

Lowland agricultural peat characteristics which influence peat hydrology: 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.

Executive summary

This research synthesised existing knowledge and data on the hydrological functioning of lowland agricultural peats to improve understanding of their hydrological behaviour and the factors that influence it. The research also aimed to inform a rewetting typology for these peatland systems to explore the implications of maintaining higher, stable water tables, as prescribed under agri-environment schemes. A literature review and data collation exercise was conducted and the data explored in the context of raising water tables in lowland agricultural peat.

The literature review has shown that water table depth and dynamics, and therefore the relative ease of maintaining raised water tables, within lowland peats are influenced by the following five factors:

- (i) Drain spacing and ditch water levels – the influence of ditch water level on the field water table depth declines with distance and increasing peat humification. The drain spacing in many existing drainage systems is too wide to maintain and control raised water tables, especially in degraded peats. Narrower effective drain spacings are likely to be necessary to enhance water ingress into the peat.
- (ii) Peat properties – drainage causes subsidence, consolidation and increased decomposition of peat (or humification). This can impact the peat through increased bulk density, reduced specific yield (affecting storage and water table variability) and reduced saturated hydraulic conductivity (affecting drainage losses, lateral inflows into the peat from the drains and the hydraulic gradient). These changes can persist after water tables have been raised.
- (iii) Groundwater seepage – groundwater seepage can be a contributor to the water balance of some lowland peatlands, helping to meet summer evapo-transpirative demands. Groundwater inflows into the drains can help maintain summer drain water levels.
- (iv) Topography – in-field topographic variation due to subsidence, peat extraction and/or drain management can influence waterlogging, water table depth, soil moisture, vegetation establishment and evapotranspiration.
- (v) Vegetation – actual evapotranspiration rates of wetland vegetation are likely to be in excess of the potential rate, with rates tending to be higher where water tables are raised within small spatial extents due to local advection.

The likely effectiveness of raising water tables will depend on several factors, including: the agro-climate (being greater in cooler and wetter areas than in drier and warmer areas); the availability of water resources to maintain high ditch water levels and support lateral flows into the peat to meet increased evapotranspiration demands; as well as peat properties and drainage management. The physical properties of drained lowland agricultural peats cannot be changed in the short-term and will therefore exert a strong influence. Consequently, a simple re-wetting typology of peat types is proposed based on:

- Trophic type: the effects of the original peat-forming vegetation on peat properties and susceptibility to degradation, and the potential for external sources of water (surface water and groundwater) to support higher water tables
- Past site usage: to account for the effects of agricultural drainage and land use on humification and peat properties.

Agri-environment scheme prescriptions under Countryside Stewardship Higher Tier (CSHT) require water tables within a narrow 20 cm range (10–30 cm or 31–50 cm below surface). Monitoring data shows that natural variability often exceeds this range, even in sites relatively unimpacted by agricultural drainage. Many peatland vegetation communities of conservation relevance and potential paludiculture crops require water table depths and fluctuations exceeding these narrow ranges. To maintain stable, higher water tables in lowland agricultural peat, and therefore comply with these agri-environment schemes, this work suggests that more active water table management interventions, such as installing field drains or digging new ditches, would be required in fields where underdrains are absent and ditch spacings are wide. This may present barriers to uptake of the schemes.

1. Introduction

Lowland (here defined as land below 200 maOD (meters above Ordnance Datum), as per Burton and Hodgson, 1987) agricultural peats occur on the drained remnants of former mires or habitats that had consistently high, but rarely above-surface, water tables. Their former wetland type can be sub-divided into bogs (acidic mires, generally with a pH less than about 5.5), fens (base-rich mires with a pH typically greater than about 5.5) and occasionally swamps (wetlands with a summer water table typically more than 25 cm above ground level) (Wheeler et al, 2009). Fen is often used as a generic term for all mires which are fed by surface- and/or groundwater in addition to precipitation, in contrast to bog that is widely used as a generic term for mires fed exclusively by precipitation. Peats in lowland raised bogs are dominated by mosses of the genus *Sphagnum*, and vascular plants adapted to waterlogged conditions such as the cotton grasses *Eriophorum* spp (JNCC, 2008), whereas peats in fens are generally composed on the remains of sedges (e.g. great fen sedge and common sedge), grasses (such as *Phragmites Australis*) and woody plants.

Within soil classification systems, peats are mostly identified by the percentage organic matter content and/or depth of organic layer. Such broad classifications are limiting at the local scale where physical and chemical characteristics influence peatland functioning (University of Leeds Peat Club 2017), leading to classifications based on hydromorphic setting (e.g. Wheeler et al, 2009) and chemical status (e.g. Rydin and Jeglum 2005). Peats are a structured and highly complex porous medium in which the pore structure comprises open, dead-end, and partially/fully closed pores that influence water storage and transport (MacAlister 2020). In pristine peats, the uppermost poorly decomposed peat layer (or acrotelm- Clymo, 1984) provide high water storage capacity which helps to regulate water table fluctuations, whilst total pore volume, average pore size, specific yield and saturated hydraulic conductivity decrease with depth due to increasing decomposition and compression from overlying peat (McCarter et al., 2021). In contrast, the hydrological properties of lowland agricultural peat are a complex consequence of human management, vegetation type and depth.

Agricultural drainage of peats has been achieved through the excavation of drainage channels/ditches in which a lowered water level is maintained through either pumping or gravity conveyance to a river. The drainage channels may be supplemented by underdrains (or pipe drains) which reduce the effective drain spacing. Environment Agency (2024) estimated the drainage density within the main lowland peat areas (Figure 1), based on the length of watercourses which are “important mainly at a local level” in the Ordnance Survey Water Course Network (which will include drainage ditches and other small surface water courses) within the lowland peat soils depicted in SoilScapes.

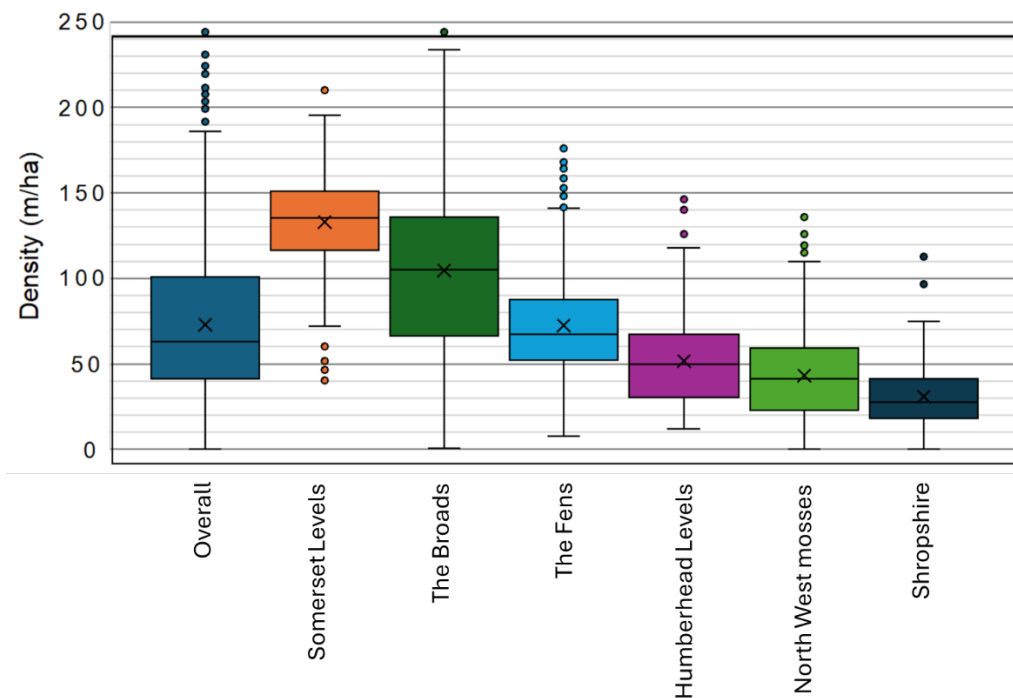


Figure 1. Box and whisker plot of the drainage ditch network density within each full or partial square kilometre of the main lowland peat areas (showing the interquartile range, median [horizontal line] total range, mean [x] and outlier points [o]) adapted from Environment Agency (2024).

The Somerset Levels (133 m/ha) and the Broads (105 m/ha) are the only lowland peat areas in Fig. 1 to have a derived drainage density that is higher than the national average (73 m/ha). However, this doesn't reflect a more intensive drainage regime in these areas but rather the general absence of under-drainage, which, where it occurs, allows larger field sizes and reduced ditch density.

Due to the important role of drainage in efficient agricultural production there was significant government assistance to farmers installing under-drainage, with such drainage activity in England peaking in the 1970s (Fig. 2). The improved drainage of soils, with waterlogging limitations to agricultural productivity during the 1970s (Fig. 2), was focused on the drier areas (in contrast to drainage in the 19th century) (Robinson 1990), and looked to have included lowland agricultural peat areas in the Fens, Lancashire coastal plain and Humberhead Levels. However, the current condition of under-drains is generally unknown.

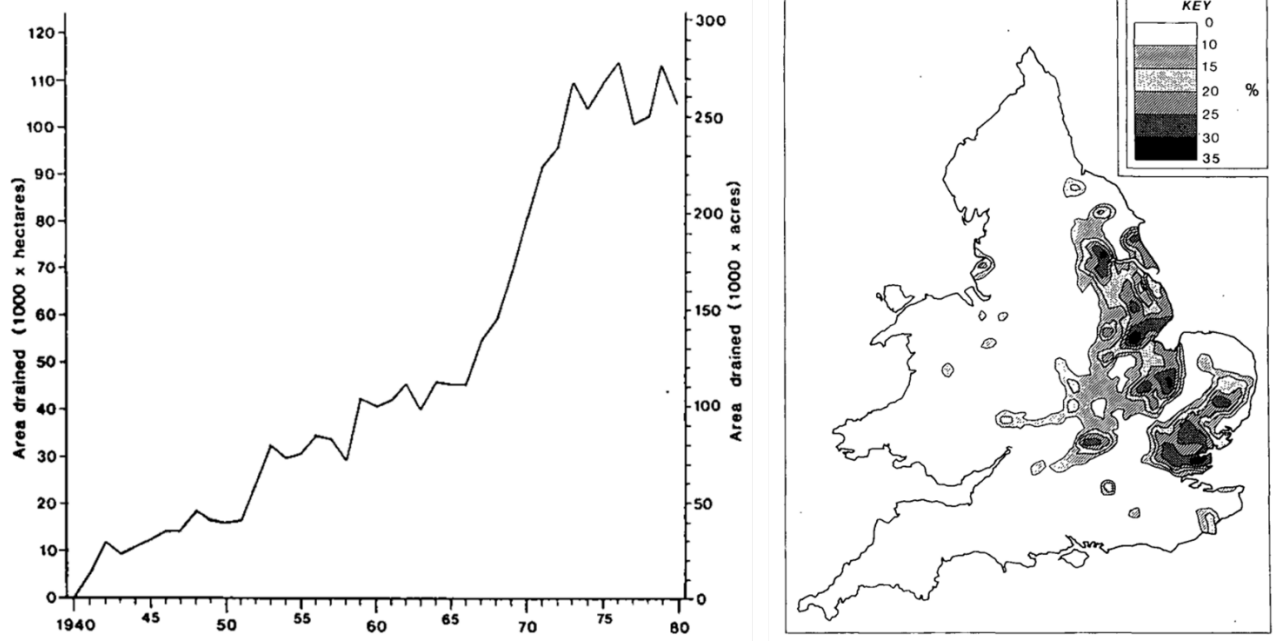


Figure 2. (left) Annual area of grant-aided underdrainage in England and Wales (1940-80) and (right) percentage of agricultural land in England and Wales underdrained in the period 1971-80 (from Robinson 1990. Copyright: Institute of Hydrology, 1990).

Drainage to lower the water table causes subsidence (Figure 3), increased decomposition of peat (or humification) and increased bulk density (Kreyling et al. 2021), particularly above the zone of water table fluctuation (Menberu et al 2021). In both lowland agricultural peats and more pristine peats, the more advanced state of decomposition of older (deeper) peat, with a greater abundance of small pores and lower saturated hydraulic conductivity, mean that water transport processes are more limited at depth (McCarter et al. 2021). Increasing dry bulk density is related to the constriction and closure of pores due to mechanical compaction and collapse, while increasing levels of decomposition leads to increasingly disconnected pores due to the biological breakdown of the peat matrix (Morris et al., 2022; McCarter et al., 2021), leading to reduced specific yield (affecting storage) and saturated hydraulic conductivity. The depositional nature of plant material and the properties of the decaying vegetation can also lead to differing susceptibility to losing large pores during decomposition and the development of preferential flow paths due to anisotropy (McCarter 2021; Baird and Gaffney, 2000), affecting water storage and movement.

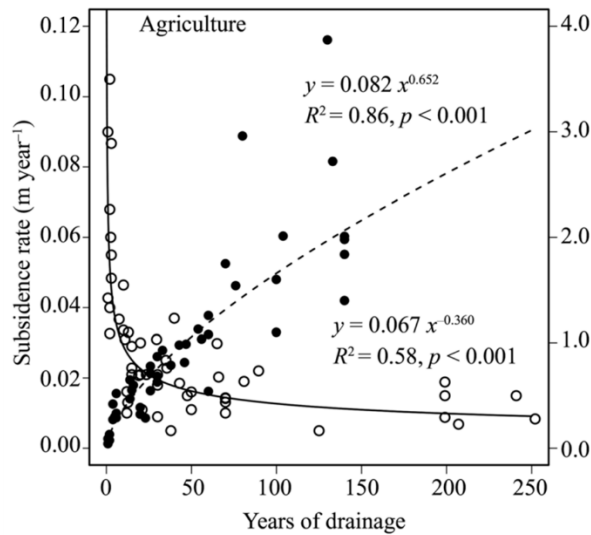


Figure 3. Scatter plots of subsidence rate and cumulative subsidence of peatlands versus time since drainage for agricultural sites (from Liu et al., 2020) [Reproduced under Creative Commons Attribution Non-Commercial License].

The water table elevation within a lowland agricultural peat is a consequence of the temporally varying balance between water inputs and water outputs within the constraints set by the drainage controls (e.g. drain water level) and peat properties (Figure 4a). Inputs of water to the peat can include precipitation, natural sub-irrigation (lateral water movement from the drains into the peat when the elevation of the water table in the peat is lower than the water level in the drain), surface irrigation and groundwater seepage (from an underlying aquifer). Soil water can leave the peat via drainage (water movement from the peat into the drains when the water table in the peat is higher than that in the drain), evapotranspiration, and seepage losses (into an underlying aquifer).

During winter, when precipitation (and in some circumstances upward groundwater seepage) is greater than actual evapotranspiration, the water table elevation increases above the drain water level to facilitate lateral saturated flow to the drains (Figure 4b). In these situations, Hooghoudt's drainflow equation indicates that the saturated hydraulic conductivity of the peat, drain/ditch water levels and drain/ditch spacing are key controls of the drainage flux, q , out of the soil profile

$$q = \frac{8Kd_e m + 4Km^2}{CL^2}$$

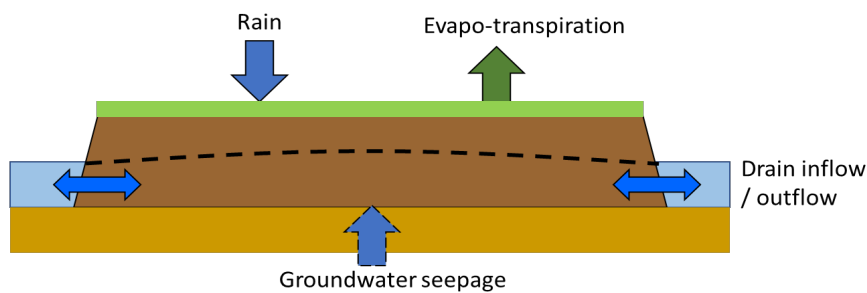
Where q is the drain flux (m/d), K is the effective saturated lateral hydraulic conductivity (m/d), d_e is the equivalent depth of the drain height above the impermeable base (m), m is the mid-drain water table height above the drain (m) and L is the distance between drains (m).

In summer, when actual evapotranspiration is greater than precipitation (and, where relevant, vertical groundwater seepage), the water table elevation declines in response to

lateral drainflow and evapotranspiration. In circumstances where capillary rise is able to move water upwards into the root zone to sustain evapotranspiration, the water table elevation is able to drop below the height of the drain water level, leading to a reversal in the direction of water flow i.e. the drain provides lateral inflow or sub-irrigation to the peat (Harpenslager et al. 2024) (Figure 4c).

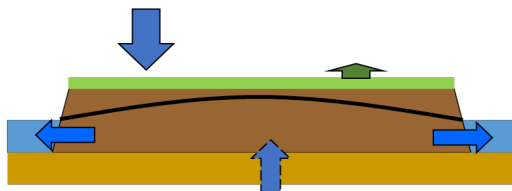
The amount that the water table changes in response to these net inputs (increase in water table level) or net outputs (decrease in water table level) is determined by the specific yield of the peat (discussed in sections 3.2 and 4.2). The specific yield (defined as the volume of water released from, or taken into storage, per unit surface area, per unit change in head e.g. water table level) is equivalent to the drainable porosity and air capacity.

(a)



(b)

Winter



(c)

Summer

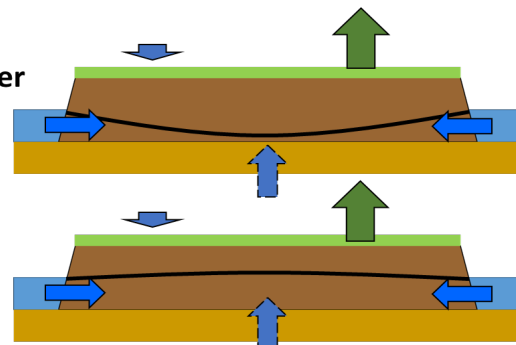


Figure 4. The (a) drivers of water table behaviours and their differences in (b) winter and (c) summer.

Bring et al. (2022) completed a systematic review of the effects of a range of wetland restoration measures on groundwater levels in temperate and boreal wetlands including peatlands. Raising ditch water levels was the most common approach to raising peat water tables. They found 111 restoration studies which were dominated by peatlands, of which most were in Finland (n=50) and the UK (n=40). However, their systematic review indicated that the magnitude of the restoration intervention (i.e. how much drain water levels are raised) is not a significant factor affecting the outcome of wetland restoration in terms of water table increase, due to the limited control that the drainage system exerts on in-field peat groundwater levels (Boonman et al., 2022). Similarly, Krayling et al. (2021) found that peatland restoration success is either reached promptly after rewetting or not at all within the three decades covered by their data. Bring et al. (2022) conclude that this points to “context-dependent effects that may be hard to predict with any precision”.

In other words, raising the water levels in agricultural drainage systems alone within lowland agricultural peat is likely to fail to deliver higher in-field water tables in many cases due to the effects of other controlling factors.

This report presents the findings of a review of the physical and site characteristics of lowland agricultural peat that influence the movement of water within lowland peats required for raising water tables (section 3). It also collates reported lowland peat physical properties and analyses shallow water table monitoring data to explore a rewetting typology for lowland peats (section 4). Finally, it evaluates the implications of inter- and intra-annual variability in water table depths within UK lowland peats for achieving and maintaining raised water tables (section 5) and presents the overall conclusions of the work (section 6).

2. Methodology

Two literature reviews were conducted to (1) review the physical and site characteristics of lowland agricultural peat that influence the movement of water required for raising water tables and (2) collate reported lowland peat physical properties. Although completed separately, the findings from these have been combined in Section 3.

The first was a review of the physical and site characteristics that influence the movement of water in lowland agricultural peat. This review used a combination of keyword searches and snowballing. Given the breadth of controls, relevant scientific literature was identified from the Web of Science database using “peat*” and a range of supplementary search terms including “drainage”, “drain* spacing”, “watertable” or “water-table”, “conductivity”, “pedotransfer”, “anisotropy”, “burning”, “hydrophobicity”, “repellenc*”, “acid sulphate”, “ochre”, “groundwater”, “Seepage”, “flux”, “re-wetting”, “restoration”, “topograph*”, “root* depth”, “irrigation”. The resulting references, including those relating to forestry on peat, were then reviewed for potential relevance through the title and abstract, excluding those not referring to lowland peat (<200 maOD.) or temperate environments. Similar keyword searches were applied using Google.

The second review collated lowland peat physical properties. Morris et al. (2022) published a very comprehensive review of saturated hydraulic conductivity (Ksat) values in northern peats based on 47 sources, including the UK, Ireland, Sweden, Finland, Canada and the US. Their work provides a useful and reliable source of information which quantifies the variability in Ksat based on some of the factors introduced in Section 1. To complement it with data from omitted studies and with other peat hydrological properties, we conducted a second literature review. Scientific literature was identified from the Web of Science using a range of search terms including “peat”, “hydraulic conductivity”, “water retention”, “field capacity”, “permanent wilting point”, “gravimetric water”, “gravimetric moisture”, “specific yield” and “Van Genuchten”. The resulting references were then reviewed for potential relevance through the title and abstract, excluding those not referring to lowland peat, raised bog and fen peat, and/or focusing on the tropics. Grey literature was identified using the same search terms in Google.

A total of 33 additional documents were returned from the searches above and used to extract data. The data extracted included vertical and horizontal saturated hydraulic conductivity, unsaturated hydraulic conductivity and related pressure head, saturation percentage volume, field capacity, wilting point, specific yield and gravimetric water content. Information was also extracted in relation to the peat context such as trophic type, vegetation, measurement method, degree of peat decomposition, sample depth, disturbance, other details (e.g., organic matter content, pH).

This extracted data is available as a separate dataset, provided as an Annex to this report as an Excel file: *Lowland agricultural peat characteristics which influence peat hydrology - data annex*.

To develop a high-level rewetting typology of lowland agricultural peats, additional information was analysed from Defra's Lowland Peat projects, the Lowland Peat Survey (Burton and Hodgson, 1987) and [LandIS - Land Information System - Cranfield Environment Centre](#). The analysis focused on trophic type, pH, landuse, vegetation type, bulk density and degree of humification. This work is reported in section 4.

Available Environment Agency water table data from shallow monitoring dipwells in peat were used to explore water table behaviour from sites under a range of land covers, ranging from natural to arable. This information was used to explore the practicality of maintaining water tables within the narrow depth ranges prescribed in Countryside Stewardship Higher Tier requirements for managing higher water tables in lowland agricultural peat. This work is reported in section 5.

3. Characteristics of lowland agricultural peat that influence the maintenance of raised water tables

Climate and climate variability plays an important role in water table dynamics within lowland peats, with higher precipitation and lower evapotranspiration being associated with higher and less variable water tables (Mitsch & Gosselink, 2000; Belyea & Malmer, 2004, Waddington et al., 2015). Raising and maintaining high water tables in lowland agricultural peat soils will therefore be more climatically challenging in drier/warmer regions of the country (e.g. East Anglia) and in dry years.

However, based on the introduction in Section 1, this section reviews the literature to understand the importance of a range of non-climate factors on the peat water table and the consequences for raising water tables. In particular, the importance of:

- Drain spacing and drain water levels
- Peat properties
- Groundwater seepage
- Topography
- Vegetation

3.1. Drain spacing and drain water levels

Although the drainage level (i.e. the surface water level in a ditch or the elevation of pipe or under-drains) sets the drainage ‘base level’, the distance into the field that the drain actually controls the water table is largely determined by the hydraulic conductivity of the peat, which is highly variable in both horizontal and vertical directions (Stratford et al., 2015). This distance of influence varies with the degree of humification of the peat (Verry et al., 2011), with Boelter (1972) suggesting it extends to only 5m in well-humified peat but extending to 50m in fibrous peat. Using dipwells on Tadham Moor, Acreman et al. (2002, cited in Stratford et al., 2015) estimated that the ditch water level only influenced the water table elevation up to 8m away. Wheeler et al. (2009) simulated water table elevations in a solid fen peat away from a river channel in The Broads and found that the “the water table response to drainage is minimal except in a very narrow strip adjacent to the river” of around 6m width, although the simulation was restricted to 30 days duration. Price et al (2003) quoted several studies suggesting that water tables might be drawn down up to 50m from the ditch in fibrous peat, but only in the immediate vicinity in very decomposed fen peat. In their assessment of the water storage benefits of raising water tables, Stachowicz et al. (2022) estimated that the maximum influence range that the ditch has on the water table rise was between 20m and 50m.

Only eight non-blanket bog peatland restoration studies were available to Bring et al. (2022) to analyse the relationship between the change in groundwater level resulting from the intervention (e.g. raising ditch water levels through ditch blocking, damming etc) and distance from the ditch (Figure 5). The reported increase in groundwater level at 1 m from the ditch, in which water levels were raised ranged from 25 to over 80 cm, but this effect rapidly diminished with distance, becoming zero at between 30 and 130m. This relationship between the change in groundwater level and distance from the ditch was well described by an exponential decay function – the water table benefit at 1m distance from the ditch was reduced to 50% after 9m [95% confidence limit (CL) of 5- 26m], and to 25% after 27m [95% CL of 12-132m].

A review of drainage design criterion for various studies conducted on peat soils (Tuohy et al 2023) found that drain depth was typically recommended in the range of 0.3m to 1m, whilst drain spacing ranged from <<10m to 60m. Cited studies showed an inverse relationship between drain spacing and mean water table depth (i.e. mean water table depth decreases with increasing drain spacing) and a positive relationship between drain depth and mean water table depth (i.e. mean water table depth increases with increasing drain depth). However, in many areas of lowland agricultural peat in England, which are only drained with ditches for low intensity or summer grazing, drain spacing will often be significantly wider.

Where existing drains are too far apart to provide appropriate water table control, an option is to increase the drainage density. LaRose et al. (1997) implemented ditch reservoirs (essentially dammed ditches with high water levels) at 3m, 5m and 8m spacing when trying to provide consistently high water tables to support the re-establishment of *sphagnum* at a former extraction site in Canada. Sub-surface Infiltration (SSI) systems consist of a series of shallow drains that are installed at least 10 cm below target ditch water level to passively irrigate drained peatlands with surface water in times of drought, to reduce groundwater level fluctuations. In the studies of Harpenslager et al (2024) and Boonman et al. (2022), these were installed at 50-70cm depth and 4-6 m spacing. Consequently, the effect of the greatly reduced drain spacing with sub-surface infiltration systems is to maintain higher water tables during the summer drought (Kechavarsi et al. 2007; Querner et al. 2012; Boonman et al. 2022). Weideveld et al (2021) found that sub-surface irrigation ditches at 5-6 m spacing had little effect compared to 100m wide ditches, with the difference in mean annual water table level being less than 5 cm across all years and locations, although the water table was 6–18 cm higher in summer and 1–20 cm lower in wet conditions.

Consequently, this suggests that many existing drainage systems will have inadequate drain spacing to maintain and control raised water tables, especially in degraded peats. Fields with pipe or under-drains may have a closer spacing than those drained solely by ditches, but their performance in many areas will be reduced due to age, lack of maintenance and blocking (e.g. by ochre). Successfully raising water tables may require changes to drainage design, from the installation of sub-surface infiltration systems or the excavation of foot drains, to enhance water ingress into the peat, although foot drains can rapidly be colonised and blocked by soft rush (Andrea Kelly, *pers. comm*).

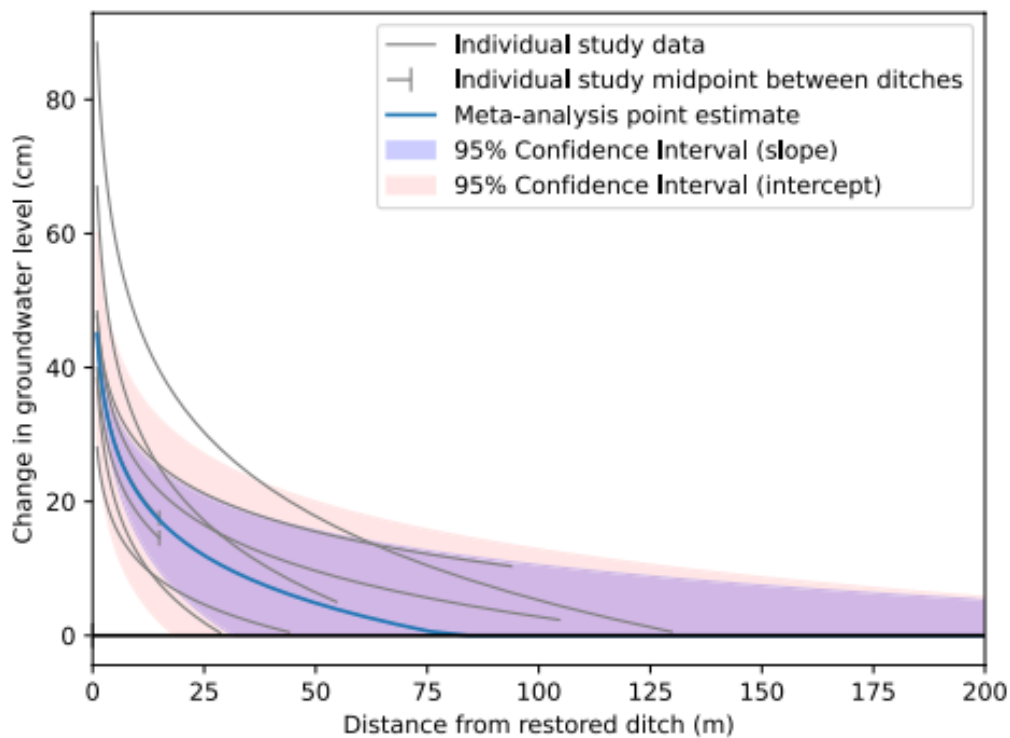


Figure 5. Restoration effects in peatlands (excluding blanket peatlands) on groundwater levels at different distances from the ditch with raised water levels, from Bring et al (2022). Each grey line corresponds to reported effects in an individual study. The blue line is the random effects (RE) model based on all studies. The purple and pink bands are the 95% confidence intervals of the RE model estimate of its two parameters. [Reproduced under Creative Commons Attribution 4.0 International License].

3.2. Peat properties

Introduction

As briefly introduced in Section 1, the relative elevation of the water table within the middle of the field in winter compared to the drain water level depends on net rainfall and the ability of the peat to transmit excess water laterally to the drain – a steeper hydraulic gradient and thus higher water table is needed where the peat is degraded and thus has lower saturated hydraulic conductivity (K_{sat}). Consequently, low K_{sat} is thought to play a central role in maintaining shallow water tables in peatlands by inhibiting lateral drainage (Morris et al., 2022). Drainage losses are thus reduced by the lower hydraulic conductivity typical in degraded peats, but the water table variability is also greater because of the generally low specific yield of degraded peats (Price et al., 2003). Specific yield is the volume of water released from, or taken into storage, per unit surface area, per unit change in head – a lower specific yield means that the water table changes by a greater amount for the same gain or loss of stored water. Price et al. (2003) considered that the effect of lowered specific yield is more important than the effects of reduced hydraulic

conductivity on controlling the water table elevation, because more water is typically lost from agricultural peatlands by evapotranspiration than lateral drainflow.

Bulk density

Agricultural drainage of peat leads to wastage, as a consequence of initial shrinkage; compression; and decomposition (biochemical oxidation), factors which are associated with an increase in bulk density. Bulk density, defined as the mass of soil per unit volume (g/cm^3), is a key physical property of peat soils that influences their hydrological functions. Peat mineralization and degradation associated with previous drainage in re-wetted sites was reflected in lower organic matter content of the top soil layer (-18%) and higher bulk density ($+61\%$) compared to undrained sites, both of which decrease porosity and hydraulic conductivity, provoking larger water table amplitudes ($+15\%$) in rewetted sites (Kreyling et al. 2021). Unlike in pristine peats, bulk density declined with depth in agricultural drained peats in Menberu et al (2021) indicating that the effects of peat compaction and decomposition occurred mostly in the upper 60cm (Figure 6).

As with the subsidence rate (Fig. 3) in newly drained peat soils (e.g. as seen in the Holme Post: Waltham, 2000), there is a rapid decline in the rate of change in bulk density (BD) (from initially high rates of increasing bulk density) within the first 10–15 years in agricultural peatlands. This is seen in a strong (non-linear) power law relationship between the rate of change in bulk density and time since the start of drainage ($R^2 > 0.69$, $p < 0.001$) (Liu et al. 2020). After this initial period, the average BD change rate for agricultural peatlands was $3.2 \times 10^{-3} \text{ g cm}^{-3} \text{ year}^{-1}$; ($p < 0.001$; t test). Liu et al. (2020) also estimated, from a subset of their data, that the BD change rate below the water table was significantly lower at $\sim 0.56 \times 10^{-3} \text{ g cm}^{-3} \text{ year}^{-1}$, consistent with the findings of Menberu et al (2021). The change in bulk density is inversely proportional to total porosity, but the macroporosity (or larger pore fraction) in Liu and Lennartz (2019) only decreased as bulk density increased within the range from 0.01 to around 0.2 g/cm^3 . This is in contrast to other studies (e.g. Schindler et al., 2003; Wallor et al., 2018) in which macroporosity initially decreased within peat degradation but then subsequently increased with further degradation.

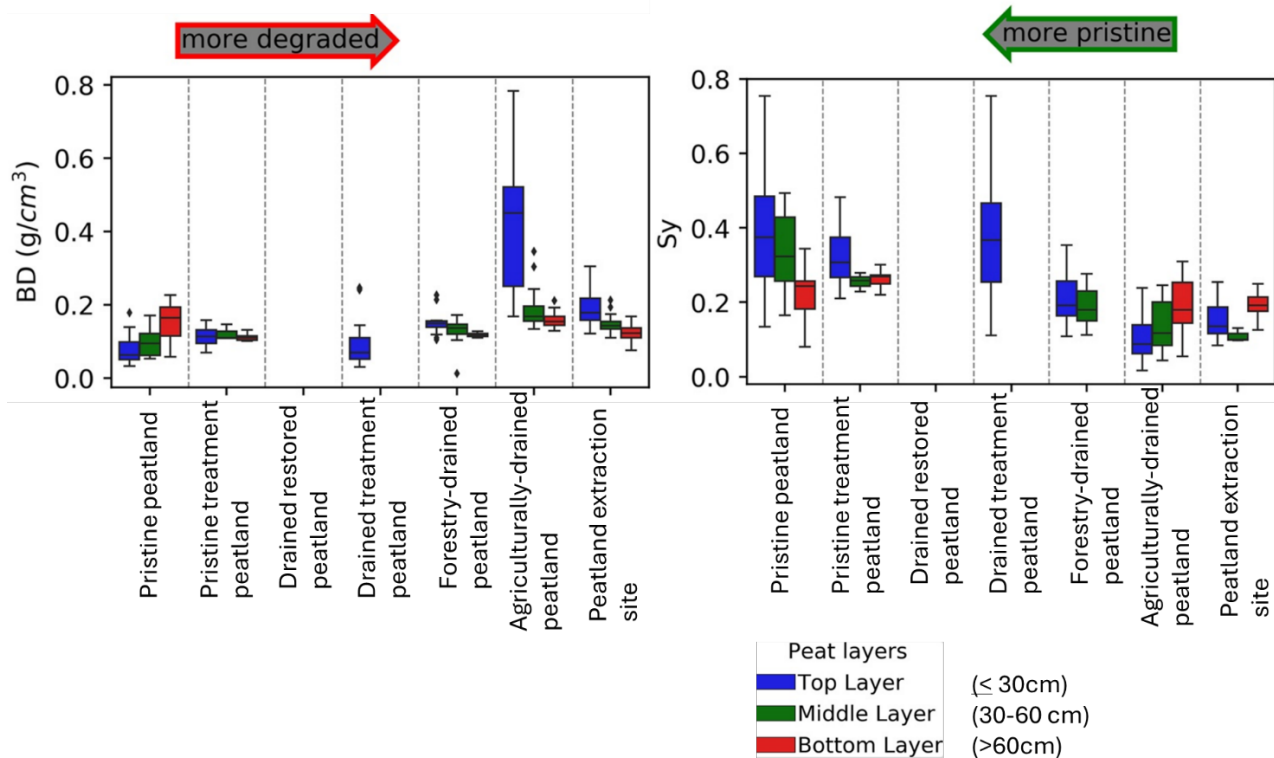


Figure 6. Boxplot of the variability in bulk density (BD) and specific yield (Sy) by depth across the degradation spectrum from pristine peatlands to agriculturally-drained and extraction sites in Finland (from Menberu et al., 2021. Reproduced under a Creative Commons Attribution-NonCommercial-NoDerivs License).

Specific yield

Specific yield is the volume of water released from, or taken into storage, per unit surface area, per unit change in head. It determines how much the water table changes for a given amount of water loss or recharge – a lower specific yield will lead to larger fluctuations in water table levels for the same change in water volume. Within peats and other soils, the soil science terms of air capacity and drainable porosity are analogous. Specific yield can be measured experimentally in a laboratory using undisturbed samples or estimated empirically by comparing the *in situ* response of the water table to known precipitation events (Bourgault et al., 2017).

The specific yield of peat can be highly variable, according to degree of humification, compaction and peat composition, with higher values typically associated with the least decomposed peat and also undisturbed or restored peat (Baird et al., 2023). There are a number of syntheses of specific yield values from the literature (i.e. Supplementary material in Baird et al., 2023; Table 1 of Rezanezhad et al., 2016) demonstrating this wide range, but not providing contextual data.

Peatland regeneration (as opposed to raising water tables) requires a high moisture content in the peat. This is facilitated in natural peatlands by the high water storage capacity and specific yield of the uppermost poorly decomposed peat layer (acrotelm)

which helps to regulate water table fluctuations. Where this has been lost or has been structurally degraded due to drainage and wastage, the remaining peat is more decomposed and has a lower water storage capacity (Price et al., 2003), resulting in larger water-table fluctuations (LaRose et al., 1997). Price et al. (2003), reporting results from Schouwenaaars (1993), described specific yield declining from 0.23-0.34 in young living *sphagnum* to 0.11-0.17 in slightly humified *sphagnum* peat (Von Post H2-3) to 0.05-0.10 in strongly humified *sphagnum* peat (Von Post H6-7). Similar results are presented in Vinny et al. (2011) for a range of peat types.

Liu et al. (2020) developed a strongly non-linear pedotransfer function for agricultural and forested peat soils to predict specific yield from bulk density data (specific yield = $0.003 \times \text{bulk density}^{-1.40}$), where specific yield was defined as the difference between total porosity and volumetric water content at -10 cm water pressure head (Figure 7). Similarly, Menberu et al. (2021) developed a multivariate regression model for specific yield using bulk density and soil depth which explained 74% of the variance and has a RMSE of 0.086:

$$\ln(Sy) = -0.696 \times \ln(BD) - 0.104 \times \ln(\text{soil depth}) - 2.64,$$

where Sy is the specific yield (decimal fraction), BD is in g cm^{-3} , and soil depth is in cm.

Although the regression model, of Menberu et al. (2021), shows that bulk density was relatively more important than soil depth for predicting specific yield (consistent with Liu et al. 2020), Bourgault et al. (2017) found a strong non-linear relationship between specific yield (measured using both laboratory and field methods) and depth in ombrotrophic peatlands in the St Lawrence Lowlands in Canada, although the specific yield was little changed below about 30cm depth (Figure 8). However, the sites of Bourgault et al. (2017) were largely undrained, with the mean specific yield of the living and slightly humified surface peat layer comprising the acrotelm being 0.69 and 0.35 depending on laboratory method, highlighting the important uncertainty due to measurement method. A similar relationship of declining specific yield with depth is seen in the samples from pristine peats in Finland of Menberu et al. (2021) in Figure 6. However, in their agriculturally drained peat sites, the specific yield is seen to increase with depth (and declining bulk density) with specific yield values below 60cm being similar to pristine sites. Consequently, the adverse effects of drainage on the specific yield of agricultural peats are greatest in the upper soils into which raised water tables would return.

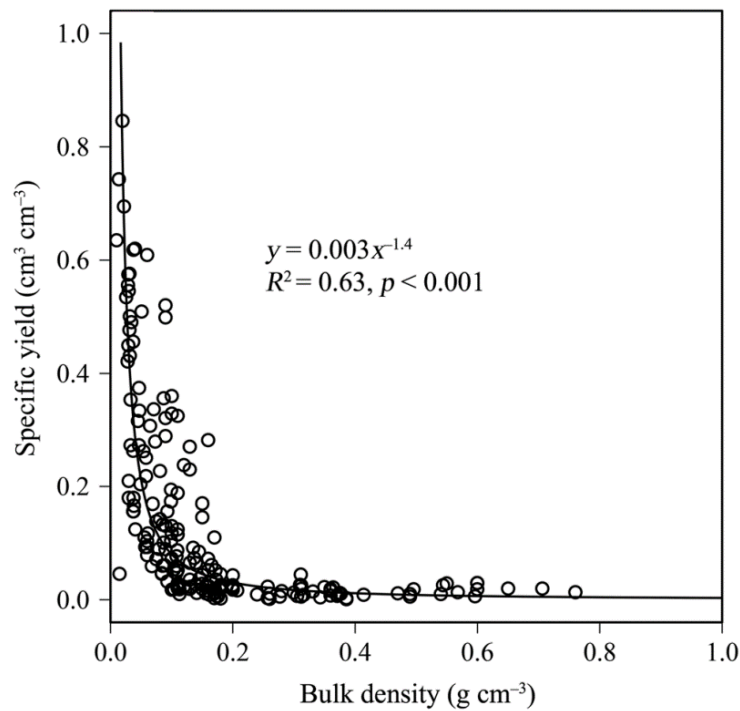


Figure 7. Relationship between specific yield and peat bulk density from Liu et al. (2020). Available under CC BY-NC 4.0 licence.

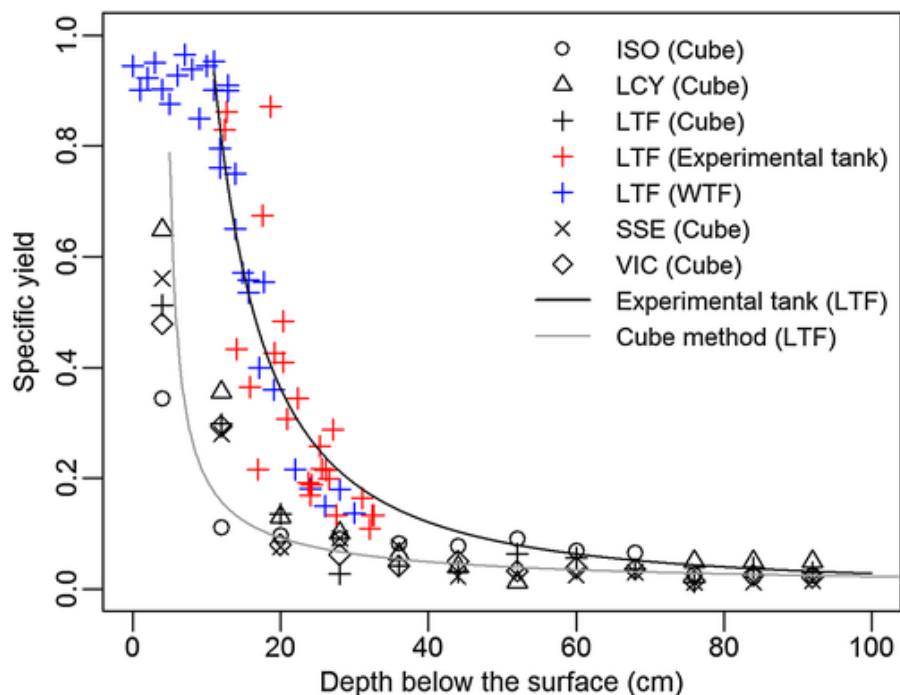


Figure 8. Variation in specific yield with depth using different laboratory methods for peatland sites in St Lawrence Lowlands in Canada [small cube (mean values are plotted for each site: e.g. LTF, LCY, ISO, SSE, VIC), experimental tank (LTF site only)] and the Water table Fluctuation (WTF) (LTF site only) method. The black line shows the SY–depth relationship using the experimental tank method (LTF site), and the grey line shows the logarithmic SY–depth relationship using the cube method (LTF) (from Bourgault et al., 2017, Available under CC BY-NC 4.0 licence).

As mentioned previously, the Water Table Fluctuation (WTF) method derives specific yield from the change in the water table level (mm) in response to rainfall (mm). Using the WTF, Menberu et al., (2018) observed significantly higher water table rises per rainfall input (of 0.36-0.85 cm/mm) in disturbed Finnish peatland conditions due to the derived lower specific yield (S_y) values (0.13-0.24) than under restored and undisturbed conditions (S_y 0.25-0.50). Similarly, Ahmad et al. (2020) found that for each unit increase in rainfall event size, the groundwater table rose by about 12 mm in a drained grassland peatland site in northern Germany compared to about 6 mm in a nearby rewetted sedge reed site, showing the effect of differing specific yield. However, the method can be susceptible to errors where local precipitation is not the sole cause of the observed water table rise – although Bourgault et al. (2017) found a mean specific yield of 0.59 in their pristine sites in the St. Lawrence Lowlands (Canada), 13% of their precipitation events generated specific yield values exceeding 1, indicating that water table response was affected by localised runoff in addition to rainfall. Their ANOVA of the subset of plausible specific yield estimates suggested that site, seasonality, and water table depth had a significant effect on specific yield. Site was considered by Bourgault et al. (2017) to be an analogue for the climate-influenced differences in water table depth, with water tables being deeper and within more humified layers in drier locations. There has been little application of the WTF method to peatlands in England, with the exception of Gilman et al. (1990) who applied it to dipwells within four transects at West Sedgemoor in the Somerset Levels, and found mean values ranged from 0.104 ± 0.028 to 0.143 ± 0.037 .

Hydraulic conductivity

The pattern and rate of flow through peat is determined by three peat hydro-physical variables (Beckwith et al., 2003). These are (i) the degree of anisotropy as given by the ratio of the horizontal (K_h) to vertical hydraulic conductivity (K_v), which influences the proportion of vertical and horizontal flow, (ii) the magnitude of both K_h and K_v , which controls the rate of lateral and vertical flow respectively, and (iii) where the distribution of hydraulic conductivity (K) is heterogeneous, the location of layers of high K_h and low K_v which control the depth of flow.

Peat layers are often anisotropic, with lateral hydraulic conductivities commonly being greater than vertical hydraulic conductivities (Wheeler et al., 2009; Rosa and Larocque, 2008), although not always (SurrIDGE, 2005). Beckwith et al. (2003) measured K_h and K_v in 400 peat samples from Thorne Moors, a cutover lowland raised bog in the Humberhead peatlands with moderately and poorly decomposed (H3 to H7 on the von Post scale) catotelm (subsoil) peat. K_h was greater than K_v in 78% of samples and there was a mean anisotropy ratio (ratio of K_h to K_v) of 3.55, similar to a bulk mean ratio of 3.72 reported by Schlotzhauer and Price (1999) in Quebec. In Canadian lowland bogs and fens, Whittington and Price (2013) found that K_h was greatest in about 90% of samples but that they had a lower anisotropy ratio of 1.8. However, these differences may be small compared to the lateral and vertical heterogeneity in peat, which reflect local variations in the depositional environment and post-depositional changes (Wheeler et al., 2009).

Morris et al. (2022) reported a meta-analysis of 2,507 northern hemisphere peat samples (including from the UK) from which they developed a number of predictive linear models for saturated hydraulic conductivity (K_{sat}). The reviewed values of K_{sat} range between 1.31×10^{-10} and 1.31×10^{-1} m/s across all locations. They found that peat K_{sat} decreases strongly with increasing depth, dry bulk density, and humification; and increases along the trophic gradient from bog to fen peat (Figure 9). The importance of bulk density is also seen in the non-linear relationship between bulk density and K_{sat} for peats in central Finland of Vinny et al. (2011). The most skilful model of Morris et al. (2022) (Model 1) explained about 75% of the variance and had the final form for the hyperoceanic climate of the UK of:

$$\log_{10}(K_{sat}) = -7.522 - 1.124(\log_{10}(\text{depth})) - 1.964(\log_{10}(\text{dbd})) - 0.139(\text{von Post}) - 1.365(\text{blanket}) + 0.675(\text{raised bog}) + 0.866(\text{poor fen}) + 2.134(\text{moderate fen}) + 1.877(\text{rich fen}) - 1.451(\text{hyperoceanic}) + 1.150(\text{hollow}) + 0.188(\text{lawn}) + 1.013(\text{hummock}),$$

where *depth* is midpoint depth below the peatland surface (m); *dbd* is peat dry bulk density (g cm^{-3}); *von Post* is von Post score expressed as an integer between 1 and 10 (dimensionless); *blanket*, *raised bog*, *poor fen*, *moderate fen*, and *rich fen* are dummy variables that can take binary values of either 0 or 1 to represent a site's trophic type (the reference category is unspecified type); and *hollow*, *lawn*, and *hummock* are dummy variables used to represent local surface microform type (the reference category is unspecified microform type).

Although Morris et al. (2022) considered the degree of hydrological disturbance of sites (Figure 9h), there is no clear influence on hydraulic conductivity, with very similar distributions of saturated hydraulic conductivity values in drained and wetted peat. However, Menberu et al. (2021) found that K_{sat} was the lowest in agricultural drained peats (Figure 10) and that there was no decline in K_{sat} with depth.

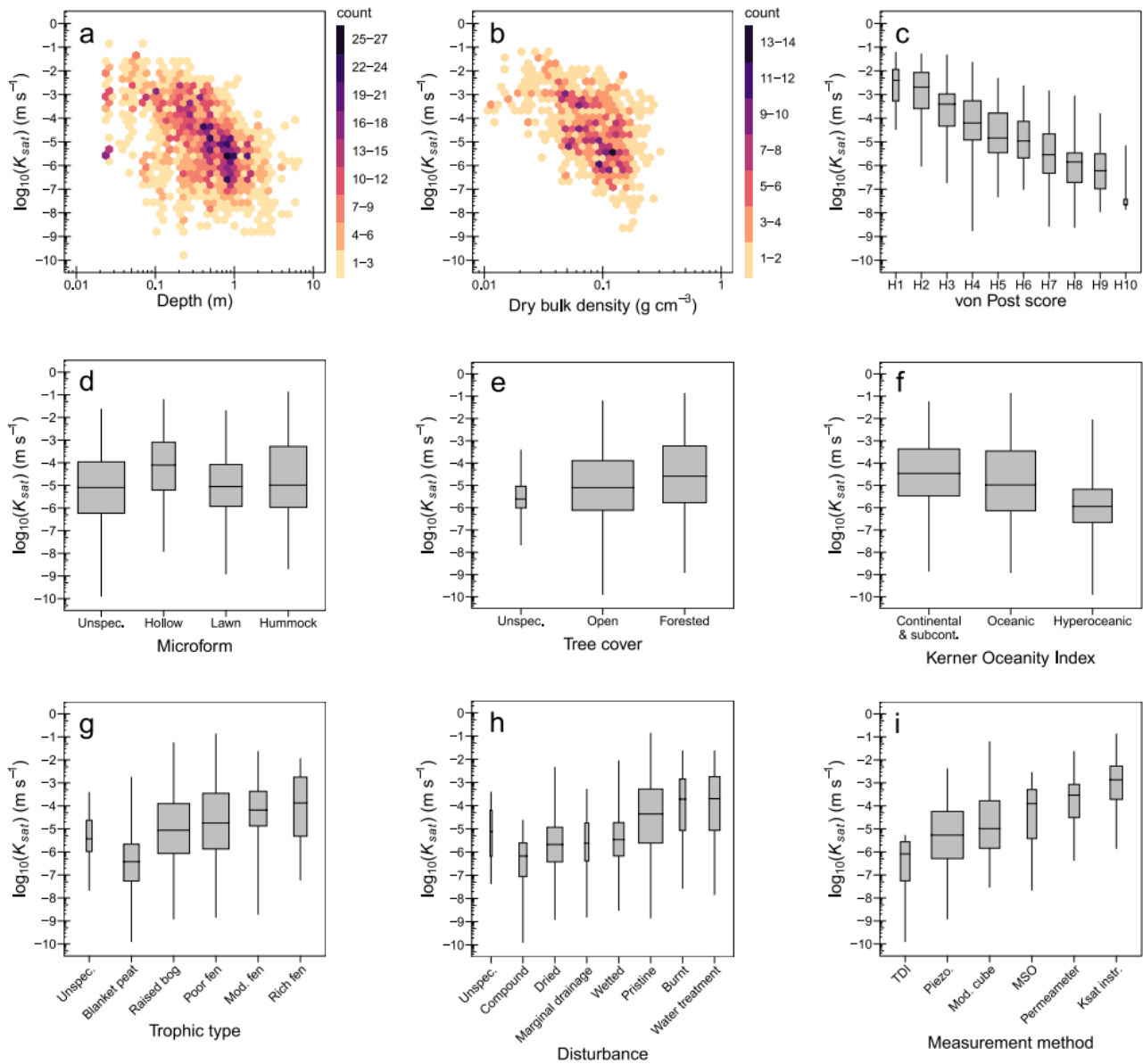


Figure 9. From Morris et al. (2022). Graphical summary of the Ksat database. Panels (a and b) are hexagonally binned point-density plots; count indicates the number of data points in each hexagonal bin. In panels (c–i), box widths are proportional to the square root of sample size. In panel (i), TDI is tension-disc infiltrometer; Piezo. is piezometer; Mod. cube is modified cube; and MSO is the multistep outflow technique. Available under CC BY 4.0 licence

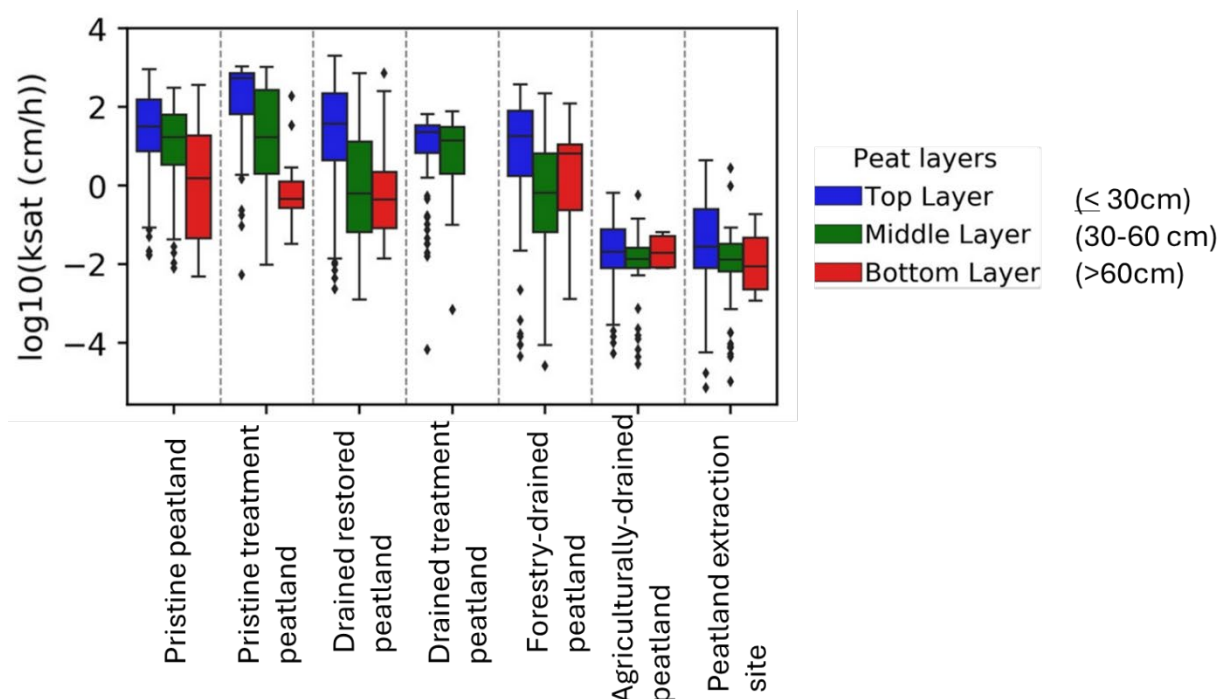


Figure 10. Boxplot of the variability in $\log_{10}(\text{Ksat} [\text{cm/hr}])$ by depth across the degradation spectrum from pristine peatlands to agriculturally-drained and extraction sites in Finland (from Menberu et al 2021. Reproduced under a e Creative Commons Attribution-NonCommercial-NoDerivs License).

The evidence from our additional LP3+ review showed good agreement with the Ksat values reported by Morris et al. (2022), including the effect of depth (Figure 11). The variability of horizontal Ksat values in this case ranges between 1.72×10^{-8} and 1.00×10^{-2} m/s which is within the overall range of Morris et al. (2022) Ksat values whilst, with many less measurement points, vertical Ksat varies between 4.86×10^{-7} and 2.01×10^{-5} m/s.

Wheeler et al., (2009) summarised hydraulic conductivity data for different peat types and conditions from a range of sources (Table 1), many from Broadland sites. However, because the data collection methods (laboratory or field-based) are unknown from the unpublished van Wirdum et al. (1997), these values are reported here separately. Nevertheless, the Broadland values of Ksat range from 1×10^{-8} m/s to 3×10^{-5} m/s, which is within the reported ranges in Morris et al. (2022) and our review. The two UK case studies in East Anglia used by Morris et al. (2022) align with these broader findings, reporting saturated hydraulic conductivity values of fen peat in a pristine site from 1.44×10^{-5} to 3.02×10^{-4} m/s (median 4.84×10^{-5} m/s), compared to 4.93×10^{-7} to 1.60×10^{-5} m/s (median 4.18×10^{-6} m/s) in a marginally drained site.

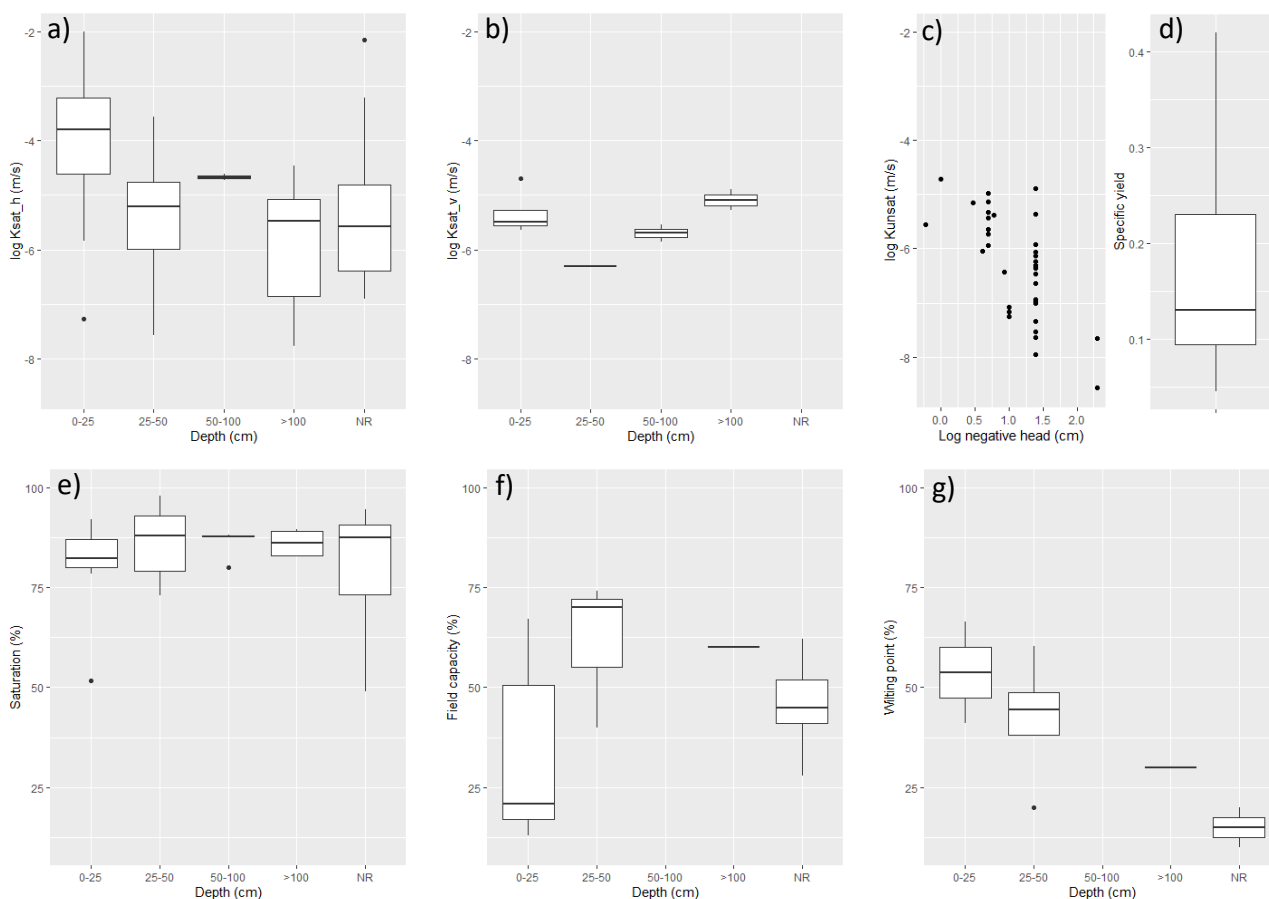


Figure 11. Reviewed values with depth for: a) horizontal saturated conductivity (n=81), b) vertical saturated conductivity (n=9), c) unsaturated hydraulic conductivity by head pressure (n=42), d) specific yield (n=20), e) saturation (n=45), f) field capacity (n=21), and g) wilting point (n=9).

Table 1. Indicative values of hydraulic conductivity in peat (adapted from Wheeler et al., 2009).

Peat type	Saturated hydraulic conductivity (m/day)	Source (full citation in Wheeler et al., 2009)
Poorly decomposed, loose peats	10^{-1} to 10^{-2}	Measurements made in Broadland sites
Loose moss peat	1.65×10^{-2}	Upton Fen, Norfolk (van Wirdum et al., 1997)
Surface fen peat	2.17	Sutton Fen, Norfolk (van Wirdum et al., 1997; Baird, Surridge and Money, 2004)

Peat type	Saturated hydraulic conductivity (m/day)	Source (full citation in Wheeler et al., 2009)
Amorphous peats	c. 10^{-3}	Measurements made in Broadland sites
Amorphous greasy peat	1.87×10^{-3}	Catfield Fen, Norfolk (van Wirdum et al., 1997; Sutton Fen, Baird et al., 2004)
Humified/brushwood peats	10^{-3} to 10^{-4}	Measurements made in Broadland sites
Humified brushwood peat	4.0×10^{-3}	Berry Hall Fen, Norfolk (van Wirdum et al., 1997)
Firm brushwood peat	5.04×10^{-3}	Catfield Fen, Norfolk (van Wirdum et al., 1997)
Humified peat with monocot and brushwood remains	9.42×10^{-4}	Reedham Marshes, Norfolk (van Wirdum et al., 1997)
Acrotelm bog peat (K_h)	10 to 10^2	Romanov (1968)
Acrotelm bog peat (K_v)	< 10	Romanov (1968)
Catotelm bog peat (K_h)	10^{-2} to 10^{-4}	Boelter (1965); Romanov (1968)
Catotelm bog peat (K_v)	10^{-2} to 10^{-4}	Boelter (1965); Romanov (1968); Beckwith et al. (2003)

The variability in the measurement of saturated hydraulic conductivity associated with field and laboratory methods is smaller than spatial variations caused by peat heterogeneity (Rosa and Larocque, 2008), which the authors considered shows the importance of systematically using and comparing both laboratory and field methods when estimating peat hydraulic parameter.

Hydrophobicity

Although there has been significant research on water repellency in peat soils associated with burning (e.g. Wilkinson et al., 2020), there are limited data on agricultural peats.

Evans et al. (1999) noted that prolonged desiccation of surface peat layers in a blanket bog may lead to the development of a hydrophobic layer which would reduce infiltration capacity of the acrotelm peat. Dekker and Ritsema (1996) found similar water repellency in grass-covered peaty clay and clayey peat soils in the Netherlands, although precipitation could rapidly infiltrate through shrinkage cracks (Schwarzel et al., 2002). Obeng et al. (2023) found that the inherent hydrophobicity of peat soils in Ireland, as measured by the ratio of hydrophobic to hydrophilic functional groups by ATR-FTIR spectroscopy, was significantly impacted by land use changes at 0-10cm depth, but not at 10-20cm depth. Their measure of hydrophobicity decreased in the order of active bog, forest plantation, industrial cutaway bog and grassland, indicating lower inherent hydrophobicity in degraded peatlands. A water drop repellency test (WDRT) on drained peat samples from a fen in Poland showed that state of peat degradation and WDPT values were not significantly correlated, but there was a tendency for higher values of water repellency at more degraded sites (Kalisz et al., 2015).

Restoration of peat hydraulic properties

In a peatland restoration (ditch blocking) experimental catchment study in Sweden (Laudon et al., 2023), the daily difference between the mean water table depth within the treatment sites (pre- and post-restoration) and control sites showed that restoration significantly decreased the median water table depth within the first 2 years after rewetting compared to pre-restoration (Figure 12). However, whilst the median difference after restoration was close to zero (i.e. there was little difference in median water table depth with the undrained control site), there was a significant range of differences of up to almost 2m from the control suggesting that resilience to dry conditions has not been restored. Similar findings on restoration success were reported by Kreyling et al. (2021) who compared 320 rewetted fen peatland sites to 243 near-natural peatland sites of similar origin across temperate Europe (Figure 13). They found that drainage-induced subsidence (from peat mineralization, consolidation and compaction) led to increased inundation after rewetting with the mean median water table being 5.0 cm above the surface in rewetted sites compared to 1.5 cm below surface in near-natural sites. In contrast, they also found that the maximum water table depth and the water table amplitude was 74% greater and 15% greater, respectively, in re-wetted sites. The findings of Kreyling et al. (2021) and Laudon et al. (2023) both reflect the ongoing adverse effects of the pre-restoration drainage on specific yield and saturated hydraulic conductivity.

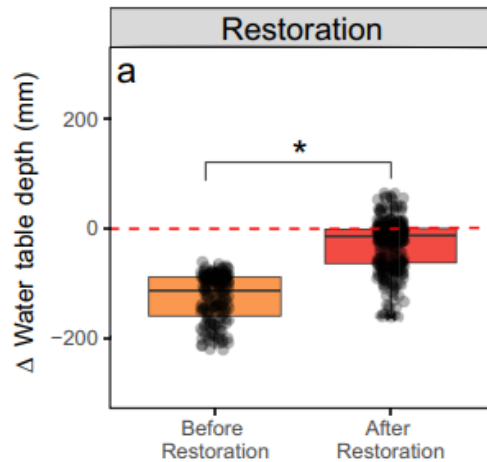


Figure 12. Differences in water table depth (WTD) between restoration and control sites before and after ditch-blocking restoration from Laudon et al. (2023). The difference was computed as restoration minus control, thus positive values indicate that the WTD is greater at the restoration site than at the control site, while negative values indicate the opposite. Black circles represent individual data points. *denotes significant difference ($p < 0.05$) between treatments. Red dotted line denotes the control zero reference. Solid line in box plots is the median value, box extents are the interquartile range (IQR) and whiskers show the 1.5IQR value [Available under Creative Commons Attribution 4.0 International License].

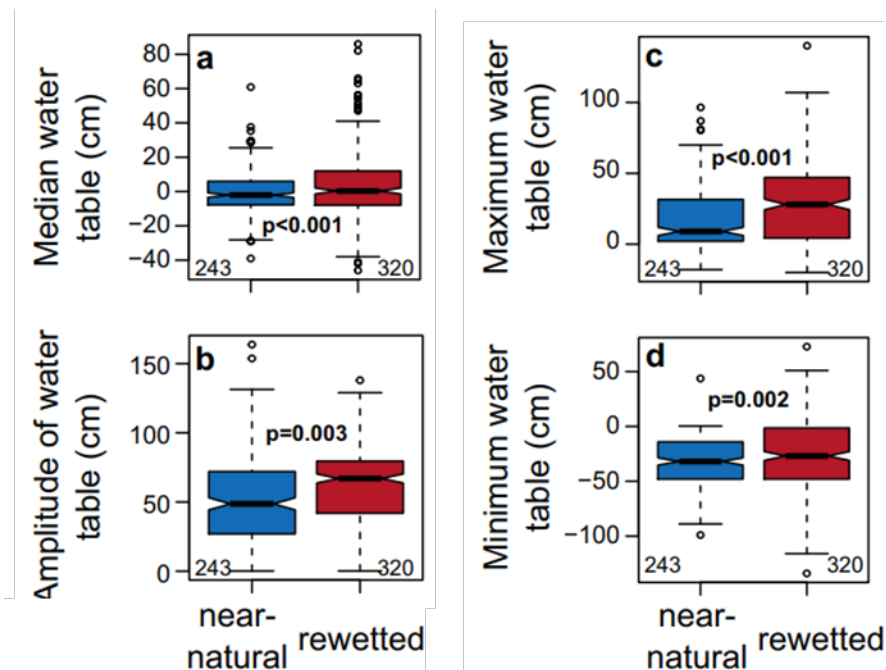


Figure 13. Comparison between rewetted (red) and near-natural (blue) sites for water table behaviour (from Krayling et al., 2021). N per group is provided as numbers next to the boxplots which display the median, quartiles (box), 1.5-fold quartile distance (whiskers) and extremes beyond the whiskers [Available under Creative Commons Attribution 4.0 International License].

3.3. Groundwater seepage

The potential importance of groundwater seepage into the water balance is recognised within the WETMECS classifications (Wheeler et al., 2009) of the major areas of lowland agricultural peats (Appendix-1). Some types of lowland agricultural peats form in the natural groundwater discharge zone (Wheeler et al., 2009) where the groundwater head within the underlying aquifer would be expected to be higher than in the peat, potentially facilitating a vertical inflow (e.g. Varmaat et al., 2016; Simpson et al., 2011a&b) into the peat and/or ditches. Such sites include those identified as the WETMEC Groups of Groundwater floodplains (WETMEC 7) and Groundwater bottoms (WETMEC 8 and 9) of Wheeler et al. (2009).

The vertical flow direction can vary spatially within peat depending on factors such as local anisotropy in hydraulic conductivity (Drexler et al., 1999), local high specific storage (Reeve et al., 2006), methane production (Romanowicz et al., 1993) and the transpiration rate of the vegetation type (Drexler et al., 1999). Hydrological studies carried out in Broadland fens (Baird et al., 2004; Surridge, 2005, van Wirdum et al., 1997 [latter two cited in Wheeler et al., 2009]) provided limited piezometric evidence for groundwater upflow into the peat, even in locations without underlying low permeability deposits, suggesting that the low vertical hydraulic conductivity of deeper peat layers or the interface between the peat and underlying deposits is acting as an effective aquitard (Wheeler et al., 2009). However, Gilvear et al. (1997) found an upward hydraulic gradient from the Crag aquifer underlying Catfield Fen in the Broads, which was particularly pronounced in the summer when high evapotranspiration rates lowered the water table in the peat, suggesting that the relatively small and fairly constant groundwater inflow of around 10mm per month provided “a degree of self-regulation”. Consequently, groundwater inflow into the peat can help meet summer evapotranspiration demands, thereby reducing losses from groundwater storage and limiting water table drawdown.

Toro et al. (2022) described upward groundwater flow into the peat during times with enhanced evapotranspiration and sufficient groundwater inflow from the larger groundwater catchment, as has been reported in other inland peatlands (e.g. Siegel and Glaser, 1987; Devito et al., 1997; Drexler et al., 1999; Fraser et al., 2001; Kopp et al., 2013). However, downward groundwater flow out of the peat occurred during winter periods of high precipitation and small ET to support coastal groundwater discharge (Toro et al. 2022).

The magnitude of upward flow into the drains can be influenced by the drainage system and its hydrogeological connectivity of ditches with the underlying aquifer, indicated by the thickness (if any) of peat and other low permeability layers between the ditch bed and the underlying aquifer (Simpson et al., 2011a&b). With raised drain water levels, the magnitude of simulated groundwater seepage directly into the drains in north-east Norfolk declined, with associated small increases in upward vertical seepage into the peat within the fields (Holman et al., 2013). In contrast, the close drain spacing seen with sub-surface infiltration (SSI) systems can inadvertently lead to the drainage of upward groundwater inflow into the peat from an underlying aquifer that would otherwise support higher water

tables during drought (Boonman et al., 2022). However, SSI systems may also sustain groundwater tables in situations or times with downward flow (Boonman et al., 2022).

Upward groundwater flow into the peat is not always beneficial. High vertical head gradients and/or high connectivity between the peat drainage systems and the underlying aquifer can lead to saline groundwater inflow (Holman et al., 2013; Simpson et al., 2011a) and, in extreme cases, saline boils (de Louw et al., 2010, 2018) in lowland drained coastal peatlands.

3.4. Topography

Changes in water table position and vegetation often co-vary across natural peatlands with changes in land surface elevation, or microtopography, from dry, raised hummocks with shrubby vegetation, to lawns with intermediate water table depth, to hollows which are seasonally to permanently inundated (Perryman et al., 2022; Price and Whitehead, 2001). In a mineral poor fen in New Hampshire, USA, Perryman et al. (2022) found a difference of 12 cm in mean water table depth between hummocks and lawns. This is likely associated within the differing K_{sat} found between surface microform type (e.g., hummocks, lawns, and hollows) at spatial scales of a few meters (e.g., Baird et al., 2016; Morris et al., 2015) that partly reflects the influence of peat floristic composition upon pore structure and water movement (McCarter et al., 2021). Microform topography in the K_{sat} ($m\ s^{-1}$) equation of Morris et al. (2022) influenced $\log_{10}(K_{sat})$ by between +0.188 (lawn) and 1.013 (hummock) compared to the reference category of unspecified microform type.

Williamson et al. (2017) evaluated the regeneration of historically drained upland blanket bog in North Wales (at an altitude of 350-500m). They concluded that the limited ecosystem response to ditch-blocking resulted from past peat subsidence within 4-5m each side of the ditches lowering the peat surface towards the new ditch-controlled water table, a process also seen in temperate lowland peat sites (e.g. Lindsay, 2010; Erkens et al., 2016; Schothorst, 1977; cited by Williamson et al., 2017). The subsided ditch lines present a risk of inundation (leading to short-term high CH_4 emissions) when ditch water levels are raised to support higher water tables. This is evident in the comparison by Kreyling et al. (2021) of 320 rewetted sites with 243 undrained near natural sites representing temperate Fens across Europe. Drainage-induced subsidence led to more frequent inundation after rewetting (mean median water table being 5.0 cm above surface in rewetted sites versus 1.5 cm below surface in near-natural sites).

However, Malloy et al., (2014) found the opposite in a cut-over peatland where harvesting had led to interior regions having lower elevation and higher available water due to a relatively shallower water table. This effect can also be seen in some agricultural lowland peats where bank elevations are raised relative to the field interior due to the effects of depositing sediment from ditch drain cleaning. Although not directly influencing water flows, the resulting effects of topographical variation on water table depth and soil moisture can influence vegetation establishment (Crowley et al., 2021) and, thus, evapotranspiration.

3.5. Vegetation

Water level is a predominant driver of peatland vegetation succession, with increasing water table depth promoting changes from hydrophyte species (that are better adapted to submergence) to *Sphagnum* mosses and graminoids (e.g., sedges, reeds), to woody plants (shrubs and trees) (Zhong et al., 2020). However, the responses of plant production to alteration in water level are not only dependent on water availability, but also to associated changes in nutrient conditions (Riutta et al., 2007). Studies of the effects of water level alteration on vegetation reveal that the relationship is complex and depends on the nature of the hydrologic change, vegetation type, and nutrient levels (Chimner et al. 2016).

Raising water tables can also prolong the root growing season by reducing soil moisture stress (Schweiger et al., 2022) and promoting plant functional types such as graminoids (e.g. sedges) which are adapted to high and prolonged water tables and can grow roots deep into the saturated peat layers (Voeselek et al., 2006). Consequently, actual evapotranspiration from lowland fens in Germany showed annual differences of up to 230 mm/yr between groundwater levels of 30 and 120 cm depth (Renger et al., 2002), leading to negative climatic water balances (Wessolek et al., 2002).

Peatland rewetting on an ex-peat extraction site in Sweden (Lundin et al., 2017) lead to decreased annual discharge, increased frequency of days with low or no runoff and decreased frequency of peak flows which were ascribed to increased evaporation from water surfaces and increased surface water storage. Similarly bogs converted to agricultural use have lower evapotranspiration than undrained bogs (Price et al. (2003)

Although different wetland vegetation types have differing water table preferences (Timmermann et al., 2009) which will affect colonisation and succession following raising water tables, the main impact being considered here is on their water demands. Crop coefficients (K_c) represents an integration of the effects of three characteristics that distinguish the vegetation from the hypothetical reference surface used to compute reference evapotranspiration: the crop height and associated aerodynamic resistance; the crop-soil surface resistance; and the albedo of the crop-soil surface (Pereira et al. (2024).

Mountford et al., (2002) provided a summary of monthly and seasonal crop coefficients taken from the literature for three classes of wetland vegetation (Table 2), which suggested that actual evapotranspiration rates in wetlands are likely to be in excess of the potential rate. Eddy correlation measurements on two wetland types – wet grassland and reedbeds – in south west England by Acreman et al. (2003) similarly showed that evapotranspiration from the reed bed exceeded that of the grassland wetland by 15% (or 50mm over the 5 months) between July to November 1999. The evapotranspiration rates at both sites exceeded the Penman Potential Evaporation estimates by 4% (wet grassland) and 19% (reedbed). In contrast, Fermor et al. (2001) and Pereira et al. (2024) report K_c values for common reed (*Phragmites australis*) that are greater and lower than one.

This variability in reported K_c will partly be a consequence of local site factors. Small stands of wetland plants may have K_c values 0.20–0.25 greater than large stands due to local advection; whilst a difference in K_c values for common reed (*Phragmites australis*) of about 0.10 is found between reeds growing in standing water and moist soil (Pereira et al. 2024; Mountford et al., 2002 – Table 3). Consequently, evapotranspiration demand will be influenced by the spatial extent of the area with raised water tables and whether this rise leads to prolonged standing water.

Table 2. Monthly crop coefficients for different vegetation types (reproduced from Mountford et al., 2002, Copyright © 2002, NERC/Centre for Ecology & Hydrology).

	Source	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec
	1	0.81	1.01	0.70	0.90	0.85	1.33	1.40	1.68	2.15	1.89	1.16	1.10
Reed: (standing water)	2	-	-	-	-	-	-	1.68	1.26	1.46	1.75	1.11	-
	3	-	-	-	1.10	1.10	1.10	1.10	1.16	0.83	-	-	-
	4	1.00	1.00	1.00	1.00	1.10	1.20	1.20	1.20	1.20	1.20	1.10	1.00
	Best est.	0.91	1.00	1.00	1.00	1.05	1.20	1.35	1.32	1.41	1.61	1.12	1.05
Reed: Moist soil/sedge fen	4	0.70	0.70	0.70	0.90	1.05	1.20	1.20	1.20	1.20	1.20	0.90	0.70
Wet grassland	2	-	-	-	-	-	-	1.40	1.30	1.27	1.27	0.99	-
Grass Pasture	4	0.40	0.40	0.40	0.95	0.95	0.95	0.95	0.95	0.85	0.85	0.40	0.40
	Best est.	0.40	0.40	0.40	0.95	0.95	0.95	1.18	1.13	1.06	1.06	0.70	0.40

Table 2. Average crop coefficients for different vegetation types (recreated from Mountford et al., 2002, Copyright © 2002, NERC/Centre for Ecology & Hydrology).

Vegetation type	Winter average	Summer average
Reed – standing water	1.12	1.22
Reed – moist soil/Sedge Fen	0.82	1.13
Wet grassland	0.56	1.04

Raising the surface water levels by 0.2m (to a depth of 0.4m in winter and 0.3m in summer) in Dutch peatlands led to a simulated 10-15% increase in water requirements due to enhanced evapotranspiration (Querner et al., 2012). In contrast, adding sub-surface drains within a surface water level of 0.5m below ground level to support lateral infiltration to maintain high water table levels increased summer water requirements by 30% owing to the more consistent and higher soil water content.

Finally, there appears to be little published evidence of the impact of scrub clearance on peatland hydrology. Price et al. (2003) reviewed evidence from the Somerset levels to suggest that birch may be associated with evapotranspiration losses up to 30% greater than nearby *Molinia* (purple moor grass) and that drained woodland may have evapotranspiration of the order of 50mm (more than 10%) greater than undrained *Sphagnum* bog.

4. Developing a re-wetting typology of LAP

4.1. Introduction

The degradation in lowland agricultural peats that has arisen from drainage occurs within the upper soil layers which would normally, in less or non-degraded systems, provide the greatest capacity for water storage and water transport within the soil profile. In conjunction with higher drain water levels, the reduced seepage losses due to this reduction in hydraulic conductivity in lowland agricultural peats (Morris et al., 2022) and their lower specific yield is expected to help to achieve raised water tables within the cooler-wetter periods of the year but also greater water table fluctuations (Price et al., 2003; LaRose et al., 1997).

Traditionally, peat has been identified by its parent material (e.g., *Sphagnum* moss, wood, and sedge/graminoid) or by its wetland classification (e.g., bog, fen, or swamp) but these descriptors alone do not provide reliable information on the hydrological properties of peat without other direct hydro-physical measurement such as bulk density, drainable porosity, saturated hydraulic conductivity (Ksat), etc. (Liu and Lennartz, 2019; MacAlister et al., 2020) or indirect measurements such as degree of humification (Morris et al., 2022). Consequently, we have first considered the characteristics that both influence peat physical properties and are typically (but not always) recorded during soil mapping to inform peat classification according to Avery (1980) and Hodgson (1997) e.g. degree of decomposition / humification; peat-forming vegetation type (e.g. *sphagnum*, grass-sedge etc); pH.

4.2. Insights from peat data within LandIS and previous Defra lowland peat projects

Trophic type, pH, land use, vegetation type and bulk density

Trophic type and pH

The trophic type of the peatland was an important fixed effects in the two Ksat models of Morris et al. (2022) with the highest performance (Model 1: $r^2=0.764$; Model 2: $r^2=0.656$). Morris et al. (2022) used the sub-divisions of wetland trophic type based on the pH ranges of the “mire water” of Rydin and Jeglum (2005) of bog pH <4.2, poor fen 4–5.5, intermediate (and moderately rich) fen pH = 5–7, and extremely rich fen pH = 6.8–9. It is important to note that “poor” and “rich” are used by Rydin and Jeglum (2005) in a floristic sense rather than a nutrient or productivity sense.

Data on peat pH, measured in water, was collected during the Lowland Peat Survey (Burton and Hodgson, 1987). Measurements from 11,965 horizons within 2180 sites were collated from LandIS. The red area in Figure 14 reflects potentially sulphuric horizons on the basis of pH alone - sulphuric horizons are defined in Avery (1980) as having a pH (in

CaCl₂) of less than 4.0 after drying; jarosite mottle of specific chroma and hue criteria and at least 0.1% sulphate-S and oxidisable S. It has been assumed that the pH in water is about 0.25 higher than in CaCl₂ (Burton and Hodgson, 1987), although this differs slightly from Davies (1971). 815 sites had a pH in water of less than 4.25 which, if in association with pyrite in seawater-influenced depositional environments, can lead to the generation of ochre where drained.

Peats with low pH were observed in parts of the Broads, Fenland, the Vale of Pickering, the Idle valley (Nottinghamshire), parts of the Lancashire coastal plain, Shropshire (Baccy, Acksea and the Weald Moors), Somerset (Queen's Sedge Moor and North Moor) (Burton and Hodgson, 1987). There is little relationship between peat pH and the pH classes of Rydin and Jeglum (2005), due to the low pH within some lowland peats being influenced by the presence of sulphides within the peat which oxidise to sulphuric acid (Burton and Hodgson, 1987), rather than by the source of water. Consequently, pH measurements of peat samples provide little value to a re-wetting typology, although they may be useful in identifying co-benefits of raised water tables such as reduced ochre formation.

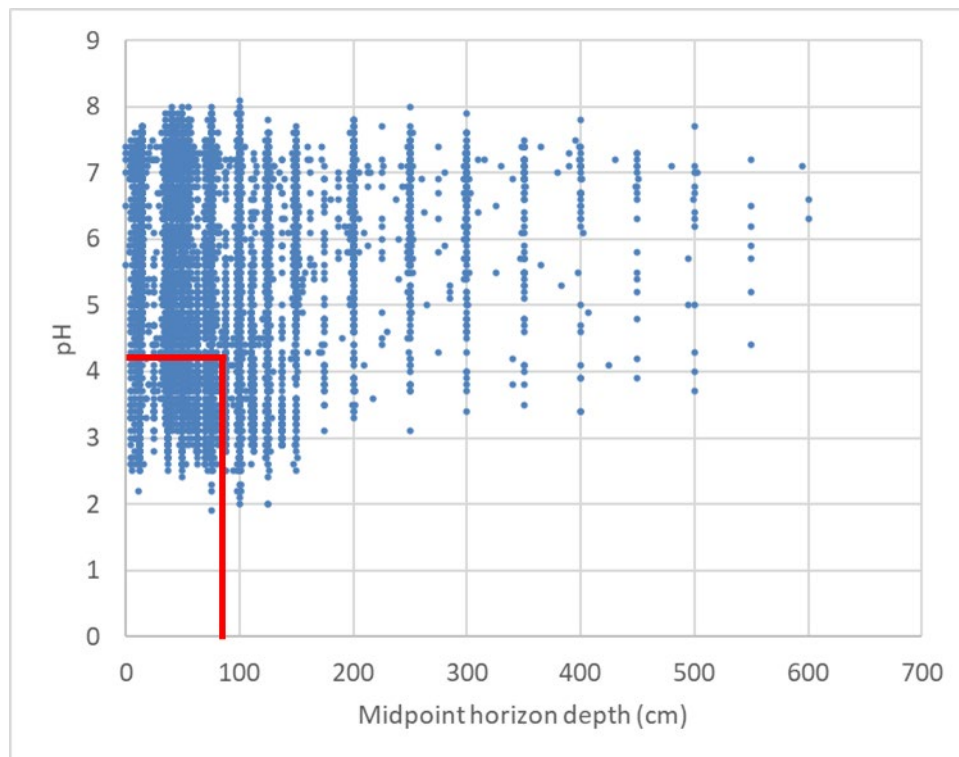


Figure 14. pH in water measurements in lowland peat horizons within the Lowland Peat Survey (Burton and Hodgson, 1987), with potentially sulphuric horizons with the red demarcated area [Soils data © Cranfield University, 2025].

Land use

The influence of land use has been considered from application of the Water Table Fluctuation (WTF) method (Bourgault et al. 2017) and analysis of LandIS soil water release data. The WTF method was applied to the high resolution water table monitoring data of UKCEH flux tower sites, where specific yield (SY) was calculated as follows:

$$S_Y = P/\Delta h$$

where P is the amount of precipitation (mm) and Δh is the water level rise (mm) following the precipitation event. The method assumes that the time lag between the end of each precipitation event and the maximum water level rise is sufficiently short to ignore evapotranspiration, net subsurface flow, and water table recession following P events and that recharge is equal to precipitation (i.e., no runoff). The box and whisker plot (Figure 15) suggests that the specific yield is generally lower in the arable and horticultural sites compared to the grassland sites, but that there is significant variability in the former. The high specific yield at the EF-WA (arable and horticultural) may reflect the influence of deep cultivation practices or the presence of an effective drainage system that limits the water table rise. Similar patterns are seen in the data from LandIS (Figure 16) which is based on soil water release data for total porosity and field capacity (5 kPa) suggesting that agricultural land use has limited utility as a direct predictor of peat hydraulic properties.

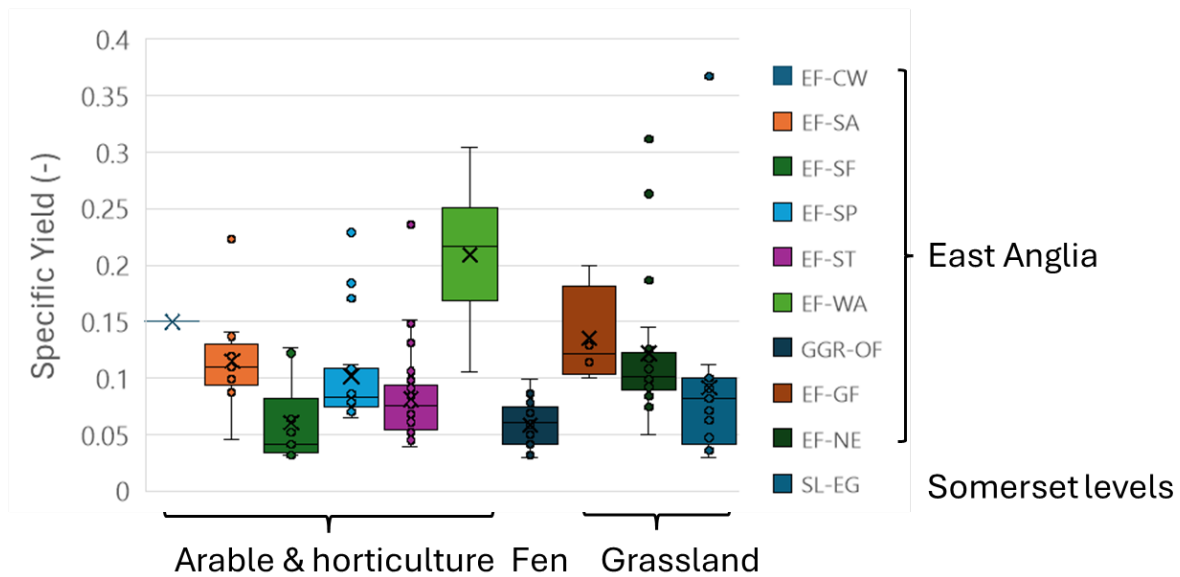


Figure 15. Box and whisker plot of specific yield estimates derived from applying the Water Table Fluctuation Method to high resolution water table data from UKCEH flux tower sites.

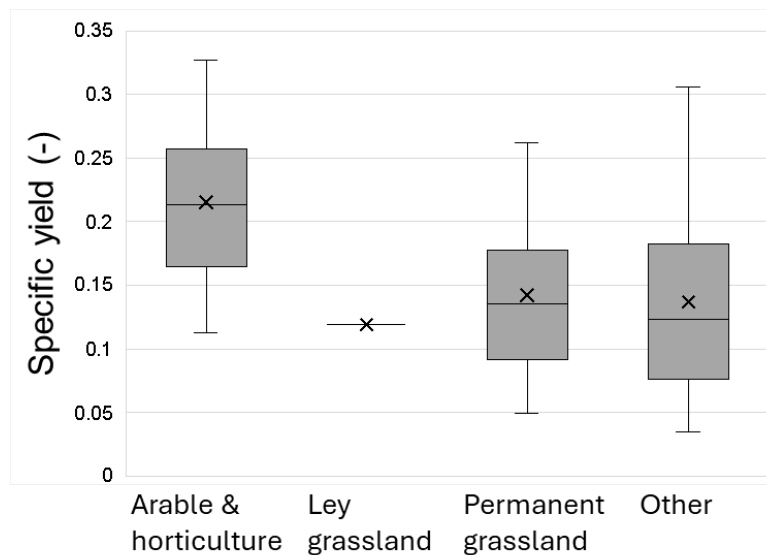


Figure 16. Box and whisker plot of specific yield estimates derived from soil water release data within LandIS [Soils data © Cranfield University, 2025].

Vegetation type and bulk density

In the two K_{sat} models of Morris et al. (2022), trophic type has an effect range on $\log_{10}(K_{sat})$ of 3.499 (Model 1) and 2.034 (Model 2). However, the effect of the more minerotrophic peat types in Models 1 and 2, (and 3) is to increase $\log_{10}(K_{sat})$, rather than reduce it, suggesting that the effect of trophic type doesn't reflect different levels of peat humification and/or compaction but may include the lower susceptibility of sedge peat, compared to *sphagnum* peat, to losing large pores during decomposition and the development of preferential flow paths in sedge peats, and by extension reeds, due to anisotropy (McCarter et al., 2021; Baird and Gaffney, 2000).

Much of our understanding of the physical structure of peat comes from *Sphagnum* peat while the pore structure of sedge or woody peats (from fen or swamps) remains less clear (Liu and Lennartz, 2019). There is therefore little knowledge on the influence that peat-forming species have on hydraulic properties (McCarter et al., 2020) with only broad descriptors of bog and fen peat and parent material (i.e., *Sphagnum*, woody, sedge, etc.) having been reliably used within pedo-transfer functions to predict peat properties (Liu and Lennartz, 2019; Morris et al., 2022). However, whilst Liu and Lennartz, 2019) found that peat type (*sphagnum*, woody or sedge) was an important control on the soil water release curve for peats with a bulk density $\leq 0.2 \text{ g/cm}^3$, they found that peat type had no significant effect on "highly degraded" peats with a bulk density $\geq 0.2 \text{ g/cm}^3$.

Within the Lowland Peat 1 sites (Evans et al., 2017), most of the degraded sites (East Anglian Fens, Manchester Mosses, Somerset Levels) had an average bulk density in the upper 50cm exceeding the critical threshold of Liu and Lennartz (2019) of 0.2 g/cm^3 (Figure 17). There are only 50 lowland peat samples within the LandIS database that have been analysed for bulk density, of which 14 are from arable and ley grassland sites and 16 are from permanent grassland with the remainder being non-agricultural. Figure 18 shows

the strong decrease in bulk density with depth in the arable and ley grassland samples, with the decline being less clear in the permanent grassland sites, but (similarly to the data from Evans et al., 2017) most samples also exceeded the critical threshold of Liu and Lennartz (2019) of 0.2 g/cm³. This suggests that peat type (*sphagnum*, woody or sedge) may not be a useful variable in characterising lowland agricultural peats.

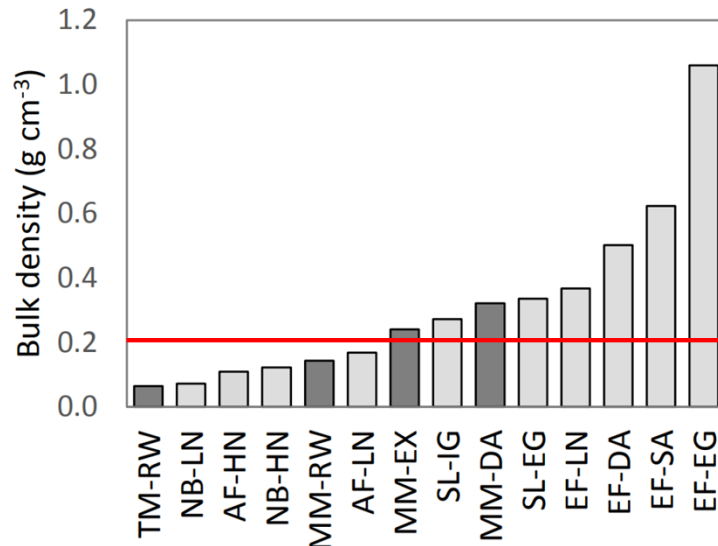


Figure 17. Average bulk density within the upper 50cm from Evans et al. (2017), with the critical threshold of Liu and Lennartz (2019) marked. Sites on raised bog peat are shown in dark grey, sites on fen peat in light grey (TM-RW: Thorne Moor re-wetted semi-natural; NB-LN: Norfolk Broad, low nutrient semi-natural; AF: Anglesey Fens- LN Low-nutrient semi-natural, HN high nutrient semi-natural; MM: Manchester Mosses- RW re-wetted semi-natural, EX peat extraction, DA deep peat arable; SL: Somerset Levels- EG extensive grassland, IG intensive grassland; EF: East Anglian Fens; - LN low nutrient semi-natural, EG extensive grassland, DA deep peat arable, SA shallow peat arable.

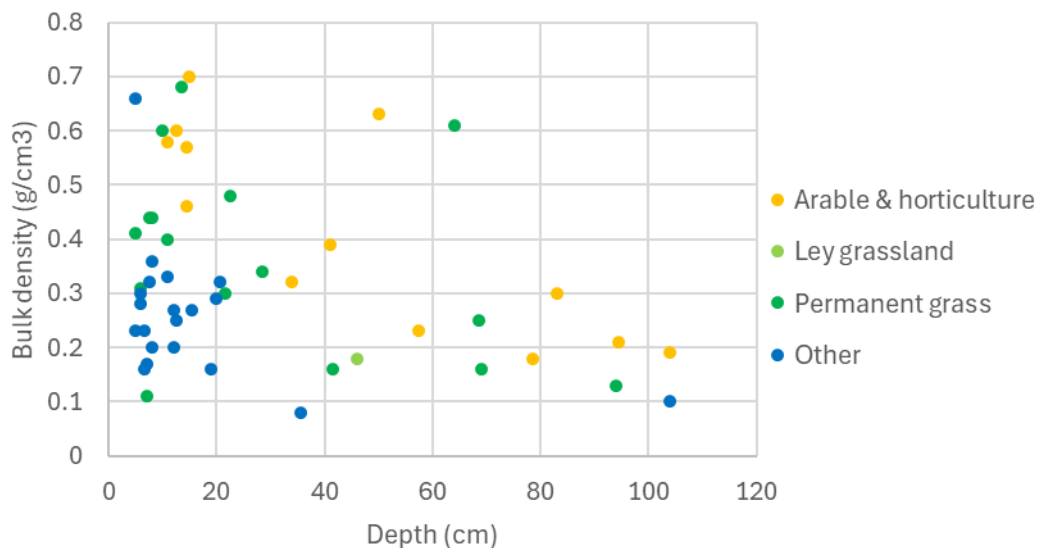


Figure 18. Measured bulk density (g/cm³) vs depth (cm) from lowland peat horizons within LandIS [Soils data © Cranfield University, 2025].

Degree of humification

Increased humification and bulk density associated with agricultural drainage leads to a reduction in hydraulic conductivity (Kreyling et al. 2021; Morris et al., 2022) and specific yield within the upper layers (Liu et al. 2020; Price et al. 2003), significantly changing the vertical distribution of these properties compared to in pristine peats (Menberu et al 2021). Morris et al. (2022) determined that dry bulk density and degree of humification are particularly important predictors of saturated hydraulic conductivity in peat soils, with Päivänen (1969) showing that the degree of humification on the 10-point Van Post scale was strongly correlated with bulk density ($r=0.918$).

In contrast to the limited number of bulk density measurements, there are far more assessments of the degree of humification within LandIS, with 9534 peat horizons within the auger bore collection which includes those from the Lowland Peat Survey (Burton and Hodgson, 1987) and 1434 peat horizons within the Somerset Levels survey of Cope and Colborne (1981). The Somerset Levels data were assigned a degree of humification based on their horizon description (amorphous $\equiv$ humified; semi-fibrous; fibrous and unspecified). The auger bore data were assigned to the same classes based on, in decreasing precedence, (1) von Post scale; (2) texture description and (3) horizon notation.

Humified peat horizons (von Post scores of H7-H10) are typically found at shallower depth than semi-fibrous and fibrous horizons (Figure 19) due to the direct (drainage-induced wastage) and indirect effects (crop offtake, wind erosion etc) of agricultural drainage but have been observed at depths far in excess of the maximum likely water table depth in drained systems, reflecting the increasing humification with age of peat (McCarter et al., 2021). However, when the thickness of peat horizons classified as humified (van Post 7-10) within a soil profile is expressed as a percentage of total recorded peat thickness, it is

apparent that there are increasing proportions under progressively more intensive agricultural systems (rough grazing; median = 54%; permanent grassland median = 94%; arable and ley grassland median = 100%) reflecting the increasing influence of progressively more intensive agricultural drainage (Figure 20). This may be partly explained by the peat type, as the remains of sedges and reeds that characterize fen peat are thought to decompose more readily than the *Sphagnum* peat found in bogs, resulting in more highly decomposed peat closer to the surface (Liu and Lennartz, 2015).

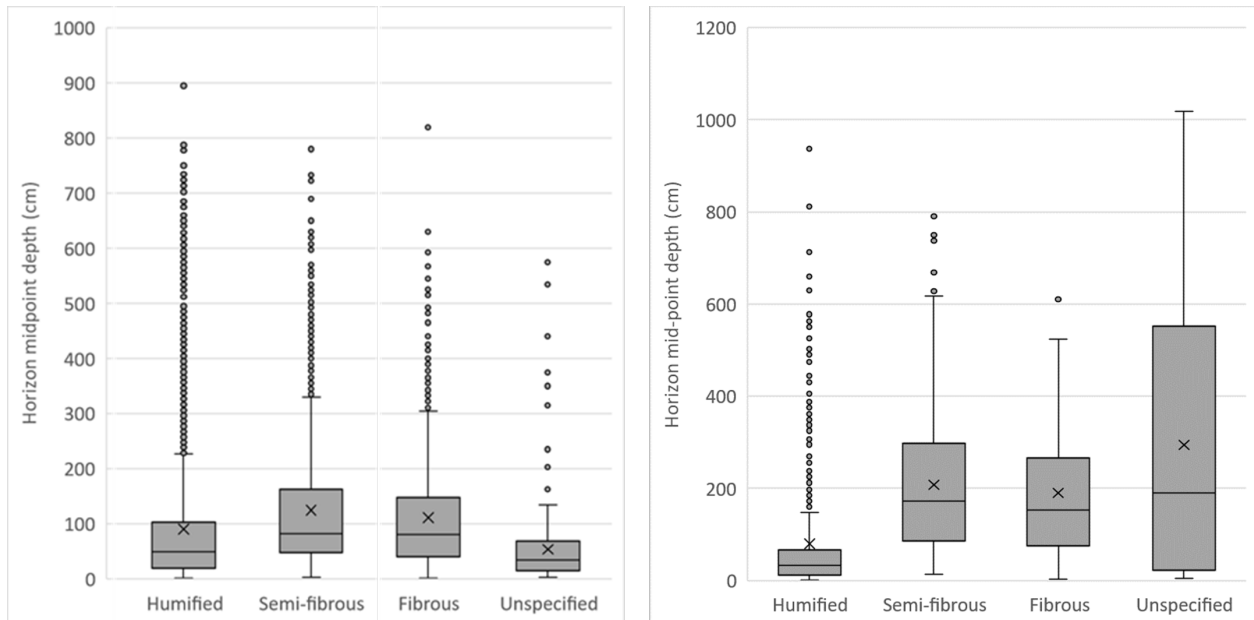


Figure 19. Box and whiskers plot of the distribution of the horizon midpoint depth (cm) for different degree of humification for (left) LandIS auger bores and (right) Somerset levels [Soils data © Cranfield University, 2025].

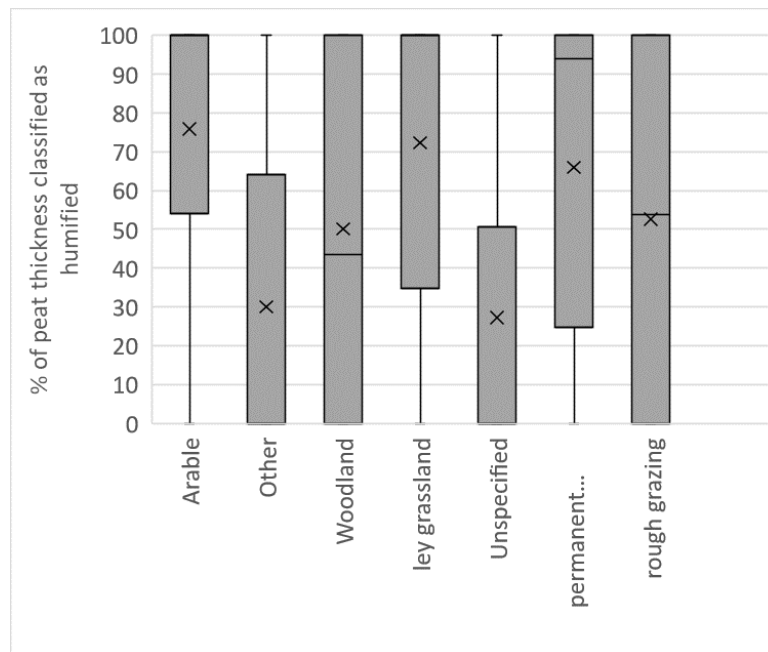


Figure 20. Distribution of the percentage of total peat thickness classified as humified (H6-10) by land use within LandIS [permanent = permanent grass] [Soils data © Cranfield University, 2025].

Finally, Figure 21 shows the thickness-weighted von Post class by land use showing the strong effect of increasing drainage management. For those horizons without a von Post assessment of decomposition, the mid-point value was given for the humified (von Post score of 8.5), semi-fibrous (5) and fibrous (2) horizons. The depth-weighted von Post class increases from a median value of 5 (inter-quartile range (IQR) of 4.1-6.7) for non-agricultural land uses excluding woodland, to 6.8 (IQR 5.0-8.5) for unmanaged grassland or rough grazing, to 7.7 (IQR = 5.9-8.5) for managed grassland (ley grassland and permanent grassland) to 8.3 (IQR= 6.8-8.5) for arable. Consequently, long-term agricultural land use is a useful a proxy for the degree of drainage-induced humification and associated peat degradation.

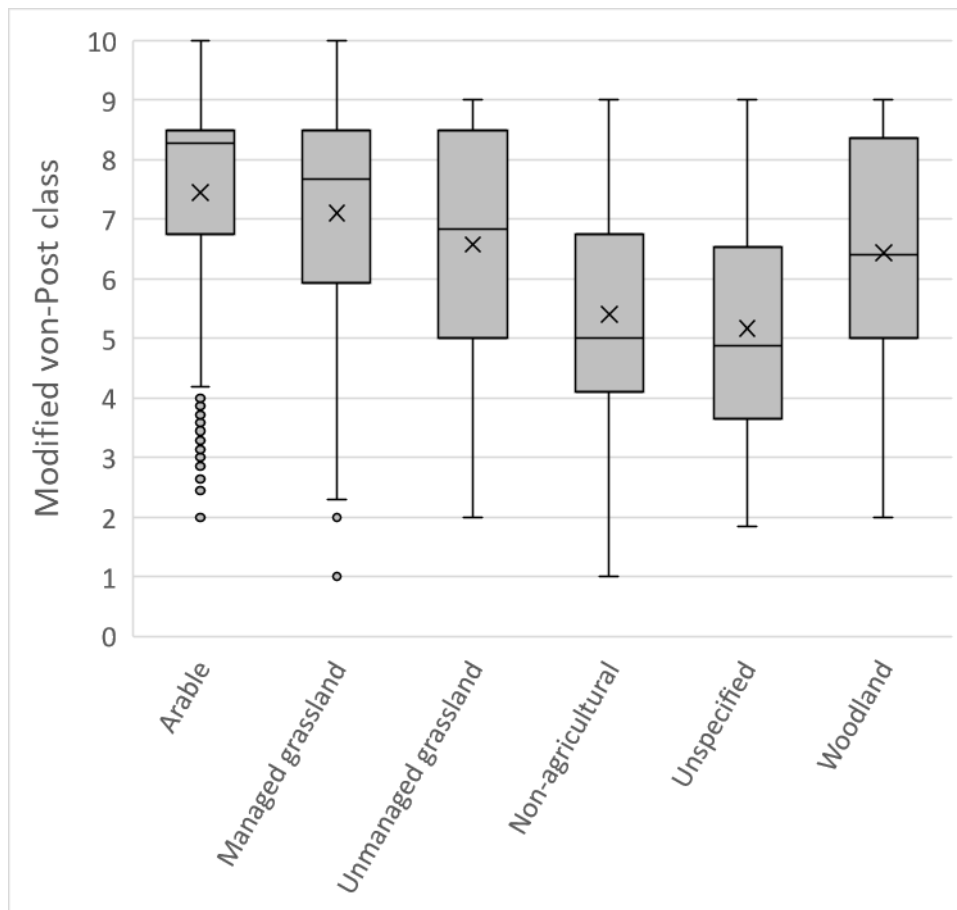


Figure 21. Distribution of thickness-weighted modified von Post score by land use [Soils data © Cranfield University, 2025].

4.3. Peat re-wetting typology

Considering Section 3 and 4.2, the likely effectiveness of raising water table in a given peat location will be dependent on a number of elements:

- Agro-climate – wetter and cooler areas of the country have higher hydrologically effective rainfall (the surplus of precipitation over evapotranspiration), will develop smaller potential soil moisture deficits during summer and thus smaller evapotranspiration-induced drops in the water table. All other things being equal, raising and maintaining higher water tables will be relatively less challenging in wetter areas (such as the Somerset Levels) than drier areas (such as the Fens), notwithstanding any concerns around flood risk;
- Available water resources – raising water tables within peats will lead to an increase in water requirements due to higher evapotranspiration (Section 3.5) associated with reduced variability in saturated and unsaturated zone water content (and reduced soil moisture stress). This can potentially be met through sub-surface irrigation (Section 3.1) or surface irrigation (Richert et al., 2000; Evans et al., 2021) The extent to which this increased demand can be met from local water resources will depend on the contribution of groundwater to the peat water balance (Section

3.3), any current or future hydrological interactions with the river system (e.g. flood overtopping of river banks), IDB summer pumping rates and the Environment Agency's water resource availability and abstraction reliability assessment (Environment Agency, 2024a) and abstraction licencing strategy (Environment Agency, 2023).

- Peat properties – the modified peat properties of lowland agricultural peats will influence the ability of water to move into / out of the peat in support of higher water tables (hydraulic conductivity: Section 3.2) and the water table response to changing short-term water storage (specific yield: Section 3.2). The degree of drainage-induced modification of these properties is related to the inherent susceptibility of different peat types to degradation, the intensity of drainage management and the loss of peat (e.g. due to wind erosion, crop off-take and peat extraction).
- Drainage management – the original purpose of agricultural drainage is two-fold – to effectively control in-field water table depth to promote agricultural production and to convey excess water away for discharge into the river network. In the absence of other interventions, it will be more effective to raise and maintain high water tables within sites with efficient water table control through increased drain water levels, due to their smaller drain spacing and higher ability to support sub-irrigation (Section 3.1). The wider drainage system also provides an important conveyance function in support of raised water tables, moving water from areas of water supply (e.g. rivers, highlands, reservoirs etc) to the raised water tables field(s).

Many of these factors (together with others identified in Section 3) that influence the water table within lowland peats can be modified through management (e.g. changing drain water levels, land vegetation type) and/or physical interventions (e.g. changing drain spacing or drain layout and connectivity; modifying field topography; constructing winter storage reservoirs etc). However, the physical properties of drained lowland agricultural peats cannot easily be changed in the short-term (notwithstanding options of removing sub-surface compaction through subsoiling or establishing deep-rooted crops- Guedessou et al., 2021) and will exert a strong influence on the future behaviour of a raised water table. Based on these findings, we propose a simple re-wetting typology of peat types (Table 4) based on:

- Trophic type: to (1) account for the effects of the original peat-forming vegetation type on peat properties and susceptibility to degradation and (2) to also reflect the potential for external sources of water (surface water and groundwater) to support higher water tables.
- Past site usage: to account for the effects of agricultural drainage, we propose subdividing past site usage based on land use as a proxy for drainage intensity, degree of humification and physical removal of peat- (1) sites under arable and horticultural management which requires strong drainage control of the water table usually seen through narrow drain spacings or wider drain spacings augmented by sub-surface

drains, and low drain water levels; (2) grassland sites which are usually less intensively drained with wider drain spacing and higher drain water levels; and (3) extractive sites where the combined effects of drainage and peat removal will have extensively degraded the peat. Although the duration of drainage has important consequence for peat degradation (Ma et al. 2022) and peat bulk density (Liu et al. 2020), data on the drainage history of individual sites is often poor and so has not been included.

Although there are many factors not considered, this typology considers that peat properties in lowland agricultural peat are more likely to be conducive for re-wetting interventions (1) in fen peats compared to raised bogs as peat K_{sat}, which is an important control on lateral sub-irrigation needed to support raised water tables, increases along the trophic gradient from bog to rich fen peat (Morris et al., 2022); whilst fen peats are more likely than raised bogs to have additional water sources (beyond precipitation) such as groundwater upwelling and/or river inputs to support water tables. And (2) in grassland systems where peat properties have been less degraded by drainage compared to arable/horticultural landuses (by drainage, cultivation, harvesting, wind erosion) or peat harvesting / extractive landuses (by drainage, peat removal, wind erosion). The monitoring sites used in Lowland Peat 1 (Evans et al., 2017) and being used in Lowland Peat 3 (LP3) and LP3+ are provisionally assigned within the classification. Although there are no sites within the “poor fen” trophic level, this is likely to reflect the limited occurrence of this trophic level within lowland peat environments, rather than a significant omission in the site selection.

Table 3. Provisional classification of Lowland Peat 1, 3 and 3+ sites within the proposed re-wetting typology according to previous land use and peat trophic type.

Trophic type ^a	Land use			
	Non-agricultural	Agricultural (grassland)	Agricultural (arable)	Extracted
Raised bog	Astley Moss (MM-RW) ¹ Thorne Moor(TM-RW) ¹	Railway View (MM) ³⁺	Little Woolden Moss (MM-DA) ¹ Wrights (MM) ³	Thorne Moor(TM-EX) ¹ Little Woolden Moss (MM-EX) ¹
Poor fen				
Moderate fen	Sutton Fen (NB-LN) ¹ Strumpshaw Fen (NB-HN) ¹	Tadham Moor (SL-IG) ¹ Langley (NB) ³ Gold Corner (SL) ³ Sedgemoor (SL) ³⁺	Redmere Farm (EF-SA) ¹ Sawtry Fen (EF) ³ Stretham Stowbridge (EF) ³ Little Common (EF) ³	
Rich fen	Wicken Sedge Fen (EF-LN) ¹	Bakers Fen (EF-EG) ¹	Rosedene Farm (EF-DA) ^{1,3}	

	Land use			
	Cors Erddreiniog (AF-LN) ¹	Tadham Moor (SL-EG) ¹		
	Cors Erddreiniog (AF-HN) ¹			

^a Trophic type based on pH data within Evans et al. (2017) and the classification of Rydin and Jeglum (2005) of bog pH <4.2, poor fen 4–5.5, intermediate (and moderately rich) fen pH = 5–7, and extremely rich fen pH = 6.8–9.

^{1, 3, 3+}: Lowland Peat 1, 3 or 3+ sites. LP3 fen sites allocated to moderate fen trophic level in absence of data

EF = East Anglian Fen; SL = Somerset Levels; NB = Norfolk Broads; AF = Anglesey Fen (see Fig 4.2 for full site code description)

Very limited peat property data are currently available to populate Table 4 based on previous lowland peat research in the UK. For example, saturated hydraulic conductivity was only measured at Cors Erddreiniog within Lowland Peat 1 with reported average values ranging from 55.4 cm/day at 80cm depth to 1.5-4.2 cm/day below 1m (AF-LN) (Evans et al., 2017). Consequently, the measurement of hydraulic conductivity and specific yield peat property data within the Somerset Moors (Fen) and Manchester Mosses (raised bog) within LP3+ will significantly augment the available data. Additional data will also be collected from the six LP3 sites.

There are also potentially large differences in estimated values by method, with field based methods in the Somerset Levels providing horizontal Ksat values of around 10⁻¹ to 10⁻³ m/day (Baird et al., 1997; Baird and Gaffney, 1994; 2000; Barber et al., 2004) compared to 10⁰ -10⁻¹ for laboratory measurements (Kechavarvarzi et al., 2010). The measurement of Ksat by both dipwell tests and laboratory measurements, and specific yield by HyProp and sand bath methods within LP3+ will enable improved characterisation of this measurement uncertainty.

5. Challenges to achieving the benefits of countryside stewardship water table prescriptions

Field measurements in agricultural peatlands have shown that high surface water levels in ditches are often unable to support high groundwater levels in the centre of the field during summer (e.g. Jansen et al., 2007, cited by Boonman et al. 2022; Kechavarzi et al., 2007), which is attributed to the high summer evapotranspiration and low horizontal conductivity of degraded agricultural peat (Boonman et al. 2022). This illustrates some of the challenges of achieving and maintaining high water tables with degraded agricultural peat soils.

It is hoped that the new Countryside Stewardship Higher Tier (CSHT) grant for “raised water level on cropped or arable land on peat soils” (CSW17¹), which is available for whole or part parcels of arable land or temporary grass, will encourage farmers to raise the water table within lowland agricultural peat soils. CSW17 requires that “water levels will be maintained between 10cm and 30cm below mean field level” which will be monitored by a monthly record of “water levels in dip wells” (Defra, 2024). CSW18² provides a similar offer for permanent grassland. It is expected that there will be a second grant offering to maintain water levels between 31cm and 50cm depth. These criteria are primarily focussed on reducing greenhouse gas emissions due to peat degradation, although they may improve “soil health, natural flood management and biodiversity”.

Given these expected benefits and the reported variability in the water table regime of restored wetlands, the potential challenges to achieving the benefits of Countryside Stewardship water table prescriptions have been assessed from two perspectives:

1. Firstly, a review of the CSHT prescriptions against the water table requirements of key vegetation communities.
2. Secondly, the observed variability in water table data from Environment Agency shallow monitoring dipwells has been evaluated to assess the practicality of maintaining a water table depth within the narrow prescribed water table range with CSW17 and 18.

¹ [CSW17: Raise water levels in cropped or arable peat soils to near the land surface - GOV.UK](#)

² [CSW18: Raise water levels in permanent grassland peat soils to near the land surface - GOV.UK](#)

5.1. Hydrological regimes of the main vegetation communities

The hydrological regimes of five National Vegetation Classification (NVC) groups are summarised to illustrate the potential impact of raising water levels on the vegetation communities in the agricultural peatlands. Hydrological regimes are largely limited to average summer and winter levels (Table 5), as time-specific data were only available for one group (MG8). These NVC groups were identified as key vegetation communities in the agricultural peatlands or nearby designated sites: MG8 (*Cynosurus cristatus* - *Caltha palustris*) grassland, M24 (*Molinia caerulea* - *Cirsium dissectum*) fen meadow, M13 (*Schoenus nigricans* - *Juncus subnodulosus*) mire, S25 (*Phragmites australis*) tall herb fen, and S2 (*Cladium mariscus*) swamp.

For more information on the NVC groups, see Rodwell (2006).

Table 4. Summary of seasonal variations in water levels associated with selected NVC classes, and impacts of increasing levels (Wheeler et al., 2004).

NVC code	NVC name	Summer water levels	Winter water levels and inundation
MG8	Grassland <i>Cynosurus cristatus</i> - <i>Caltha palustris</i>	Observed levels: mean -30 cm, ranging -20 to -45 cm. Increasing levels: prolonged waterlogging will shift vegetation to mire or swamp types, e.g. nutrient depletion (M23) nutrient enrichment (S6 and S5).	Observed levels: water at or near the surface. Increasing levels: limited impact of waterlogging outside of the growing season.
M24	Fen meadow <i>Molinia caerulea</i> - <i>Cirsium dissectum</i>	Observed levels: wide range, mean -25 cm, down to -50 cm Increasing levels: Low fertility: vegetation shifts to M13; High fertility: vegetation shifts to M22 or common species (<i>Agrostis</i> sp.)	Observed levels: high winter water tables are observed Increasing levels: prolonged inundation will shift community towards 'swamp' like forms of herbaceous fen.
M13	Mire <i>Schoenus nigricans</i> - <i>Juncus subnodulosus</i>	Observed levels: best examples of M13 have summer levels at or near surface (e.g. -9 cm), but not inundated. Deeper water levels will degrade quality.	Observed levels: existing M13 sites have winter water levels at or near the surface (-5 to +1 cm) Increasing levels: reduced quality of M13 vegetation.

NVC code	NVC name	Summer water levels	Winter water levels and inundation
		Increasing levels: decrease in M13 quality is possible.	
S24	Tall herb fen <i>Phragmites australis</i>	Observed levels: large variability, -14 cm on average, but at greater and shallow water levels when associated with other vegetation. Increasing levels: reduction in the species richness of the community and may shift to more wetland species.	Observed levels: inundation is observed in many S24 sites, but is not considered essential. Increasing levels: prolonged inundation may lead to species loss.
S2	Swamp <i>Cladium mariscus</i>	Observed levels: - 15 to + 40 cm Increasing levels: prolonged high-water levels or inundation can shift communities to <i>Phragmites</i> or open water habitats	Observed levels: inundation in the winter may protect the plant from frost damage. Increasing levels: water levels >40 to 50 cm may result in loss of sedge cover, especially if mown late

MG8 (*Cynosurus cristatus* – *Caltha palustris*) Grassland

The MG8 grassland was selected to represent a wet grassland community for this report because it is a diverse plant community found on grazing land with a high density of artificial drainage, particularly in southern England (Wheeler et al., 2004). The community is typically composed of a mixture of grass, sedge and rush species. It is found on well drained geology, e.g. gravel or chalk, thus could be associated with groundwater-related WETMECs (7-9), but is also found on ditch-drained degraded ombrogenous peats (WETMEC 4; e.g. Somerset Levels) (Figure 22).

Further details on the main WETMECS of the main lowland agricultural peat areas are provided in Appendix-1.

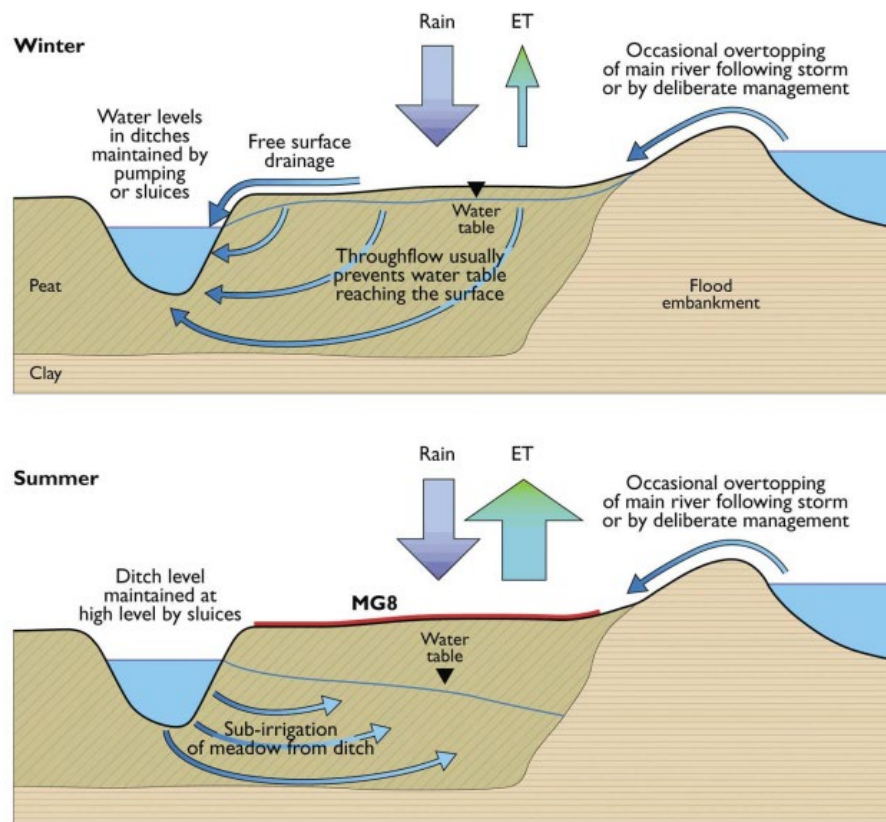


Figure 22. A representation of the hydrology of a MG8 grassland (red line) in a ditch-drained peat (Wheeler et al., 2004).

The community is found on soils with water levels within -50 cm of the surface (Wheeler et al., 2004). Summer levels must be lower than the root zone to ensure sufficient aeration (e.g. -30 cm) but can be higher in the winter (Figure 23). However, deviations from reference levels exist based on the hydrological mechanisms. Elevated water levels will negatively impact the MG8 community, shifting it towards mire or swamp communities depending on the degree of waterlogging and nutrient supply. Slight increases in waterlogging can shift the community to grasslands MG10 (*Holcus lanatus* – *Juncus effusus* pasture rush), or where associated with nutrient exhaustion, M23 (*Juncus effusus/acutiflorus* - *Galium palustre* rush-pasture). Greater waterlogging will shift the community to S6 (*Carex riparia* swamp) or, with nutrient enrichment, S5 (*Glyceria maximum* swamp). Year-round grazing and hay cutting by mid-summer is needed to control growth of tall rushes.

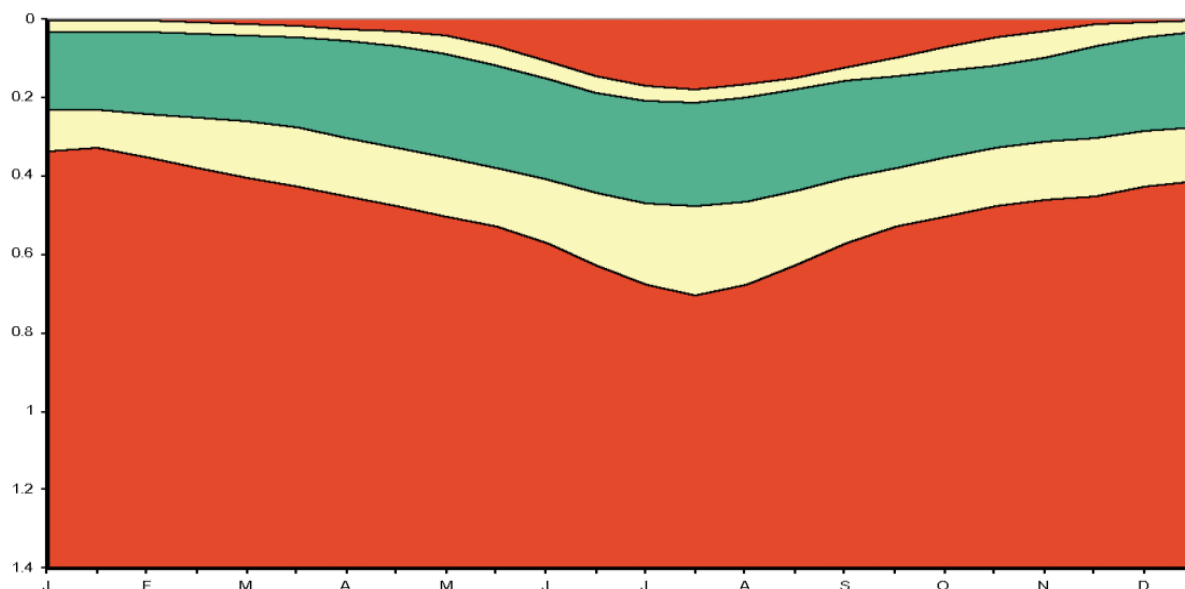


Figure 23. Water table level variations for MG8: green – optimal water levels, amber – water levels that if happened every year could cause a vegetation change, red – a change likely if water levels occurred in a single year (Wheeler et al., 2004).

M24 (*Molinia caerulea* – *Cirsium dissectum*) Fen meadow

The M24 vegetation community is found across a variety of habitats and soil types, including wet grasslands and fens on organic or humic soils. It can be found in groundwater fed basins (Wheeler et al., 2004), and is identified as having conservation value in WETMECs 7, 8 and 9. (Wheeler et al., 2009). Former examples were likely found in summer ‘dry’ floodplains (WETMEC 5) before conversion to agriculture (Wheeler et al., 2009) (Figure 24).

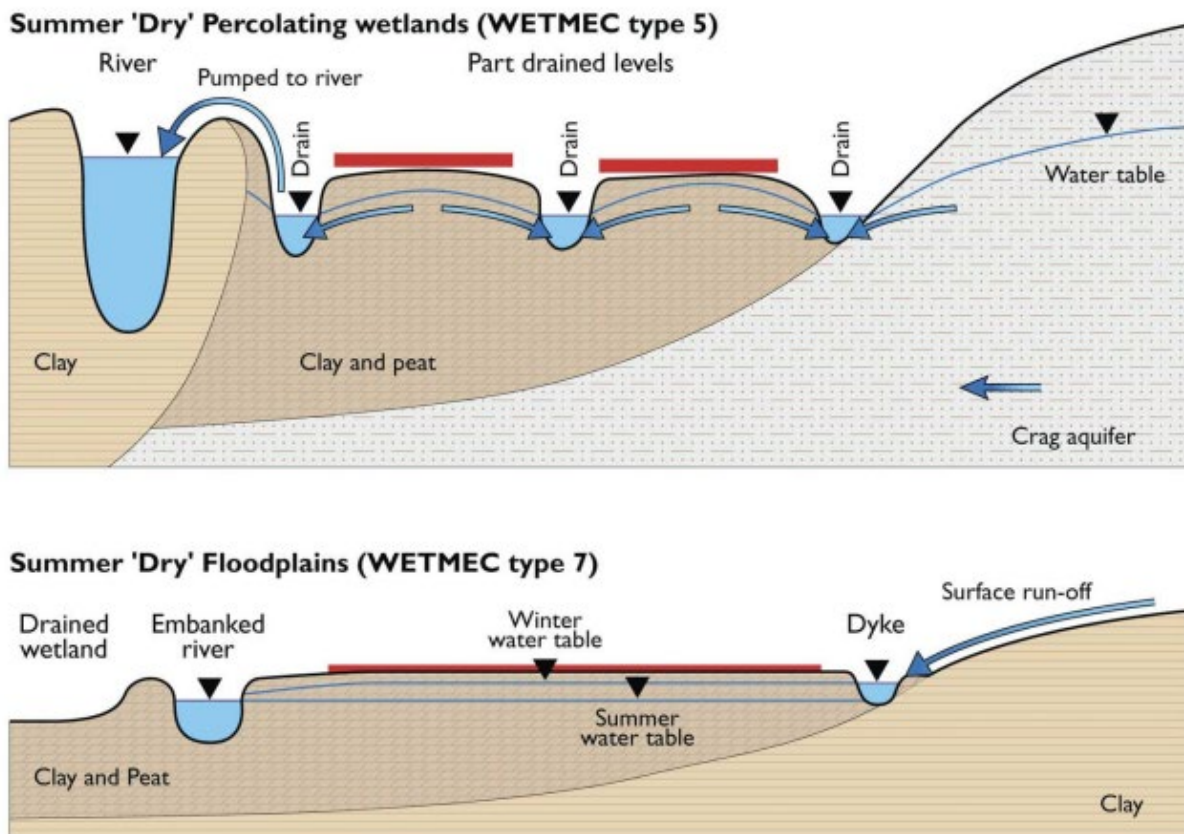


Figure 24. A representation of the hydrology and location of a M24 community (red line) in a ditch-drained floodplain (Wheeler et al., 2004). Note: the WETMEC numbers differ from Wheeler et al. (2009); Summer 'Dry Floodplain is WETMEC type 6.

No time-specific data on water levels were available to specify a hydrological regime, thus a general summary of water levels is provided (Wheeler et al, 2004). The community exists in locations with a wide range in summer water levels, down to -50 cm. High summer water levels may shift the community to M13 in low fertility basins but to M22 or common species (*Agrostis* sp.) in high fertility basin. Winter water levels may be high, but inundation is not common except in briefly in winter. Prolonged periods of surface inundation will favour 'swamp-line forms of herbaceous fen' (Wheeler et al., 2004). Management is required to maintain this vegetation community, mowing or grazing.

M13 (*Schoenus nigricans* – *Juncus subnodulosus*) Mire

The M13 vegetation community is currently limited in distribution geographically, though widespread in East Anglia (Wheeler et al., 2009). Patches are associated with WETMEC 9 – groundwater-fed bottoms. No time-specific data on water levels were available to specify a hydrological regime, thus a general summary of water levels is provided (Wheeler et al, 2004). Current M13 communities are typically located in areas with water tables that are very close to or at the surface. Inundation in winter may impact the quality of M13 stands. Observed summer levels are variable, though the best quality M13 sites have water levels that are below the surface (ca. -10 cm). However, groundwater 'flushing' appears to

support these 'high grade' sites', which are located on slopes that may prevent surface water accumulation and reducing conditions, whilst maintaining wet soil conditions.

S24 (*Phragmites australis*) tall-herb fen

The S24 vegetation community is a tall-herb fen community that is currently found in limited locations in the UK. However, it is associated with WETMECS groups that might exist in the future with raised water levels in the agricultural peatlands (Figure 25). There is considerable variability in the hydrological conditions associated with current S24 communities (Wheeler et al., 2004). Summer water levels for sites in East Anglia were below the soil surface (mean = -15 cm). Deeper water levels are reported of subcommunities with *Glyceria maxima* (S24b; -26.1 cm) and *Symphytum officinalis* (S24c; -47.5 cm), shallower water levels are reported when associated with *Cicuta virosa* (S24e, -3.4 cm). Communities found on solid peat have a lower water level (-23.3 cm). Winter inundation is common at S24 sites but may not be an important factor. Spring or summer inundation may reduce species diversity and cause a shift to wetland species.

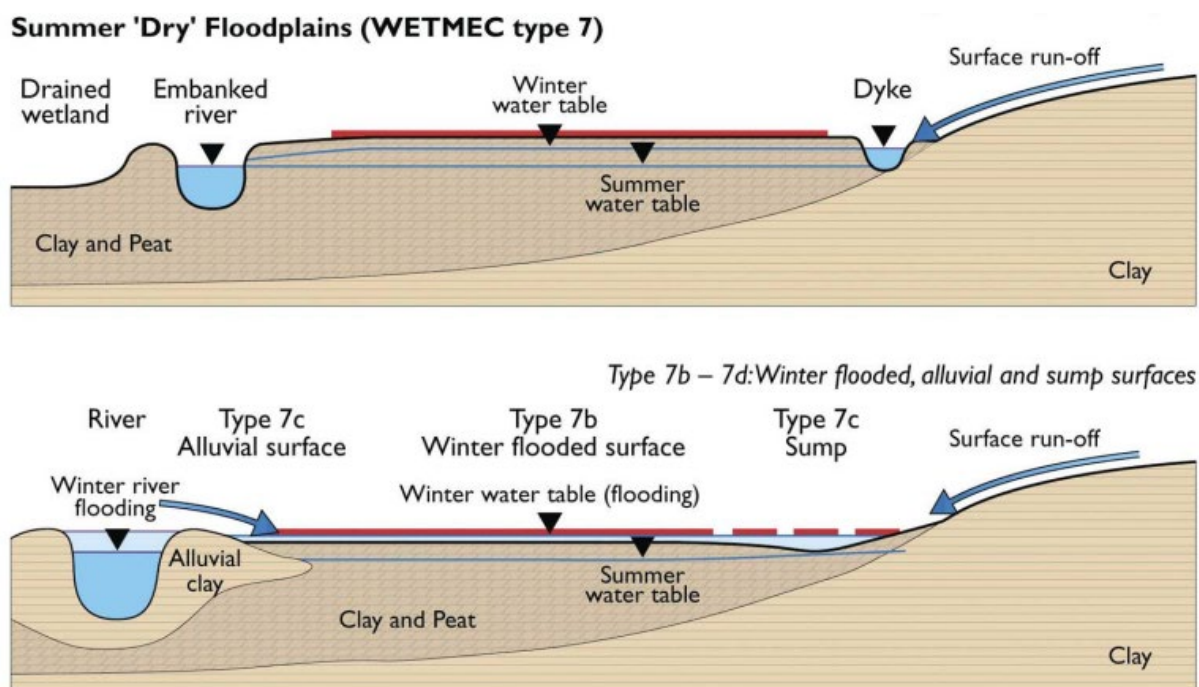


Figure 25. A representation of the hydrology and location of a S24 community (red line) in three types of floodplain (Wheeler et al., 2004). Note: the WETMEC numbers differ from Wheeler et al. (2009); Summer 'Dry Floodplain is WETMEC type 6.

S2 (*Cladium mariscus*) swamp

The S2 vegetation community is a tall sedge community found in wet fens and swamps, often in association with *Carex elata* swamp (S1) (Wheeler et al., 2004). Current examples of S2 communities are found in areas of extracted peat, but the paleaecological studies, presented above, suggest that it was a common component of the uppermost layer of peat in the Fens and Broadlands. There is considerable variation in the hydrological conditions

of current S2 communities. The community requires water levels above the surface at least part of the year, and appears to grow best when water levels are between -15 cm to 40 cm. Deeper water levels may support aquatic plant species. Water levels in excess of 40 to 50 cm may result in the loss of sedge cover, being replaced by open water or, with nutrient enrichment, *Phragmites*, *Typha* and *Glyceria* (e.g. S4). The community is not normally managed but is often adjacent to sedge and reed beds that are cut. It is more susceptible to winter high water levels if it has been mown in late summer.

Paludiculture

Eighty-eight native UK wetland species have been identified with promising potential for energy, food, fodder, medicinal use, and raw material provision (Natural England 2022). Most exploratory effort to date has focused on Bulrush (*Typha latifolia*) and farmed *Sphagnum*. It is suggested that *T. latifolia* is most suited to conditions where water levels are >30 cm above the ground surface in winter and close to the surface in summer (Timmerman, 2003, cited by Mulholland et al., 2020). However, Haldan et al. (2022) found maximum productivity in a mesocosm experiment with the watertable at 22 cm below soil surface, whilst the biomass production of *T. latifolia* exceeded *T. angustifolia* to 45cm depth.

In contrast, several studies have shown that the growth rate of most *Sphagnum* species is highest at water tables just below the capitula (top of the plant), independent of the species (Gaudig et al., 2020). In glasshouse experiments, a constant high water table remaining at 2 cm below the capitula during growth led to highest productivity for all tested species, with significantly lower productivity with water table fluctuations between 2 and 9 cm below capitulum during growth (Gaudig et al., 2020). However, growth rates observed at German sites reported by Mulholland et al. (2020) had almost 100% *Sphagnum* cover and a carpet thickness of 5-9 cm achieved within 1.5 years at one site where the water table was maintained within 10 cm of the ground surface and gaps in cover were filled at the end of the first year; while the second site, with a more variable water table and no gap-filling, achieved more than 90% cover within 3.5 years (Mulholland et al., 2020).

From Table 5 and Section 5.1, it is apparent that vegetation communities of conservation relevance and potential paludiculture crops can require water table depths and fluctuations that are beyond the current specification of CS17/18, reducing ability to achieve biodiversity and agricultural co-benefits from raised water tables.

5.2. Evaluation of the lowland peat water table variability within monitoring data

Environment Agency monitoring data

Groundwater level monitoring data were provided by the Environment Agency for shallow monitoring sites which are either located within the area of peat soils as indicated on the National Soil Map of England and Wales (in the absence of the new Natural England peat

map) or in which peat was mentioned within the borehole metadata. Borehole depth, where known, was limited to 5m based on Koch et al. (2023). Figure 26 shows the very uneven distribution of monitoring sites, with most being in the Broads. The landcover at each monitoring borehole was identified from the UKCEH Land Cover Class within the Land Cover Map 2020 (UKCEH 2022) for the site's given grid reference.

Groundwater levels had been collected by both manual measurement and high-resolution loggers, with some sites having both. Data on "dipped" groundwater depth (m) was used. Individual sites had between six and over 700,000 measurements. The data was quality-assured with the following changes applied:

- Negative dip depths of $>-0.1\text{m}$, which suggested significant depths of standing water above the measurement datum at the time of observation, were removed. In many of the EA monitoring sites where data were available, the measurement datum is around 0.5 to 1.0 metre above ground level.
- Dip depths of < 0 to -0.1m , which suggested small depths of standing water above the measurement datum at the time of observation, were edited to zero.
- Sites in which all, or almost all, measurements were negative (and appeared to be the reverse of the expected seasonal patterns) were converted to positive readings, assuming that the sign of the data had been incorrectly entered.
- Individual dip depths within high-resolution (15 minute or hourly) logged data that were excessively low/high compared to neighbouring data points were removed.
- Dip depths that were excessively deep compared to expected borehole depths were removed.

Two analyses were performed using this data, depending on the monitoring point metadata:

1. For monitoring points which lacked a ground level elevation such that groundwater levels could be not expressed as depth below ground level, the edited dip depth observations were expressed as a deviation from the median depth to evaluate the variability in water table levels. Figure 27 shows the distribution in the average deviation from the median dip depth by week number (week 1 = 1st week of January) for boreholes with different landcovers.
2. For those logged boreholes for which datum and ground level elevations were available (which are all in the Eastern region), the dip depths were converted to depth below ground level. Time series of groundwater depth by year were produced for grassland (Figure 28), woodland (Figure 29) and fen sites (Figure 30) to demonstrate the inter- and intra-annual variability in absolute water table depth.

Fig. 27 shows that, as would be expected, positive deviations (water table is above the median observed dip depth) are more common in winter and negative deviations (water table is below the median observed dip depth) are more common in summer. The data

also shows that there is a wide range of variability in water table depth within and between landcovers. Arable sites have relatively low variability reflecting the efficient water table control from intensive drainage. Improved grassland, deciduous woodland and fen have greater variability which is likely to reflect weaker water table control from drainage and relatively high evapotranspiration. The low variability within the heather and bog classes may reflect a combination of factors including low site numbers (4 for heather), wetter climates (bog sites are located at Wem and Thorne Mosses), lower evapotranspiration (sphagnum-dominated bogs), wetland restoration management etc. Nevertheless, the range of variability in most land uses and sites is in excess of the 20cm range of SW17/18.

Figures 28 to 30 demonstrate the variability in water table depth (and surface flooding) between years, sites and habitats. Both the grassland and deciduous woodland sites span a range of hydrological conditions but only one wet woodland site maintains a water table depth at less than 50cm below ground level through the years, even when winter water tables are with 20cm of the surface through the winter. The fen sites show generally shallower water table depths, but all but one fail to maintain a water table at less than 30cm depth through all years.

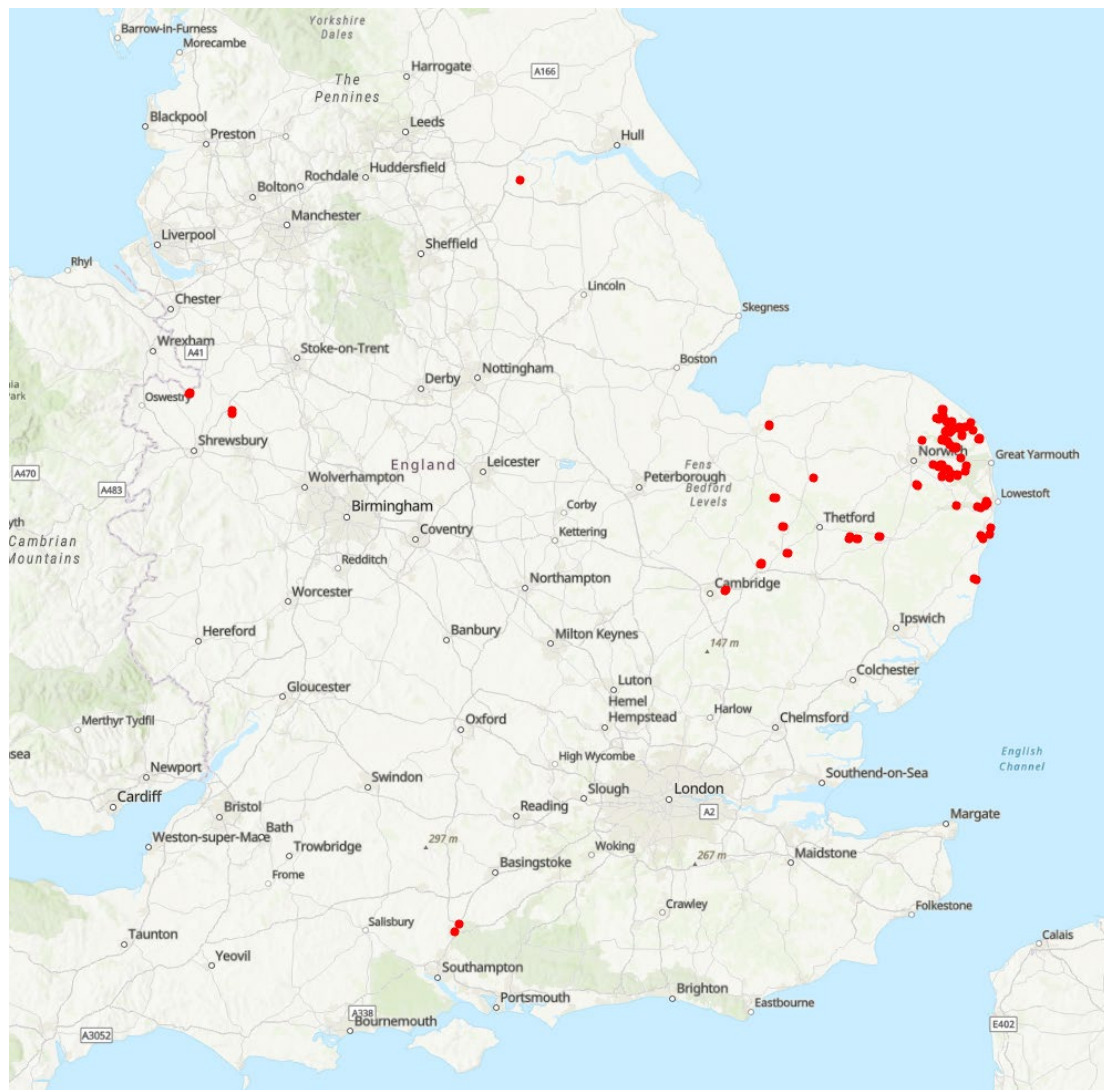


Figure 26. Location of the 265 shallow Environment Agency monitoring boreholes within potential peatland areas [NB many points are overlapping due to proximity]. Map data from OpenStreetMap (<https://www.openstreetmap.org/copyright>).

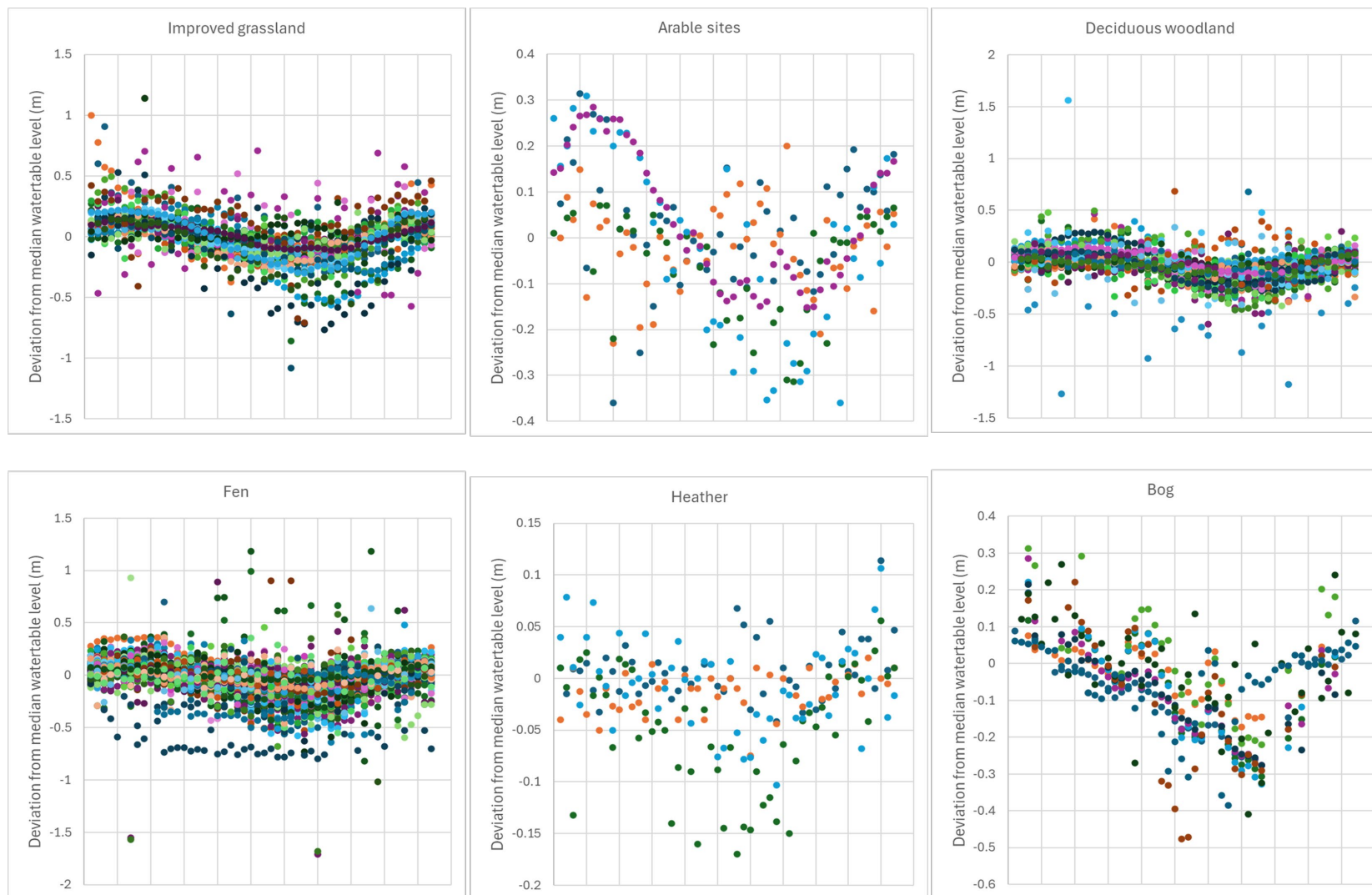


Figure 27. Weekly average deviation from median water table level within dipwells within different landcovers (note different y axes). x-axis is week number from 1 to 52.

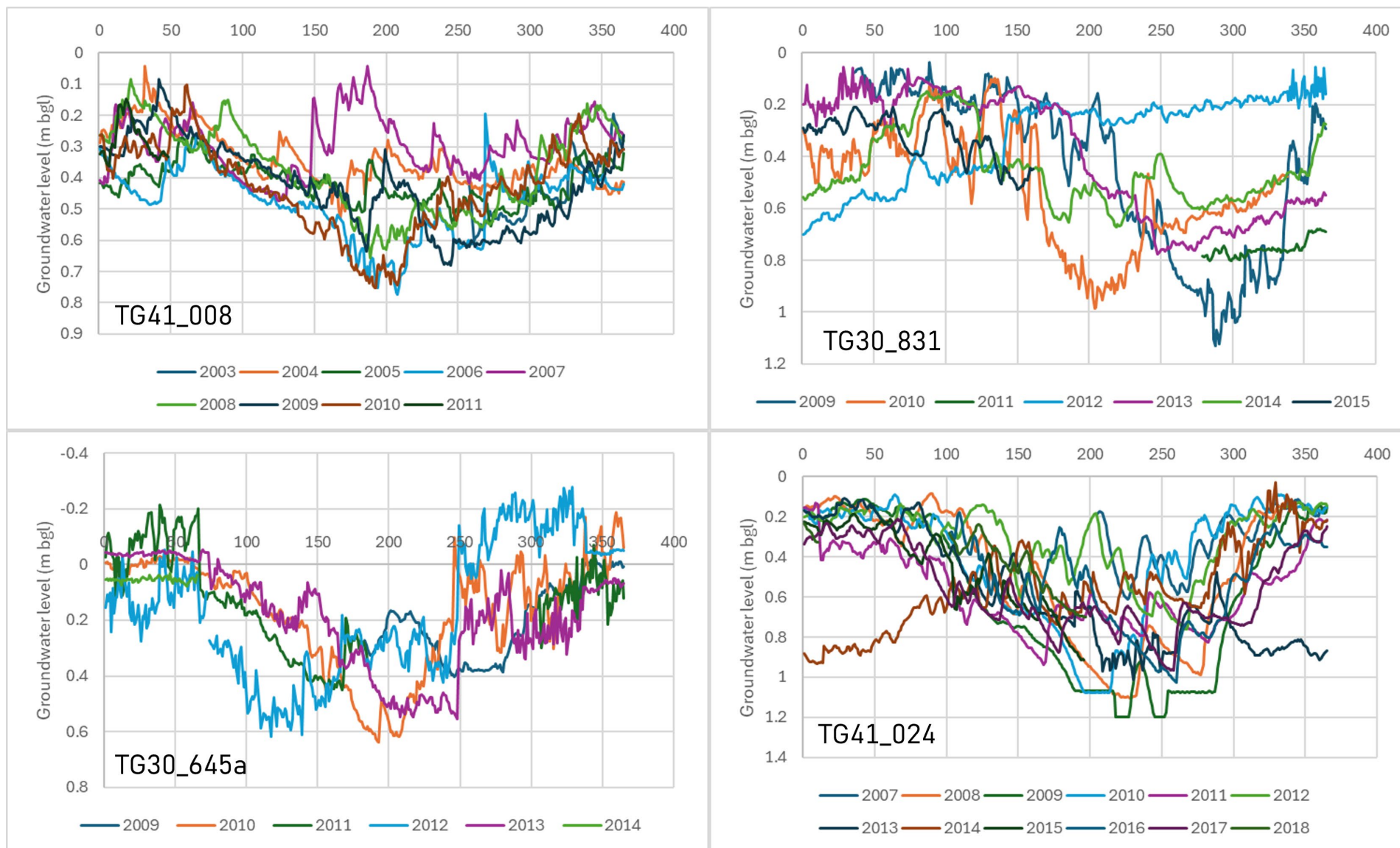


Figure 28. Daily average water table depth (metres below ground level) against day number within logged dipwells within grassland sites (negative values indicate standing water). X-axis indicates day of year.

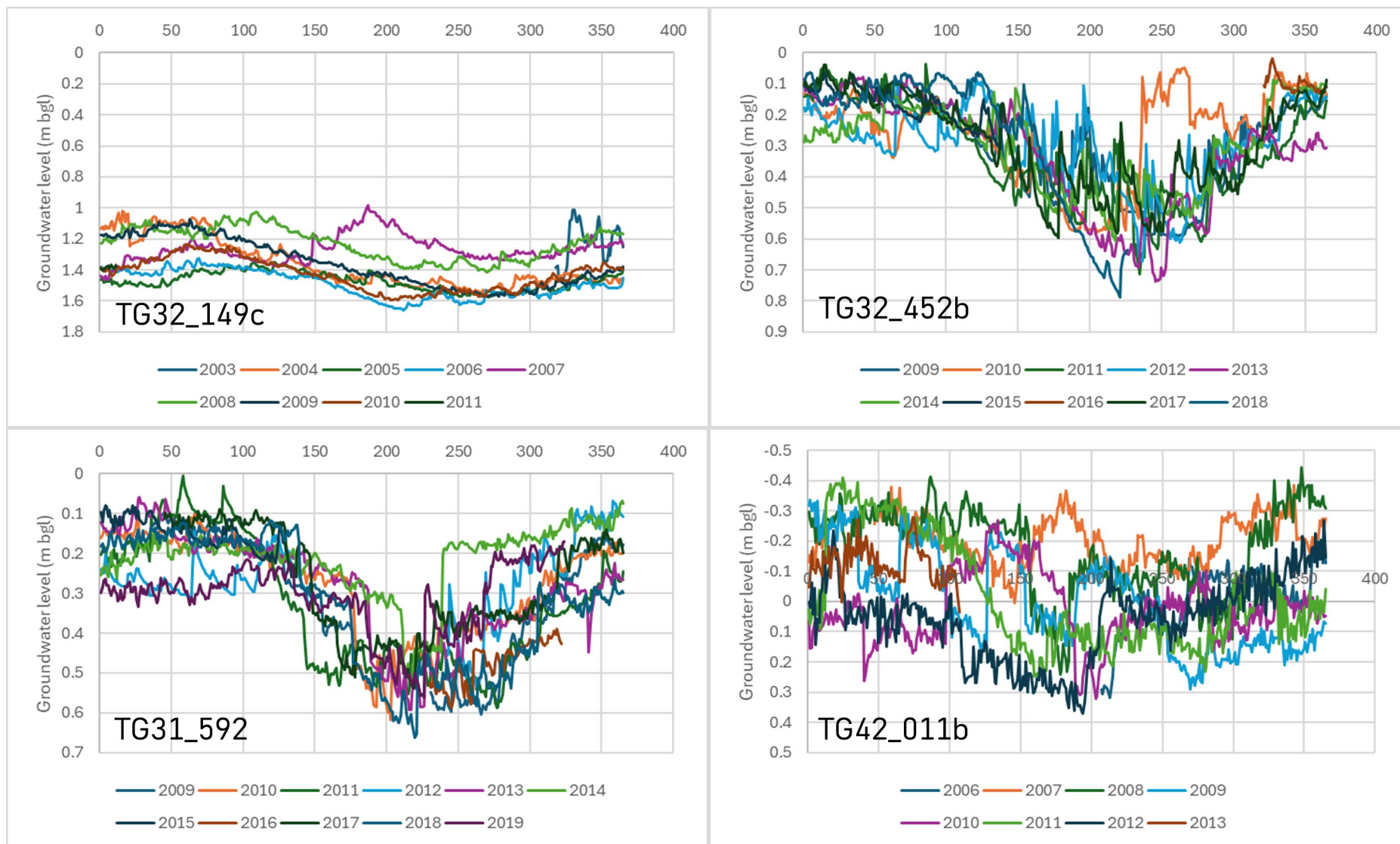


Figure 29. Daily average water table depth (metres below ground level) by day number within logged dipwells within woodland sites. X-axis indicates day of year.

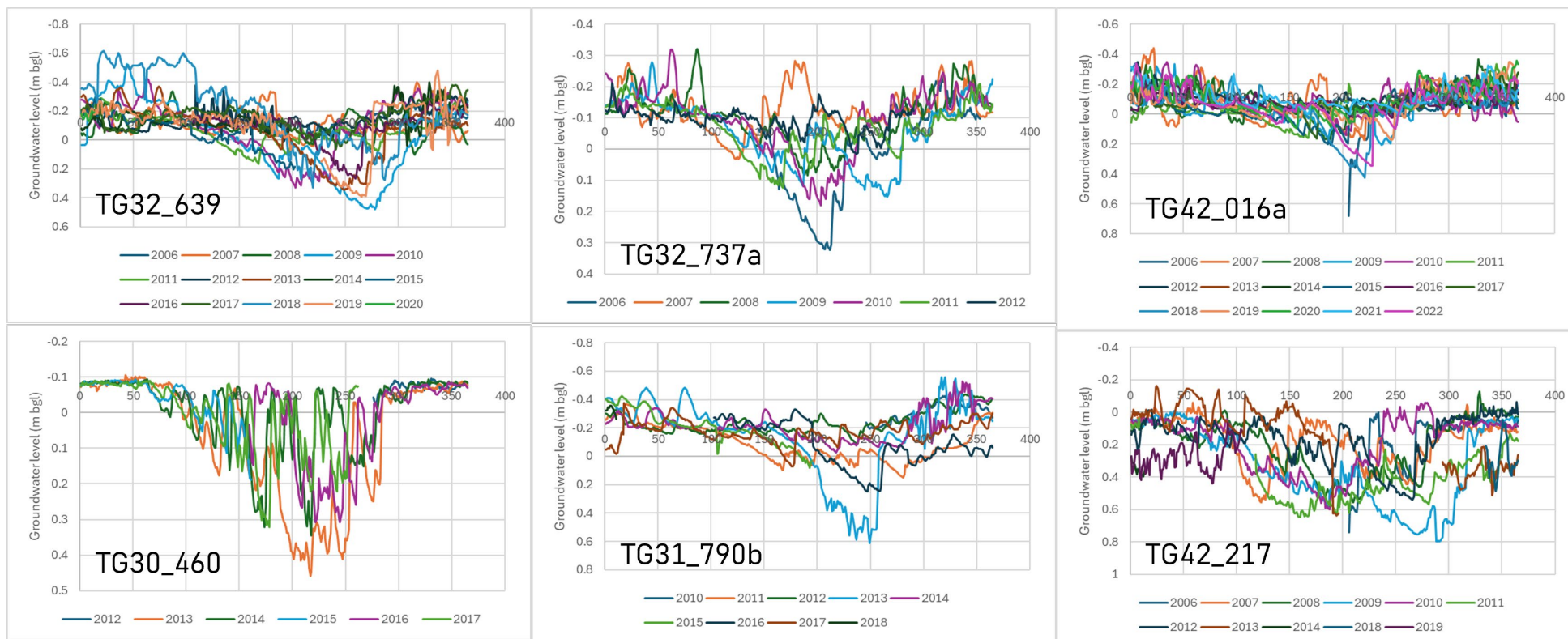


Figure 30. Daily average water table depth (metres below ground level) by day number within logged dipwells within fen sites. X-axis indicates day of year.

UKCEH flux tower data

High resolution water table monitoring data has been analysed from ten UKCEH flux tower monitoring stations, spanning arable and horticulture, grassland and fen landcovers. The data has been averaged to a daily time scale for clarity (Figure 31). Of the sites, only three have an inter-quartile range of 30cm or less (i.e. low variability in the water table depth), of which two are arable/horticulture sites (EF-SP, EF-ST of which the former is now a paludiculture trial) and one is fen (GGR-OR). This demonstrates the importance of both drain water level management and having an effective drainage system for managing and maintaining water table depth through drainage and sub-irrigation/lateral inflows.

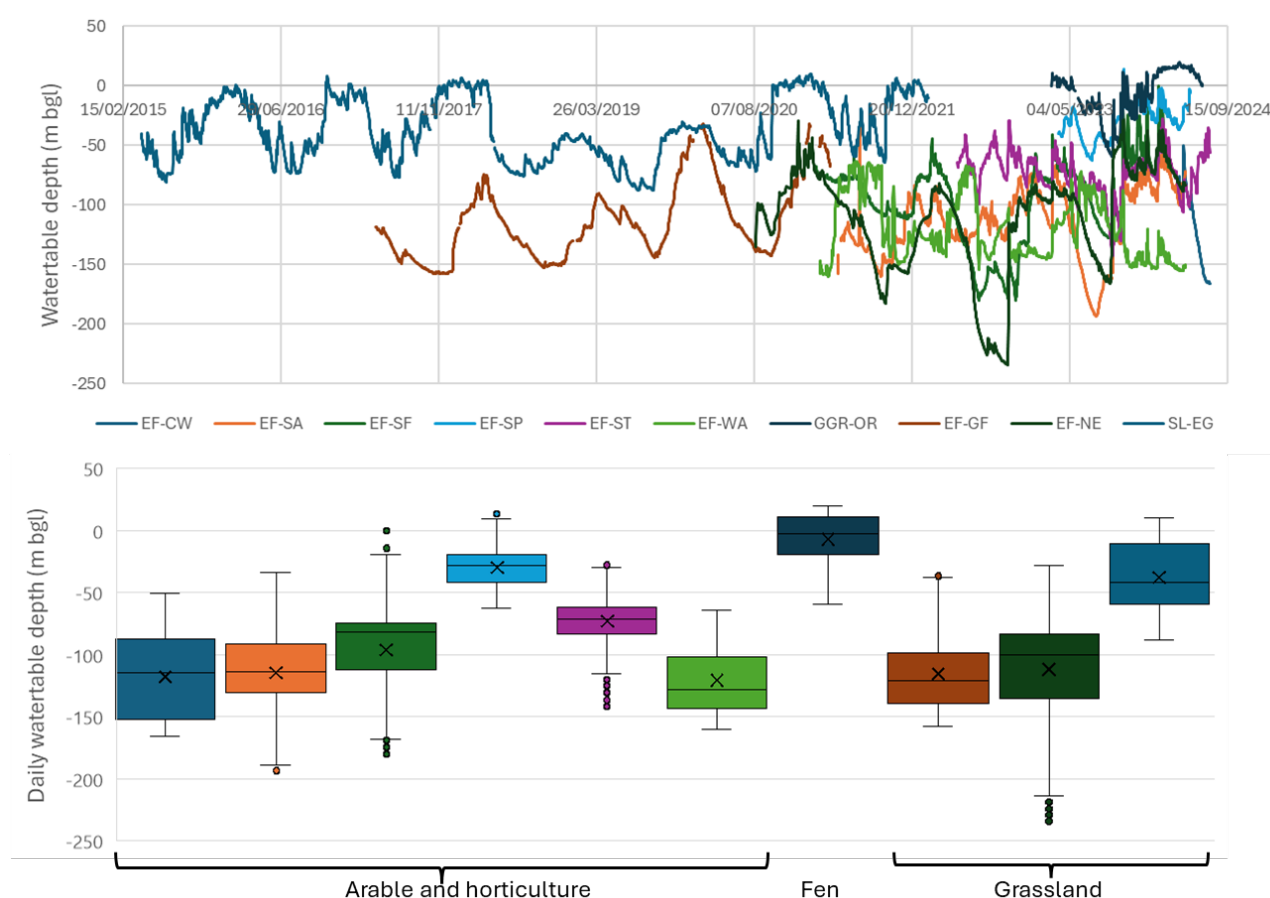


Figure 31. Time series (upper) and box and whisker plot (lower) of daily average water table depth (metres below ground level) within logged dipwells at UKCEH flux tower sites.

In summary, the analysis of Environment Agency and UKCEH water table monitoring data has shown the significant intra- and inter-annual variability in the water table depth at most sites, including those with semi-natural landcovers that would be expected to be relatively unimpacted by agricultural drainage. It therefore demonstrates the likely challenge of maintaining consistently raised high water tables with little variability, unless there is either low effective drain spacing (e.g. closely spaced under-drains) or regular additions of surface water (e.g. in river-connected fens). Unless the new CSW17/18 and CWS19/20

and other agri-environment schemes more clearly address this natural variability within water table behaviour within their water table prescriptions, it either risks the penalisation of well-meaning landowners or the need for more active and expensive water table management interventions in many fields with relatively wide ditch spacings and the absence of under-drains. In these situations, the additional costs of digging new ditches (Grobe et al., 2021; Brust et al., 2018), installing new under-drains (Harpenslager et al., 2024; Boonman et al. 2022) or installing solar-powered irrigation systems (Pouliot et al., 2015) may be a significant barrier to uptake.

6. Conclusions

The main findings of this report are:

1. The water table depth within a lowland agricultural peat is the result of the time-varying balance between water inputs and water outputs within the controls set by the drainage system (e.g. drain water level and spacing) and peat properties.
2. The hydrological properties of lowland agricultural peat are a complex consequence of human management, the original peat-forming vegetation type and peat depth. Drainage to lower the water table causes increased decomposition of peat (or humification) and bulk density, and reduced specific yield (affecting storage) and saturated hydraulic conductivity (affecting flow).
3. The observed water table within lowland peats varies within and between years, even within sites that are relatively unimpacted by agricultural drainage. However, some arable/horticultural sites are able to achieve low water table variability due to the performance of well-maintained and closely spaced under-drained systems.
4. Raising ditch water levels is the most common approach to raising peat water tables reported in the literature. However, whilst raising ditch water levels was shown to increase mean water table depth, the water table variability and maximum water table depth remained greater than in control (near-natural) sites, suggesting that water table resilience to dry conditions is not restored due to the ongoing adverse effects of drainage on specific yield and saturated hydraulic conductivity.
5. Consequently incentivising higher water table levels in agricultural drainage systems alone will mostly fail to deliver consistently higher in-field water tables due to the controlling factors of the peat properties and the drainage system. Maintaining a high water table within a narrow 20cm depth range during summer, as currently prescribed by CSW17/18 and CWS19/20, will not be achieved in many lowland agricultural peat fields even with high surface water levels in ditches, due to high summer evapotranspiration and the low hydraulic conductivity of degraded agricultural peat.
6. Such agri-environment schemes should explicitly allow for the natural variability within water table behaviour within their raised water table prescriptions. Alternatively, more active and expensive water table management interventions will be required in many fields to try to ensure compliance, such as digging new ditches, installing new under-drains or irrigation systems. Unless costed into the scheme design, these are likely to be a significant barrier to uptake.

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7. Appendices

7.1. Appendix-1: WETMEC summary

This appendix summarises the wetland functional water supply mechanisms (WETMEC) groups³ that best describe the main agricultural peats in England and the hydrological regimes for their dominant wetland plant communities. The following agricultural peat regions are included: Fens, Norfolk Broads, Somerset Levels, Humberhead Levels, Lancashire Mosses and Shropshire Mosses.

The WETMECs describe the inter-relationships between water source, water quantity, water quality and vegetation type in sites supporting herbaceous wetland vegetation in lowland England and Wales. Consequently, they describe the original or past ecohydrology of drained lowland agricultural peats rather than their current functioning. Nevertheless, and despite the adverse impacts of drainage on peat wastage and peat properties, they can provide useful insights into hydrological supply mechanisms and interactions.

Agricultural Peat WETMECs

Two main sources of information were used to identify the WETMEC group(s) that best describe the agricultural peat areas: (i) designated site information and (ii) the scientific literature describing the vegetation composition and stratigraphy of the peatlands. This two-pronged approach was required because the former vegetation communities were removed and hydrological regimes altered when the peatlands were converted to agricultural land.

Nationally designated wetland sites within and surrounding the agricultural peat areas were identified using Magic Map (Natural England, 2024a). The search focused on Special Areas of Conservation (SACs) and Sites of Special Scientific Interest (SSSIs). SACs are designated under the Habitats Directive for the protection of habitats and species that have a priority status. Only habitats that relate to wetlands were included in the search, focusing on those in the 'raised bogs, mires and fens' category, though several sites with wet heaths were also identified, which fall under the 'temperate heath and scrub' (Annex 1 habitat types; JNCC, 2024). The scientific literature was searched using the name of the agricultural peatland areas and a variety of key words (e.g. peat, stratigraphy, vegetation, and paleoecology). Additional academic literature and government reports were identified from reference lists in the articles, recent literature citing those articles, and

³ <https://www.gov.uk/government/publications/wetland-functional-mechanisms-a-synopsis-of-wetland-water-supply-mechanisms-wetmecs>

recommendations from wetland specialists involved in the project. The WETMEC group was identified from Wheeler et al. (2009) based on the type of designated habitat, its condition, and the vegetation community that is currently found at the site (Natural England, 2024b). Whilst the peatlands that existed prior to conversion to agriculture might have been spatial complex, containing a variety of WETMEC groups, only WETMEC groups that are typically found over large areas >10 ha in flat or gently sloping areas were included in this report.

The Fens

There is large uncertainty in the WETMEC groups for the agricultural peats in the Fens due to conversion of most of the peatland to agriculture and changes to topography and hydrology, e.g. drained and levee construction. However, studies conclude that the much of the Fens were floodplain fens with high groundwater levels (Mountford et al., 2002). Palaeoecological evidence suggests that open eutrophic vegetation communities existed across much of the Fens, alternating in some locations with fen carr (Waller and Kirby, 2021). There is little evidence of the progression beyond meso-oligotrophic conditions, except in basins that are remote from rivers, e.g. Holme-Walton and Wood Fen (Waller and Kirby, 2021). Areas next to rivers have would have had a greater likelihood of inundation by surface water, whilst areas above the chalk would have base-rich groundwater, creating conditions that are less conducive to raised bog formation (Waller 1992). However, there was likely a greater concentration of raised bogs in the west of the Fenland basin, which received drainage from the Jurassic clays.

There are few sites in the Fenlands designated for wetland habitat. The Fenland SAC contains refers to several sites that cover a total of 612 hectares, which are mostly covered with wetland vegetation (Natural England, 2024b). There are two Annex I habitats within the SAC:

1. Molinia fen-meadows (6410) with *Molinia caerulea* – *Cirsium dissectum* (M24)
2. Calcareous fens (7210) that range from species-rich to species-poor, dominated by *Cladium* sp. Roydon Common and Dersingham bog is described as a lowland mixed valley mire with ‘communities grading from wet acid heath through valley mire to calcareous fen’.

The Fens
5: Summer ‘Dry’ Floodplains (e.g. Wicken Fen)
6: Surface Water Percolation Floodplains
4. Drained Ombrogenous (minor)
4a bog: western edge, e.g. Holme Fen;
4b fen, Cambridge, e.g. Woodwalton

The Broads

There is also uncertainty about the WETMEC groups that best describe the agricultural peats in the Norfolk Broads. Most sources describe the Broads peatlands as ‘floodplain mire’ containing a rich fen vegetation, though how this translates to the WETMECs depends on the local topographical and geological setting. The bedrock is largely the Norwich Crag formation, a sedimentary rock of marine origin, but the London Clay is present on the River Yare and located only 0.9-1.3 m below the surface. There are extensive superficial deposits of sand and gravel (e.g. glacial till) and alluvium. The underlying Chalk, which is exposed in the headwaters, influences water levels and chemistry (alkaline). Palaeoecological research in floodplain mires on the River Yare noted herbaceous peat in the upper layer with vegetative fragments and seeds dominated by the sedge *Cladium mariscus* and the reed *Phragmites australis* (Wells, 1988).

Sites in the Norfolk broads that are designated for their habitats include The Broads SAC (UK0013577) and Norfolk Valley Fens SAC (UK0012892) (Natural England, 2024b). These SACs contain three Annex-1 wetland habitats:

1. calcareous fens with *Cladium mariscus* and species of the *Caricion davallianae* (7210)
 - a. *Cladium mariscus* swamp and sedge beds (S2)
2. alkaline fens that group in areas with a high water table and base-rich waters (7230)
 - a. *Schoenus nigricans* – *Juncus subnodulosus* mire (M13)
 - b. *Carex rostrata* – *Calliergon cuspidatum/giganteum* mire (M9)
 - c. *Phragmites australis* – *Peucedanum palustris* fen (S24)
3. transition mires and quaking bogs (rare isolated and associated with former peat-cuttings)
 - a. *Carex rostrata* – *Sphagnum squarrosum* mire (M5)
 - b. *Carex rostrata* – *Calliergon cuspidatum/giganteum* mire (M9)
 - c. *Carex rostrata* – *Potentilla palustris* tall-herb fen (S27)

The sites also contain North Atlantic wet heaths with *Erica tetralix* (4010), which grows on nutrient-poor, acidic substrates in headwaters (Norfolk Valley Fens), and *Molinia* sp. meadows (6410), which grown on moderately base-rich soils (6410) (Natural England, 2024b).

Much of the floodplain mires would have been WETMEC 5 (summer-dry floodplains) prior to peat cutting and conversion to agriculture (Wheeler et al., 2009). In areas with peat extraction, WETMEC 6 (surface water percolation floodplain) is now found. The high

permeability of the bedrock suggests that WETMECS 7, 8 and 9 may have also been present. Wheeler et al. (2009) identified sites in East Anglia associated with WETMEC 8 (groundwater-fed bottoms with Aquitard) and WETMEC 9 (groundwater-fed bottoms). WETMEC 7 may have also been present, especially in the more chalk influenced river floodplains (e.g. River Bure); *Cladium* is often associated with groundwater-fed floodplains (Hibbert 1978, p. 91).

Norfolk Broads
5: Summer ‘Dry’ Floodplains
6: Surface Water Percolation Floodplains (in areas of peat extraction)
9: Groundwater-Fed Bottoms
8: Groundwater-Fed Bottoms with Aquitard
7: Groundwater floodplains

Somerset Levels

The agricultural peats in the Somerset Levels were most likely ombrogenic bogs. Paleaocological studies conclude that the uppermost layers are composed of *Sphagnum* moss or *Sphagnum* moss and the heather *Calluna* (Clapham and Godwin, 1948, p. 238-244). Vegetation communities dominated by *Ericaceae* and *Sphagna* prefer acidic conditions, which are found in rain-fed raised bogs.

The Somerset Levels do not contain any SACs, though the Somerset Levels and Moors Special Protection Area (SPA) and numerous SSSI sites cover large portions of the area (Natural England, 2024b). The SSSIs are generally protected for their grasslands, though there are some mire-type communities containing sedge and rush species, e.g. *Carex panicea*, *C. disticha*, *C. hirta*, and *C. lepidocarpa* and *Juncus* spp. (e.g. *Juncus effusus*).

Somerset Levels
4. Drained Ombrogenous

Humberhead Levels

The agricultural peats in Humberhead Levels are believed to be a former raised peat bog formed through the infilling of a large glacial lake. There are several sites designated for wetland habitats in the Levels, including Thorne Moor SAC (UK0012915) and Hatfield Moor SSSI (UK0030166), which are described as raised bogs (Natural England, 2024b). Thorne Moor is the largest area of raised bog in England. It is primarily classified as degraded raised bog still capable of natural regeneration (7120), though improved management has increased species richness to include *Sphagnum* spp, *Eriophorum*

angustifolium and *E. vaginatum*, heather *Calluna vulgaris*, and cross-leaved heath *Erica tetralix* (JNCC, 2024). Similarly, Hatfield Moor SSSI is considered a remnant of the once extensive lowland raised bog that covered the area.

Humberhead Levels

4. Drained Ombrogenous

Lancashire Mosses

The agricultural peatland in the coastal plains stretching south from Lancaster to Liverpool and east to Manchester are largely described as former lowland raised bogs (IUCN-UK, 2024). All the designated sites in the region are described as raised bogs, including two SACs and a SSSI in the south (Manchester Mosses SAC, Roudsea Wood and Mosses SACs, and Bryn Marsh and Ince Moss SSSI) and a SSSI to the north (Windmarleigh Moss SSSI) (Natural England, 2024b). Manchester Mosses SACs is classified as degraded raised bog (7120), where past drainage has shifted the vegetation community to *Molinia caerulea*, *Pteridium aquilinum*, and birch. Roundsea has areas of active raised bog (7110) and degraded raised bog (7120). There may have been more complexity in the wetland types in the Lancashire Mosses. The SSSI citation for Highfield Moss describes site as having the 'best remaining example of the raised mires which once covered large areas of Greater Manchester and Merseyside, but also describe the major habitat as 'mixed valley mire communities on the peat' (Natural England, 2024b).

Lancashire / South Cumbrian Mosses

4. Drained Ombrogenous

Shropshire Mosses

The agricultural peatlands of the Shropshire are likely former raised bogs for the most part. Weald Moors to the north of Telford are described as forming from the infilling of a large post-glacial lake, creating raised bogs that were converted for agricultural use. Similarly, the Meres and Mosses landscape in North Shropshire contains several sites designated for their wetlands that contain active raised bogs: Marches Mosses (Fenn's, Whixall, Bettisfield, Wem and Cadney Mosses SAC and SSSI), the West Midland Mosses SAC (Abbots Moss SSSI, Chartley Moss SSSI, Clarepool Moss SSSI, Wybunbury Moss SSSI), and Fenmere SSSI (Natural England, 2024b). Though, the large number of open waterbodies (i.e. meres) in the area suggest that there may have been a considerable variety of wetland types. For example, Brownheath Moss SSSI and Ramsar has fen and carr habitats, which may represent the transition from open water to peat bog that existed in the Mosses.

Shropshire Mosses

4. Drained Ombrogenous

List of abbreviations

CS	Countryside Stewardship
dbd	Peat dry bulk density
EA	Environment Agency
Kc	Crop coefficient [relating to a reference evapotranspiration (ET_0)]
Kh	Horizontal hydraulic conductivity
Ksat	Saturated hydraulic conductivity
Kv	Vertical hydraulic conductivity
LandIS	Land Information System (Cranfield University)
LAP	Lowland Agricultural Peat
LCM	Land Cover Map
maOD	metres Above Ordnance Datum
SAC	Special Areas of Conservation
SSSI	Sites of Special Scientific Interest
SY	Specific yield
TDI	Tension-disc infiltrometer
UKCEH	UK Centre for Ecology & Hydrology
WDRT	Water drop repellency test
WETMECS	Wetland functional water supply mechanisms
WTF	Water table fluctuation

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