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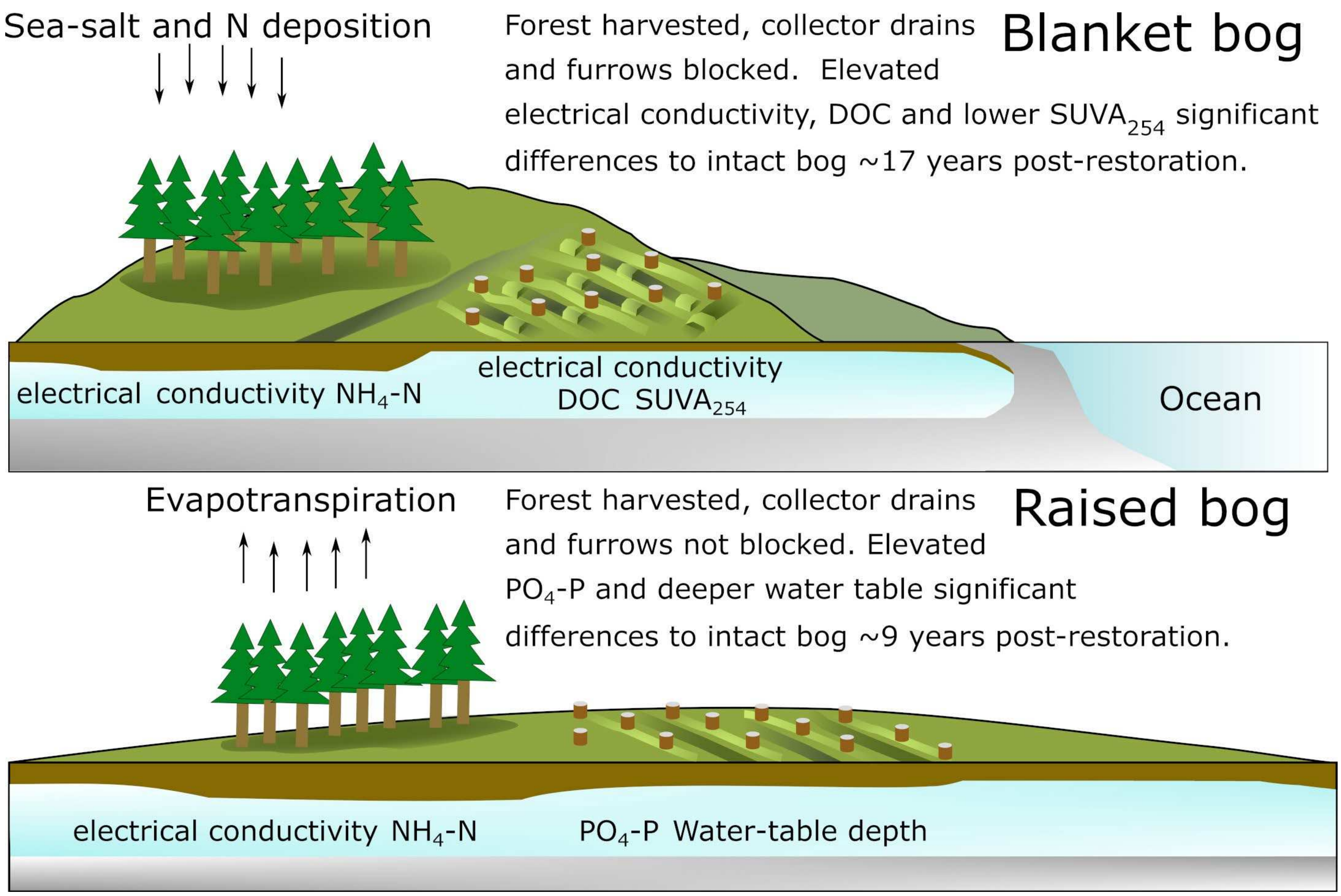
Sea-salt and N deposition
Forest harvested, collector drains and furrows blocked. Elevated electrical conductivity, DOC and lower $SUVA_{254}$ significant differences to intact bog ~17 years post-restoration.
Blanket bog
electrical conductivity NH_4-N
electrical conductivity
DOC $SUVA_{254}$
Ocean
Evapotranspiration
Forest harvested, collector drains and furrows not blocked. Elevated PO_4-P and deeper water table significant differences to intact bog ~9 years post-restoration.
Raised bog
electrical conductivity NH_4-N
PO_4-P Water-table depth

A comparison of porewater chemistry between intact, afforested and restored raised and blanket bogs

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Abstract

Afforestation is a significant cause of global peatland degradation. In some regions, afforested bogs are now undergoing clear-felling and restoration, often known as forest-to-bog restoration. We studied differences in water-table depth (WTD) and porewater chemistry between intact, afforested, and restored bogs at a raised bog and blanket bog location. Solute concentrations and principal component analysis suggested that water-table drawdown and higher electrical conductivity (EC) and ammonium (NH_4-N) concentrations were associated with afforestation. In contrast, higher dissolved organic carbon (DOC) and phosphate (PO_4-P) concentrations were associated with deforestation. Drying-rewetting cycles influenced seasonal variability in solute concentrations, particularly in shallower porewater at the raised bog location. WTD was significantly deeper in the oldest raised bog restoration site (~9 years post-restoration) than the intact bog (mean difference = 6.2 cm). However, WTD in the oldest blanket bog restoration site (~17 years post-restoration), where furrows had been blocked, was comparable to the intact bog (mean difference = 1.2 cm). When averaged for all porewater depths, NH_4-N concentrations were significantly higher in the afforested than the intact sites (mean difference = 0.77 mg L^{-1}) whereas significant differences between the oldest restoration sites and the intact sites included higher PO_4-P (mean difference = 70 µg L^{-1}) in the raised bog and higher DOC (mean difference = 5.6 mg L^{-1}), EC (mean difference = 19 µS cm^{-1}) and lower $SUVA_{254}$ (mean difference = 0.13 L mg^{-1} m^{-1}) in the blanket bog. Results indicate felled waste (brash) may be a significant source of soluble C and PO_4-P. Mean porewater PO_4-P concentrations were between two and five times higher in furrows and drains in which brash had accumulated compared to other locations in the same sites where brash had not accumulated. Creating and maintaining brash-free buffer zones may therefore minimise freshwater impacts.

1 Introduction

Peatlands are highly important ecosystems, responsible for a third of the global soil carbon pool (Yu *et al.*, 2010, Scharlemann *et al.*, 2014) and essential for a range of other ecosystem services (Bonn *et al.*, 2016), despite covering less than 3% of the Earth's land area (Xu *et al.*, 2018). Historically, the condition of peatlands has been influenced by land management, with an estimated 15% of peatlands globally now in a non-natural state (Joosten, 2016). Large-scale deforestation of naturally forested peatlands or afforestation of treeless peatlands with non-native trees for timber (Päivänen and Hånell, 2012) or palm oil production (Joosten, 2016) are significant sources of peatland degradation (Ramchunder *et al.*, 2012, Menberu *et al.*, 2016). More than half of Finland's formerly accumulating peatlands have been forestry-drained, mainly between 1960 and 1990 (Strack, 2008) and in the UK, non-native coniferous trees have been planted on previously open peatlands since the 1940s with up to ~190,000 ha of deep peat afforested between 1950 and the 1980s (Cannell *et al.*, 1993, Hargreaves, 2003).

Recognition of the biodiversity value and the carbon sequestration (Apps *et al.*, 1993, Simola *et al.*, 2012) potential of peatlands has led to increased efforts to protect and restore these ecosystems in the UK (Parry *et al.*, 2014, Andersen *et al.*, 2017) and globally (Rochefort and Andersen, 2017). Attempts to restore previously afforested fen and bog peatlands have occurred in many parts of Europe and some areas of North America (Anderson *et al.*, 2016, Chimner *et al.*, 2016, Andersen *et al.*, 2017). Earlier restoration in Scandinavia (Komulainen *et al.*, 1999, Haapalehto *et al.*, 2011) predates much of the work carried out in the UK, but forest-to-bog restoration is still a relatively new practice. Therefore, there has been limited opportunity to study the long-term effects on water-table depth (WTD) and water quality of large-scale deforestation to support peatland restoration. Also, it is not clear if different peatland types respond similarly or not to forest-to-bog restoration.

Conifer plantations on peatlands lower the water table through drainage and evapotranspiration (Anderson, 2001). Restoration (Figure 1) requires forest clearance to raise the water table, which is a critical factor for the hydrological functioning of bogs (Anderson, 2001, Holden *et al.*, 2004, Price *et al.*, 2016). Forest clearance alone may not result in sufficient change in water-table levels to bring about restoration in the short term (Anderson and Peace, 2017). Therefore, drainage ditches and furrows may be blocked to assist in the recovery of the water table (Haapalehto *et al.*, 2011, Haapalehto *et al.*, 2014, Anderson and Peace, 2017). However, few UK studies report peatland restoration after conifer felling results in water-table levels that are similar to those in undisturbed peatlands.

There are a range of forest clearance methods that can result in different amounts of forest biomass being left on the site, which potentially affects water quality. At some restoration sites, usually, those where the forest is being felled early for peatland restoration, the trees are left to decompose naturally on the ground or have even been compressed into furrows and drains to slow the flow of water (Muller *et al.*, 2015). At others, most of the timber and felling debris (i.e. branches and tops) has been removed using low impact techniques (Shah and Nisbet, 2019). Residues from decaying forest debris can be an important source of nutrients and organic matter (Muller and Tankéré-Muller, 2012, Muller *et al.*, 2015, Gaffney, 2017, Gaffney *et al.*, 2018, Shah and Nisbet, 2019) entering adjacent watercourses with the potential to affect sensitive aquatic species such as Atlantic salmon, *Salmo salar*, and freshwater pearl mussel, *Margaritifera margaritifera*, both legally protected species (Shah and Nisbet, 2019). Mulching of whole trees is an alternative to conventional harvesting and is sometimes used where the trees have little or no commercial value and where extraction could cause further damage to soil and water. However, little is known about the effects of mulching on water quality.

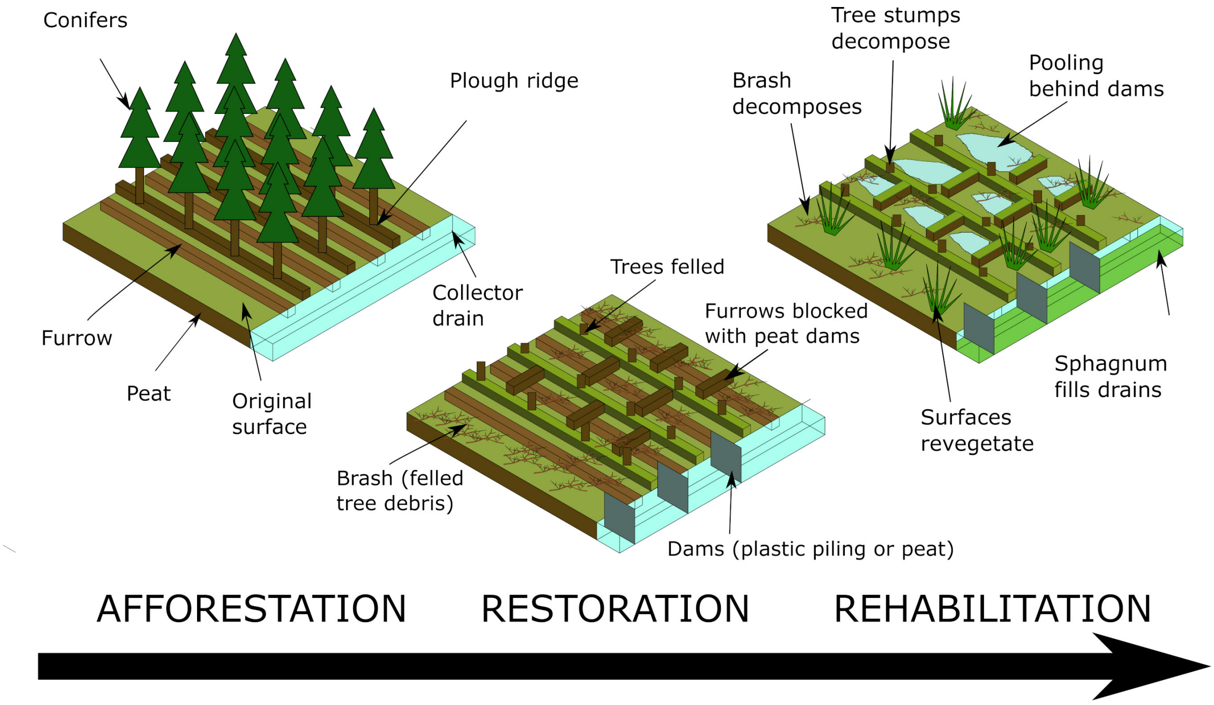

Figure 1 - Typical forest-to-bog restoration process - i) afforestation of peatland for commercial gain and forest matures (30-50 years); ii) forest is felled; ideally, timber is harvested, drains, and furrows are typically blocked to raise water-table levels; iii) peatland is left to rehabilitate to restore pre-afforestation ecosystem function.

Most forest-to-bog restoration studies in the UK have occurred on blanket bogs, and those that have examined water quality have focussed mainly on streamwater (Table 1). As bogs are ombrotrophic (i.e. receive nutrients mainly via precipitation), internal nutrient cycling is essential, and changes in nutrient cycling are tightly coupled to the carbon cycle (Keller *et al.*, 2006, Oviedo-Vargas *et al.*, 2013, Gaffney *et al.*, 2018). In the UK, only Gaffney *et al.* (2018) has looked at differences in porewater quality between forested, intact and restored sites finding the lasting legacies from afforestation were elevated NH_4-N and acidity 17 years after felling as well as incomplete WTD recovery. The influence of forestry on soil water pH is well established although it has become less of an issue in recent times (Harriman and Morrison, 1982, Fowler *et al.*, 1989, Nisbet *et al.*, 1995, Drinan *et al.*, 2013, Nisbit and Evans, 2014). However, the conclusions of Gaffney *et al.* (2018) were based on a limited programme of sampling on three occasions during the growing season and only on a blanket peatland. Thus, there is a need to expand the range of peat types and the frequency and duration of study post-felling to get a better understanding of the processes controlling porewater chemistry at different times.

Table 1 – Peatland conifer plantation restoration studies examining water quality. MF = minerotrophic fen; PF = nutrient poor fen / bog; BB = Blanket bog; RB = Raised bog; SS = soil samples; SW = streamwater; PW = porewater; TF = trees felled; TM = trees mulched; DD = drains dammed; DI =drains infilled; CF = brash compacted in furrows; LIH = low-impact harvesting; WTD = water-table depth.

Study	Location	Peatland type	Sample type	Restoration techniques	Time since felling	Key findings
Haapalehto *et al.* (2011)	Finland	MF/RB	SS	TF/DD/DI	0-10 years (if growth sufficient)	Elemental concentrations of Ca, K, Mg, Mn, and P were comparable to pristine peatlands 10 years after restoration.
Koskinen *et al.* (2011)	Finland	MF/PF	SW	TF/DD/DI	0-6 years (PF only)	MF leached more N, less dissolved organic carbon (DOC) and P than PF.
Sallantaus and Koskinen (2012)	Finland	MF/PF	SW	TF/DD/DI	0-7 years (PF only)	P elevated for 6 years in the PF. 6 times higher after 6 years in MF.
Muller and Tankéré-Muller (2012)	Scotland	BB	SW	TF/CF	0.5-1.5 years	Al and Mn influenced by felling. Forest buffer strips counteract mobilisation of DOC, K, and Fe.
Haapalehto *et al.* (2014)	Finland	MF/RB	PW	TF/DD/DI	0-10 years (if growth sufficient)	Long-term decrease in DOC and nutrient leaching were observed, but temporary increases in N and P for the first 5 years.
Muller *et al.* (2015)	Scotland	BB	SW	TF/TM/CF	~2 years	Spikes in DOC, Al, Fe, K, Mn, P year following felling. DOC-4, K-6, and P-15 times higher than near-natural bog after two years.
Koskinen *et al.* (2017)	Finland	MF/PF	SW	TF/DD/DI	0-4 years	Elevated DOC, N, and P 4 years after restoration. Less of an issue in PF.
Gaffney *et al.* (2018)	Scotland	BB	PW/SW	TF/TM/DD/CF	0-17 years.	WTD, pH and NH_4^+ in PW main barriers to restoration success.
Shah and Nisbet (2019)	Scotland	RB	SW	TF/LIH	0–9 years	Elevated phosphate returned to pre-felling levels after 3-5 years. DOC elevated 4 years after-restoration. pH impacts varied with a significant increase at one site.

Although conifer afforestation on peat can increase dissolved organic carbon (DOC) concentrations in porewater (Grieve and Marsden, 2001) through increased mineralisation as a result of lower water tables and increased litter production, the most significant increases in streamwater concentrations of DOC have been reported after felling (Evans *et al.*, 2005). Several studies where clear-felling and drain-blocking have taken place documented increases in DOC and nutrients (Koskinen *et al.*, 2011, Muller and Tankéré-Muller, 2012, Sallantaus and Koskinen, 2012, Muller *et al.*, 2015, Koskinen *et al.*, 2017) as have studies limited to clear-felling and forest harvesting (Rodgers *et al.*, 2010, Asam *et al.*, 2014b, Palviainen *et al.*, 2014, Clarke *et al.*, 2015, Nieminen *et al.*, 2015, Shah and Nisbet, 2019). However, time frames for recovery to pre-felling levels vary from 3-5 years (Shah and Nisbet, 2019) to greater than 10 years (Palviainen *et al.*, 2014) for N and P. Shah and Nisbet (2019) recommended that less intensive harvesting techniques can help reduce these negative impacts. However, more information is required to understand the transport mechanisms of DOC and nutrients from the point of source to watercourses.

Given the lack of studies that have compared the impact of forest-to-bog restoration on porewater quality and WTD at different peatland types over time, the objectives in this study were to:

i. Determine whether significant differences in WTD and porewater chemistry exist between intact, afforested, and restored bog sites.
ii. Investigate whether differences exist in the response of porewater chemistry to forest-to-bog restoration at different depths in the peat (20 to 80 cm)
iii. Quantify seasonal variability in WTD and porewater chemistry in intact, afforested, and restored bog sites and determine whether significant differences exist.
iv. Compare and contrast the impact of forest-to-bog restoration on porewater DOC and nutrients at a raised bog and blanket bog peatland.

2 Methods

2.1 Study sites

The blanket bog (BB) sites in this study are located at Forsinain in the 'Flow Country' area of northern Scotland (Figure 2), the largest blanket peatland in Europe (c. 4000 km^2). The land was previously owned by the Forestry Commission and has subsequently been acquired by the Royal Society for the Protection of Birds (RSPB) as part of the Forsinard Flows National Nature Reserve (NNR) in Sutherland (58° 24'43.2 "N, 3° 52'25.0 "W). The raised bog (RB) sites are located at Flanders Moss, part of a group of lowland raised bogs formed on the Carse of Stirling in Central Scotland (56° 08'10.5 "N, 4° 19'28.7 "W) and are managed by Forestry and Land Scotland (previously Forestry Commission Scotland) and Scottish National Heritage. The annual mean rainfall, between 1981-2010, (Met Office *et al.*, 2018) was 1444 mm at Flanders Moss and 1097 mm at Forsinain. The annual mean temperature was 8.7 °C at Flanders Moss, and 7.4 °C at Forsinain, over the same period.

Standing forestry plantation sites (hereafter referred to as afforested bog (AB)) and open, near-natural bog (hereafter referred to as intact bog (IB)) were included to represent the different land-management types. Two restored (R) sites of different ages since restoration (R1 > R2), and using slightly different restoration techniques, were selected at both locations (Table 2). At each location, all sites were broadly comparable in terms of slope, and the afforested sites were carefully chosen so that the whole area was under canopy cover.

Table 2 - Site characteristics at Flanders Moss (RB) and Forsinain (BB) where CA = catchment area (ha); Nest labels = unique sampling location IDs (referred to later). Felled-to-waste = trees felled, but timber and brash not extracted.

Site	Description	CA (ha)	Nest labels	Tree clearance dates	Restoration method	Furrow spacing	Planting year
RBIB	Intact raised bog	6.0	24,26,28,32				
RBAB	Afforested raised bog	0.2	72,73,74,75			1.4 m	~1965
RBR1	Restored raised bog	2.5	10,11,12,13	24/11/2009-09/12/2009 01/08/2011-18/10/2011	Part conventional harvesting; part low impact harvesting and removal of brash and logs.	1.4 m	~1965
RBR2	Restored raised bog	26.2	16,17,21,22	01/10/2013-31/03/2014	Conventional harvesting (i.e. fell, debranch, extract timber, leave brash).	1.4 m	~1965
BBIB	Intact blanket bog	1.6	45,46,47,48				
BBAB	Afforested blanket bog	5.1	63,64,65,66			1.9 m	~1980
BBR1	Restored blanket bog	1.6	33,34,35,36	2002-2003	Felled-to-waste/furrows & main drain blocked.	1.4 m	~1980
BBR2	Restored blanket bog	2.3	37,38,39,40	2014-2015	Mulched/main drain blocked.	2.3 m	~1980

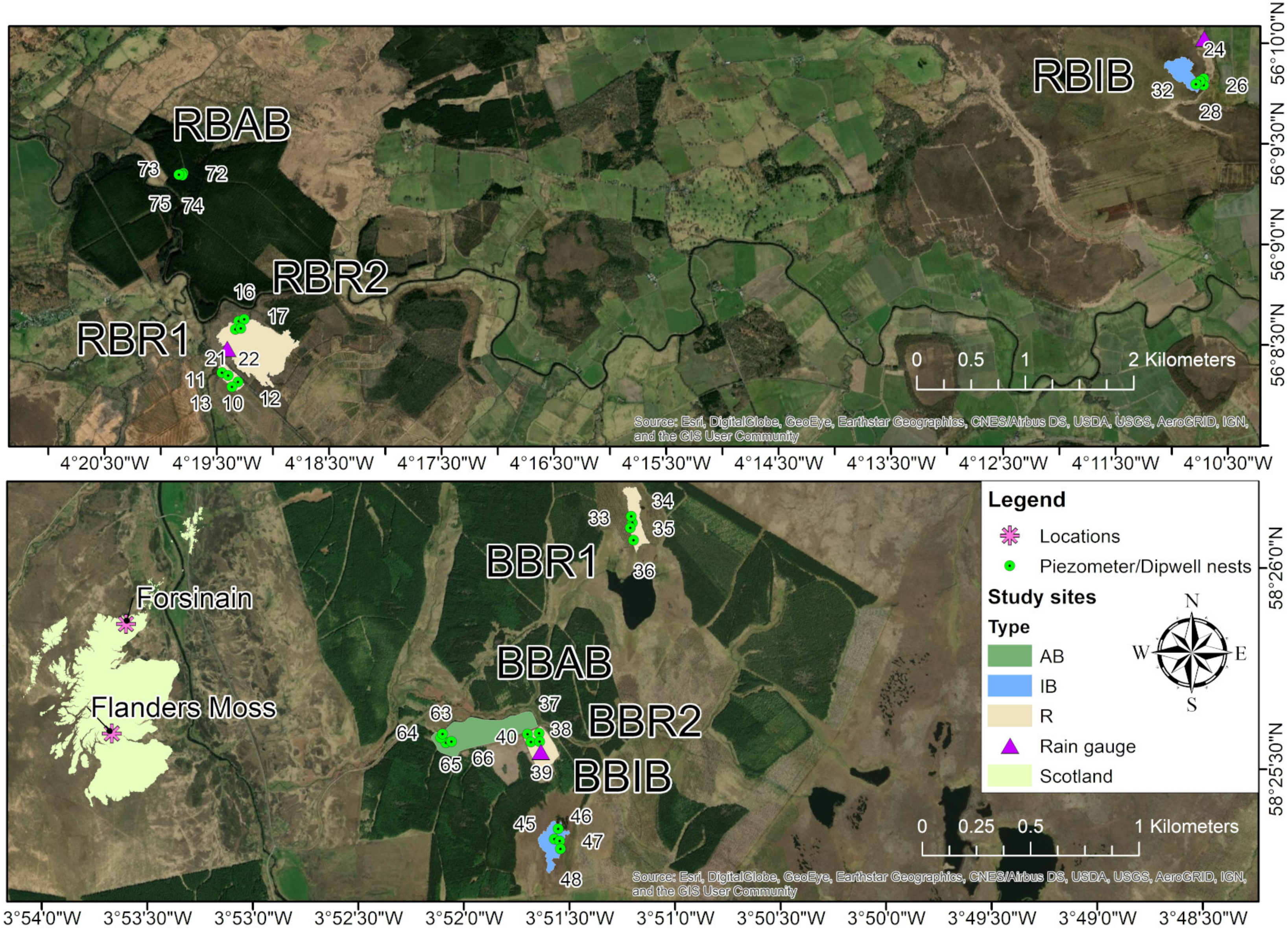

Figure 2 - Study site experimental design at Forsinain (BB) and Flanders Moss (RB); AB = afforested bog; IB = intact bog; R1 =oldest restoration site; R2 = most recent restoration site. The numbers represent instrument nest labels.

RBIB represented the best example of intact bog in the area with a mosaic of sphagnum mosses (including some nationally scarce species: *S. austinii*, *S. fuscum* and *S. molle*), sedges, ericaceous shrubs, and sundews (*Drosera spp.*). RBAB, RBR2 and RBR1 were drained in the 1920s to improve conditions for grouse shooting, and in the 1960s and 1970s were ploughed and planted with lodgepole pine (*Pinus contorta*) and Sitka spruce (*Picea sitchensis*). Records of fertiliser application at the time of planting are not available, but an application of NPK fertiliser was customary in afforestation schemes of that period (Shah and Nisbet, 2019). The first rotation forest at RBR2 was harvested over 6 months between 2013-14 using a conventional harvester and forwarder (Shah and Nisbet, 2019). The tree stems were extracted, but much of the brash, comprising branches and tops, was left on site to decompose. RBR1 was felled in two phases: the first in the winter of 2009 (15%) and the remainder in summer/autumn 2011. The first phase of felling was carried out using standard forest harvester, and forwarder techniques with forest materials, including brash, left *in situ*. The second phase was carried out by hand felling and winching the main stems out using an overhead Skyline (Shah and Nisbet, 2019), after which all useable timber and brash were removed and chipped for biomass. Neither drain nor furrow blocking had taken place at the raised bog restoration sites at the time of the study.

The vegetation at BBIB is similar in composition to that at RBIB with the addition of liverworts, bog asphodel (*Narthecium ossifragum*) and bogbean (*Menyanthes trifoliata*) in pools. At the southern end of BBIB, there was evidence peat cutting had once taken place and forestry had been planted to the east and west, but mainly it is a good example of near-natural bog, typical to the area, with natural pool complexes in the north. In the 1980s, the blanket bog was drained and planted with the same mixture of tree species as the raised bog, but there was a difference in the ploughing/planting phase with the furrows being 50 and 90 cm further apart in BBAB and BBR2 respectively. However, the furrows were similarly spaced to those at the raised bog in BBR1 (Table 2). It is likely but cannot be assumed that a standard application of NPK fertiliser would have been used at the time of planting.

BBR2 differed to the other restoration sites in that it was the only site where the trees had been 'whole-tree mulched' in 2014, most likely using a mechanical masticator mounted on the arm of an excavator (Moffat *et al.*, 2006, Muller *et al.*, 2015). Whole-tree mulching is an alternative to conventional felling where the trees are essentially chipped from standing, often used when the forest is being felled early, growth has been so weak that harvesting would entail a net cost, there are access constraints and the potential for site damage resulting in environmental impacts. It has the practical advantages of not requiring timber extraction, leaving less coarse debris on the surface of the peat and can potentially reduce soil and water damage as there is reduced machine trafficking. The main drain at BBR2 was also blocked with a sequence of three plastic piling dams close to the outflow, and additional peat dams were added at regularly spaced intervals on 23rd March 2019. BBR1 was felled in 2002/3 when the trees were still young (~20 years old), but any felled material was not extracted (felled-to-waste). Instead, it was compressed into the furrows, which were later blocked with peat dams in 2015/16, the same time as the main drain.

2.2 Field sampling and measurements

Within each site, four nests, consisting of four piezometers at 20, 40, 60 and 80 cm depths to collect soil water, and a dipwell for monitoring WTD (Figure 2), were carefully inserted into the peat after a hole had been augured of a slightly smaller diameter than the tubes. The piezometers were constructed from 19.05 mm internal diameter PVC tubing, cut to length, with 0.5 cm holes drilled in a ring at the sampling depth, and two further rings of holes drilled ±1 cm either side. Therefore, the porewater was sampled over a ~2.5 cm range at each depth. There was a 5 cm reservoir in the bottom of the piezometer to collect water. Air holes were drilled well above the surface to allow venting but prevent the ingress of overland flow, and a flush-fitting plug formed a watertight seal at the base. The dipwells were constructed from similar PVC tubing, generally > 1 m in length and with 0.5 cm holes drilled at 3.5 cm intervals throughout the length of the tube with four holes at each interval. The base was sealed with a PVC plug. Caps were fitted to the tops of both piezometers and dipwells to prevent debris and insect ingress.

The piezometer-dipwell nests were allocated random locations within each site, using the "Create Random Points" tool in ArcGIS (ESRI, 2017), generally > 30 m apart, and stored as waypoints in a handheld GPS. Each piezometer was labelled with a unique nest number followed by a letter A – D representing the sampling depth (where A = 20, B = 40, C = 60, D = 80 cm). The nest locations represented a range of surface features associated with afforestation and natural bog microforms (restored and afforested - ridges, furrows and original surface; intact - hollows, hummocks and lawns) and different mixtures of vegetation, which were recorded during the installation. A tipping bucket rain gauge was installed at the blanket bog, where the sites were relatively close to each other (BBR2: Davis 6465 + HOBO UA00364 event logger). Two tipping bucket rain gauges were installed at the raised bog (RBR1: Davis 7852 + HOBO H07-002-04 event logger; RBIB: Davis 6465 + HOBO UA00364 event logger), to account for any localised rainfall differences between the sites. Air temperature observations were taken from the 1 km HadUK-Grid (Met Office *et al.*, 2018). The closest weather stations with continuous records of air temperature for the study period were Bishopton, Glasgow for Flanders Moss (27.3 km), and Kinbrace for Forsinain (17.5 km).

The porewater chemistry and WTD were monitored at each site from April 2018 until November 2019. On each site visit, manual dipwell readings were taken using a steel capillary tube with a self-adhesive scale. All piezometers were emptied of any water and sampled the following day into a 50 mL centrifuge tube using a plastic syringe connected to a 1 m PVC hose rinsed with deionised water between samples. During dry periods, there was not always a collectable sample at 20 and 40 cm, particularly in the afforested sites. All porewater samples were packed with ice packs in an insulated box for refrigerated transport back to the laboratory.

Porewater samples were collected monthly from April to August 2018 and then at two-monthly intervals thereafter until November 2019 (n = 12; 1164 samples), except for a gap during winter due to site inaccessibility (December 2018 – March 2019). Measurements of pH and electrical conductivity (EC) were made on return to the laboratory using a HANNA 9124 pH meter and HORIBA B-173 EC meter.

2.2 Chemical analysis

Samples were vacuum filtered through 0.45 µm cellulose acetate filters usually within 48 hrs and then analysed for nutrients using colourimetry (Skalar San++ colourimetric auto-analyser) and dissolved organic carbon (DOC) by combustion (Analytik Jena Multi N/C 2100C combustion analyser). The following nutrients were determined using the auto-analyser: dissolved ammonium-nitrogen (NH_4-N), soluble reactive phosphate as phosphorus (PO_4-P), total oxidised nitrogen (TON) and nitrite-nitrogen (NO_2-N) with detection limits of 0.01, 0.005, 0.16 and 0.002 mg L^{-1}, respectively. Nitrate-nitrogen (NO_3-N) concentrations were determined by subtracting NO_2-N from TON. Additionally, water colour was measured by absorbance at 254 nm, 465 nm and 665 nm using a spectrophotometer (Jasco V-630 double beam spectrophotometer). Absorbance readings were converted to standardised water colour measurements of absorbance units per metre (abs m^{-1}).

Humic and fulvic acids are the dominant components of DOC and absorb light at different wavelengths, in different quantities. As a result, the ratio of absorption at 465 nm and 665 nm, known as the E4:E6 ratio, gives an indication of the proportion of humic and fulvic acids and hence the degree of humification as humic acids are more mature than fulvic acids (Grayson and Holden, 2011, Strack *et al.*, 2015). Thurman (1985) observed that humic acids from soils had an E4:E6 of 2 to

5, whereas fulvic acids had a ratio of 8 to 10. However, in some waters, little absorption occurs at 665 nm, so absorption at 254 nm, when normalised to the DOC concentration, has been used instead of E4:E6 as an indicator of aromaticity (Weishaar *et al.*, 2003, Helms *et al.*, 2008). The result, known as specific UV absorption ($SUVA_{254}$) was found by Weishaar et al. (2003) to correlate strongly with DOC aromaticity, as determined by 13C-nuclear magnetic resonance (13C-NMR). Higher values indicated greater aromaticity and therefore, greater hydrophobicity.

2.3 Data analysis

Solute boxplots were used as a visual comparison of the spread of the data using the ggplot2 package (Wickham, 2016) in R Studio (RStudio Team, 2016) for shallow (20-40 cm) and deep (60-80 cm) sampling depths at each site. Although peat depths were generally greater than 1 m, four equal depth increments were chosen in the top 80 cm to account for water-table drawdown in the forestry and to ensure no mineral material below the peat was disturbed. Time-series data were produced by taking the month and year of the sampling date, and statistical summaries were used for plotting mean monthly values and standard errors.

Other statistical analyses were performed in SPSS (IBM Corp., 2016), by firstly testing for normality and homogeneity of variance, and where possible parametric ANOVA tests of differences in the mean values of each group were used to test any hypotheses and identify any interactions between sites, location (Flanders Moss/Forsinain) and sampling depth. Where the data deviated from a normal distribution or homogeneity of variance was not satisfied, it was transformed in SPSS, or non-parametric Kruskal-Wallis tests were used. *Post-hoc* tests were used to determine significant differences for parametric tests, and pairwise comparisons were used for the same purposes for non-parametric Kruskal-Wallis tests. Spearman's rank coefficients (r_s) were calculated in SPSS to assess any non-parametric correlations between variables. Mann-Whitney U tests were used for non-parametric analysis in testing for differences between the locations. Generalised Linear Mixed

Models were used in SPSS to assess the independence of repeated measurements using 'Compound Symmetry' as the covariance type, the unique piezometer identifier as the subject and the sampling month as the repeated variable.

Principal component analysis (Jolliffe and Springer-Verlag, 2002) was carried out on the porewater variables at both locations using the three main treatments (intact, afforested, restored) as groups. Scree plots were produced to examine the variances of the principal components selecting all nutrient variables, DOC, WTD, air temperature, pH, and EC. Biplots, using the 'ggbiplot' package in R Studio (Vu, 2011), were generated to examine any clustering of observations with respect to the variable loadings and the first two principal components. The piezometer-dipwell nest label was used to identify individual observations to assess any outliers. The variable loadings gave a visual representation of their significance for the three different treatment groups and any relationships they may have. Any solute values that were below the detection limits of the instruments were substituted by the detection limit divided by the square root of two (Croghan and Egeghy, 2003). Outliers were preserved.

3 Results

3.1 Climate conditions during the study

The total monthly rainfall and mean monthly air temperatures from April 2018 until the end of November 2019 are shown in Figure 3 for both the raised bog and the blanket bog locations. Over the study period, the blanket bog was over a degree cooler, and there was 36 mm less precipitation than the raised bog. For 2018, the annual precipitation from the on-site rain gauges was 1001 mm at the raised bog and 742 mm at the blanket bog, which is considerably less than the mean annual figures of 1444 mm and 1097 mm at the raised bog and the blanket bog, respectively (Met Office *et al.*, 2018). Mean monthly temperatures during the study ranged from 3.4 to 16.6 °C at the raised bog and 2.4 to 15.4 °C at the blanket bog. The period between April 2018 and August 2018 was an unusually hot and dry period at both locations, and at the blanket bog, no rain was recorded for 36 consecutive days between the 15th June and 21st July. 2018 was one of the hottest summers on record with a longer-lasting drought at the blanket bog location. However, in the 2019 study year, the blanket bog rain gauge recorded 146 mm higher total precipitation than the RBIB rain gauge.

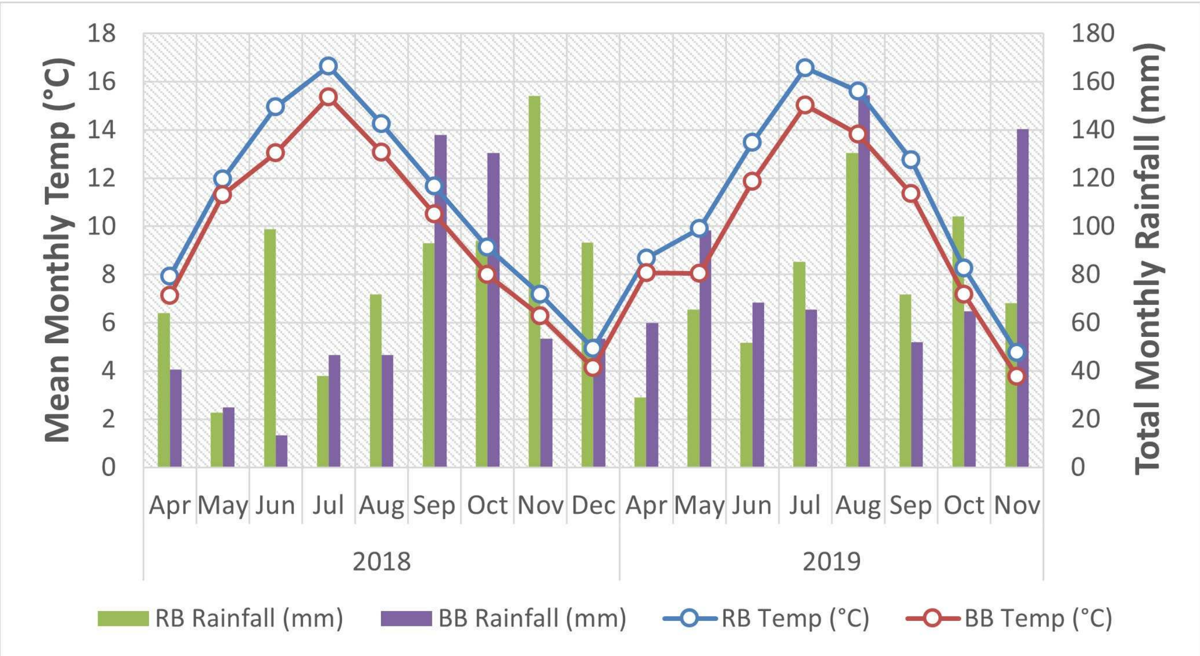

Figure 3 - Temperature and rainfall at Forsinain (BB) and Flanders Moss (RB) over the study period. Rainfall for the raised bog is taken from the RBIB rain gauge.

3.2 Water-table depth

Water-table drawdown in the forestry sites was evident at both raised and blanket bog locations (Figure 4). There was a significant difference ($p < 0.001$, one-way ANOVA) in WTD between the afforested (deepest) and the intact bog (shallowest) sites. The mean WTD at RBAB was 30.6 cm compared to 9.8 cm at RBIB. The average WTD was also significantly deeper ($p < 0.05$, one-way ANOVA) at RBR1 (16.0 cm) and RBR2 (20.7 cm) than RBIB. The mean WTD at BBAB was 25.6 cm compared to 9.6 cm at BBIB. However, the mean WTD for both BBR1 (8.4 cm) and BBR2 (11.9cm) were not significantly different ($p = 0.855$, one-way ANOVA) to BBIB. Overall, the mean WTD was significantly deeper (p = 0.002, one-way ANOVA) at the raised bog (18.0 cm) than the blanket bog (13.8 cm) location, and there was a significant interaction (p < 0.001, two-way ANOVA) between the location and the sampling month which highlighted significant seasonal differences existed between the two locations (Figure 4).

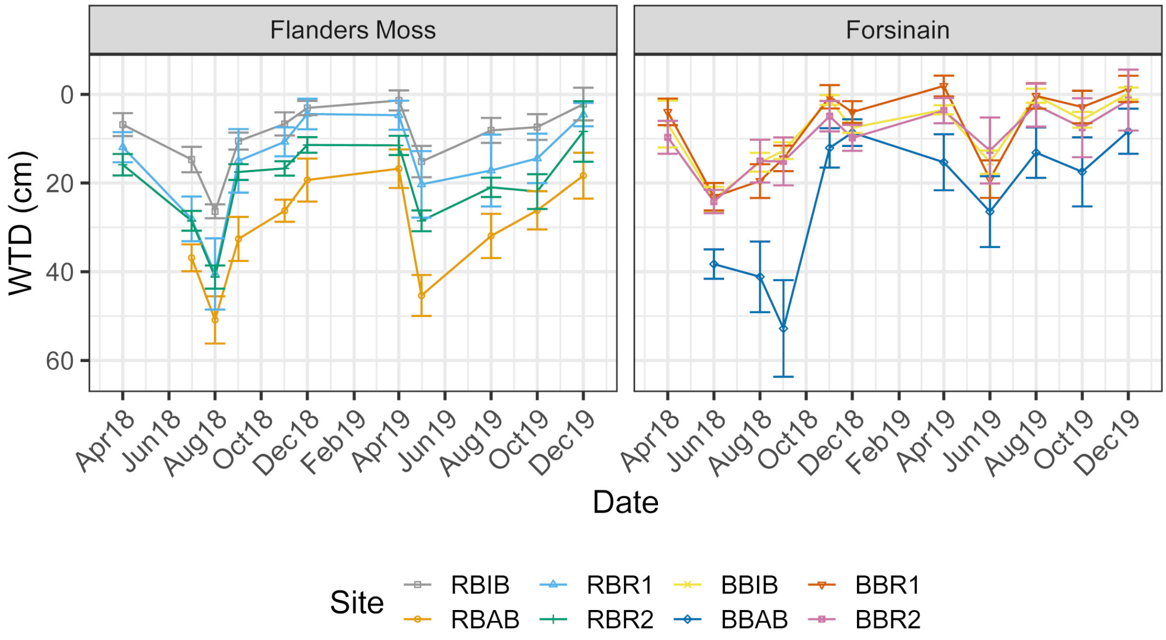

Figure 4 - Time series of water-table depth (WTD) ± SE for the sites at both locations. RB = raised bog; BB = blanket bog; IB = intact bog; AB = afforested bog; R1 = oldest restoration site; R2 =most recent restoration site.

WTD displayed a strong seasonal pattern at both locations (deeper in summer and shallower in winter) reflecting the rainfall and evapotranspiration patterns over the study period (Figure 4). On average, WTD was 0.2 cm deeper at RBIB than BBIB and 5.0 cm deeper at RBAB than BBAB, but the differences were not statistically significant. The difference in the WTD between the afforested site and the other sites was larger in the unusually dry period of spring/summer 2018 at the blanket bog, and in May 2019 at the raised bog location. The water table at BBAB receded beyond that of RBAB in the 2018 summer drought but remained shallower during the following summer (Figure 4). In wetter periods, the differences in WTD between the treatments decreased, especially at the blanket bog location. There was a divergence between BBAB and the other blanket bog sites, in July and August 2018, where rainfall was sufficient to raise the water table in the restored and intact sites, but not in the afforested site.

3.3 Porewater chemistry

Boxplots of the main porewater variables for each study site are presented in Figure 5. A small proportion of NH_4-N, PO_4-P and NO_2-N (0.3%, 7.0% and 14.0%, respectively) concentrations were below detection limits whereas the majority (98.4%) of TON concentrations were below the detection limit. NH_4-N concentrations at RBAB (mean = 1.48 mg L^{-1}) were significantly higher ($p < 0.001$, Kruskal-Wallis test) than for the other raised bog sites with the lowest mean concentration at RBIB (0.66 mg L^{-1}). RBR2 had the second-highest mean NH_4-N concentration (1.11 mg L^{-1}), while RBR1 (0.50 mg L^{-1}) was not significantly different from RBIB. Given that the majority of TON concentrations were below the detection limit, and NO_2-N concentrations were generally lower than 0.02 mg L^{-1}, both NO_2-N and NO_3-N are not presented. Mean PO_4-P concentrations for RBIB, RBAB, RBR1 and RBR2 were, 0.05 mg L^{-1}, 0.29 mg L^{-1}, 0.12 mg L^{-1} and 0.40 mg L^{-1}, respectively. PO_4-P concentrations at RBR2 were significantly higher ($p < 0.02$, Kruskal-Wallis test) than the other sites and although they were less at RBR1 they were still significantly higher ($p < 0.001$, Kruskal-Wallis test) than at RBIB.

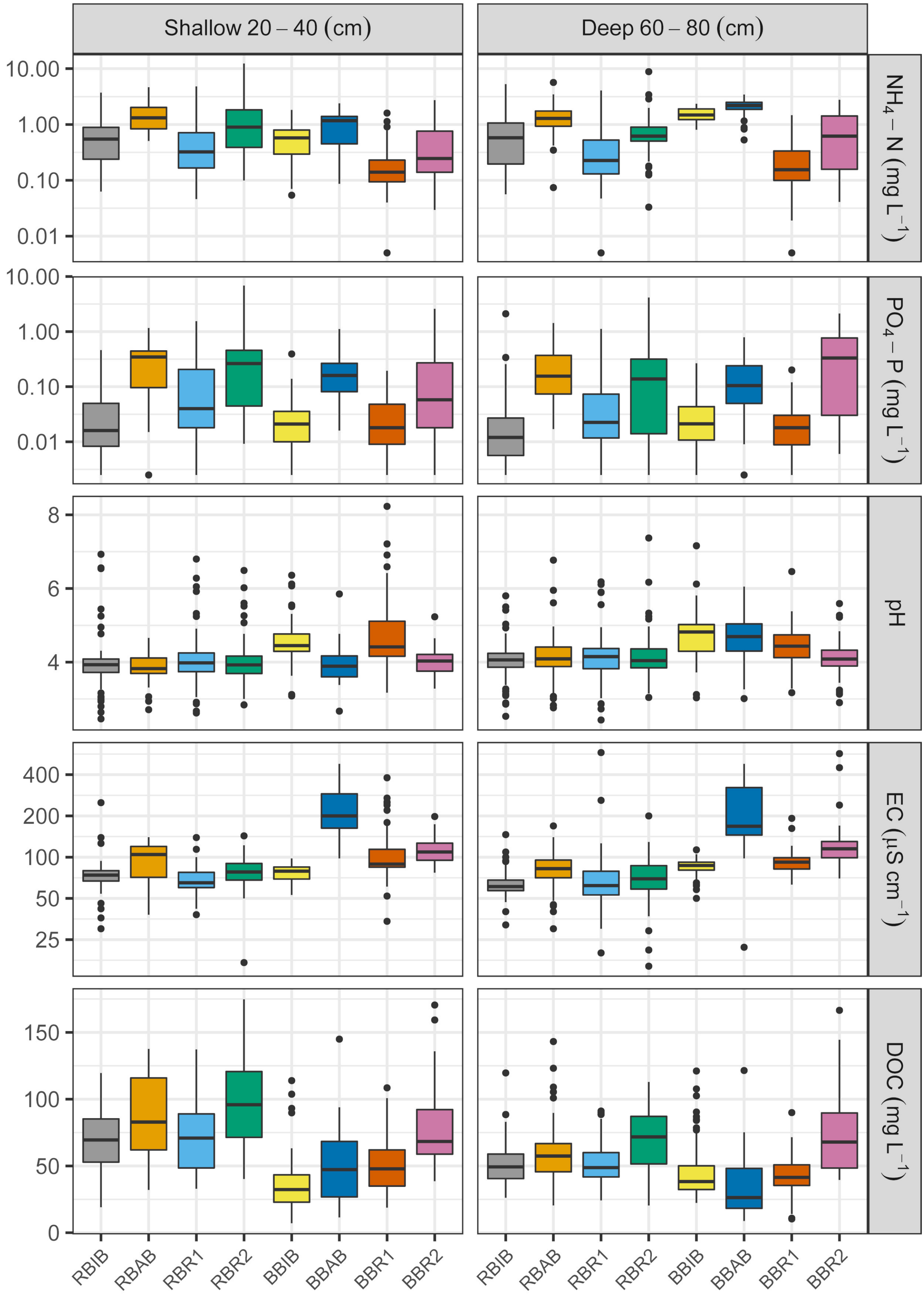

Figure 5 - Porewater variables for shallow and deep porewater. Log scales were used for NH_4-N, PO_4-P and EC to aid readability. RB = raised bog; BB = blanket bog; IB = intact bog; AB = afforested bog; R1 = oldest restoration site; R2 =most recent restoration site.

There was no significant difference in the pH between the raised bog sites although the means were fractionally higher (~0.1 units) in the two restoration sites than at RBIB and RBAB. The mean EC was 19 µS cm^{-1} higher ($p < 0.001$, Kruskal-Wallis test) in RBAB than the intact bog, but it was lower at the two restoration sites with no significant difference between RBR1 (72 µS cm^{-1}) and RBIB (69 µS cm^{-1}). Mean DOC at RBR2 (77.8 mg L^{-1}) was significantly higher ($p < 0.05$, Kruskal-Wallis test) than at RBAB (67.2 mg L^{-1}), RBR1 (58.5 mg L^{-1}) and RBIB (59.2 mg L^{-1}). On average, E4:E6 ratio and $SUVA_{254}$ values, at 20-40 cm depths, (Table 3) were significantly lower ($p < 0.05$, Kruskal-Wallis test) at RBAB than RBIB. However, no significant difference was found for DOC, E4:E6 and $SUVA_{254}$ between RBR1 and RBIB.

*Table 3 - Comparison of E4:E6 ratio (unitless) and $SUVA_{254}$ (L mg^{-1} m^{-1}) means ± SE for the study sites (20-40 cm depths). RB = raised bog; BB = blanket bog; IB = intact bog; AB = afforested bog; R1 = oldest restoration site; R2 =most recent restoration site. Significant differences are taken from Kruskal-Wallis pairwise comparisons. Significance levels are denoted as: *** < 0.001; ** < 0.01; * < 0.05.*

				E4:E6			**$SUVA_{254}$**	
Location	**S1**	**S2**	**S1 Mean**	**S2 Mean**	**Sig.**	**S1 Mean**	**S2 Mean**	**Sig.**
RB	RBAB	RBR2	6.62 ± 0.18	8.95 ± 0.30	0.000 ***	3.43 ± 0.11	3.31 ± 0.04	0.077
	RBAB	RBR1	6.62 ± 0.18	9.63 ± 0.44	0.000 ***	3.43 ± 0.11	3.39 ± 0.03	0.242
	RBAB	RBIB	6.62 ± 0.18	9.90 ± 0.31	0.000 ***	3.43 ± 0.11	3.50 ± 0.04	0.010 *
	RBR2	RBR1	8.95 ± 0.30	9.63 ± 0.44	0.192	3.31 ± 0.04	3.39 ± 0.03	0.002 **
	RBR2	RBIB	8.95 ± 0.30	9.90 ± 0.31	0.040 *	3.31 ± 0.04	3.50 ± 0.04	0.000 ***
	RBR1	RBIB	9.63 ± 0.44	9.90 ± 0.31	0.600	3.39 ± 0.03	3.50 ± 0.04	0.168
BB	BBAB	BBR2	8.23 ± 0.47	8.60 ± 0.40	0.461	3.60 ± 0.06	3.34 ± 0.03	0.000 ***
	BBAB	BBR1	8.23 ± 0.47	7.64 ± 0.24	0.438	3.60 ± 0.06	3.52 ± 0.06	0.971
	BBAB	BBIB	8.23 ± 0.47	8.42 ± 0.31	0.390	3.60 ± 0.06	3.88 ± 0.08	0.004 **
	BBR2	BBR1	8.60 ± 0.40	7.64 ± 0.24	0.108	3.34 ± 0.03	3.52 ± 0.06	0.000 ***
	BBR2	BBIB	8.60 ± 0.40	8.42 ± 0.31	0.964	3.34 ± 0.03	3.88 ± 0.08	0.000 ***
	BBR1	BBIB	7.64 ± 0.24	8.42 ± 0.31	0.060	3.52 ± 0.06	3.88 ± 0.08	0.001 **

At the blanket bog location, mean concentrations of NH_4-N were also highest in the afforested site (1.82 mg L^{-1}), but BBR1 (0.26 mg L^{-1}) and BBR2 (0.81 mg L^{-1}) had significantly lower ($p < 0.002$, Kruskal-Wallis test) concentrations than BBIB (1.11 mg L^{-1}). PO_4-P concentrations at BBR2 (mean = 0.51 mg L^{-1}) and BBAB (mean = 0.17 mg L^{-1}) were significantly higher ($p < 0.001$, Kruskal-Wallis test) than the other blanket bog sites, yet BBR1 and BBIB both had means of 0.03 mg L^{-1}. The mean pH was highest at BBIB (4.63) and lowest at BBR2 (4.07), which was significantly lower ($p < 0.001$, one-way ANOVA) than for the other sites at this location. EC was significantly higher ($p < 0.001$, Kruskal-Wallis test) at BBAB than any other study site and significantly higher ($p < 0.001$, Kruskal-Wallis test) at BBR1 and BBR2 than BBIB. Mean DOC was highest at BBR2 (74 .2 mg L^{-1}) and lower at BBR1 (46.3 mg L^{-1}), yet a Kruskal-Wallis test showed BBR1 had significantly higher ($p = 0.001$) DOC concentrations than at BBIB (40.7 mg L^{-1}). $SUVA_{254}$ values (20-40 cm depths) at BBIB were significantly higher than all the other blanket bog sites ($p < 0.005$, Kruskal-Wallis test) and the means at the two restored sites were lower than BBAB.

There was a significant difference ($p < 0.001$, Mann-Whitney U test) in DOC concentrations between locations with the mean concentration 31.6% higher at the raised bog. The E4:E6 ratios and $SUVA_{254}$ (Table 3) suggest the blanket bog peat is more humified and aromatic than that of the raised bog ($p < 0.05$, Mann-Whitney U test). Both pH and EC were also significantly higher at the blanket bog location ($p < 0.001$, one-way ANOVA and Mann-Whitney U test, respectively). Tests for the independence of the repeated measurements for the site and sampling month and variations in the porewater chemistry with the different surface features are given in the supplementary information (Tables S1 and S2, respectively).

3.3.1 Variations with sampling depth

At the raised bog, there was a negative correlation between DOC and sampling depth ($r_s = -0.375$, $p < 0.001$, $N = 548$) with significantly higher ($p < 0.001$, Kruskal-Wallis test) concentrations at shallow (20-40 cm) depths. DOC concentrations at shallow depths were significantly higher ($p < 0.001$, Kruskal-Wallis test) in RBR2 than RBIB. Higher PO_4-P concentrations were also observed at RBR2 at shallow depths, and they were more strongly correlated with DOC ($r_s = 0.476$, $p = 0.001$, $N = 46$) than at deeper depths. Positive correlations were observed for $SUVA_{254}$ ($r_s = 0.270$, $p < 0.001$, $N = 403$) and pH ($r_s = 0.207$, $p < 0.001$, $N = 556$) with sampling depth, and greater variability existed between sites at shallow depths. Averaged out for all sampling depths $SUVA_{254}$ was not significantly different between sites, yet at shallow depths, it was significantly higher at RBIB ($p < 0.05$, Kruskal-Wallis test) than the other raised bog sites except RBR1. The mean pH was 0.13 units lower at RBAB than RBIB at shallow depths, with little difference when averaged out for all sampling depths.

At the blanket bog location, the E4:E6 ratio was negatively correlated ($r_s = -0.373$, $p < 0.001$, $N = 432$) and NH_4-N concentrations were positively correlated ($r_s = 0.341$, $p < 0.001$, $N=514$) with sampling depth. Higher PO_4-P concentrations at BBR2 were observed at deeper sampling depths (60-80 cm), and there was a stronger positive correlation between DOC and PO_4-P ($r_s = 0.497$, $p < 0.001$, $N = 79$) than at shallow depths. The mean pH at BBAB was significantly ($p<0.001$, one-way ANOVA) lower than at BBIB at shallow depths (20-40 cm), by 0.59 units, but it was least acidic at the deepest depth (80 cm). At shallow depths, the mean pH at BBR1 was 0.65 units higher than BBR2, whereas BBR1 and BBIB were not significantly different.

3.3.2 Seasonal variability

Figure 6 shows the temporal variations in porewater chemistry at the raised bog location. There was greater seasonal variability in NH_4-N, PO_4-P and DOC concentrations at shallow depths (20-40 cm) than at deeper depths (60-80 cm). Seasonal peaks in the shallow porewater occurred most frequently at the afforested and the restoration sites. Peaks were observed at RBR2 for NH_4-N in July 2019 (4.19 mg L^{-1}), PO_4-P (1.03 mg L^{-1}) in April 2018 and DOC (115.84 mg L^{-1}) in September 2019. Winter peaks at RBR2 were limited to a spike in PO_4-P (0.93 mg L^{-1}) in the final sampling month. Other near-surface porewater peaks were observed at RBIB for pH (5.68) in the first sampling month, and NH_4-N (> 2.5 mg L^{-1}) at RBAB, in the autumn.

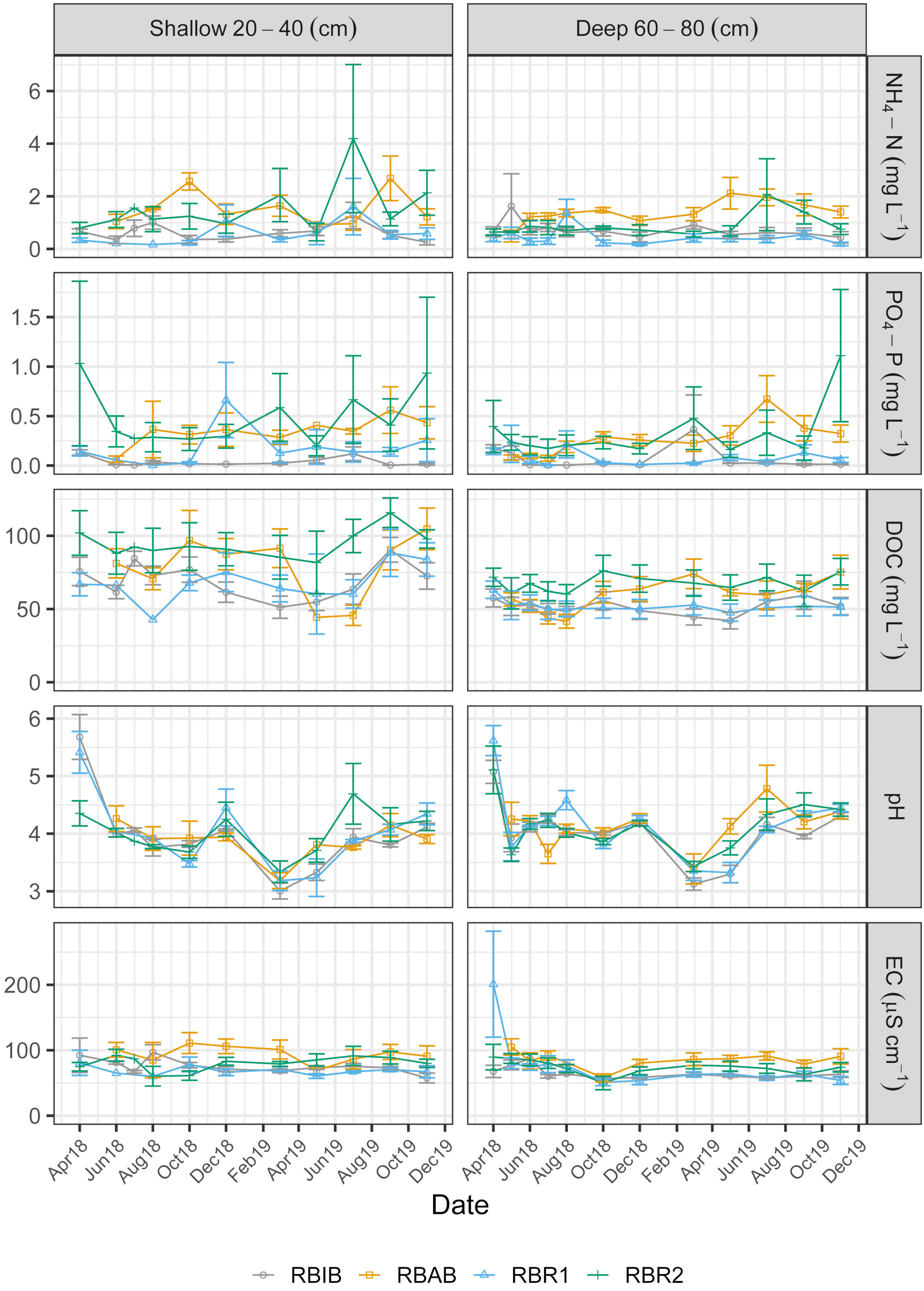

Figure 6 - Time series data of mean porewater concentrations ± SE for shallow and deep porewater for the raised bog location. IB = intact bog; AB = afforested bog; R1 = oldest restoration site; R2 =most recent restoration site.

Although there was less variability in the porewater solutes at deeper sampling depths at the raised bog location, they still displayed some seasonality. A peak in PO_4-P at RBR2, in the shallow porewater, saw a corresponding peak (1.11 mg L^{-1}) in the deeper porewater (November 2019). Peaks in PO_4-P (0.67 mg L^{-1}) and pH (4.78) at RBAB were observed in July 2019. NO_3-N was typically present in low concentrations, but minor peaks were observed, usually in dry periods.

Figure 7 shows the temporal variations in porewater chemistry at the blanket bog location. BBAB experienced two peaks in NH_4-N at shallow depths in March 2019 (1.56 mg L^{-1}) and September 2019 (1.48 mg L^{-1}). At BBR2, NH_4-N peaked (shallow = 0.82 mg L^{-1}; deep = 1.50 mg L^{-1}) at the same time as PO_4-P (shallow = 0.72 mg L^{-1}; deep = 1.05 mg L^{-1}) and DOC (shallow = 101.80; deep = 81.77 mg L^{-1}) in the autumn after the dry summer of 2018. At BBAB, EC fell to 66 µS cm^{-1} in the dry period of July 2018 in the deeper porewater, rising to a peak of 285 µS cm^{-1} in the autumn of 2018 in the shallow porewater. NO_3-N was found in similarly low concentrations to the raised bog, apart from minor peaks usually associated with dry periods.

Except for BBAB, where EC was elevated beyond the other sites, seasonal patterns in pH and EC were similar at the two locations, with greater variability in pH. At both locations, the highest mean pH was recorded in April 2018 and the lowest in March 2019 with similar seasonal trends at both shallow and deeper depths. E4:E6 ratio and $SUVA_{254}$ varied little between sampling dates and are, therefore, not presented.

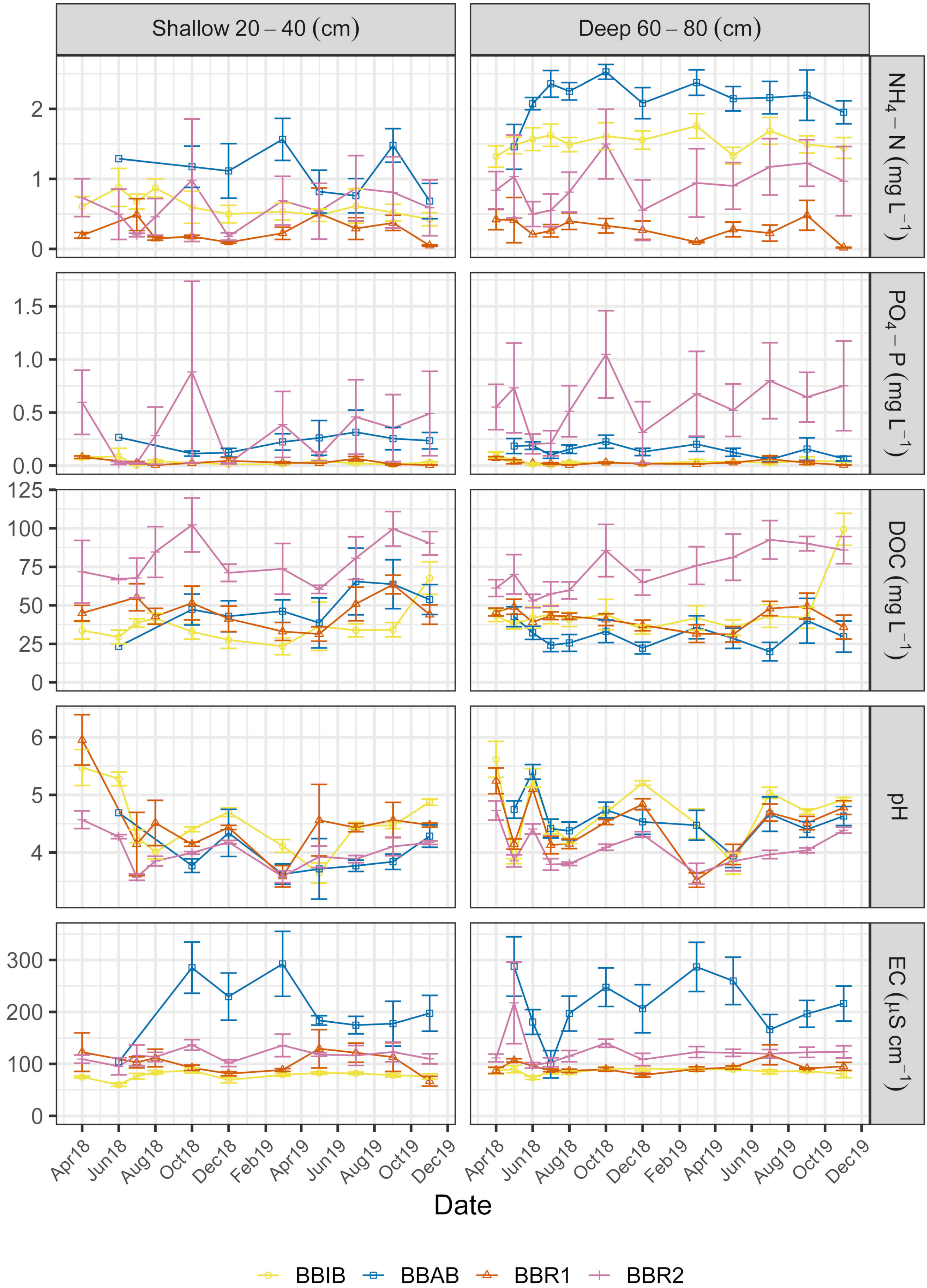

Figure 7 –Time series data of mean porewater concentrations ± SE for shallow and deep porewater for the blanket bog location. IB = intact bog; AB = afforested bog; R1 = oldest restoration site; R2 =most recent restoration site.

3.3.3 Principal component analysis

Principal component analysis (PCA) was used to highlight the associations between variables for the three treatment groups (intact, afforested, restored) at each location (Figure 8). NH_4-N, EC and WTD appeared to be associated with the forestry at both locations, but the associations were stronger at the blanket bog location. High concentrations of PO_4-P, NO_2-N and DOC were more strongly associated with the blanket bog restoration sites, and nests 16, 17 (RBR2) and 40 (BBR2) were consistently outside the normal probability for the clusters of observations. The chemical composition of the porewater at the raised bog restoration sites is closer to the other treatment groups whereas at the blanket bog there is a distinct difference. Overall, PCA highlighted the difference in the spread of porewater observations for the afforested and restored treatment groups with the smallest spread occurring in the intact bog at each location. However, it is important to note that measurement uncertainties can affect the outcome of PCA (Gortler *et al.*, 2020) hence the broader groupings into the three main treatment groups; afforested, intact and restored.

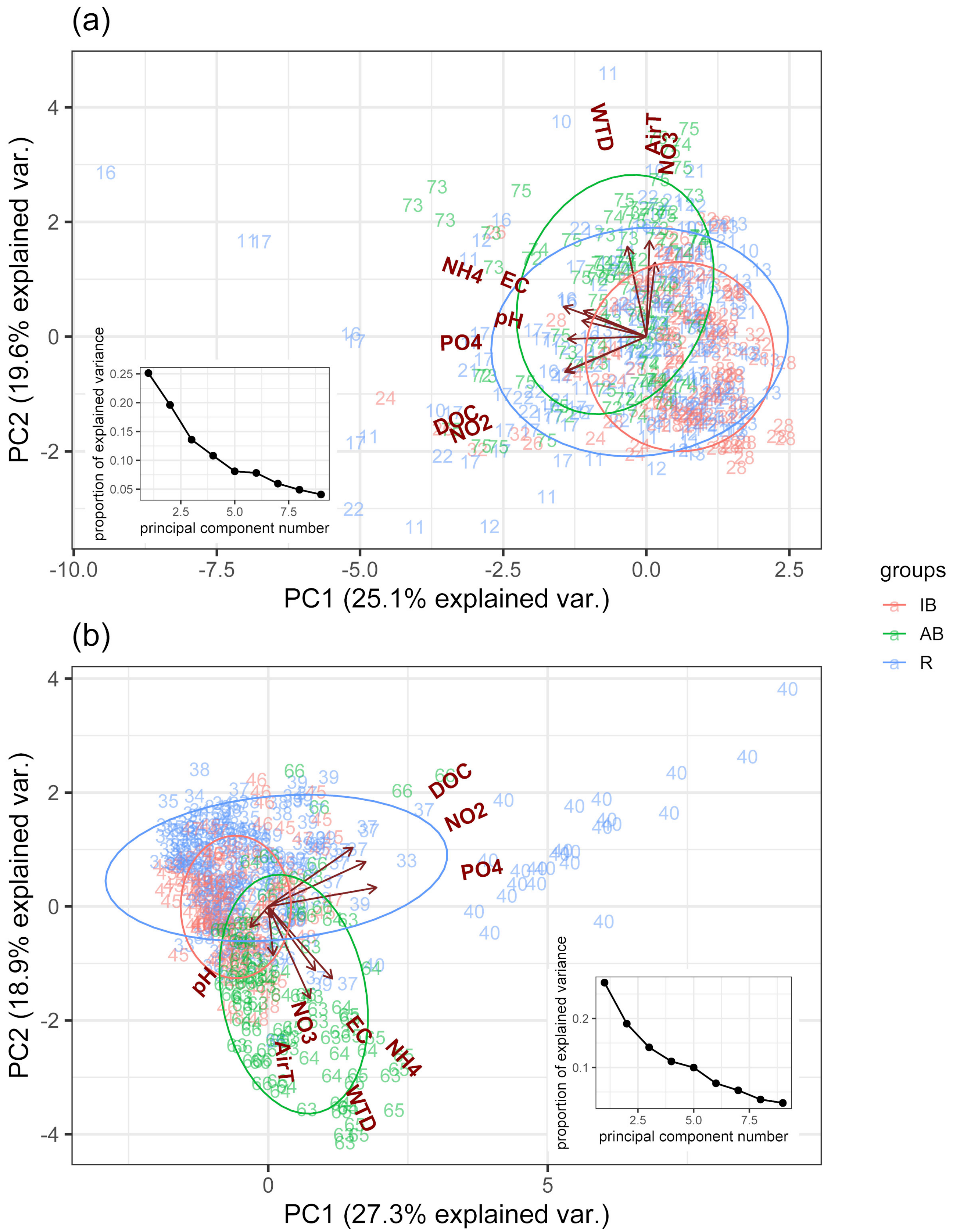

Figure 8 - Biplots of porewater quality at (a) the raised bog and (b) the blanket bog with respect to the first two principal components (PC). The normal range for clusters of observations is denoted by the ellipses (probability 0.68). The proportion of explained variance for each PC is plotted in the embedded scree plots. The observations are plotted with their nest labels. The closer the loadings (arrows) are to the PC axes, and the further they are from the origin indicates a higher spread of data for that PC. Arrows close together are positively correlated; arrows at 90° are uncorrelated, and those at 180° are negatively correlated.

4 Discussion

4.1 Water table

The results show that afforestation was associated with greater water-table drawdown in comparison to the intact bog for both the raised bog and blanket bog locations. The afforested sites also had significantly deeper water tables than the restoration sites. Unlike for the raised bog location, WTD in the blanket bog restoration sites was not significantly different from that in the intact bog, and where both furrows and drains had been blocked at the oldest blanket bog restoration site, we found the mean WTD was shallower than the intact bog (~17 years post-restoration) by 1.2 ± 2.1 cm. Water table recovery after forest-to-bog restoration work has been reported in other studies (Muller *et al.*, 2015, Andersen *et al.*, 2017, Gaffney *et al.*, 2018), but Gaffney *et al.* (2018) and Anderson and Peace (2017) found the recorded levels had not reached near-natural conditions. Gaffney *et al.* (2018) reported the oldest restoration site (17 years post-restoration) where drains and not furrows had been blocked, had a mean WTD that was 8 cm deeper than that of the intact sites. Gaffney *et al.* (2020) also suggest that the effects of drain blocking alone can be quite localised and had previously suggested that furrow blocking may also be required to assist water-table recovery (Gaffney *et al.*, 2018). However, the fact that the mean WTD was similar to the intact bog in both blanket bog restoration sites in this study suggests additional local factors such as slope or microtopography are also likely to be important in controlling water-table recovery following forest-to-bog restoration. It should also be noted that this study was not specifically designed to test for the effects of drain and furrow blocking on water tables.

In dry periods, the water-table drawdown at both locations was much more pronounced in the afforested than the intact bog, and drawdown in the restoration sites was similar to the intact afforested bog. Anderson and Peace (2017) found a slight water-table drawdown in restored sites 5 m outside the former forest edge in dry conditions at a blanket bog location. Our study found higher water-table drawdown in the restoration sites compared to the intact bog during drought periods, but the drawdown effect was less for the oldest restoration sites. Outside of drought conditions, there was less difference between the WTD in the restoration sites and the intact bog, but more difference between the most recently restored sites and those restored earlier.

4.2 Restoration impacts on porewater chemistry

The porewater chemistry and the PCA highlighted that higher concentrations of DOC and PO_4-P were associated with restoration, particularly at the blanket bog location. The mean PO_4-P concentration at BBR2 (0.51 mg L^{-1}) was ~3 times higher than BBAB (0.17 mg L^{-1}) and ~17 times higher than BBR1 (0.03 mg L^{-1}). PO_4-P concentrations were significantly higher at RBR2 than RBIB and RBR1 at all depths. However, at shallow depths (20-40 cm), the mean PO_4-P concentration was 0.17 mg L^{-1} higher at RBR2 than BBR2, both of which were restored at a similar time. At deeper depths, PO_4-P concentrations were higher at BBR2 than RBR2. No mineral deposits were detected in the top 1 m of peat taken from BBR2. Therefore, we are unable to explain why higher concentrations of PO_4-P at BBR2 were detected at deeper sampling depths.

Averaged across both locations, the mean concentrations of PO_4-P at the shallowest depths (20 cm) were ~10 times higher in the afforested (0.33 ± 0.07 mg L^{-1}) than the intact bog (0.03 ± 0.01 mg L^{-1}), which could be due to fallen needles and other forest litter (Moore *et al.*, 2005, Asam *et al.*, 2014a). Historical fertiliser applications may be another potential source of elevated PO_4-P concentrations which have been found to persist in surface waters for up to 10 years (Kenttämies, 1982) and could persist for longer in porewater. However, it might be expected that the trees would have sequestered any excess P, given their relative planting dates (Drinan *et al.*, 2013). PO_4-P concentrations were often low (< 0.1 mg L^{-1}, 67.1% of the time) at most piezometer nests, but high values (> 2 mg L^{-1}) were detected in some nests (i.e. nests 17 and 40) suggesting some local effects.

Nests 17 (RBR2) and 40 (BBR2) were in well-defined furrows providing preferential flow paths for runoff; these furrows contained higher forest biomass, compared to other nests, suggesting that the forest biomass was the major source of PO_4-P. Furthermore, we hypothesise that the accelerated decay of the mulched material at BBR2, which would likely decompose more readily than coarser forest debris, contributed to the higher PO_4-P concentrations at this site.

Some studies on UK and Scandinavian peatlands (Rodgers *et al.*, 2010, Asam *et al.*, 2014b, Kaila *et al.*, 2014, Clarke *et al.*, 2015, Koskinen *et al.*, 2017, Shah and Nisbet, 2019) have suggested increases in porewater and surface water concentrations of PO_4-P, as a result of forestry operations, to be a relatively short-term effect (3–5 years). Others have suggested timeframes > 10 years (Palviainen *et al.*, 2014, Gaffney, 2017, Gaffney *et al.*, 2018). In this study, porewater concentrations of PO_4-P at RBR2 and BBR2 were significantly higher than the intact bog 5-6 years post-restoration. The only restoration site where PO_4-P concentrations were not significantly different from the intact bog was BBR1, 17 years after restoration. Elevated PO_4-P concentrations in the porewater may persist for longer periods than surface water from the catchment outlets studied by Shah and Nisbet (2019) with porewater concentrations at RBR1 70 µg L^{-1} higher than the intact bog, 9 years after restoration. After dilution in surface waters, they are unlikely to be cause for concern, but the higher concentrations detected by Shah and Nisbet (2019) shortly after clear-felling suggest caution should be applied in ecologically sensitive waters (e.g. upstream of freshwater pearl mussel populations or lochs).

Other studies have reported that forest residues are a primary source of organic matter and thus nutrients and DOC (Muller *et al.*, 2015, Shah and Nisbet, 2019). Elevated DOC and PO_4-P concentrations, attributed to forest residues, have been reported in both streamwater (Rodgers *et al.*, 2010, Koskinen *et al.*, 2011, Kaila *et al.*, 2014, O’Driscoll *et al.*, 2014, Räsänen *et al.*, 2014, Koskinen *et al.*, 2017) and porewater (Asam *et al.*, 2014b, Gaffney *et al.*, 2018). Our results found a significant positive correlation between DOC and PO_4-P at RBR2 and BBR2, which was higher at the mulched site (BBR2). Other factors that influence DOC production are WTD, temperature and pH (Clark *et al.*, 2009). Lower water tables through drainage have been found to stimulate enzymes responsible for peat decomposition and increased DOC production (Peacock *et al.*, 2015). Temperature can increase DOC production directly by stimulating microbial activity within the peat (Kane *et al.*, 2014) or indirectly by increased plant productivity (Freeman *et al.*, 2004). The solubility of DOC in soil solution is widely known to increase with increasing pH (Clark *et al.*, 2005). Neither WTD nor pH proved to be a strong control of DOC in this study, although there was more variability in concentrations at shallow depths where most of the fluctuations in WTD occurred, suggesting WTD had some influence. Despite the influence of WTD and pH on DOC, seasonal temperature changes can sometimes be more important (Koehler *et al.*, 2009). At the restoration sites, the DOC may be derived from the above-ground biomass (Don and Kalbitz, 2005) or decomposition of bare peat (Qassim *et al.*, 2014) following clear-felling. Therefore, it is likely the influence of the WTD and pH on DOC at the restoration sites is masked by the input of DOC from the tree litter or surface peat decomposition, particularly after dry periods.

As observed in other peatland restoration studies (Urbanová *et al.*, 2011, Gaffney *et al.*, 2018), the dominant form of inorganic nitrogen was NH_4-N, with low NO_3-N concentrations (Urbanová *et al.*, 2011, Gaffney, 2017, Shah and Nisbet, 2019) at both locations. Other studies have found nitrate leaching to be more of an issue in minerotrophic fens than bogs (Koskinen *et al.*, 2011, Koskinen *et al.*, 2017). Shah and Nisbet (2019) observed modest increases of NO_3-N in streamwater at the raised bog following felling, where porewater NH_4-N would be readily oxidised to NO_3-N (Daniels *et al.*, 2012) in streams, but concentrations never exceeded 0.5 mg L^{-1}.

4.3 Legacy effects from afforestation

Solute concentrations and principal component analysis suggested that water-table drawdown and higher EC and NH_4-N concentrations were associated with afforestation. Average NH_4-N concentrations were 0.83 mg L^{-1} higher at RBAB and 0.71 mg L^{-1} higher at BBAB ($p < 0.001$), where WTD was more drawn down, than the intact bog. Our findings were similar to those of Gaffney *et al.* (2018). However, they found elevated NH_4-N concentrations after restoration persisted for > 17 years. We found concentrations were significantly less in the oldest restored sites at both locations, compared to the intact bog. Therefore, it was not found to be a long-lasting legacy effect in this study. The deeper water table in the afforested bog may increase the mineralisation of organic matter within aerobic peat (Sapek, 2008, Daniels *et al.*, 2012) enhancing porewater concentrations of NH_4-N. Previous studies have found higher concentrations of NH_4-N in peatlands with water-table drawdown and particularly after drainage (Holden *et al.*, 2004, Daniels *et al.*, 2012, Gaffney, 2017, Gaffney *et al.*, 2018). Furthermore, conifer trees have been found to capture 40-60% of atmospheric nitrogen deposition in the canopy (Schulze, 1989, Pryor and Klemm, 2004), and this could, to a certain extent, explain the higher NH_4-N concentrations in the afforested bog.

EC was significantly higher at the blanket bog location, particularly at BBAB, which is likely a consequence of being nearer the coast. Forest scavenging of sea-salts has been reported in other studies (Monteith *et al.*, 2007, Dunford *et al.*, 2012) and surface water from a nearby forest drain was found to contain significantly higher Na^+ and Cl^- concentrations than all the other sites in the study, which supports this argument. Therefore, the significantly higher EC observed at BBR1 and BBR2 compared to the intact bog is likely a legacy effect of sea-salt scavenging from the former forestry. The effect was less noticeable on the raised bog, which was further inland in comparison, but EC was significantly higher at RBAB than the other the raised bog sites.

4.4 Differences between location

Overall, the water-table comparisons between restored, afforested and intact sites between the raised bog and the blanket bog in this study appear similar. However, there was a closer correspondence in water table between the restored and intact blanket bog sites than those at the raised bog, despite the lower annual rainfall at the blanket bog location. Therefore, the fact that the trees were less mature at the blanket bog location, coupled with the blocking of the drains, at both restoration sites, and furrows at BBR1 would likely explain why the water table at the blanket bog restoration sites was more similar to the intact bog. Prolonged water-table drawdown at the raised bog as a result of afforestation may have led to more peat degradation, providing new voids and pathways for flow within the peat, and greater hydrophobicity (Holden and Burt, 2002, Worrall *et al.*, 2007), which could all account for a potentially slower recovery in WTD at the restoration sites at the raised bog compared to at the blanket bog location.

The mean DOC concentration was 15.7 mg L^{-1} higher in the raised bog than the blanket bog. Other studies have reported very high porewater DOC concentrations in raised bogs (Grau-Andrés *et al.*, 2019) which could be due to the higher plant productivity and warmer climatic due to their lowland setting (Freeman *et al.*, 2001, Freeman *et al.*, 2004) or the relative immobility of DOC (Glatzel *et al.*, 2019). Elevation and air temperature differences were not strong, but the blanket bog was significantly cooler ($p < 0.001$, one-way ANOVA) than the raised bog over the study period. Therefore, the higher DOC in the raised bog could be through higher temperatures stimulating plant, microbial and enzyme activity (Kane *et al.*, 2014) and the significantly deeper WTD in the raised bog. DOC may also be less concentrated in the porewater of the blanket bog system due to the greater mobility of solutes (Glatzel *et al.*, 2019). Mean $SUVA_{254}$ values for the sites also suggested DOC was naturally more hydrophobic at the blanket bog location. The E4:E6 ratio and $SUVA_{254}$ values were closer to the intact site at the raised bog location, suggesting DOC quality, in terms of lability and degree of humification, may have been quicker to respond to restoration than at the blanket bog sites. However, these results and lower variability in solute concentrations between the raised bog sites are unexpected, given the greater variability in WTD compared to the blanket bog.

4.5 Implications for management

The results of this study have several important implications for management. However, it is important to note that we did not undertake a before-after time-series approach with each site and to note that the restoration methods differed between the two locations, local environmental conditions may have affected the results, and the sites were restored at different times. Management implications include:

i. For both the raised and blanket bogs, porewater DOC and PO_4-P concentrations were significantly higher at the most recently restored sites than the afforested and the intact bog. The increases in DOC and PO_4-P are most likely related to leaching from forest residues and soil disturbance following clear-felling. Therefore, to ensure forest-to-bog restoration has minimal impact on water quality, we suggest clear felling of the trees is carried out in phases to reduce the likelihood of high peaks in PO_4-P, particularly for large sites.
ii. Given that WTD was most similar to the intact bog in the restored sites where drain and furrow blocking had taken place, and the fact Gaffney *et al.* (2020) reported drain-blocking to have a localised impact on WTD, this suggests that both drains and furrows should be blocked to encourage more rapid water-table recovery for whole forest blocks.
iii. We observed that PO_4-P concentrations were between two and five times higher in porewater taken from furrows and drains in which brash had accumulated, either deliberately as part of the restoration, or naturally, than from other locations at the same sites where it had not. Therefore, we recommend that brash is not allowed to accumulate in furrows or drains and that ideally it is removed from forest-to-bog restoration sites.

iv. Highest concentrations (mean = 0.51 mg L^{-1}) of porewater PO_4-P were observed at BBR2 where the trees had been mulched, and the material spread over the site. Highest porewater DOC concentrations (mean = 74.2 mg L^{-1}) for the blanket bog were also observed at this site. These results suggest that that mulched debris is a major source of water-soluble C and PO_4-P which is leached from drainage waters mixing with the fresh/senescent forest biomass and transferred from the vegetation to the peat and subsequently surface waters (Wickland *et al.*, 2007). However, a focused study on the impacts of mulching with replication would be necessary to fully determine the effects on porewater and surface water.

5 Conclusions

We found significant differences in the WTD and porewater chemistry between intact, afforested, and restored bog sites at both the raised bog and blanket bog locations. Forest-to-bog restoration sites were associated with much shallower water tables than afforested sites, and WTD was closest to near-natural conditions in the blanket bog restoration sites where drain and furrow blocking had taken place. Elevated porewater concentrations of NH_4-N, higher EC and deeper WTD are more associated with the afforested bog at both locations. In contrast, elevated porewater concentrations of PO_4-P and DOC are more associated with the restoration processes and the impact of clear-felling. There were few differences in porewater chemistry between intact bog and the oldest restoration sites in this study. However, PO_4-P concentrations were significantly higher at the raised bog site that had been restored nine years earlier, than in the nearby near-natural bog. For the blanket bog system, DOC concentrations and EC were significantly higher in the site which had been restored seventeen years earlier than the intact bog. Elevated porewater PO_4-P concentrations were found where brash had accumulated in drains and furrows, and where forest materials were mulched on site. Therefore, we recommend that brash is not allowed to accumulate in furrows or drains and that ideally it is removed from restoration sites to reduce the impact of forest-to-bog restoration on downstream water quality.

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8 Supplementary information

Table S1 - Generalised Linear Mixed Model fixed effects for site and sampling month interactions using 'Compound Symmetry' as the covariance type. Subject = unique piezometer identifier; Repeated variable = sampling month.

Target	Transformed	Distribution	Link function	F	df1	df2	Sig.
NH_4-N	Log	Normal	Identity	13.196	8	971	<0.001
PO_4-P	Log	Normal	Identity	13.733	8	958	<0.001
pH	Untransformed	Normal	Identity	13.261	8	1115	<0.001
EC	Log	Normal	Identity	46.072	8	1109	<0.001
DOC	Log	Normal	Identity	10.759	8	1117	<0.001

Table S2 - Porewater chemistry means and standard deviations (SD) for the different afforested and restored surface features (Furrows; Original surface; Ridges), intact bog microforms (Hollows; Hummocks; Lawns) and the land-use type (AB = afforested bog; IB = intact bog; R = restored).

Type	Microform		DOC ($mg\ L^{-1}$)	E4:E6	$SUVA_{254}$ ($L\ mg^{-1}\ m^{-1}$)	PO_4-P ($mg\ L^{-1}$)	NH_4-N ($mg\ L^{-1}$)	pH (pH units)	EC ($\mu S\ cm^{-1}$)
AB	Furrow	Mean	45.30	6.01	3.82	0.15	1.66	4.30	156
		N	95	73	74	80	80	94	94
		SD	32.68	2.20	0.75	0.18	0.83	0.62	74
	Original	Mean	56.77	5.80	3.58	0.25	1.82	4.30	192
		N	110	87	87	98	98	112	109
		SD	27.72	2.28	0.57	0.26	0.82	0.73	126
	Ridge	Mean	58.74	5.82	3.50	0.34	1.15	3.98	70
		N	34	26	26	31	31	34	34
		SD	15.15	0.99	0.39	0.25	0.64	0.40	15
	Total	Mean	52.49	5.88	3.66	0.23	1.66	4.26	160
		N	239	186	187	209	209	240	237
		SD	28.99	2.11	0.64	0.24	0.83	0.65	105
IB	Hollow	Mean	43.73	9.55	3.63	0.06	0.62	4.29	74
		N	86	61	61	86	86	85	85
		SD	19.34	3.00	0.25	0.23	0.54	0.74	16
	Hummock	Mean	55.08	9.17	3.63	0.05	1.12	4.24	75
		N	116	82	82	82	82	113	113
		SD	24.20	2.52	0.45	0.08	0.63	0.62	22
	Lawn	Mean	48.51	8.71	3.63	0.03	0.92	4.43	77
		N	129	90	90	119	119	128	127
		SD	23.38	3.09	0.51	0.04	0.68	0.74	15
	Total	Mean	49.57	9.09	3.63	0.04	0.89	4.33	76
		N	331	233	233	287	287	326	325
		SD	23.07	2.89	0.43	0.14	0.65	0.70	18
R	Furrow	Mean	64.37	7.50	3.47	0.48	0.84	4.13	101
		N	230	162	162	211	211	227	229
		SD	31.40	2.21	0.31	0.84	1.05	0.53	52
	Original	Mean	62.25	8.81	3.53	0.12	0.57	4.31	89
		N	285	206	212	251	249	288	285
		SD	25.71	3.04	0.48	0.26	1.03	0.79	49
	Ridge	Mean	67.26	8.04	3.40	0.10	0.49	4.10	84
		N	61	39	42	46	46	62	61
		SD	18.63	2.18	0.16	0.24	0.41	0.67	22
	Total	Mean	63.63	8.21	3.49	0.27	0.67	4.22	93
		N	576	407	416	508	506	577	575
		SD	27.53	2.72	0.40	0.60	1.01	0.69	48