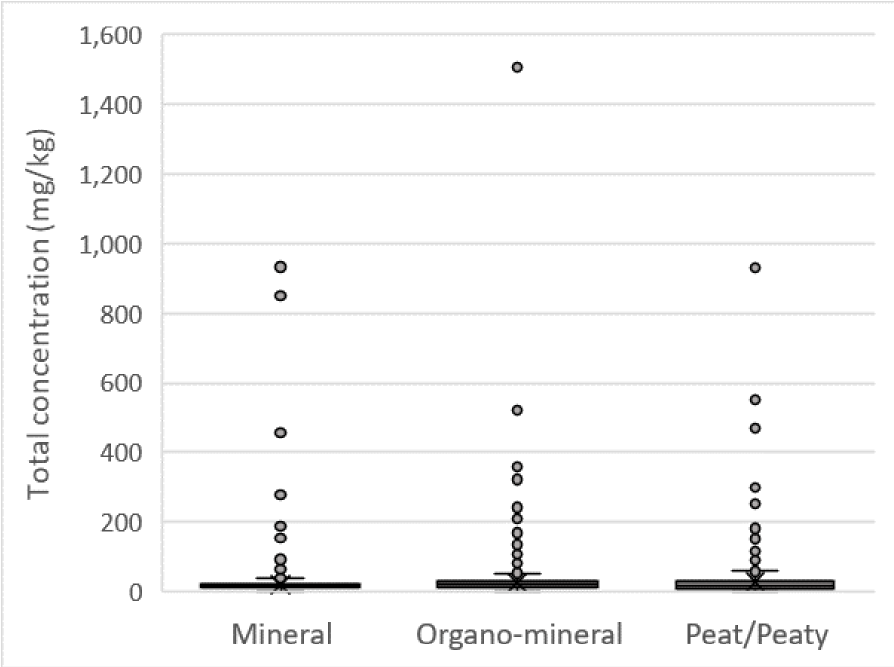

Figure 13. Boxplot showing total Cu concentrations across soil classes. Medians are: mineral = 17.2 mg/kg, organo-mineral = 20.2 mg/kg, peat/peaty = 16.6 mg/kg.

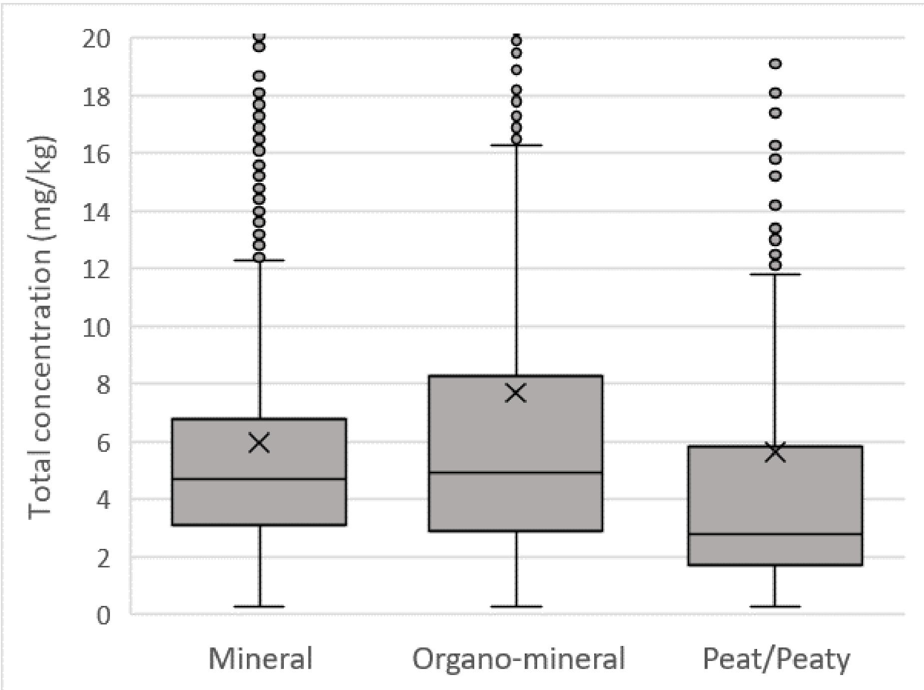

Figure 14. Boxplot showing extractable Cu concentrations across soil classes (outliers not shown for clarity). Medians are: mineral = 4.7 mg/kg, organo-mineral = 4.9 mg/kg, peat/peaty = 2.8 mg/kg.

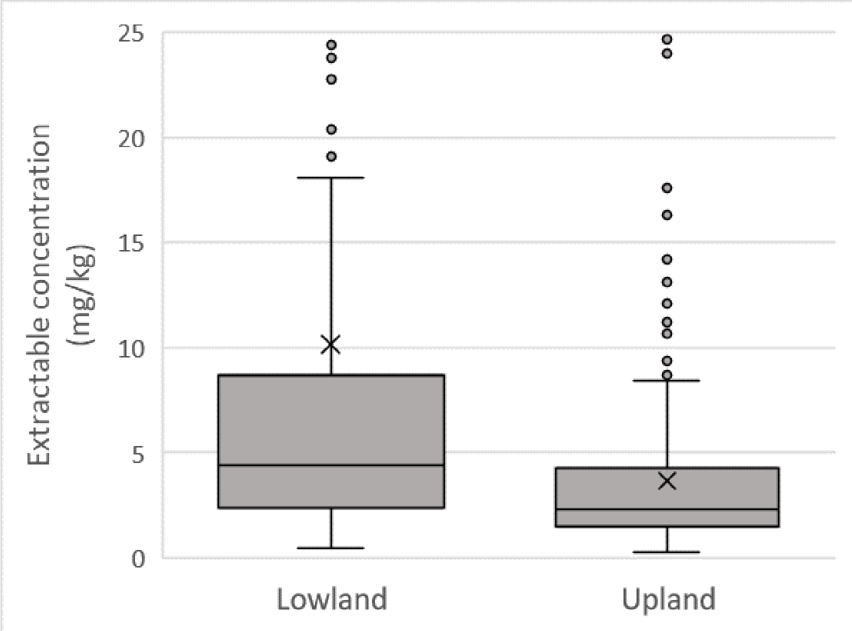

Figure 15. Boxplot showing extractable Cu concentrations for peat classes (outliers not shown for clarity). Medians are: lowland = 4.4 mg/kg, upland = 2.3 mg/kg.

Iron (Fe) and ochre

As discussed in Section 3.3, Fe dynamics are heavily implicated in the P response to rewetting and are therefore important when considering eutrophication. Additionally, Fe contributes to water colour (Kritzberg & Ekström, 2012) and is therefore relevant for drinking water treatment. Fe also has direct and indirect effects on aquatic ecosystem functioning, including reduction in periphyton and benthic invertebrate abundance, and damage to fish gills (Vuori, 1995). Concentrations of Fe have been increasing for several decades in European and UK surface waters (Björnerås et al., 2017, Neal et al., 2008) and, in part, peatland drainage is implicated in these increases (Heikkinen et al., 2022). In the collated data, Fe concentrations were available for 13 sites, with pore and surface water means of 15 (0.3-35) and 1.3 (0.1-3.6) mg l^{-1}. Zak et al. (2010) show a clear effect of drainage on Fe concentrations in a study of 17 German and Polish fens; peat Fe content was 6.2 mg dm^{-3} in heavily drained agricultural fens, compared to 0.5 and 0.1 mg dm^{-3} in weakly drained non-agricultural fens and near-natural fens respectively. This translated into anoxic pore water Fe concentrations of 47 mg l^{-1} for heavily drained sites, compared to 0.6 and 0.3 mg l^{-1} for weakly drained and near-natural fens.

The higher Fe concentrations in drained than undrained sites is due to the high amounts of redox sensitive Fe bound in peat, in particular in highly decomposed peat, that can be released under redox conditions (Zak et al., 2010). A mesocosm study using cores from four Finnish mires (two bog, two fen) and two Irish blanket bogs found that raising water tables promoted the release of Fe but only at nutrient-rich sites (Kaila et al., 2016), and Dijk et al. (2004) showed that rewetted Dutch grassland fens led to elevated pore water Fe concentrations. It is therefore conceivable that rewetting UK lowland peats could lead to increased Fe in surface waters, but this will be dependent on site histories (e.g. degree of drainage-induced degradation) and types (e.g. fen or bog). In, fens which often received Fe-rich groundwater, rewetting often mobilizes Fe due to changes in redox conditions. Conversely, ombrotrophic bogs typically have lower iron content, and rewetting may result in less Fe mobilization. In the NSI data, total Fe concentrations were lowest for peat soils (Fig. 16).

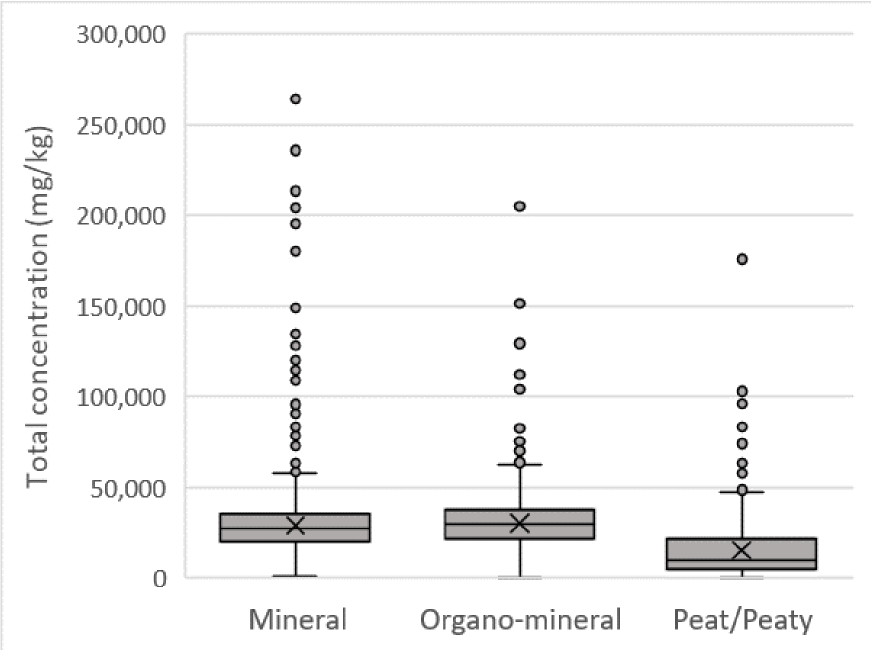

Figure 16. Boxplot showing total Fe concentrations across soil classes. Medians are: mineral = 27,100 mg/kg, organo-mineral = 29,600 mg/kg, peat/peaty = 9,800 mg/kg.

Another Fe issue in drained peatlands is the formation of ochre. When pyritic (FeS_2) soils are drained and oxygenated SO_4 is released, pH falls, and ochre forms (Bloomfield, 1972). This is of concern for several reasons: aquatic life can be directly harmed by the highly acidic conditions (pH 2-3) and by associated increased solubility of aluminium under low pH; low pH can have negative effects on crop yields; and ochre can physically clog drainage systems and corrode metallic water management infrastructure (pumps, culverts, abstraction points) (Burton & Hodgson, 1987, Cranfield Environment Centre, no date, Page et al., 2020).

Ecologically, precipitation of ochre can smother ditch and river beds, aquatic plants, and gills of aquatic animals; and in severe cases can render the water column virtually opaque, with severe consequences for the aquatic biota. For the UK, the potential for acid sulfate soils and ochre to develop is particularly high on the fen peat soils of East Anglia, Norfolk and Lincolnshire (Dawson et al., 2010, Cranfield Environment Centre, no date). There is an absence of data and reports concerning the effects of rewetting or raising water tables on ochre formation. High surface water ochre concentrations (6.7 mg l^{-1} Fe) are a major concern due to the drainage of acid sulfate soil peatlands in the Broads (Brograve catchment), where raising the water levels was considered as a potential remediation solution (Harding et al., 2005). Pyritic soils only become acid sulfate soils once drained due to oxygen ingress, and thus we assume that rewetting will reverse this and “shut off” the FeS_2 oxidation process, thus preventing ochre formation.

Mercury (Hg)

Freshwater pollution by Hg is a concern due to its high toxicity, environmental persistence, and potential for bioaccumulation through trophic chains (Rodrigues et al., 2013). Hg has been relatively well-studied in boreal forests and mires. In these ecosystems, Hg dynamics are strongly coupled to those of organic matter (Eklöf et al., 2012), so any land use change that affects organic matter (e.g. peatland drainage) will likely also affect Hg (Grigal, 2003). A study of 22 sites in Latvia, including near-natural, extracted, forested and agricultural peatlands reported total Hg concentrations in peat from < 1 μg kg^{-1} to ~200 μg kg^{-1}. For drained sites Hg concentrations increased towards surface layers, whilst for near-natural bogs no change with depth was observed (Bārdule et al., 2022). Preliminary results following the rewetting of two forestry drained fens in northern Sweden show significant increases in total Hg concentrations in streams from 0.0041 to 0.0057 μg l^{-1} (Laudon et al., 2023). These values are well below the safe exposure limit for 95% of freshwater organisms of 3.16 μg l^{-1} (Rodrigues et al., 2013).

In the UK some lowland rivers have suffered direct Hg pollution (e.g. River Yare, Norfolk) from effluent from sewage works (Edwards et al., 1999) but aquatic concentrations in many waterways are low (Robson & Neal, 1997). It is likely that indirect Hg pollution via atmospheric deposition has affected lowland peatlands (Blair et al., 2009), and peatlands in general, as demonstrated by higher Hg concentrations in peat soils in the NSI (Fig. 17). However, it is currently an unknown if, and how, these Hg stocks will be affected or mobilised by raising water tables.

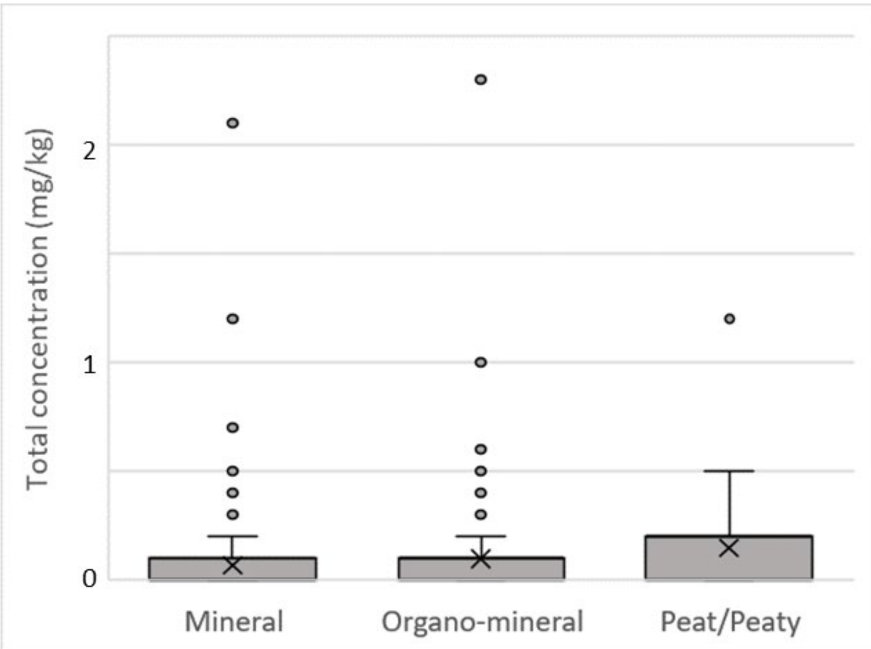

Figure 17. Boxplot showing total Hg concentrations across soil classes. Medians are: mineral = 0.0 mg/kg, organo-mineral = 0.1 mg/kg, peat/peaty = 0.2 mg/kg.

Lead (Pb)

Similarly to Cu (see earlier in Section 3.5), Pb, derived from industrial sources, can also accumulate in fen and bog peat (Osborne et al., 2024). For the UK, the majority of research examining peatland Pb dynamics has focused on upland bogs (e.g. Rothwell et al., 2010), due to their proximity to large industrial cities, and these ecosystems show large Pb concentrations in the NSI data (Figs. 18-20) (Rawlins et al., 2012). Higher surface water Pb concentrations are also associated with industry (Robson & Neal, 1997), whilst pore water Pb concentrations can be small at sites that are distant from industry (e.g. frequently below detection limits for Flow Country blanket bogs; Gaffney et al., 2024).

There is a lack of research concerning effects of drainage/rewetting on Pb, but Broder & Biester (2015) found that Pb concentrations correlated with DOC in drainage waters of a German ombrotrophic mountain bog, suggestive of Pb-DOC binding. However, drought/rewetting (following storm events) caused this correlation to break down, and led to a flushing effect of Pb into surface waters. The authors suggested that this was due to rising water tables mobilising additional pools of Pb in surface layers. Assuming these findings translate to lowland peat, it is possible that rewetting drained peatlands could lead to increased concentrations into surface waters, but this will be site specific (e.g. will only apply to sites with a history of Pb pollution) and could either be a short-term pulse (e.g. Nabuyanda et al., 2019) or a sustained effect.

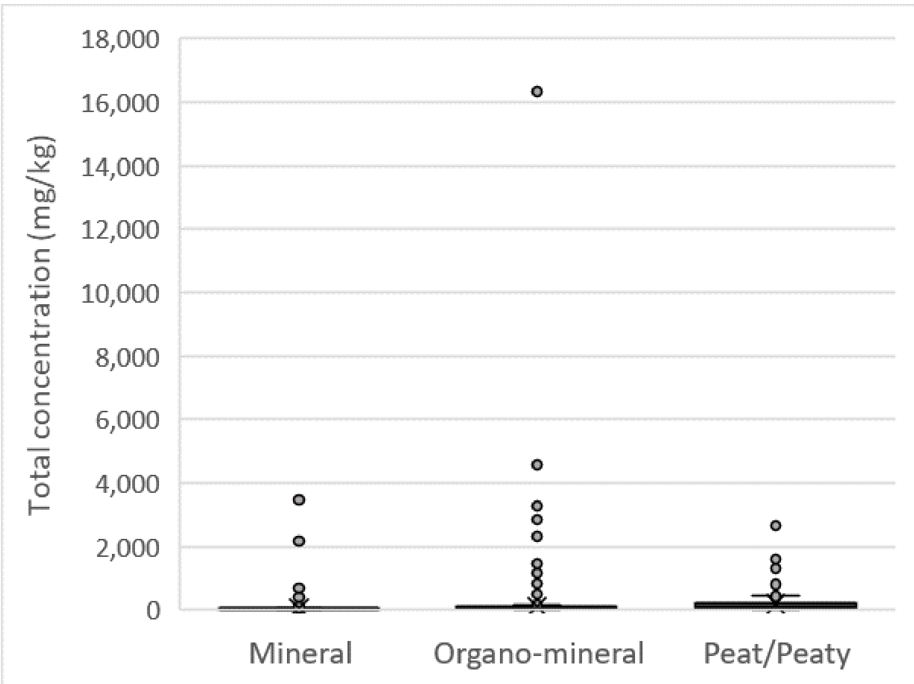

Figure 18. Boxplot showing total Pb concentrations across soil classes. Medians are: mineral = 33 mg/kg, organo-mineral = 50 mg/kg, peat/peaty = 119 mg/kg.

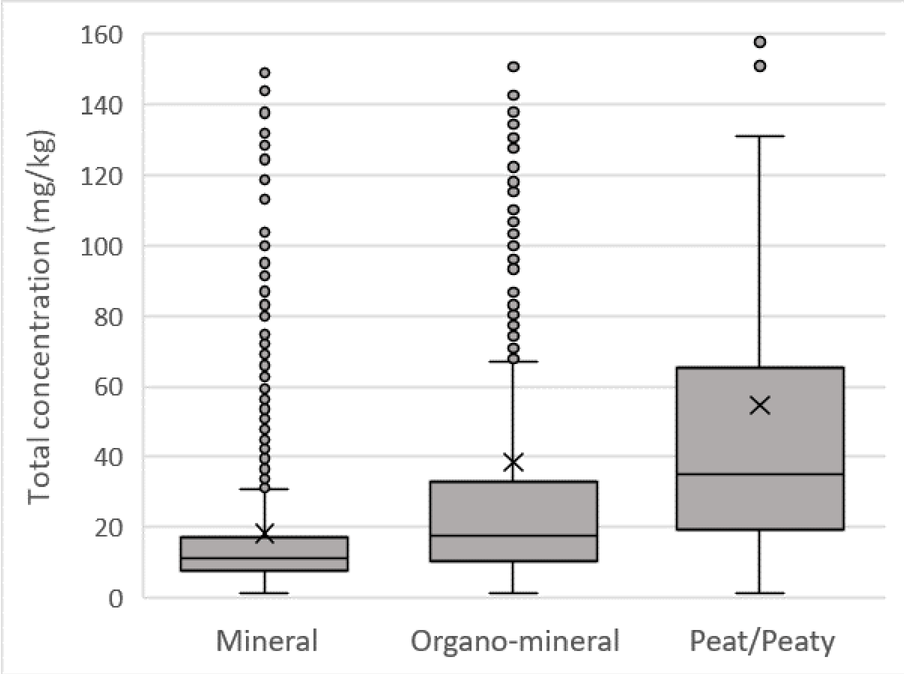

Figure 19. Boxplot showing extractable Pb concentrations across soil classes (outliers not shown for clarity). Medians are: mineral = 11.1 mg/kg, organo-mineral = 17.4 mg/kg, peat/peaty = 34.9 mg/kg.

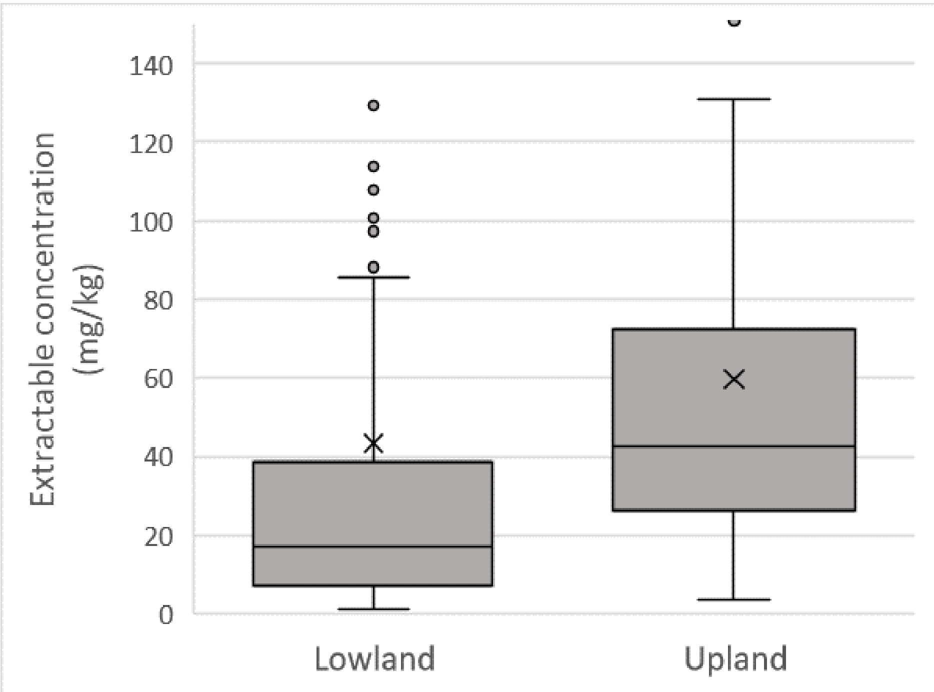

Figure 20. Boxplot showing extractable Pb concentrations for peat classes (outliers not shown for clarity). Medians are: lowland = 17.3 mg/kg, upland = 42.5 mg/kg.

Magnesium (Mg)

In our data collation, Mg was measured for 13/80 sites with mean concentrations of 9.7 (3.8-21) and 11 (0.7-25) mg l^{-1} for pore and surface water. As for Ca (see earlier in section

3.5), Mg in lowland fen peatlands is indicative of natural groundwater inputs (Bootsma & Wassen, 1996) but can also be an indicator of degradation in raised bogs, where surface loss and subsidence expose underlying fen peat (Pschenyckyj et al., 2023). This possibly explains the higher extractable Mg concentrations in lowland peat in the NSI (Fig. 21). Mg and Ca were significantly related in the collated data (rho = 0.71, p = 0.02).

Few studies address the effects of drainage and rewetting on Mg, although Lundin et al (2017) reported higher peat Mg concentrations following rewetting of an extracted nutrient-rich Swedish bog. In the NSI data peatlands had lower concentrations of total Mg compared to other soil classes (Fig. 22) but similar extractable Mg (Fig. 23).

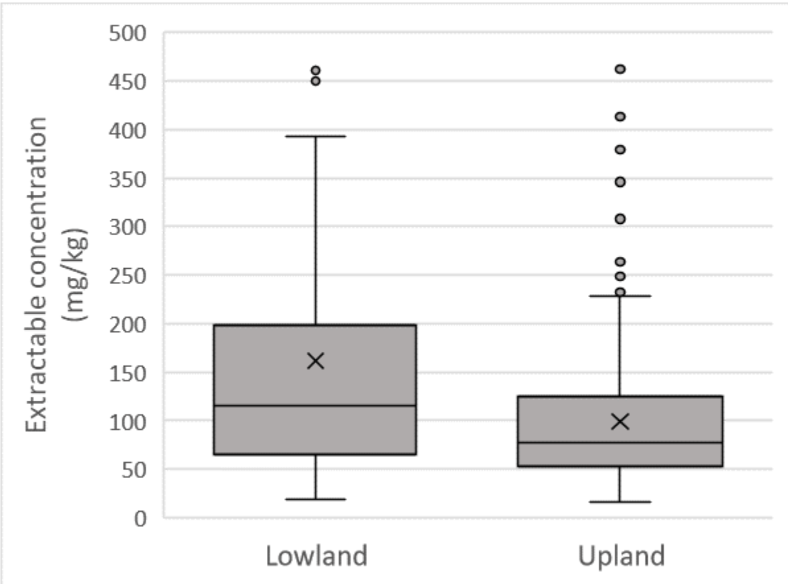

Figure 21. Boxplot showing extractable Mg concentrations for peat classes (outliers not shown for clarity). Medians are: lowland = 115 mg/kg, upland = 78 mg/kg.

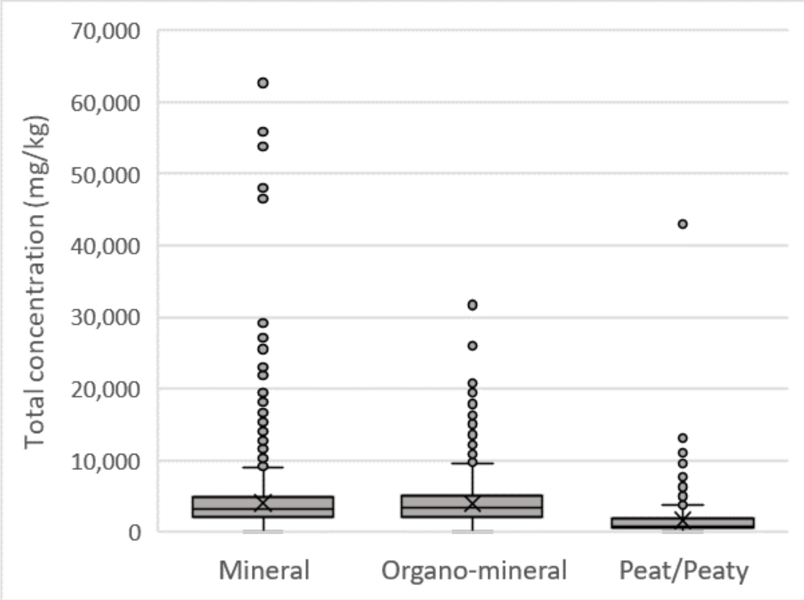

Figure 22. Boxplot showing total Mg concentrations across soil classes. Medians are: mineral = 3,130 mg/kg, organo-mineral = 3,360 mg/kg, peat/peaty = 860 mg/kg.

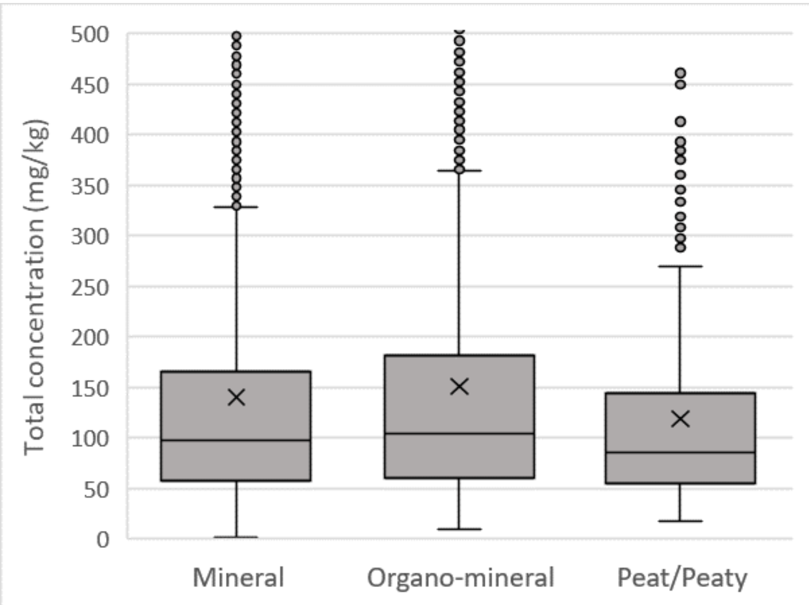

Figure 23. Boxplot showing extractable Mg concentrations across soil classes (outliers not shown for clarity). Medians are: mineral = 97 mg/kg, organo-mineral = 104 mg/kg, peat/peaty = 85 mg/kg.

Manganese (Mn)

Mn is an essential element for organisms, however, in excess (e.g. above 0.15 mg l^{-1}, the naturally occurring concentration in freshwaters) it can cause toxic effects such as impaired functioning of the nervous system (Niemiec and Wisniowska-Kielian, 2015). Mn was occasionally measured and reported in lowland peat studies, with concentrations of 1-3 mg l^{-1} in pore water of coastal German fens (Wen et al., 2018) and 0.08-0.36 mg l^{-1} in surface waters (Lundin et al., 2017, Hendriks et al., 2024, Bockermann et al., 2024).

Lundin et al. (2017) showed post-rewetting decreases from 1.09 to ~0.2 mg l^{-1} and from 0.14 to 0.07 mg l^{-1} in two extracted Swedish mires.

Mn concentrations vary according to redox potential, but this relationship in fen peat is not always clear, due to other determinands (e.g. pH, carbonates) also influencing Mn (Boonman et al., 2024). In the NSI, peat soils had lower total and extractable Mn (Figs. 24, 25), and lowland peats had higher extractable Mn compared to upland peats (Fig. 26).

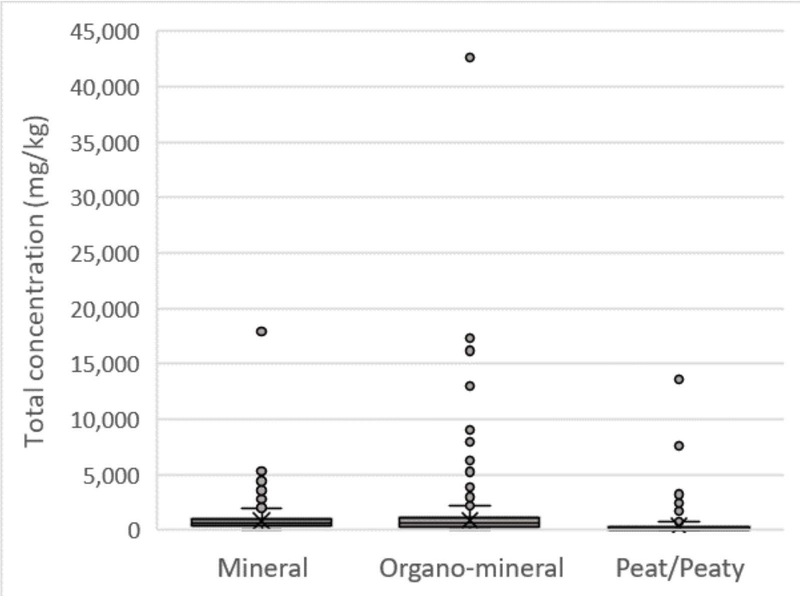

Figure 24. Boxplot showing total Mn concentrations across soil classes. Medians are: mineral = 631 mg/kg, organo-mineral = 599 mg/kg, peat/peaty = 76 mg/kg.

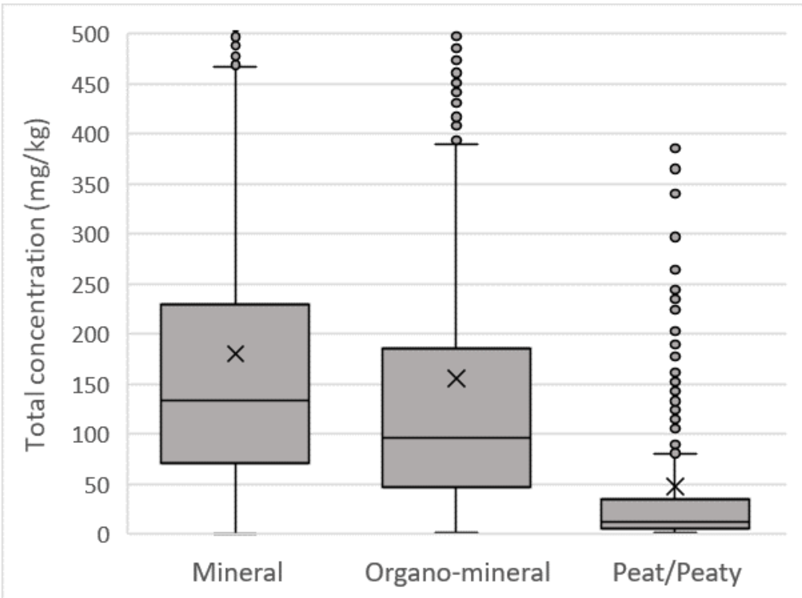

Figure 25. Boxplot showing extractable Mn concentrations across soil classes (outliers not shown for clarity). Medians are: mineral = 134 mg/kg, organo-mineral = 96 mg/kg, peat/peaty = 12 mg/kg.

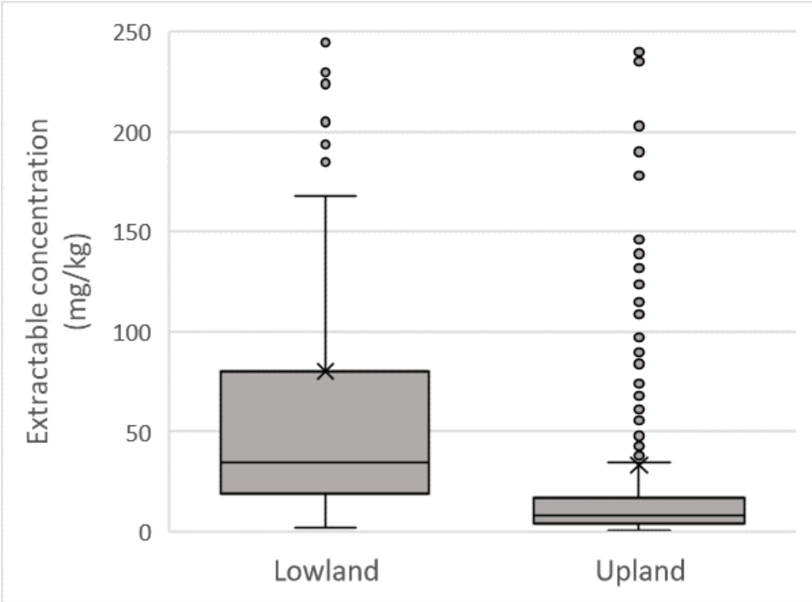

Figure 26. Boxplot showing extractable Mn concentrations for peat classes (outliers not shown for clarity). Medians are: lowland = 34.5 mg/kg, upland = 8.0 mg/kg.

Potassium (K)

K concentrations, mainly in surface waters, were captured by our data collation, where the mean concentration was 9.0 (0.46-45) mg l^{-1}. K correlated strongly with Ca (rho = 0.8, p = 0.01) and may be derived from groundwater (Giller & Wheeler, 1986) or fertiliser inputs (Bootsma & Wassen, 1996).

Some studies have found increases in pore water and extractable K following fen rewetting (Koerselman et al., 1993, van der Laan et al., 2024), with increased leaching to surface waters (van der Riet et al., 2013). Others have not (Venterink et al., 2002); so it seems likely that the response to rewetting will be site dependent and potentially linked to e.g. clay particle adsorption (van der Laan et al., 2024).

Sodium (Na)

Na concentrations were almost exclusively reported for surface waters with a mean of 17.1 (2.8-58) mg l^{-1}. Na correlated strongly with chloride concentrations (rho = 0.93, p = 0.003) suggesting that sea salt deposition (or occasionally road salting, see Pschenyckyj et al., 2023) is a major controller of Na in lowland peatlands.

The NSI data showed no clear difference in soil Na concentrations between soil classes. For two Swedish extracted bogs, Lundin et al. (2017) found a post-rewetting Na decrease for one site (from 5.4 to 3.5 mg l^{-1}) but no change at a second. For small streams in near-natural and degraded Irish bogs Pschenyckyj et al. (2023) found no significant difference in Na concentrations.

Na can also be present due to saline intrusion associated with drainage of lowland agricultural peats (Simpson et al., 2011), particularly in parts of the Broads, as improved drainage increases the hydraulic gradient between the sea and drains and (potentially) improves the hydraulic connectivity with the underlying aquifer. Raising water tables within lowland agricultural peats is expected to reduce (but not stop) the saline inflow into the drainage systems over time (Holman et al., 2013), but it is uncertain whether the reduced mass flux of salt will lead to reduced salt concentrations in drainage systems due to the associated changes in drain hydrology and water balance.

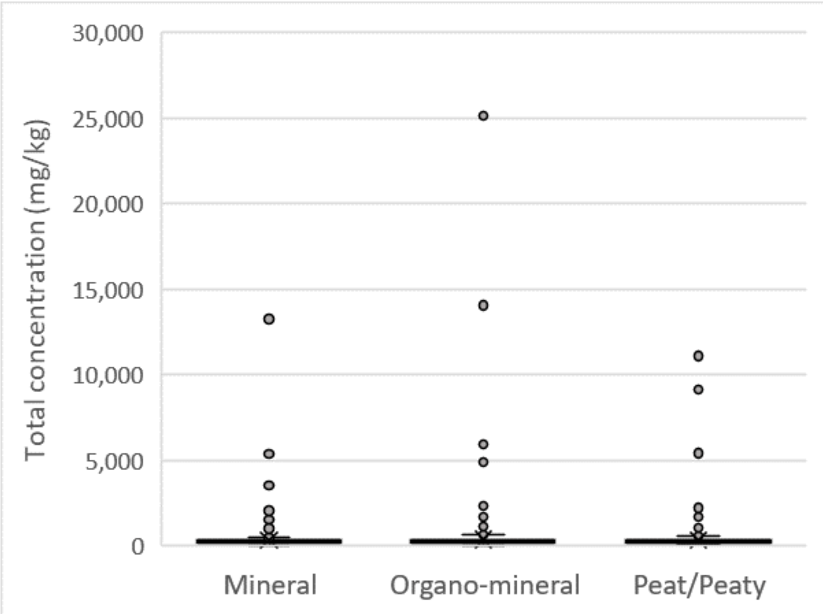

Figure 27. Boxplot showing total Na concentrations across soil classes. Medians are: mineral = 241 mg/kg, organo-mineral = 255 mg/kg, peat/peaty = 218 mg/kg.

3.6. Pesticides and herbicides

In general, peatlands and pesticide/herbicide dynamics appears to be an under-researched area. In their peatland conservation synthesis, Taylor et al. (2020) identify only one partially relevant study; that of Barry et al. (2008) where herbicides were used to reduce invasive species in Ohio fens (although Barry et al. only mention herbicides in passing). Literature searches reveal a handful of other peatland studies, as follows. Bortolozo et al. (2016) show that agricultural pesticides can accumulate within the pore water and soil of a farmed upland peatland in Brazil. In a study of a 50 km^2 German catchment, which included drained peat and gley soils, Ulrich et al. (2021) found herbicides in the water of under-field drains, streams, shallow groundwater and drinking wells. Similarly, Bech et al. (2023) measured pesticides in a stream in a 106 km^2 lowland Danish catchment that included peat soils. However, these studies do not provide evidence relevant to peatland land management. Effects of lowland peat drainage and rewetting on pesticides and herbicides is currently a clear knowledge gap.

A general review by Gramlich et al. (2018) of agricultural drainage on hydrology and substance flux includes pesticides and herbicides, aims to disaggregate effects between organic and mineral soils. The authors suggest that concentrations in surface waters of drained landscapes peak soon after application following rainfall, but that exact concentrations into receiving waters will be determined by site-specific and substance-specific characteristics.

The majority of references cited within Gramlich et al. (2018) focus on mineral soils, but some are relevant for this review. For instance, in a study of small Swiss gley soil catchments it was found that wetter sites had higher herbicide losses due to surface runoff and preferential flow to under-field drains (Leu et al., 2004). This might suggest that raising water tables, whilst continuing to apply pesticides, could result in higher losses to surface waters. However, pesticides and herbicides can also be adsorbed by organic matter (Vereecken, 2005). For example, a study in southern Norway reported lower pesticides losses from a field with loam/silt loam (3.2% organic carbon) compared to two fields with silty clay loams (1.1-1.5% organic carbon) (Riise et al., 2004). Contrary to this, a synthesis found positive correlations between organic carbon content and pesticide losses, but the highest organic carbon content was only 4.5% (Brown & van Beinum, 2009). Clearly, losses into drainage waters will be substance specific and depend on sorption coefficients (Jones et al., 2000). Gramlich et al. (2018) point to a range of other site characteristics and management interventions that can affect pesticide and herbicide fluxes, including soil clay content, drainage intensity, ploughing, and time between application and rainfall.

In a European review of pesticides in under-field drains, concentrations ranged from zero to 1570 µg l^{-1}, and 55/97 studies recorded losses < 0.1% compared to field application, and 28/97 were 0.1-1% (Brown & van Beinum, 2009). However, the majority of studies are for mineral soils and so whether their findings are applicable to lowland peatlands is unknown.

3.7. Emerging contaminants

Emerging contaminants are briefly considered in this section, including per- and polyfluoroalkyl substances (PFAS), persistent organic pollutants (POPs), micropollutants and microplastics. The knowledge base for these emerging contaminants is extremely limited for peatlands, and for wetlands as a wider ecosystem class (Overton et al., 2023). A study of a drained arable fen in Germany used 1:10 mix of treated wastewater to surface water for rewetting purposes, and then measured ditch and pore water for 67 micropollutants (including artificial sweeteners, pharmaceuticals and industrial compounds).

Of the 31 micropollutants that were detected in the treated wastewater, 15 were present in pore water and none were present in ditch water, with the exception of one sample where diclofenac (a pharmaceutical) was measured at a concentration of 0.11 µg l^{-1}. The authors suggested that diclofenac cannot bind with organic matter and is thus persistent under anaerobic conditions. They concluded by stating that peatlands could buffer receiving waters against the negative effects of micropollutants (Maasen et al., 2017).

Others have noted that some micropollutants (e.g. pharmaceuticals, polycyclic aromatic hydrocarbons, polychlorinated biphenyls) can bind to peat-derived organic matter (e.g. Aristilde & Sposito, 2013) and be retained within peat soils (particularly bogs) (Berset et al., 2001, Dreyer et al., 2005, Richter et al., 2015), and potentially broken down (Overton et al., 2023). In contrast, for PFAS, it has been shown that these compounds are mobile within the peat mass, and are not well retained (Dreyer et al., 2012). Microplastics accumulate within peat soils and can be derived from local sources or atmospheric deposition (Allen et al., 2021, Nguyen et al., 2022). This accumulation may be particularly pronounced in lowland peatlands where agricultural amendments (e.g. sewage sludge) are applied (Nizzetto et al., 2016), and microplastics within drained agricultural soils may then be leached to surface waters (Bigalke et al., 2022).

Although concentrations of emerging contaminants are likely to be affected by peatland water table management, at present a lack of studies and data means no firm conclusions can be made.

4. Conclusions

The findings of this review can be used to inform field monitoring programmes through considerations on what types of parameters to measure, the effects of peatland type and land cover, as well as the nature of rewetting.

Of the water quality variables included in this review, the most commonly measured variable was DOC concentration (74% of sites), followed by nutrients including NH_4 (45%) and NO_3 (41%). Other more commonly measured variables (measured at 20-35% of sites) included DIC concentration, PO_4, DOC flux, SO_4, DIC flux, Cl, and Ca. Less commonly measured variables, measured at only 1-5% of sites, included Cu, Br, Zn; Si, Al, and F. Slightly more common (measured at 7 to 16% of sites) were Mn, NO_2, Na, POC flux, K, TN, POC concentrations, Mg, Fe, and TP.

This data availability reflects current relative understanding of these variables. For instance, the effects of drainage (increases) and rewetting (decreases) on DOC concentration and fluxes are well understood for peatlands. While POC and DIC concentrations and fluxes have less meaningful contributions to peatland carbon budgets and are less essential to measure in the context of rewetting.

It was found that land cover influenced DOC, with higher pore water DOC concentrations at grassland sites compared to rewetted and near-natural sites, but no effect was observed for surface water. This demonstrates the importance of sampling both surface and pore waters, while surface water samples were collected more frequently (24 studies) than pore water samples (14 studies).

The collated data had an even distribution of bog (36) and fen (37) sites. The distinction between bogs and fens was important for DOC, where there is greater uncertainty extrapolating from all peatland classes to fens, due to their hydrology. Thus, the response of DOC (and possibly other variables) to rewetting is likely to differ between peatland types and should therefore be an important consideration for sampling.

Besides peatland type, another important consideration for rewetting is soil type, and the potential for the development of acid sulfate soils, which can mobilise Al and ochre. Fen peat soils in East Anglia, Norfolk and Lincolnshire are at particular risk of acid sulfate soil development Dawson et al., 2010, Cranfield Environment Centre, no date).

Nutrients stimulating eutrophication (N, P, SO_4) were frequently measured, with average surface water TN and PO_4-P concentrations exceeding ecological limits (WFD-UKTAG, 2016). These nutrients are important to measure, as rewetting can alter their dynamics (e.g. lower TN, mobilise P, increase SO_4).

Considering the water source for rewetting and its nutrient concentrations is another important variable to note. To mitigate post-rewetting N and P mobilisation into drainage waters, a partial rewetting (WT 20 cm below the surface) has been suggested (van der Laan et al., 2024). Additionally, TSR has been used to minimise post-rewetting N, P, and

DOC increases (Zak et a., 2018, Emsens et al., 2015, Harpenslager et al., 2015, Quadra et al., 2023), although it may have undesirable consequences for carbon storage.

Compared to C and nutrients, there were notably fewer studies measuring metals, anions and cations in peatlands following rewetting. These chemicals can have ecological implications including harmful effects on aquatic ecosystem functioning.

Based on the findings of the review, of notable concern are Al, As, Cu, Fe, Hg, Pb due to their potential mobilisation following rewetting and risks to aquatic ecosystems. Pesticides, herbicides, and other novel contaminants are another under-researched area in peatland rewetting.

The Environment Agency proposes a list of priority chemicals under the Water Framework Directive, based on their potential exposure in the aquatic environment (based on use and monitoring data) and their hazard to aquatic life (based on persistence, bioaccumulation and toxicity) (Wilkinson et al., 2007). Sixteen priority chemicals were identified for environmental water quality standards development. Several of these priority chemicals overlap with those included in this report: Al, As, Cl, Cu, Fe, Mn, Zn. Where resources allow, the other WFD priority chemicals should also be considered in rewetting studies: ammonia, chromium, cyanide, cypermethrin, diazinon, permethrin, toluene, tetrachloroethane, and phenol.

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List of abbreviations

AAS	[Hydride] Atomic Absorption Spectrometry
DEFRA	Department for Environment, Food & Rural Affairs
DIC	Dissolved inorganic carbon
DOC	Dissolved organic carbon
EA	Environment Agency
ICP	Inductively Coupled Plasma Emission Spectrometry
IPCC	Intergovernmental Panel on Climate Change
ND	Not detected
NSI	National Soil Inventory (Cranfield University)
POC	Particulate Organic Carbon
TN	Total nitrogen
TP	Total phosphorus
TSR	Top soil removal
UKCEH	UK Centre for Ecology & Hydrology
WHO	World Health Organisation
WT	Water table

5. Appendices

Table 1. Country, land cover and peat type for each data source. 'n.d.' indicates data not determined.

Country	Land cover	Peatland type	Reference
UK	Near-natural, Arable, Extracted, Grassland, Rewetted	Fen and bog	Evans et al., 2016b
UK	Near-natural	Fen	Peacock et al., 2015
UK	Near-natural	Fen	Menichino et al., 2019
UK	Near-natural	Bog	Ridley, 2014
UK	Near-natural	Fen	Giller & Wheeler, 1986
UK	Rewetted	Bog	Bottrell et al., 2004
UK	Near-natural, Afforested	Bog	Howson et al., 2021, Shah & Nisbet, 2019
Ireland	Rewetted	Bog	Lally et al., 2012
Ireland	Rewetted	Bog	Regan et al., 2020
Ireland	Drained, Rewetted	Bog	Wilson et al., 2022
Ireland	Grassland	Bog	Renou-Wilson et al., 2014
Ireland	Extracted, Near-natural	Bog	Pschenyckyj et al., 2023
Netherlands	Grassland	n.d.	Vermaat et al., 2016
Netherlands	Grassland	n.d.	Hendriks et al., 2024
Netherlands	Grassland, Near-natural	n.d.	Veraart et al., 2017

Country	Land cover	Peatland type	Reference
Netherlands	Grassland	Fen	Best et al., 1995
Netherlands	Grassland, Near-natural	Fen	Whatley et al., 2014
Germany/Poland	Rewetted, Near-natural	Fen	Zak et al., 2010
Germany	Rewetted	Fen	Zak et al., 2018
Germany	Grassland	Fen	Tiemeyer et al., 2007
Germany	Grassland	Fen	Scholz & Trepal, 2004
Germany	Grassland	Bog	Frank et al., 2017
Germany	Grassland, Rewetted, Near-natural	Bog	Frank et al., 2014
Germany	Grassland	Fen	Tiemeyer & Kahle, 2014
Germany	Rewetted	Bog	Welpelo et al., 2024
Germany	Near-natural, Paludiculture	Bog	Oestmann et al., 2022
Germany	Rewetted, Arable	Fen	Höll et al., 2009; Fiedler et al., 2008
Germany	Rewetted	Fen	Unger et al., 2021; Wen et al., 2018
Germany	Grassland, Rewetted	Fen	Köhn et al., 2021
Germany	Paludiculture	Bog	Daun et al., 2023, Vroom et al., 2020
Germany	Grassland	Fen	Bockermann et al., 2024
Sweden	Rewetted	Bog	Lundin et al., 2017

Country	Land cover	Peatland type	Reference
Sweden	Near-natural	Fen	Wallin et al., 2015
Sweden	Cropland	Fen	Berglund & Berglund, 2011
France	Near-natural	n.d.	Bougon et al., 2011

Table 2. Metals of interest within the National Soil Inventory and analytical methods. 'n.d.' indicates not determined.

Metal	Total concentration (mg/kg)	Extractable concentration (mg/kg)
Aluminium	Inductively Coupled Plasma Emission Spectrometry (ICP) in an aqua regia digest	n.d.
Arsenic	Hydride Atomic Absorption Spectrometry (AAS), extracted into hydrochloric acid after digestion with nitric acid and ashing with magnesium nitrate	n.d.
Calcium	Inductively Coupled Plasma Emission Spectrometry (ICP) in an aqua regia digest	n.d.
Copper	Inductively Coupled Plasma Emission Spectrometry (ICP) in an aqua regia digest	Inductively Coupled Plasma Emission Spectrometry (ICP) after shaking 10ml of soil with 50ml of 0.05M EDTA at pH 7.0 for 1h at 20 deg. C and then filtering
Iron	Inductively Coupled Plasma Emission Spectrometry	n.d.

Metal	Total concentration (mg/kg)	Extractable concentration (mg/kg)
	(ICP) in an aqua regia digest	
Lead	Inductively Coupled Plasma Emission Spectrometry (ICP) in an aqua regia digest	Inductively Coupled Plasma Emission Spectrometry (ICP) after shaking 10ml of soil with 50ml of 0.05M EDTA at pH 7.0 for 1h at 20 deg. C and then filtering
Magnesium	Inductively Coupled Plasma Emission Spectrometry (ICP) in an aqua regia digest	shaking 10ml of air dry soil with 50ml of 1.0M ammonium nitrate for 30mins, filtering and then measuring the concentration by flame photometry
Manganese	Inductively Coupled Plasma Emission Spectrometry (ICP) in an aqua regia digest	Inductively Coupled Plasma Emission Spectrometry (ICP) after shaking 10ml of soil with 50ml of 0.05M EDTA at pH 7.0 for 1h at 20 deg. C and then filtering
Mercury	Hydride Atomic Absorption Spectrometry (AAS), digested in a nitric/sulfuric acid mixture	n.d.
Sodium	Inductively Coupled Plasma Emission Spectrometry (ICP) in an aqua regia digest	n.d.

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