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RESEARCH PAPER

A comparison of peat properties in intact, afforested and restored raised and blanket bogs

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Abstract
Recognition of peatlands as a key natural store of terrestrial carbon has led to new initiatives to protect and restore them. Some afforested bogs are being clear-felled and restored (forest-to-bog restoration) to recover pre-afforestation ecosystem function. However, little is known about differences in the peat properties between intact, afforested and restored bogs. A stratified random sampling procedure was used to take 122 peat cores from three separate microforms associated with intact (hollows; hummocks; lawns), afforested and restored bogs (furrows; original surface; ridges) at two raised and two blanket bog locations in Scotland. Common physical and chemical peat properties at eight depths were measured in the laboratory. Differences in bulk density, moisture and carbon content between the afforested (mean = 0.103 g cm^{-3}, 87.8% and 50.9%, respectively), intact (mean = 0.091 g cm^{-3}, 90.3% and 51.3%, respectively) and restored bogs (mean = 0.095 g cm^{-3}, 89.7% and 51.1%, respectively) were small despite their statistical significance. The pH was significantly lower in the afforested (mean = 4.26) and restored bogs (mean = 4.29) than the intact bogs (mean = 4.39), whereas electrical conductivity was significantly higher (mean: afforested = 34.2, restored = 38.0, intact = 25.3 μS cm^{-1}). While significant differences were found between treatments, effect sizes were mainly small, and greater differences in pH, electrical conductivity, specific yield and hydraulic conductivity existed between the different intact bogs. Therefore, interactions between geographic location and land management need to be considered when interpreting the impacts of land-use change on peatland properties and functioning.

KEYWORDS
carbon sequestration, decomposition, drainage, forestry, peat, restoration

1 | INTRODUCTION

A third of global soil carbon is thought to be stored in peatlands (Scharlemann et al., 2014; Yu et al., 2010), but their ability to act as carbon sinks depends on their condition. Worldwide, 12% of peatlands are degraded and potentially act as a carbon source for the atmosphere (Joosten, 2016). Continued global industrial development has led to the loss of former peat accumulating landscapes through land management and climate change. In particular, afforestation has led to significant peatland habitat loss worldwide with the planting of non-native species on treeless or naturally forested peatlands for timber (Anderson et al., 2016; Strack, 2008) or palm oil production (Joosten et al., 2016; Sangok et al., 2017). Palviainen et al. (2004) estimated, at the time, that 15 million hectares of natural peatlands had been drained for forestry worldwide, and so, values are now likely to be even greater. Up to 600,000 ha (Cannell et al., 1993; Paavilainen & Päivänen, 1995) of UK peatlands may have been drained for forestry, with 9% of the total area of deep peat planted with coniferous trees between the 1950s and 1980s (Cannell et al., 1993; Hargreaves et al., 2003).

There has been considerable investment in restoration initiatives following interest in the carbon storage potential of peatlands and wider ecosystem service benefits (Anderson, 2001; Anderson et al., 2016). Forest-to-bog restoration is one example, where trees are felled and drains blocked to raise the water table in an attempt to restore peatland functions (Anderson, 2001; Anderson & Peace, 2017; Gaffney, 2017; Gaffney et al., 2018; Muller et al., 2015; Muller & Tankéré-Muller, 2012; Shah & Nisbet, 2019). Forestry on peatlands lowers the water table due to drainage and increased evapotranspiration from growing tree stands (Anderson, 2001). Felling has been found to reverse this process to a degree, but the blocking of drains and furrows is usually required to restore water-table levels, so they are similar to those of intact bogs (Anderson & Peace, 2017; Howson et al., 2021a; Koskinen et al., 2011; Koskinen et al., 2017; Menberu et al., 2016). However, little is known about the effects on peat properties following prolonged water-table draw-down associated with plantation forestry or restoration after felling. Post-drainage consolidation of the peat, shrinkage of dried peat near the surface and wastage through oxidation after drainage, compaction from the weight of the trees (Anderson & Peace, 2017; Liu et al., 2020) and the presence of tree roots may mean that the properties of peat several years after restoration are still significantly different to those found in intact systems.

The effects of afforestation on peat properties are difficult to predict since several interacting factors exist. Compression from the weight of the trees may be expected to increase bulk density and decrease hydraulic conductivity as soil pores collapse (Silins & Rothwell, 1998). Alternatively, the top layers of drying peat may experience desiccation cracks, increasing hydraulic conductivity (Holden et al., 2004). Such changes may also have implications for the peat's water storage capacity (Price & Schlotzhauer, 1999) and carbon content (Simola et al., 2012). Higher bulk densities, reduced moisture content and subsidence have been attributed to peat compression and oxidation, but the magnitude of change in each property is unclear (Anderson & Peace, 2017; Price & Schlotzhauer, 1999; Sloan et al., 2019). Oxidation of surface peat also leads to greater humification and loss of carbon to the atmosphere. Mustamo et al. (2016) found that hydraulic conductivity, bulk density and humification were strongly co-dependent, but there was considerable spatial variability, which they suggested could be due to the dominance of macropores. Studies on the carbon content of afforested peatlands are scarce. A study on tropical peatlands found reduced soil carbon concentrations in mature palm oil plantations compared with near-natural peat swamp forests (Tonks et al., 2017). Simola et al. (2012) estimated forestry drained peatlands in Finland to be losing 1.5 t C ha^{-1} $year^{-1}$ after analysing the carbon content of peat cores taken from 37 different locations previously sampled in the 1980s. However, carbon stored in tree roots and litter layers was ignored. Hargreaves et al. (2003) recorded similar rates of C loss, 1 t C ha^{-1} $year^{-1}$, for a closed-canopy forest on drained peat in Scotland using eddy covariance flux measurements. In contrast, Hommeltenberg et al. (2014) estimated higher rates of C loss, 3 t C ha^{-1} $year^{-1}$, from a peatland spruce plantation in Germany, but their estimate relied on the assumption that 50% of peat volume loss due to subsidence was oxidative wastage. In each of these studies, carbon loss estimates did not consider losses through leaching, which Dinsmore et al. (2010) found comprised 25% of net ecosystem exchange C uptake.

Anderson and Peace (2017) found an apparent recovery in blanket peat bulk density and moisture content 10 years after clear-felling and blocking furrows. As a result, a limited reversal in subsidence was observed, suggesting that some peat mass swelling can occur in response to restoration. In turn, this swelling may increase the peat permeability as pore spaces open up. However, studies to assess this hypothesized mechanism have never been undertaken. We could not identify other studies that have looked at the impact of forest-to-bog restoration effects on peat properties. Studies from peatlands restored after severe disturbances such as peat harvesting (Price, 1996; Price & Schlotzhauer, 1999) may offer insights into potential forest-to-bog impacts on peat properties, but further work is required to establish whether such findings apply to restoration after forestry operations. Studies on previously harvested peatlands in Quebec (González et al., 2014; McCarter & Price, 2015; Price, 1996; Price & Schlotzhauer, 1999) found spontaneous recolonized vegetation in abandoned sites and a shift towards wetland favouring species 3–17 years after restoration. However, where restoration had allowed for the regeneration of *Sphagnum* mosses, differences in physical properties between the surface layers and the underlying peat often meant near-natural peatland function did not return in the short term (McCarter & Price, 2015). McCarter and Price (2015) concluded that additional structural growth, decomposition and consolidation of the regenerated *Sphagnum* would be necessary before the previously harvested bog would return to a favourable status.

Afforested peatlands typically have little understory vegetation due to the lack of light under conifer forest canopies and reduced moisture conditions. Hancock et al. (2018) found a trajectory towards open bog plants in the first 6 years after restoration, followed by a slower rate of change thereafter. Anderson and Peace (2017) found a similar pattern with quick colonization of *Eriophorum* in the first 3–4 years after restoration, but different species compositions to intact bogs after 10 years, which may be related to differences in peat properties between furrows, ridges and the original surface. Hancock et al. (2018) suggested a greater shift away from natural bog vegetation on plough ridges and steeper slopes, whereas Anderson and Peace (2017) found a greater divergence away from natural bog vegetation in furrows.

Given the lack of forest-to-bog studies on peat properties, our main aim was to quantify common physical and chemical characteristics to determine differences between intact, afforested and restored sites on blanket and raised bogs. We hypothesized that the peat properties in the afforested bogs would significantly differ from those found in intact systems, but the properties in forest-to-bog restoration sites would lie somewhere between those found in afforested and intact bogs.

2 | METHODS

2.1 | Study sites

The sites were chosen from two raised bog and two blanket bog locations in Scotland, each containing areas of undisturbed peatland (hereafter referred to as intact bog - IB), forestry (hereafter referred to as afforested bog - AB) and forest-to-bog restoration (hereafter referred to as restored bog – R). Where two restoration sites were sampled at the same location, the older site was referred to as R1 and the younger site as R2. The two blanket bog locations, Forsinain and Talaheel, were situated in the RSPB Forsinard Flows National Nature Reserve, northern Scotland (Figure S1). The reserve is part of the Flow Country, Europe's largest expanse of blanket bog. The raised bogs were located in central and southern Scotland (Figure S1), one at Flanders Moss, which is part of a series of lowland raised bogs formed on the uplifted former estuary of the River Forth, and the second at Ironhirst Moss, part of the Lochar Mosses, Dumfries & Galloway. Examples of intact bog, first rotation afforested bog and restored bog were selected for taking peat cores at each location where peat depths were greater than 1 m. Table 1 provides further characteristics of the chosen sites at each location.

The different locations were representative of blanket bog and raised bogs typically found in the UK and were all classified as deep peat. The afforested bogs had been planted according to normal practice at the time, with the same mixture of Sitka spruce (*Picea sitchensis*) and lodgepole pine (*Pinus contorta*) and their forest canopies were largely closed at the time of the study. The intact bogs were dominated by *Sphagnum*, sedges and ericaceous shrubs, with occasional sundews (*Drosera* spp.), bog asphodel (*Narthecium ossifragum*), bog myrtle (*Myrica gale*) and bogbean (*Menyanthes trifoliata*) in natural pools. Although the restored bogs were felled at different times, and restoration methods differed, all standing trees had been removed. At Ironhirst Moss and Flanders Moss, the first rotation forest was felled and not restocked, and no additional restoration was carried out, although drain blocking and other re-wetting treatments are scheduled for the latter. At the four locations, brash was either removed and chipped for biomass, left on site to decompose, or compressed into furrows and drains. The trees were mulched from standing at the youngest restoration site at Forsinain using a mechanical masticator, and the woodchips spread on the peat surface. At Forsinain and Talaheel, drain and furrow blocking had also taken place.

2.2 | Sample collection

A random stratified sampling procedure (Figure 1) was adopted where 60 × 60 m grids were selected at each site, and three individual 10 × 10 m cells were selected at random for obtaining peat cores. In each cell, cores were taken from three different microforms: hollows, hummocks and lawns in the intact bogs, and ridges, furrows and the area between furrows and ridges (hereafter referred to as the original surface) in afforested and restoration sites. Therefore, nine peat cores were taken from each site using a 50 cm long, 5 cm inner diameter Russian corer after removing any litter layer, and each core's location was recorded using GPS. Two 50 cm peat cores were taken from the same hole to sample the 1 m profile at a given location, ensuring the Russian corer cover plate was carefully aligned in the same orientation for each sample. Each peat core was placed in a 1 m section of PVC guttering for protection and wrapped with cellophane to form an airtight seal. On return to the laboratory, the cores were refrigerated at 4°C before analysis, usually within a month of collection. Overall, 122 peat cores were taken from the 126 selected cells; it was impossible to take three

TABLE 1 Characteristics of chosen sites

Location	Site	Description	Bog classification	Region	Lat/Lon	Mean peat depth (m)	Mean annual WTD (cm)	Felling date	Planting date	Coring date
Forsinain	FOAB	Afforested bog	Blanket bog	Flow Country	58°25′30.6″N 3°52′08.0″W	2.3	28.7		1980	16/12/2018
(FO)	FOIB	Intact bog			58°25′15.9″N 3°51′42.6″W		9.1			06/08/2019
	FOR1	Old restored			58°26′01.3″N 3°51′21.3″W		4.6	2002/03		06/08/2019
	FOR2	Young restored			58°25′30.0″N 3°51′46.4″W		10.0	2014/15		16/12/2018
Talaheel	TAAB	Afforested bog	Blanket bog	Flow Country	58°24′02.8″N 3°48′33.2″W	2.3	–		1981	26/09/2019
(TA)	TAIB	Intact bog			58°24′39.9″N 3°48′09.6″W		–			25/09/2019
	TAR1	Restored			58°24′46.6″N 3°48′22.2″W		–	1997/98		25/09/2019
Flanders Moss	FMAB	Afforested bog	Raised bog	Stirlingshire	56°09′07.1″N 4°19′53.5″W	4.5	21.3		1965	25/06/2019
(FM)	FMIB	Intact bog			56°09′47.4″N 4°10′54.0″W		9.1			25/06/2019
	FMR1	Old restored			56°08′07.9″N 4°19′26.1″W		13.8	24/11/2009–09/12/2009 01/08/2011–18/10/2011		13/12/2019
	FMR2	Young restored			56°08′26.5″N 4°19′25.2″W		19.6	01/10/2013–31/03/2014		13/12/2019
Ironhirst Moss	IRAB	Afforested bog	Raised bog	Dumfries and Galloway	55°01′38.3″N 3°30′00.7″W	5.0	–		2005	14/11/2019
(IR)	IRIB	Intact bog[a]			55°01′36.8″N 3°30′01.1″W		–			14/11/2019
	IRR1	Restored			55°01′20.5″N 3°29′35.9″W		–	1999/00	1971	14/11/2019

[a]Due to site restrictions, intact cores were taken from a wide, unplanted gap in the forest.

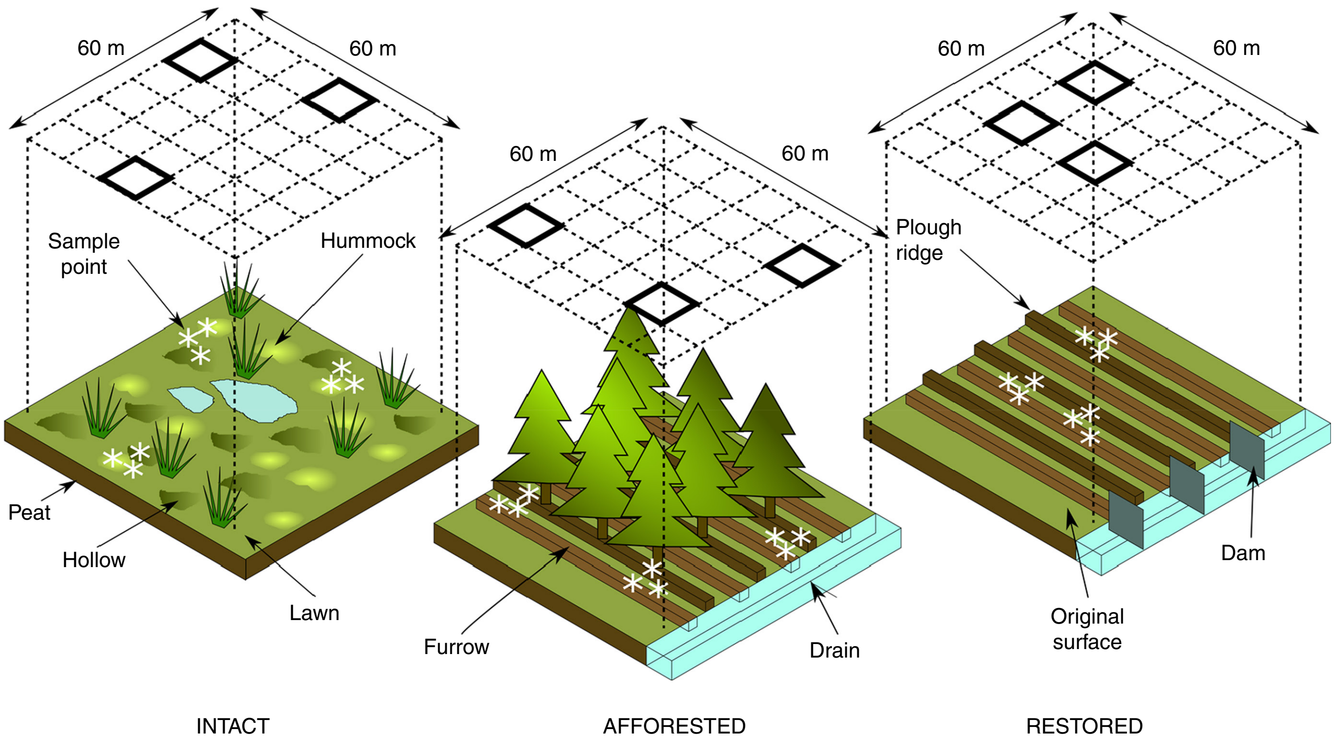

FIGURE 1 Stratified random sampling procedure for peat core selection at each site—three 10 × 10 m grid squares were selected at random, and three 1 m cores were taken from the different microforms in each grid square (Intact: hummocks, hollows, lawns; Afforested and restored: furrows, ridges and the original surface)

cores from the afforested Talaheel site and one core from the afforested Ironhirst Moss site after the Russian corer sustained damage from particularly compressed peat. Cores were not taken in drought periods, and water-table positions at all locations were similar between sampling dates.

2.3 | Laboratory analysis of peat

Each peat core was split into eight depths (0–10, 10–20, 20–30, 30–40, 40–50, 50–60, 60–80, 80–100 cm) for subsequent analysis using a sharp kitchen knife. A 32 mm diameter cylindrical cutter was used to take a subsample of peat from each section's deepest end to analyse specific yield (S_y). The remaining peat was then placed in airtight bags, labelled and returned to refrigerated storage between analyses. The subsamples were weighed along with a thin cotton muslin cloth square (supplied by Wilko) and used to wrap the sample to prevent peat loss when measuring S_y. The muslin cloth square was wrapped around the subsample of peat and held in place by elastic bands to form a parcel of peat. Samples were placed in a tank of rainwater, collected from a garden water butt and allowed to soak for 24 h. They were then weighed and placed on a sieve, covered with a lid and allowed to drain for 24 h before reweighing. An estimate of S_y was calculated using Price's (1996) method, which is determined as the difference in saturated and drained peat weights divided by the saturated weight. A regression relationship between the dry and wet weights of the muslin cloth after soaking for 24 h, followed by draining for 24 h, found 1 g of dry muslin cloth to retain 1.4 g of water ($R^2 = .93$), which was used as a correction factor.

After calculating S_y, the samples were transferred to crucibles, making sure not to lose any peat, and oven-dried at 105°C until they were at a constant weight. The oven-dried weight was recorded for each sample before being placed in a muffle furnace for 16 h at 550°C. After cooling in a desiccator, the remaining ash was weighed to determine the loss of organic matter. Loss on ignition (LOI) was calculated from Equation 1:

$$\%\,\mathrm{LOI} = \left(\frac{\text{oven dried mass (g)} - \text{ashed mass(g)}}{\text{oven dried mass (g)}}\right) \times 100 \quad (1)$$

where %LOI is percentage loss on ignition, equivalent to the percentage of organic matter in a sample. The method has a detection limit of 0.05%, and the balances used for weighing had an accuracy of 0.04 g and were calibrated between weighing batches of samples. The bulk density was calculated by dividing the sample's oven-dried mass by the initial volume. Moisture content was determined from the difference in weight between the fresh and oven-dried samples divided by the fresh sample weight, measured before the samples were analysed for S_y.

Peat acidity and electrical conductivity were determined by measuring pH and conductivity in a suspension of fresh peat in deionized water at a 1:10 ratio of wet peat mass to the solution (Rowell, 1994) using a HANNA 9124 pH meter with automatic temperature compensation and a HORIBA B-173 conductivity meter, respectively. Before batches of readings, a two-point calibration was used to calibrate the pH probe using buffer solutions of pH 4 and 7. The conductivity meter was calibrated by first soaking the sample well with deionized water and then using a 1.41 mS cm^{-1} calibration solution. The sample well on the probe was rinsed with deionized water and dried with tissue paper to prevent dilution between readings. The peat-water mixture was stirred and then placed on a shaking table for 1 h before taking pH and electrical conductivity readings. The humification of each sample was assessed by squeezing the peat and using the amount of amorphous material that passes through the fingers, plant remains and the colour of the expelled water to estimate the degree of humification. Humification was quantified using the 10-point von Post scale, where 1 equals 10% or undecomposed plant material and 10 equals 100% decomposed (von Post, 1922).

The percentage carbon content (hereafter referred to as carbon content) was determined from the regression equation derived between LOI and %C by Bol et al. (1999) and used by Garnett et al. (2001) and Parry and Charman (2013) for moorland soils, which included deep peat, and is given by Equation 2.

$$\%C = (\%\mathrm{OM} \times 0.526) - 0.167 \tag{2}$$

Carbon density was calculated by multiplying the carbon content, as a fraction, by the dry bulk density for each sample. The result was then multiplied by the depth increment to give the carbon weight per unit area. However, it is impossible to compare the carbon stock between the sites as the peat in the afforested sites has experienced peat volume changes due to shrinkage, compression (Price & Schlotzhauer, 1999) and oxidative wastage (Anderson & Peace, 2017;Shotbolt et al., 1998; Sloan et al., 2019), resulting in a lowering of the peat's surface, e.g. by 40–80 cm 30–50 years after forest ploughing and planting (Shotbolt et al., 1998; Sloan et al., 2019). The assumption that the upper 1 m of peat at the AB, IB and R locations at a site were originally similar, validating their comparison, does not hold true where differential subsidence has occurred, bringing peat originally below 1 m into the sampled layer. The degree of oxidative wastage over the decades since forestry operations began is not known. Therefore, this study only presents carbon content (% C) and carbon density (g C cm^{-3}) and does not calculate carbon accumulation or loss rates since any inference of these based on our data would be misleading (Young et al., 2019).

2.4 | Field tests

Saturated hydraulic conductivity (K_s) was measured in the field at Flanders Moss and Forsinain using a combination of piezometer slug tests (Baird et al., 2004; Baird et al., 2008; Surridge et al., 2005) where water tables were shallow and mini-disc tension infiltrometer tests where water tables were drawn down (Zhang et al., 2016). Infiltrometer tests were performed at three locations in each site ($n = 6$) at 20 and 40 cm depths. All the piezometers had an inner diameter of 2.9 cm and 10 cm intakes precisely machined, so they were comparable. The slug tests were conducted at 20 ± 5 cm and 40 ± 5 cm depths. A hole was augured in the peat to the piezometer diameter, and the piezometers were inserted into the hole with an internal cylindrical blocker held in place over the intake screen to prevent peat entry on insertion. Once the piezometers were at the correct depth, the blockers were removed and they were developed to remove any peat that may be obscuring the intakes. The piezometers were developed by removing all the water with a dosing syringe and then leaving them to refill. Once refilled, enough water was sampled to test for particulates. The process was repeated until the water was not cloudy, indicating any peat obscuring the intakes had been removed. Level-Troll 500 pressure transducers were inserted into the piezometers, and the water level was allowed to stabilize. Slug tests at 20 and 40 cm depths were carried out by adding 30 ml of water with the dosing syringe, and the piezometers were left until water levels had returned to the resting level. The pressure transducers recorded the water level every 5 s throughout the process. Additional slug tests were carried out at 60 and 80 cm depths at some sites where 30 mL of water was removed from the piezometers. K_s for the piezometer slug tests was calculated using the Hvorslev (1951) equation given by Equation 3:

$$K_s = -\frac{A}{Ft}\log_e\left(\frac{h}{h_0}\right) \tag{3}$$

where A is the inside cross-sectional area of the piezometer, F is the shape factor of the piezometer calculated by equation 9 in Brand and Premchitt (1980), t is time, h is the difference in head between the piezometer and the soil around the intake and h_0 is the initial head difference.

In the afforested sites, where the water table was drawn down and piezometer tests could not function, METER group mini-disc infiltrometers (Holden, 2009; Holden et al., 2001; Zhang et al., 2016) were used to estimate K_s. Peat was exposed or carefully excavated using a trowel for 0 cm, 20 cm and 40 cm depths and a fine layer of moist sand spread between the disc and the peat. A suction head of −0.5 cm was used since it was the closest attainable to zero. Three measurements were taken from 0, 20 and 40 cm depths from afforested sites and additional surface measurements from one restoration site. The van Genuchten alpha value for the peat and the suction head were used to calculate other van Genuchten soil parameters from the van Genuchten equation (van Genuchten, 1980). The alpha value was estimated from the relationship between bulk density and organic matter content for *Sphagnum* peat (Liu & Lennartz, 2019). K_s was calculated by dividing C_1 (the slope of cumulative infiltration (cm) against the square root of time (s)) by A (Zhang, 1997). A related to the van Genuchten values for the peat at the given suction and radius of the infiltrometer disc as calculated by the van Genuchten equation for the given alpha value (van Genuchten, 1980).

2.5 | Data analysis

The distributions of peat property variables were tested in Minitab (Minitab 19 Statistical Software, 2020). S_y was closest to a log-normal distribution, K_s to an exponential distribution, and pH was closest to a normal distribution. However, pH was the only variable to fit a specific distribution confidently. Therefore, any significant differences between treatments for pH were determined from one-way ANOVA tests followed by post hoc analysis, not assuming equal variances. Nonparametric tests for differences between treatments, which did not assume a specific distribution, were performed on the original values using Kruskal–Wallis and pairwise comparisons in SPSS (IBM Corp, 2016) for all other variables. Nonparametric tests were also used to test differences between the locations for all variables. The effect size was calculated using rstatix (Kassambara, 2020) in R Studio (2016). Eta squared (η^2) was calculated for one-way ANOVA tests, whereas the effect size for nonparametric tests (r) was calculated from the Z statistic divided by the square root of the sample size (N). Mann–Whitney U tests were used to test differences between the two bog classifications (raised bog; blanket bog). Spearman's rank correlation coefficients (r_s) were calculated for testing relationships between variables. Variables were plotted for the different treatments and sites over the depth profile using ggplot 2 (Wickham, 2016) in R Studio. Descriptive statistics were calculated in SPSS and Minitab. Differences between the three main treatments (intact, afforested, restored) were first tested, followed by differences between bog type, location and microtopographic levels. Statistical analyses were also performed on the different methodologies used for K_s measurements.

3 | RESULTS

3.1 | Differences in peat properties between treatments

3.1.1 | Entire core

Bulk density was slightly higher in the afforested bogs ($p < .05$, Kruskal–Wallis, $r > .07$) than in the intact and restored bogs (Table 2). However, no statistically significant difference was found in bulk density between the intact and restored bogs at the 95% confidence interval ($p = .063$, Kruskal–Wallis). There were statistically significant differences in moisture content between all three treatments ($p < .001$, Kruskal–Wallis, $r > .10$), lowest in afforested and highest in intact bogs. In contrast, carbon content was significantly lower in afforested and highest in intact bogs ($p < .001$, Kruskal–Wallis, $r > 0.30$). Carbon density was highest in the afforested bogs ($p < .05$, Kruskal–Wallis, $r > .08$), where the bulk density was highest. However, the effect sizes were small in each case.

While the mean specific yield (S_y) was lowest in intact and highest in restored bogs (Table 2), there were no significant differences between the three treatments ($p = .497$). The geometric mean of saturated hydraulic conductivity (K_s) was 1.7×10^{-4} cm s^{-1} across all sampled depths, and surprisingly, no significant difference was observed between treatments ($p = .616$, Kruskal–Wallis). The pH was significantly higher in the intact than the afforested and restored bogs ($p < .005$, one-way ANOVA, $\eta^2 > .02$), while electrical conductivity was significantly lower in the intact bogs ($p < .01$, Kruskal–Wallis, $r > .4$). However, no significant difference was observed between afforested and restored bogs for pH ($p = .460$, one-way ANOVA) and electrical conductivity ($p = .850$, Kruskal–Wallis), respectively.

3.1.2 | Variation with depth

Variation with depth for the measured peat properties is given in Figure 2 except for K_s, which is omitted since it was only measured at 20 and 40 cm depths for all sites. Overall, bulk density decreased with depth ($r_s = -.173$, $p < .001$, $N = 965$) for all three treatments except for the deepest sampling depth. Moisture content generally increased with depth ($r_s = .314$, $p < .001$, $N = 965$) except for fluctuations in the restored and afforested bogs,

TABLE 2 Descriptive statistics for the three main treatments for all measured variables over the whole 1 m peat core

Variable	Type	CLD	*N*	Mean	*SE*	Min	Median	Max
Bulk density (g cm^{-3})	AB	a	256	0.1034	0.0026	0.0091	0.0959	0.2500
	IB	b	286	0.0913	0.0015	0.0147	0.0883	0.2038
	R	b	422	0.0952	0.0013	0.0323	0.0921	0.2729
Moisture (%vol)	AB	c	257	87.75	0.39	32.89	89.15	98.56
	IB	a	286	90.30	0.16	78.87	90.58	98.44
	R	b	422	89.70	0.14	64.14	90.05	96.67
C (% of dry mass)	AB	c	248	50.899	0.079	44.132	51.224	52.407
	IB	a	282	51.263	0.082	37.185	51.574	52.411
	R	b	405	51.120	0.073	35.894	51.415	52.369
S_y	AB	a	257	0.0827	0.0028	0.0036	0.0749	0.3432
	IB	a	286	0.0778	0.0022	0.0055	0.0738	0.2425
	R	a	422	0.0871	0.0030	0.0033	0.0745	0.6093
von Post	AB	b	257	5.6	0.099	1.1	5.4	9.5
	IB	a	286	6.0	0.101	1.1	6.3	9.5
	R	ab	423	5.9	0.078	1.1	6.2	9.4
pH	AB	b	257	4.26	0.026	3.54	4.16	5.64
	IB	a	286	4.39	0.020	3.53	4.39	5.10
	R	b	423	4.29	0.014	3.38	4.30	5.15
EC (μS cm^{-1})	AB	a	257	34.21	1.36	11	31	280
	IB	b	286	25.26	1.47	11	21	350
	R	a	423	38.01	1.75	12	31	330
Infiltrometer K_s (cm s^{-1})	AB	a	18	0.001317	0.000226	0.000001	0.001421	0.002634
	R	a	3	0.001876	0.000403	0.001381	0.001572	0.002674
Piezometer K_s (cm s^{-1})	AB	a	2	0.000042	0.000028	0.000014	0.000042	0.000070
	IB	a	12	0.000608	0.000252	0.000010	0.000111	0.002210
	R	a	20	0.000130	0.000044	0.000005	0.000039	0.000795

Note: The arithmetic mean was taken for K_s across all measured depths for the two different methods used.

Abbreviations: CLD, compact letter display (the same letters signify no significant difference between the treatments at the 95% confidence interval); EC, electrical conductivity; *N*, sample size; *SE*, standard error of the mean; von Post estimates are given as 1–10 for the degree of humification.

flattening off at depths greater than 60 cm. Carbon content declined at the 40 cm sampling depth in the intact bogs and 60 and 80 cm depths for the afforested and the restored bogs, respectively. S_y was not significantly correlated with sampling depth, although a weak negative correlation existed between S_y and bulk density ($r_s = -.116$, $p = .001$, $N = 965$). Including all sites, a decline in S_y for afforested bogs over 0–40 cm depths was followed by a spike at 50 cm and a steady increase between 60 and 100 cm depths. A consistent drop in S_y at 40 cm depths was evident at the two afforested raised bog locations but not at the blanket bog locations. S_y in the restored bogs for all sites remained relatively constant from the surface until 40 cm depth, followed by a general increase in S_y towards 100 cm depth. There was a sharp decline in S_y between 10 and 20 cm depth in the intact bogs, and then, it remained relatively constant until 100 cm depth. Correlations between the measured variables and depth below the surface are given in Figure 3. The pH generally increased with depth ($r_s = .342$, $p < .001$, $N = 966$), whereas electrical conductivity decreased ($r_s = -.281$, $p < .001$, $N = 966$). Humification had the strongest positive correlation with depth ($r_s = .671$, $p < .001$, $N = 966$), changing from the lowest point on the von Post scale (H1 – completely undecomposed peat) to near the highest (H9 – practically fully decomposed peat), at deeper depths. After removing one outlier where we were not confident the piezometer intake was at the correct depth, a Kruskal–Wallis test found K_s to be significantly lower ($p < .01$, $r > .69$) in the restoration sites, at 20 cm depths, than at the intact and afforested sites.

3.2 | Differences between bog types and locations

There were significant differences in peat properties between the two different bog types (Table 3).

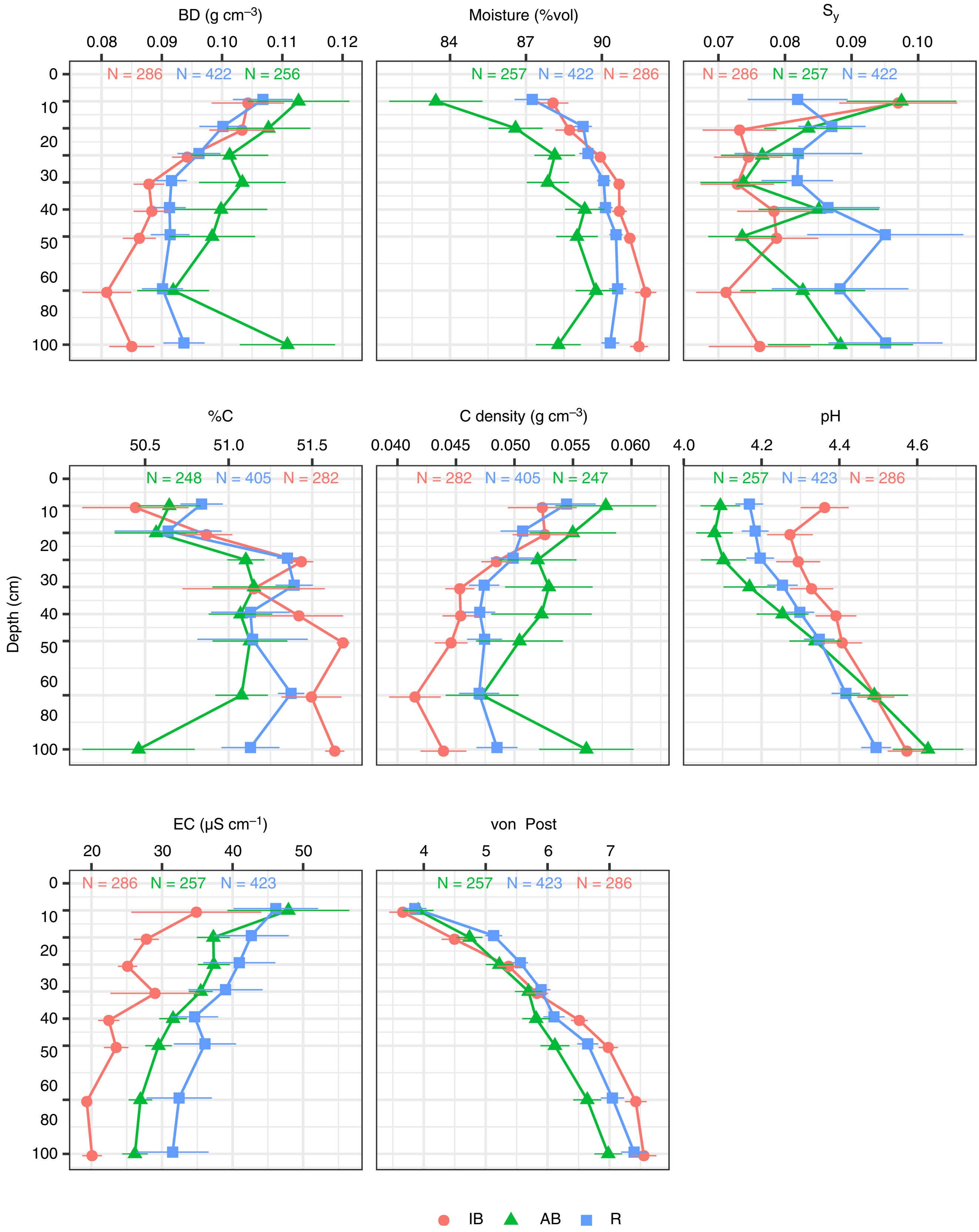

FIGURE 2 Peat property means for the three main treatments (IB = intact bog, AB = afforested bog and R = forest-to-bog restoration) by depth (cm) ± standard errors. BD, bulk density; EC, electrical conductivity. The sample size for each treatment is given for each variable

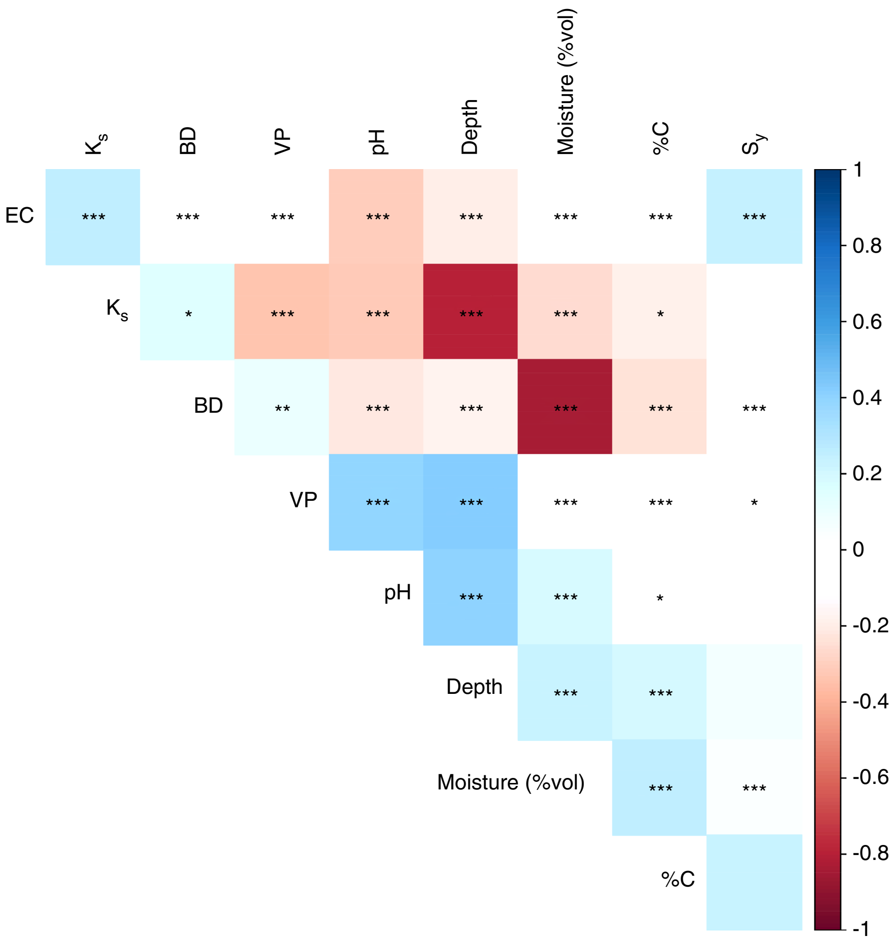

FIGURE 3 Spearman rank correlation matrix between the measured variables and depth at 0.005 (***), 0.01 (**), 0.05 (*) significance levels

Mann–Whitney tests showed that the blanket bog sites had higher bulk density (p = .013, r = .09), lower moisture content (p = .027, r = .02), lower S_y ($p <$ .001, r = .01) and more highly decomposed peat ($p <$ .001, r = .21) than raised bogs, although effects were small. The pH was significantly lower ($p <$ .001, Mann–Whitney, r = .73), and the electrical conductivity significantly higher (p = .006, Mann–Whitney, r = .36) in the raised bog locations, with large and medium effect sizes, respectively. Across all treatments, there were no significant differences in K_s between the four locations. However, S_y was significantly different between all locations ($p <$ .05, Kruskal–Wallis, $r >$ .008). Flanders Moss had the highest S_y, and Talaheel had the lowest. Figure 4 displays the depth profiles for the measured variables for the intact bog sites from each of the four locations, excluding K_s for the reasons previously explained. Differences in the peat properties of intact bog between locations showed greater overall effects for some variables than the land-use treatments (compare Figures 2 and 4). The largest effects were observed for bulk density ($p <$ .001, Kruskal–Wallis, r = .686) and moisture content ($p <$ .001, Kruskal–Wallis, r = .716) between the two blanket bog locations, and pH ($p <$ .001, one-way ANOVA, η^2 = .327) between all locations.

3.3 | Microtopographic differences

There were no significant differences in the peat properties measured between the intact bog microforms. The depth profiles are given in the supplementary information (Figure S2). In the afforested bogs (Figure S3), the peat in the furrows was significantly ($p <$ .001, Kruskal–Wallis, $r >$ .32) more humified than in ridges or the original surface. No significant difference was found in humification between the ridges and the original surface. In the restoration sites (Figure S4), the pH, moisture content, S_y, and humification were significantly higher in the furrows than in ridges ($p <$ .01, Kruskal–Wallis, $r >$ .15), and the bulk density was significantly lower ($p <$ .005, Kruskal–Wallis, r = .17), although the effects were small.

4 | DISCUSSION AND CONCLUSIONS

4.1 | Differences between treatments

Our spatial comparison follows similar patterns to the time-series study by Anderson and Peace (2017), who observed a small decrease in bulk density (~0.01 g cm^{-3}),

TABLE 3 Descriptive statistics for the blanket bog (BB) and raised bog (RB) classification for all treatments and depths up to 1 m

Variable	Class	CLD	N	Mean	SE	Min	Median	Max
BD (g cm^{-3})	BB	a	477	0.0979	0.0014	0.0100	0.0942	0.2500
	RB	b	487	0.0945	0.0014	0.0091	0.0900	0.2729
Moisture (%vol)	BB	b	478	89.35	0.20	32.89	90.03	98.56
	RB	a	487	89.37	0.18	64.14	90.06	98.55
C (% of dry mass)	BB	a	467	51.260	0.054	37.928	51.497	52.407
	RB	a	468	50.951	0.072	35.894	51.334	52.411
S_y	BB	b	478	0.0755	0.0020	0.0048	0.0688	0.3718
	RB	a	487	0.0907	0.0026	0.0033	0.0819	0.6093
von Post	BB	a	478	6.2	0.0813	1.1	6.3	9.5
	RB	b	488	5.5	0.0631	1.1	5.3	9.4
pH	BB	a	478	4.38	0.0144	3.54	4.40	5.15
	RB	b	488	4.25	0.0163	3.38	4.21	5.64
EC (μS cm^{-1})	BB	b	478	28.29	0.93	11	24	280
	RB	a	488	38.05	1.65	11	31	350
Infiltrometer K_s (cm s^{-1})	BB	a	12	0.001580	0.000226	0.000008	0.001542	0.002674
	RB	a	9	0.001151	0.000366	0.000001	0.000475	0.002634
Piezometer K_s (cm s^{-1})	BB	a	14	0.000355	0.000156	0.000005	0.000113	0.002210
	RB	a	20	0.000250	0.000131	0.000007	0.000039	0.002014

Note: The arithmetic mean was taken for K_s across all measured depths for the two different methods used.

Abbreviations: BD, bulk density; CLD = compact letter display (the same letters signify no significant difference between the bog types at the 95% confidence interval); EC, electrical conductivity; *N* = sample size; *SE* = standard error of the mean; von Post estimates are given as 1–10 for the degree of humification.

particularly in the top 10 cm, and an increase in moisture content of blanket bog 10 years after clear-felling and furrow blocking. Recovery was attributed to renewed buoyancy after re-wetting previously unsaturated near-surface peat and overburden pressure release from clear-felling (Anderson & Peace, 2017). However, we observed the highest variability for bulk density and moisture content in the afforested bogs. Mean bulk density ranged from 0.084 g cm^{-3} to 0.138 g cm^{-3} and moisture content from 83.0% to 89.0% from the afforested blanket bog sites, which appeared independent of plantation age. Given the variability and range of errors, it could be argued that the magnitude of differences between treatments for bulk density, moisture and carbon content were small despite being significant.

It is important to note that sampling fewer locations more intensively than more locations less intensively may increase the risk of pseudoreplication and, therefore, affect the validity of statistical tests. However, in most cases, the sampling locations from the different sites were sufficiently far away that the samples taken from the different treatments would have been unlikely to influence each other. They were also randomly selected through ArcGIS. The degrees of freedom for each statistical test were typically well below the sample sizes at the subsample level, an indicator that pseudoreplication was not problematic in our study (Waller et al., 2013). Furthermore, where pseudoreplication is an issue, Type I errors are likely to increase with the number of treatment replicates, increasing the chance of rejecting the null hypothesis (Hurlbert, 1984). Therefore, the sampling scheme is unlikely to be responsible for the small statistical responses encountered. However, it is also important to note that differences between locations may make it more difficult to ascribe statistical differences to the treatments.

The slightly lower carbon content in the afforested bogs could suggest some oxidative losses of CO_2 from the peat due to increased decomposition rates in aerobic peat through deeper water tables (Clark et al., 2009), but it was surprising that larger differences were not found between treatments. The water table in the afforested site at Flanders Moss (FMAB) dropped to below 60 cm, and at the afforested site at Forsinain (FOAB), the water table was below 50 cm in the summer drought of 2018 (Howson et al., 2021b). However, the water table may have been lower outside our study period, which may explain the lower carbon content throughout the 1 m profile for afforested peat (if further oxidation had occurred). The difference in carbon content between FMAB and FMIB was greater than that between the afforested and intact sites at the other locations, which may be related to that site having an earlier planting date. Higher electrical

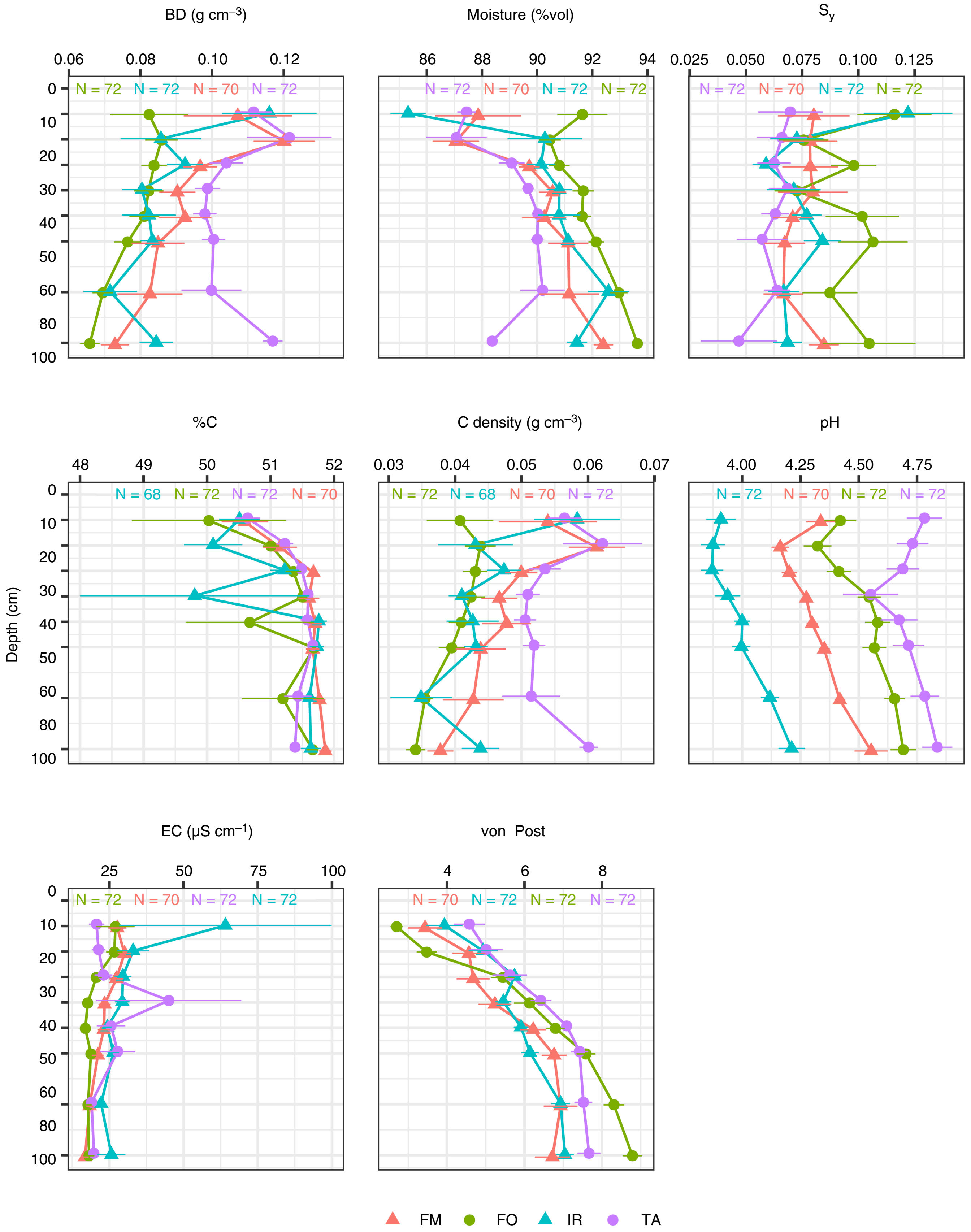

FIGURE 4 Peat property means throughout the depth profile at the intact sites ± standard errors where FM = Flanders Moss; FO = Forsinain; IR = Ironhirst; TA = Talaheel. Triangles represent the raised bog points, and circles represent the blanket bog points. BD = bulk density; EC = electrical conductivity. The sample size for each location is given for each variable

conductivities at the afforested and restoration sites could be due to legacy effects of acid interception and sea salt scavenging (where sites were near the coast) by forest canopies (Curtis et al., 2014; Harriman & Morrison, 1982; Howson et al., 2021a), which enhances the inputs of solutes to the peat from throughfall and stemflow (Gaffney et al., 2018; Howson et al., 2021a; Neary & Gizyn, 2011). However, the pH was variable between afforested sites, despite the significant difference between treatments, which probably reflects different acid deposition rates between locations, the impact of which will be influenced by tree age. It is also possible that tree litter and felled waste may be the primary source of enhanced solutes and lower pH in afforested and restored bogs. Electrical conductivity was significantly higher in the afforested bogs at the blanket bog locations, most likely due to sea salt scavenging from their maritime proximity. High electrical conductivity was also observed in the raised bog restoration sites at Flanders Moss, but it is important to note that one restoration site had high calcium inputs (the source of which was unidentified), which may have influenced the results.

4.2 | Differences between bog types and locations

There was no significant difference in bulk density, moisture content, carbon content, S_y, or K_s between intact raised and blanket bogs. The pH and degree of humification were significantly lower and electrical conductivity higher in the raised bogs than in the blanket bogs with large, small and moderate effect sizes, respectively. Atmospheric deposition and land use are likely controls on pH and electrical conductivity. Therefore, geographic location impacts the degree to which pH declines and electrical conductivity increases through differences in acid and sea salt deposition rates (Dunford et al., 2012; Evans et al., 2005; Harriman et al., 2003). The correlation between pH and humification may suggest decomposition, pH and water-table depth are interconnected. Less humified peats usually indicate shallower water tables; however, the substratum's pH can also influence the degree of decomposition (Drzymulska, 2016). Studies have shown that lower pH can inhibit microbes associated with decomposition (Bridgham & Richardson, 1992; Ivarson, 1977), but the process may differ between anaerobic and aerobic peats (Bridgham & Richardson, 1992; Preston et al., 2012). Therefore, the higher pH at the intact and restored blanket peatland sites may have enhanced decomposition. Conversely, the lower pH in the afforested bogs may have inhibited decomposition (Killham & Wainwright, 1981), possibly explaining why differences in peat carbon content were less than expected. However, the pH did not vary considerably between treatments, and it was quite variable between the different locations. Therefore, there may have been other influencing factors. The afforested site at Forsinain had unusually higher pH than the intact bog, and the oldest restoration site at Flanders Moss had unexplained Ca inputs at the catchment outlet, which may indicate the presence of a spring or other mineral influences. It is also possible those catchments were limed as part of forestry establishment, although there are no records of this occurring, and the practice would have been extremely unusual.

4.3 | Differences between microforms

No significant differences were found between the intact bog microforms in the measured peat properties. However, the degree of humification was significantly higher in the furrows than in the other afforested microforms and S_y significantly lower in the furrows than the original surface, in the top 60 cm at the afforested blanket bog sites. Carbon and moisture content were significantly lower, and bulk density and electrical conductivity were significantly higher in ridges than the furrows in the top 20 cm of afforested raised bogs.

Bulk density, moisture content, humification, S_y and pH differed significantly throughout the depth profile between microforms in the raised bog restoration sites, whereas only humification, moisture and carbon content differed significantly in the blanket bog restoration sites. Significant differences in compressibility, bulk density and K_s have previously been observed between intact microforms at specific depths, but high overall variability is typically reported (Baird et al., 2016; Branham & Strack, 2014; Waddington et al., 2010). Baird et al. (2016) found K_s, bulk density and humification highly variable between microforms, although it was suggested that ridges were more highly decomposed with higher bulk densities than hollows. Baird et al. (2016) found that natural microforms persisted for 1200 and 1400 years. Their results indicated significant differences in K_s between hollows and ridges at 50 cm but not at 90 cm. However, our study suggests that any differences between intact microforms in the top metre of peat were insignificant compared with changes brought about by forest ploughing and restoration.

Forest ploughing likely explains the differences in humification profiles between afforested microforms where the top 30–50 cm of peat in the furrows is removed, and the resulting ridges would be a mixture of the top layers from the furrows. The top layers in furrows immediately after ploughing would be equivalent to 30–50 cm below the original surface. In the restored bogs, the humification below ridges and the original surface was closer to that below furrows than in the afforested bogs. Since planting, furrows can infill with mosses and tree litter, potentially explaining the convergence in humification profiles. As might be expected, the moisture content below furrows was higher, particularly at shallow depths, in the restored bogs than in the afforested bogs. However, differences in pH between the furrows and the other microforms in the restored bogs were greater than in the afforested bogs.

The greater differences in pH and total concentrations of solutes (electrical conductivity) between furrows and the other microforms in the restored bogs could contribute to the divergence Anderson and Peace (2017) found away from natural bog vegetation. They had suggested furrows may be deeper and steeper-sided than natural depressions found in intact bogs, and nutrient levels may be higher. The higher electrical conductivity found in our peat samples in the furrows and nutrient concentrations found in the porewater supports this argument, especially where felled brash has accumulated (Howson et al., 2021a). Hancock et al. (2018) reported more of a trajectory away from natural bog vegetation in the plough ridges after restoration, which may be influenced by the higher bulk densities and reduced moisture content for the microform. These results would support the use of new restoration methods such as 'stump flipping' and ground smoothing as a mandatory step in the restoration process of forest-to-bog sites (Andersen et al., 2017; Anderson, 2017).

4.4 | Implications of findings for hydrological functioning

The flow of water and solute transport in peatlands are closely linked with the physical peat properties (Rezanezhad et al., 2016). In this study, K_s ranged from 2.67×10^{-3} cm s^{-1} to 5.53×10^{-7} cm s^{-1}, similar to those reported in other peatland studies (Baird et al., 2016; Lewis et al., 2012), with no significant difference observed between treatments, except at the shallowest depth measured (20 cm) where the forest-to-bog restoration sites had significantly lower hydraulic conductivities than the intact and afforested sites. This finding could have resulted from the collapse of larger pores in the near-surface peat from the weight of heavy machinery used during forestry operations. Interestingly, mean bulk density was lower in the restoration sites, where K_s was measured, than in the intact sites but not by a significant margin. There were no significant differences at the other measured depths. The spatial variance was similar between the different treatments at depths other than 20 cm, which suggests land-use change did not increase spatial variance.

Overall, despite the difference in bulk density and lower near-surface K_s at the restoration sites, subsurface flow through the upper layers of peat may not differ significantly between treatments if tree roots and cracks from desiccation and forestry disturbances provide new flow paths. The result may explain why the streamflow duration curves reported by Howson et al. (2021b) were comparable for the intact, afforested and restored sites at Forsinain, although *c*. 80% of the flow in blanket bogs has been shown to occur across the peat surface (Holden & Burt, 2003). Surface infiltration rates were also not significantly different between the afforested sites, suggesting other factors, such as tree age, may influence how high water tables rise during storm events, as reported by Howson et al. (2021b). It is hypothesized that changes in the peat properties resulting from afforestation would not affect the hydrology of peatlands as much as the fundamental change to the water budget brought about by the presence of trees through increased evapotranspiration, which would be more pronounced where the trees are mature.

5 | CONCLUSION

Overall, the hypothesis that peat properties in the afforested bogs would be significantly different from those in the intact bogs, while peat properties for restored bogs would lie somewhere in between, was accepted. Therefore, the trajectory of some variables towards values found in natural bogs may suggest ecosystem recovery after re-wetting. However, differences in bulk density, specific yield, humification, moisture and carbon content between land-use types were small, and the minimum effect size (η^2 and r) for pH, electrical conductivity, specific yield and saturated hydraulic conductivity was greater between the different intact bogs than those observed between land-use treatments. Acid deposition rates and sea salt scavenging may have influenced the pH and peat decomposition, although there may have been interactions with the underlying geology. Therefore, interactions between geographic location, peatland type and land-use treatments need to be considered when interpreting land management impacts on peatland properties and functioning.

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CONFLICT OF INTEREST

The authors confirm that they have no conflicts of interest, which may have influenced the findings of this study.

DATA AVAILABILITY STATEMENT

The data is available upon reasonable request.

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REFERENCES

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

Anderson, A. R. (2001). *Deforesting and restoring peat bogs: A review* (Vol. *32*). Forestry Commission. https://www.forestresearch.gov.uk/research/archive-deforesting-and-restoring-peat-bogs-a-review/

Anderson, A. R. (2017). Forest-to-bog restoration in Scotland: Recent advances, gaps and barriers. IUCN-UK peatland programme: Forest Research. http://www.iucn-uk-peatlandprogramme.org/sites/www.iucn-uk-peatlandprogramme.org/files/Forest-to-bog%20restoration%20in%20Scotland%20-%20Russell%20Anderson.pdf

Anderson, R., & Peace, A. (2017). Ten-year results of a comparison of methods for restoring afforested blanket bog. *Mires and Peat*, *19*(6), 1–23. https://doi.org/10.19189/MaP.2015.OMB.214

Anderson, R., Vasander, H., Geddes, N., Laine, A., Tolvanen, A., O'Sullivan, A., & Aapala, K. (2016). Afforested and forestry-drained peatland restoration. In A. Bonn, T. Allott, H. Joosten, & R. Stoneman (Eds.), *Peatland restoration and ecosystem services: Science, policy and practice* (pp. 213–233). Cambridge University Press. https://doi.org/10.1017/CBO9781139177788.013

Baird, A. J., Eades, P. A., & Surridge, B. W. J. (2008). The hydraulic structure of a raised bog and its implications for ecohydrological modelling of bog development. *Ecohydrology*, *1*(4), 289–298. https://doi.org/10.1002/eco.33

Baird, A. J., Milner, A. M., Blundell, A., Swindles, G. T., & Morris, P. J. (2016). Microform-scale variations in peatland permeability and their ecohydrological implications. *Journal of Ecology*, *104*(4), 531–544. https://doi.org/10.1111/1365-2745.12530

Baird, A. J., Surridge, B. W. J., & Money, R. P. (2004). An assessment of the piezometer method for measuring the hydraulic conductivity of a Cladium mariscus—Phragmites australis root mat in a Norfolk (UK) fen. *Hydrological Processes*, *18*(2), 275–291. https://doi.org/doi:10.1002/hyp.1375

Bol, R. A., Harkness, D. D., Huang, Y., & Howard, D. M. (1999). The influence of soil processes on carbon isotope distribution and turnover in the British uplands. *European Journal of Soil Science*, *50*(1), 41–51. https://doi.org/10.1046/j.1365-2389.1999.00222.x

Brand, E. W., & Premchitt, J. (1980). Shape factors of cylindrical piezometers. *Géotechnique*, *30*(4), 369–384. https://doi.org/10.1680/geot.1980.30.4.369

Branham, J. E., & Strack, M. (2014). Saturated hydraulic conductivity in sphagnum-dominated peatlands: Do microforms matter? *Hydrological Processes*, *28*(14), 4352–4362. https://doi.org/10.1002/hyp.10228

Bridgham, S. D., & Richardson, C. J. (1992). Mechanisms controlling soil respiration (CO_2 and CH_4) in southern peatlands. *Soil Biology and Biochemistry*, *24*(11), 1089–1099. https://doi.org/10.1016/0038-0717(92)90058-6

Cannell, M. G. R., Dewar, R. C., & Pyatt, D. G. (1993). Conifer plantations on drained peatlands in Britain: A net gain or loss of carbon? *Forestry: An International Journal of Forest Research*, *66*(4), 353–369. https://doi.org/10.1093/forestry/66.4.353

Clark, J. M., Ashley, D., Wagner, M., Chapman, P. J., Lane, S. N., Evans, C. D., & Heathwaite, A. L. (2009). Increased temperature sensitivity of net DOC production from ombrotrophic peat due to water table draw-down. *Global Change Biology*, *15*(4), 794–807. https://doi.org/10.1111/j.1365-2486.2008.01683.x

Curtis, C. J., Battarbee, R. W., Monteith, D. T., & Shilland, E. M. (2014). The future of upland water ecosystems of the {UK} in the 21st century: A synthesis. *Ecological Indicators*, *37*(part B), 412–430. https://doi.org/10.1016/j.ecolind.2013.10.012

Dinsmore, K. J., Billett, M. F., Skiba, U. M., Rees, R. M., Drewer, J., & Helfter, C. (2010). Role of the aquatic pathway in the carbon and greenhouse gas budgets of a peatland catchment. *Global Change Biology*, *16*(10), 2750–2762. https://doi.org/10.1111/j.1365-2486.2009.02119.x

Drzymulska, D. (2016). Peat decomposition – Shaping factors, significance in environmental studies and methods of determination; a literature review. *Geologos*, *22*(1), 61–69. https://doi.org/10.1515/logos-2016-0005

Dunford, R. W., Donoghue, D. N. M., & Burt, T. P. (2012). Forest land cover continues to exacerbate freshwater acidification despite decline in sulphate emissions. *Environmental Pollution*, *167*, 58–69. https://doi.org/10.1016/j.envpol.2012.03.022

Evans, C. D., Monteith, D. T., & Cooper, D. M. (2005). Long-term increases in surface water dissolved organic carbon: Observations, possible causes and environmental impacts. *Environmental Pollution*, *137*(1), 55–71. https://doi.org/10.1016/j.envpol.2004.12.031

Gaffney, P. (2017). *The effects of bog restoration in formerly afforested peatlands on water quality and aquatic carbon fluxes*. (Doctoral thesis). The University of Aberdeen.

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

Garnett, M. H., Ineson, P., Stevenson, A. C., & Howard, D. C. (2001). Terrestrial organic carbon storage in a British moorland. *Global Change Biology*, *7*(4), 375–388. https://doi.org/10.1046/j.1365-2486.2001.00382.x

González, E., Henstra, S. W., Rochefort, L., Bradfield, G. E., & Poulin, M. (2014). Is rewetting enough to recover sphagnum and associated peat-accumulating species in traditionally exploited bogs? *Wetlands Ecology and Management*, *22*(1), 49–62. https://doi.org/10.1007/s11273-013-9322-6

Hancock, M. H., Klein, D., Andersen, R., & Cowie, N. R. (2018). Vegetation response to restoration management of a blanket bog damaged by drainage and afforestation. *Applied Vegetation Science*, *21*(2), 167–178. https://doi.org/10.1111/avsc.12367

Hargreaves, K. J., Milne, R., & Cannell, M. G. R. (2003). Carbon balance of afforested peatland in Scotland. *Forestry: An International Journal of Forest Research*, *76*(3), 299–317. https://doi.org/10.1093/forestry/76.3.299

Harriman, R., & Morrison, B. R. S. (1982). Ecology of streams draining forested and non-forested catchments in an area of Central Scotland subject to acid precipitation. *Hydrobiologia*, *88*, 251–263. https://doi.org/10.1007/BF00008505

Harriman, R., Watt, A. W., Christie, A. E. G., Moore, D. W., McCartney, A. G., & Taylor, E. M. (2003). Quantifying the effects of forestry practices on the recovery of upland streams and lochs from acidification. *Science of The Total Environment*, *310*(1), 101–111. https://doi.org/10.1016/S0048-9697(02)00626-5

Holden, J. (2009). Flow through macropores of different size classes in blanket peat. *Journal of Hydrology*, *364*(3–4), 342–348. https://doi.org/10.1016/j.jhydrol.2008.11.010

Holden, J., & Burt, T. P. (2003). Runoff production in blanket peat covered catchments. *Water Resources Research*, *39*(7), 1191. https://doi.org/10.1029/2002WR001956

Holden, J., Burt, T. P., & Cox, N. J. (2001). Macroporosity and infiltration in blanket peat: The implications of tension disc infiltrometer measurements. *Hydrological Processes*, *15*(2), 289–303. https://doi.org/10.1002/hyp.93

Holden, J., Chapman, P. J., & Labadz, J. C. (2004). Artificial drainage of peatlands: Hydrological and hydrochemical process and wetland restoration. *Progress in Physical Geography: Earth and Environment*, *28*(1), 95–123. https://doi.org/10.1191/0309133304pp403ra

Hommeltenberg, J., Schmid, H., Drösler, M., & Werle, P. (2014). Can a bog drained for forestry be a stronger carbon sink than a natural bog forest? *Biogeosciences*, *11*(13), 3477–3493. https://doi.org/10.5194/bg-11-3477-2014

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

Howson, T., Chapman, P. J., Shah, N., Anderson, R., & Holden, J. (2021b). The effect of forest-to-bog restoration on the hydrological functioning of raised and blanket bogs. *Ecohydrology*, *14*(7), e2334. https://doi.org/10.1002/eco.2334

Hurlbert, S. H. (1984). Pseudoreplication and the design of ecological field experiments. *Ecological Monographs*, *54*(2), 187–211. https://doi.org/10.2307/1942661

Hvorslev, M. (1951). *Time lag and soil permeability in groundwater observations, Waterways Exper*. Sta. Corps of Engrs, U.S. Army. https://www.csus.edu/indiv/h/hornert/geol_210_summer_2012/week%203%20readings/hvorslev%201951.pdf

IBM Corp. (2016). *IBM SPSS statistics for windows (version 24.0)*. IBM Corp. https://www.ibm.com/uk-en/products/spss-statistics

Ivarson, K. C. (1977). Changes in decomposition rate, microbial population and carbohydrate content of an acid peat bog after liming and reclamation. *Canadian Journal of Soil Science*, *57*(2), 129–137. https://doi.org/10.4141/cjss77-017

Joosten, H. (2016). Peatlands across the globe. In A. Bonn, T. Allott, H. Joosten, & R. Stoneman (Eds.), *Peatland restoration and ecosystem services: Science, policy and practice* (pp. 19–43). Cambridge University Press. https://doi.org/10.1017/CBO9781139177788.003

Joosten, H., Sirin, A., Couwenberg, J., Laine, J., & Smith, P. (2016). The role of peatlands in climate regulation. In A. Bonn, T. Allott, H. Joosten, & R. Stoneman (Eds.), *Peatland restoration and ecosystem services: Science, policy and practice* (pp. 63–76). Cambridge University Press. https://doi.org/10.1017/CBO9781139177788.005

Kassambara, A. (2020). Pipe-friendly framework for basic statistical tests (version 0.6.0). https://CRAN.R-project.org/package=rstatix

Killham, K., & Wainwright, M. (1981). Deciduous leaf litter and cellulose decomposition in soil exposed to heavy atmospheric pollution. *Environmental Pollution Series A, Ecological and Biological*, *26*(1), 79–85. https://doi.org/10.1016/0143-1471(81)90100-8

Koskinen, M., Sallantaus, T., & Vasander, H. (2011). Post-restoration development of organic carbon and nutrient leaching from two ecohydrologically different peatland sites. *Ecological Engineering*, *37*(7), 1008–1016. https://doi.org/10.1016/j.ecoleng.2010.06.036

Koskinen, M., Tahvanainen, T., Sarkkola, S., Menberu, M. W., Laurén, A., Sallantaus, T., Marttila, H., Ronkanen, A.-K., Parviainen, M., Tolvanen, A., Koivusalo, H., & Nieminen, M. (2017). Restoration of nutrient-rich forestry-drained peatlands poses a risk for high exports of dissolved organic carbon, nitrogen, and phosphorus. *Science of The Total Environment*, *586*, 858–869. https://doi.org/10.1016/j.scitotenv.2017.02.065

Lewis, C., Albertson, J., Xu, X., & Kiely, G. (2012). Spatial variability of hydraulic conductivity and bulk density along a blanket peatland hillslope. *Hydrological Processes*, *26*(10), 1527–1537. https://doi.org/10.1002/hyp.8252

Liu, H., & Lennartz, B. (2019). Hydraulic properties of peat soils along a bulk density gradient—A meta study. *Hydrological Processes*, *33*(1), 101–114. https://doi.org/10.1002/hyp.13314

Liu, H., Price, J., Rezanezhad, F., & Lennartz, B. (2020). Centennial-scale shifts in hydrophysical properties of peat induced by drainage. *Water Resources Research*, *56*(10), 1–14. https://doi.org/10.1029/2020WR027538

McCarter, C. P. R., & Price, J. S. (2015). The hydrology of the Bois-des-bel peatland restoration: Hydrophysical properties limiting connectivity between regenerated sphagnum and remnant vacuum harvested peat deposit. *Ecohydrology*, *8*(2), 173–187. https://doi.org/10.1002/eco.1498

Menberu, M. W., Tahvanainen, T., Marttila, H., Irannezhad, M., Ronkanen, A.-K., Penttinen, J., & Kløve, B. (2016). Water-table-dependent hydrological changes following peatland forestry drainage and restoration: Analysis of restoration success. *Water Resources Research*, *52*(5), 3742–3760. https://doi.org/10.1002/2015WR018578

Minitab 19 Statistical Software. (2020). (Version 19.2020.1). Minitab, Inc. https://www.minitab.com

Muller, F. L. L., Chang, K.-C., Lee, C.-L., & Chapman, S. J. (2015). Effects of temperature, rainfall and conifer felling practices on the surface water chemistry of northern peatlands.

Biogeochemistry, *126*(3), 343–362. https://doi.org/10.1007/s10533-015-0162-8

Muller, F. L. L., & Tankéré-Muller, S. P. C. (2012). Seasonal variations in surface water chemistry at disturbed and pristine peatland sites in the flow country of northern Scotland. *Science of The Total Environment*, *435–436*, 351–362. https://doi.org/10.1016/j.scitotenv.2012.06.048

Mustamo, P., Hyvärinen, M., Ronkanen, A. K., Kløve, B., & Moffat, A. J. (2016). Physical properties of peat soils under different land use options. *Soil Use and Management*, *32*(3), 400–410. https://doi.org/10.1111/sum.12272

Neary, A. J., & Gizyn, W. I. (2011). Throughfall stemflow chemistry under deciduous coniferous forest canopies in south-Central Ontario. *Canadian Journal of Forest Research*, *24*(6), 1089–1100. https://doi.org/10.1139/x94-145

Paavilainen, E., & Päivänen, J. (1995). *Peatland forestry: Ecology and principles*. Springer-Verlag. https://doi.org/10.1007/978-3-662-03125-4

Palviainen, M., Finér, L., Kurka, A.-M., Mannerkoski, H., Piirainen, S., & Starr, M. (2004). Release of potassium, calcium, iron and aluminium from Norway spruce, scots pine and silver birch logging residues. *Plant and Soil*, *259*(1/2), 123–136. https://doi.org/10.1023/B:PLSO.0000020938.78215.bd

Parry, L. E., & Charman, D. J. (2013). Modelling soil organic carbon distribution in blanket peatlands at a landscape scale. *Geoderma*, *211–212*, 75–84. https://doi.org/10.1016/j.geoderma.2013.07.006

Preston, M., Smemo, K., McLaughlin, J., & Basiliko, N. (2012). Peatland microbial communities and decomposition processes in the James Bay lowlands, Canada. *Frontiers in Microbiology*, *3*(70), 1–15. https://doi.org/10.3389/fmicb.2012.00070

Price, J. S. (1996). Hydrology and microclimate of a partly restored cutover bog, Quebec. *Hydrological Processes*, *10*(10), 1263–1272. https://doi.org/10.1002/(SICI)1099-1085(199610)10:10%3C1263::AID-HYP458%3E3.0.CO;2-1

Price, J. S., & Schlotzhauer, S. M. (1999). Importance of shrinkage and compression in determining water storage changes in peat: The case of a mined peatland. *Hydrological Processes*, *13*(16), 2591–2601. https://doi.org/10.1002/(SICI)1099-1085(199911)13:16%3C2591::AID-HYP933%3E3.0.CO;2-E

Rezanezhad, F., Price, J. S., Quinton, W. L., Lennartz, B., Milojevic, T., & Van Cappellen, P. (2016). Structure of peat soils and implications for water storage, flow and solute transport: A review update for geochemists. *Chemical Geology*, *429*, 75–84. https://doi.org/10.1016/j.chemgeo.2016.03.010

Rowell, D. L. (1994). *Soil science: Methods and applications*. Longman Scientific & Technical. https://doi.org/10.1002/jsfa.2740660423

RStudio Team. (2016). *RStudio: Integrated development environment for R*. RStudio, Inc. https://www.rstudio.com/

Sangok, F. E., Maie, N., Melling, L., & Watanabe, A. (2017). Evaluation on the decomposability of tropical forest peat soils after conversion to an oil palm plantation. *Science of The Total Environment*, *587–588*, 381–388. https://doi.org/10.1016/j.scitotenv.2017.02.165

Scharlemann, J. P. W., Tanner, E. V. J., Hiederer, R., & Kapos, V. (2014). Global soil carbon: Understanding and managing the largest terrestrial carbon pool. *Carbon Management*, *5*(1), 81–91. https://doi.org/10.4155/cmt.13.77

Shah, N. W., & Nisbet, T. R. (2019). The effects of forest clearance for peatland restoration on water quality. *Science of The Total Environment*, *693*, 133617. https://doi.org/10.1016/j.scitotenv.2019.133617

Shotbolt, L., Anderson, A. R., & Townend, J. (1998). Changes to blanket bog adjoining forest plots at bad a' Cheo, Rumster Forest, Caithness. *Forestry: An International Journal of Forest Research*, *71*(4), 311–324. https://doi.org/10.1093/forestry/71.4.311

Silins, U., & Rothwell, R. L. (1998). Forest peatland drainage and subsidence affect soil water retention and transport properties in an Alberta peatland. *Soil Science Society of America Journal*, *62*(4), 1048–1056. https://doi.org/10.2136/sssaj1998.03615995006200040028x

Simola, H., Pitkänen, A., & Turunen, J. (2012). Carbon loss in drained forestry peatlands in Finland, estimated by re-sampling peatlands surveyed in the 1980s. *European Journal of Soil Science*, *63*(6), 798–807. https://doi.org/10.1111/j.1365-2389.2012.01499.x

Sloan, T. J., Payne, R. J., Anderson, A. R., Gilbert, P., Mauquoy, D., Newton, A. J., & Andersen, R. (2019). Ground surface subsidence in an afforested peatland fifty years after drainage and planting. *Mires and Peat*, *23*(6), 1–12. https://doi.org/10.19189/MaP.2018.OMB.348

Strack, M. (Ed.). (2008). *Peatlands and climate change*. International Peat Society.

Surridge, B. W. J., Baird, A. J., & Heathwaite, A. L. (2005). Evaluating the quality of hydraulic conductivity estimates from piezometer slug tests in peat. *Hydrological Processes*, *19*(6), 1227–1244. https://doi.org/10.1002/hyp.5653

Tonks, A. J., Aplin, P., Beriro, D. J., Cooper, H., Evers, S., Vane, C. H., & Sjögersten, S. (2017). Impacts of conversion of tropical peat swamp forest to oil palm plantation on peat organic chemistry, physical properties and carbon stocks. *Geoderma*, *289*, 36–45. https://doi.org/10.1016/j.geoderma.2016.11.018

van Genuchten, M. T. (1980). A closed-form equation for predicting the hydraulic conductivity of unsaturated soils. *Soil Science Society of America Journal*, *44*(5), 892–898. https://doi.org/10.2136/sssaj1980.03615995004400050002x

von Post, L. (1922). Swedish geological peat survey with the results obtained so far. *Sven. Mosskult. Tidskr*, *36*, 1–27.

Waddington, J. M., Kellner, E., Strack, M., & Price, J. S. (2010). Differential peat deformation, compressibility, and water storage between peatland microforms: Implications for ecosystem function and development. *Water Resources Research*, *46*(7), article W07538. https://doi.org/10.1029/2009WR008802

Waller, B. M., Warmelink, L., Liebal, K., Micheletta, J., & Slocombe, K. E. (2013). Pseudoreplication: A widespread problem in primate communication research. *Animal Behaviour*, *86*(2), 483–488. https://doi.org/10.1016/j.anbehav.2013.05.038

Wickham, H. (2016). *ggplot2: Elegant graphics for data analysis*. Springer-Verlag. https://ggplot2.tidyverse.org

Young, D. M., Baird, A. J., Charman, D. J., Evans, C. D., Gallego-Sala, A. V., Gill, P. J., Hughes, P. D. M., Morris, P. J., & Swindles, G. T. (2019). Misinterpreting carbon accumulation rates in records from near-surface peat. *Scientific Reports*, *9*(1), 17,939. https://doi.org/10.1038/s41598-019-53879-8

Yu, Z., Loisel, J., Brosseau, D. P., Beilman, D. W., & Hunt, S. J. (2010). Global peatland dynamics since the last glacial maximum. *Geophysical Research Letters*, *37*(13), Article L13402. https://doi.org/10.1029/2010gl043584

Zhang, R. (1997). Determination of soil sorptivity and hydraulic conductivity from the disk infiltrometer. *Soil Science Society of America Journal*, *61*(4), 1024–1030. https://doi.org/10.2136/sssaj1997.03615995006100040005x

Zhang, Z., Lin, L., Wang, Y., & Peng, X. (2016). Temporal change in soil macropores measured using tension infiltrometer under different land uses and slope positions in subtropical China. *Journal of Soils and Sediments*, *16*(3), 854–863. https://doi.org/10.1007/s11368-015-1295-z

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