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Restoration of afforested peatland: effects on pore- and surface-water quality in relation to differing harvesting methods

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Abstract

The restoration of drained and afforested peatlands is carried out by removing trees and blocking forestry drains to reinstate key ecosystem services such as peatland habitat provision to support specialised biodiversity assemblages and carbon storage. Although restoration is a slow process, in the short-term, management interventions result in physical and biogeochemical disturbances, impacting upon pore- and surface-water quality. These impacts may vary depending on the restoration techniques used. Here, we compare the effects of two restoration techniques that both include blocking of main forestry drains but that vary in the amount of tree material removed from afforested sites: stem only harvesting ("standard harvesting", STD) and whole tree harvesting ("enhanced harvesting", ENH). We measured their short-term (0-1 year) effects on pore- and surface-water quality in a replicated BACI study. We assessed pore- and surface-water quality using principal response curves, comparing restoration treatments with afforested and open bog controls.

Our results showed the largest effects in surface- and shallow pore-water after restoration, where DOC, PO_4^{3-}, NH_4^+ and K showed the greatest concentration increases, in both ENH and STD plots, relative to controls. Lesser increases were measured for Al, Ca, Fe and Mn concentrations. Meanwhile, in deep pore-water, water quality changed comparatively little during the study. We also found a post-restoration (mean) rise in water table of 10 cm in both ENH and STD treatments.

In general, ENH plots showed larger concentration increases in surface-water, likely related to the additional disturbance caused during brash removal (i.e. more intensive management). However, concentrations had begun to decline for many parameters by the end of the study, which was attributed to the removal of a major source of nutrients (brash) by this technique. In contrast, STD plots showed greater concentration increases in shallow pore-water, given (on average) a 5 cm shallower water table and with all brash remaining on site. Although longer-term monitoring would be required to test this, our results suggest that there remains merit in removing brash (compared to stem-only harvest) when considering longer-term recovery to bog.

Keywords

water table; blanket bog; whole tree harvest; stem only harvest; DOC; logging residues

1. Introduction

Across Europe and beyond, many peatlands have been managed for forestry, either by afforestation of open peatlands or by drainage to increase natural tree cover (Anderson et al., 2016). These practices have several detrimental effects on the function and ecosystem services of peatlands, including loss of bog vegetation (Laine et al., 1995), declines in breeding bird numbers (Wilson et al., 2014), damage to carbon sequestration capacity (Cannell et al., 1993) and effects on water quality (Nieminen et al., 2017).

Collectively, these detrimental impacts have influenced forestry and peatland management policy and encouraged land managers to restore afforested peatlands (Andersen et al., 2016), which has been ongoing in Europe since the 1980s, but in recent times has recently increased markedly in scale (Anderson et al., 2016). Restoration involves felling or removal of planted trees and blocking of drains created for forestry. The restoration process, while aimed at reviving peatland function (Hancock et al., 2018; Hermans et al., 2019), can however have several short-term ecological/environmental impacts, which can be termed ‘hot moments’ (McClain et al., 2003). These include altered greenhouse gas exchange, hydrology and water quality, due to the physical and biogeochemical disturbances involved (Komulainen et al., 1999; Muller et al., 2015; Anderson and Peace, 2017; Gaffney et al., 2018; Hermans et al., 2018; Shah and Nisbet, 2019). Thus, these short-term altered biogeochemical processes (hot moments) are ecologically important and occur immediately following rewetting at the terrestrial aquatic interface (e.g. the newly rewetted zone; McClain et al., 2003).

In the U.K., the rate of restoration of blanket bogs (drained and afforested during the 1970s and 1980s with non-native conifer species) has reached >500 ha yr^{-1} (Anderson et al., 2016). Here, restoration aims to create a functioning open bog habitat by sufficiently rewetting a restoration area, allowing bog vegetation to recover; often termed “forest-to-bog” restoration (Hancock et al., 2018). Optimal methods of forest-to-bog restoration are developing over time, with the ultimate aim being to find the most cost-effective methods which also achieve good ecosystem recovery (Andersen et al., 2016; Hermans et al., 2019). Current options include different forms of tree harvesting (harvest of stems only, or whole trees); different degrees of drain blocking (blocking of main forestry drains only, or main drains plus plough furrows); and methods altering microtopography (stump flipping and/or surface smoothing) (Andersen et al., 2016; Anderson and Peace, 2017; Artz et al., 2018). However, a key consideration is the tree harvesting required, as this determines the amount of tree material typically left on site, which then subsequently decomposes and releases nutrients (Palviainen et al., 2004). Indeed, conifer harvesting on blanket bogs has been found to rapidly increase nutrient concentrations (including nitrogen and phosphorus) in pore- and surface-water under areas where tree brash (tree tops, branches and needles) is left in-situ (Finnegan et al., 2012; Asam et al., 2014b), while peat rewetting can also increase pore-water dissolved organic carbon (DOC) and iron (Fe) levels (Fenner et al., 2001).

In a long-term chronosequence study, the strongest effects of restoration on pore- and surface-water were found to occur in the first three years following restoration (Gaffney et al., 2018). These changes may in turn affect microbial and vegetation communities (Bragazza et al., 2012; Larmola et al., 2013), likely impacting longer-term functioning in restoration sites (Gaffney et al., 2018; Hancock et al., 2018), and could also produce deleterious impacts downstream in streams, lochs and rivers (Drinan et al., 2013; Gaffney et al., 2021).

With 190 000 ha of deep peat (>50cm) afforested across the UK (Hargreaves et al., 2003) and a similar area in Ireland (Black et al., 2008), there is significant potential for increased restoration in coming decades (Forestry Commission Scotland, 2015; Vanguelova et al., 2018). However, there is currently a lack of knowledge as to how different harvesting methods (for forest-to-bog restoration) affect water quality, particularly in the short-term (0-1 years), where the greatest biogeochemical disturbances can occur (Shah and Nisbet, 2019). Therefore, the aim of this study was to measure the short-term effects (years 0-1) of restoration of drained afforested blanket bogs on pore- and surface-water quality, comparing two different harvesting methods (stem only harvest and whole tree harvest) in relation to unimpacted control sites. We hypothesised that restoration would increase pore- and surface-water concentrations of DOC, nutrients and dissolved metals, with the greatest increases likely in stem only harvest areas, where more material (brash) is left on site.

2. Methods

2.1 Site description

This study was undertaken on the Forsinard Flows National Nature Reserve (Scotland) managed by the Royal Society for the Protection of Birds (RSPB). This area is part of the Flow Country http://www.theflowcountry.org.uk/, widely recognised as one of the most important areas of intact blanket bog in Europe (Lindsay et al., 1988), which includes many designated Special Areas of Conservation (SACs), under European Union designation. On the reserve, there are areas of intact open blanket bog and formerly drained, afforested areas undergoing restoration towards blanket bog (forest-to-bog restoration). Around the reserve, some drained areas with intact plantation forestry remain.

Our main study site was Dyke Forest (58.415, -3.953; lat, long), an area of blanket bog afforested with non-native conifers (Sitka spruce (*Picea sitchensis*) and Lodgepole pine (*Pinus contorta*)) between 1982 and 1989. Here, forest-to-bog restoration commenced in October 2014, across three replicated blocks (DF1, DF2, DF3; Figure 1) utilising two different methods of tree removal: 1) stem only harvesting - where trees were felled and stems harvested but brash (tree tops and branches) was stripped and arranged in "mats" (for machinery to drive on) and left in-situ (hereafter *standard harvesting*); and 2) whole tree harvesting, where standard harvesting took place but brash mats were also later removed (hereafter *enhanced harvesting*). Each block (DF1, DF2, DF3) contained two plots restored by *standard harvesting* and two restored by *enhanced harvesting*. Additionally, in DF1 and DF3, some areas (38% and 18%, respectively) within the *standard harvesting* plots underwent whole tree mulching, where trees too small to yield timber value were simply mulched in-situ. For the purposes of this study, these areas were also considered as *standard harvesting* (i.e., very similar to the situation when brash alone is left in-situ). Prior to tree removal, all main forestry drains were blocked (using plastic piling dams 2 m apart in triplicate) prior to the drain entering a water course. As an additional measure, to retain sediment, Hytex® geotextile silt traps were also installed upstream of dams.

Each of these blocks were under standing forestry at the beginning of this study (May 2013). Restoration in DF1 and DF3 commenced (with tree removal) in October 2014, and in March 2015 for DF2. All felling and harvesting was complete by August 2015.

The sampling design here also included three standing forest control plots ("afforested controls") and three open bog control plots. Afforested controls were similar in nature to the restoration areas, in terms of peat depth, slope, drainage pattern, tree species and age while open bog controls represented comparable examples of intact blanket bog from the region (Table S1). Hill drains (spaced ≥25 m apart, and blocked in 1996 and 2004), were present in some open bog controls. These sites were typical of near natural bogs in the region and are designated SAC blanket bog sites. *Sphagnum* spp. dominated open bog control plots, while other mosses and vascular plants were dominant ground flora in afforested and restoration plots.

During the study period (2013-2015), mean annual temperature was 6.8°C, 8.9°C and 7.3°C and annual precipitation was 848 mm, 1059 mm and 719 mm in the years 2013, 2014 and 2015 respectively, (measured with a Davis Vantage Pro2Plus weather station, situated close to site BOG2). This compares to the 1981–2010 long term average of 11.4°C and 970.5 mm for Kinbrace weather station (~ 20 km distance; Met Office, 2021). All study areas were situated between 130 m and 180 m altitude with a mean slope of between 1.5 and 3.2 degrees.

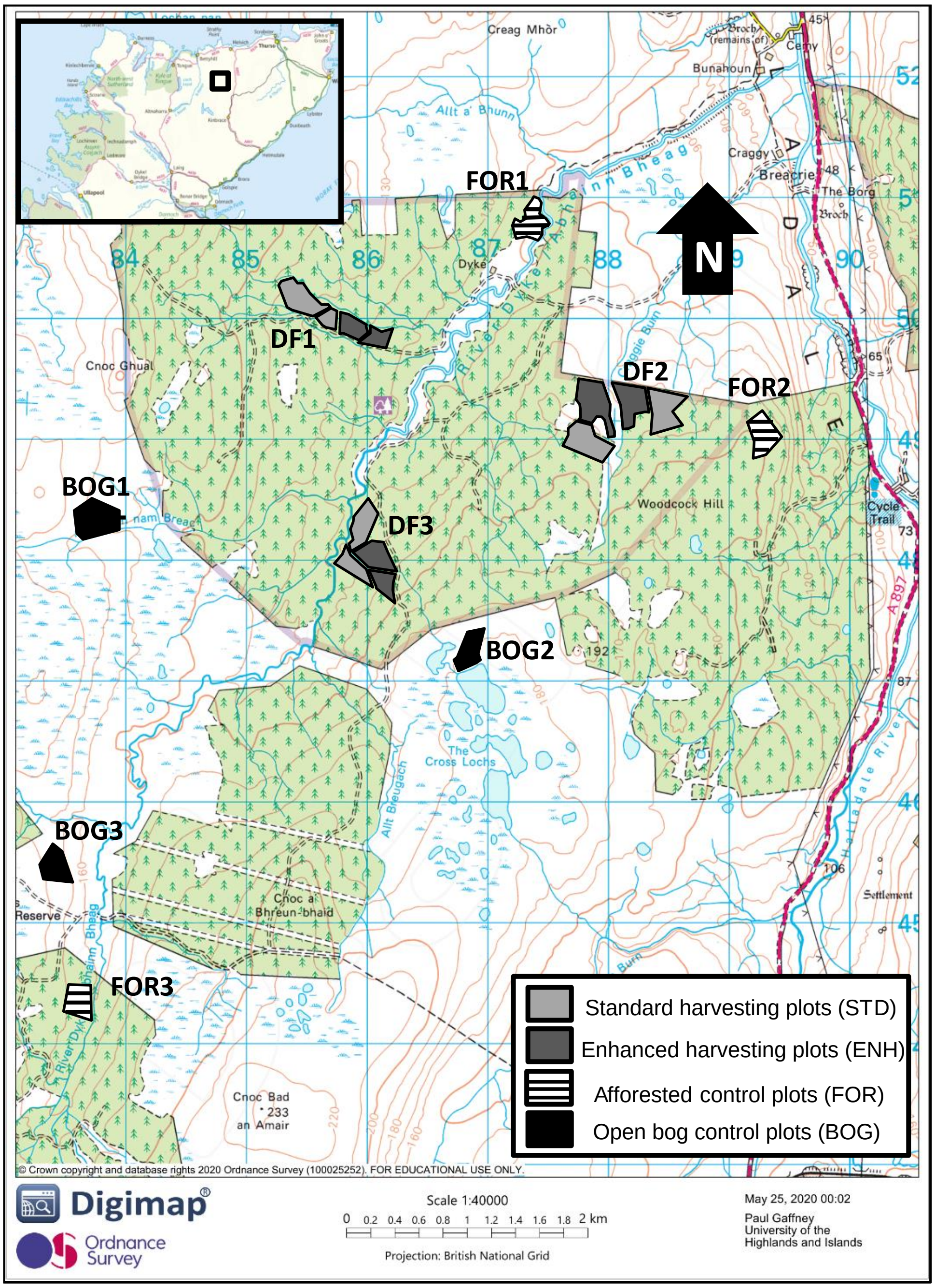

Figure 1: Forest-to-bog restoration and control plots for Dyke Forest (DF), where pore- and surface-water were monitored. Light and dark grey plots were *standard harvesting* (STD) and *enhanced harvesting* (ENH) restoration areas respectively (felled/harvested between October 2014 and August 2015) – and each restoration block (DF1, DF2, DF3) contained two STD and two ENH plots (see section 2.1). Hashed plots were afforested controls (FOR) and black plots were open bog controls (BOG). This map shows the maximum extent of forestry in the area; since this study took place large areas have been removed for restoration, which is ongoing.

2.2 Pore- and surface-water sampling

Pore-water was sampled using piezometers (polypropylene piping), incorporating a drilled 10 cm sampling zone (Wallage et al., 2006). One transect of four piezometer 'pairs', was installed in the direction of the slope (from upslope to downslope) in each plot (Table S1). For each pair, one piezometer was shallow (sampling a 10 cm zone between 10 and 45 cm below the surface), and one was deep (sampling a 10 cm zone between 50 and 80 cm below the surface) (Gaffney et al., 2018). Shallow and deep piezometers were intended to sample the temporally aerated and waterlogged peat zone respectively at each site. Each transect also included one dipwell - a fully drilled pipe to measure water table level (WTL), which was measured manually on each sampling occasion.

Pore-water sampling commenced in May 2013 and took place at six-week intervals until October 2013 (pre-restoration). From October 2013 to December 2015, sampling frequency reduced to every three-four months, with a seven-month gap in monitoring during the main restoration period (as access was not permitted when machinery was operating; Table S1). In total, there were 10 pore-water sampling rounds, six before restoration management and four afterwards.

Piezometers were first emptied four-six days prior to water quality sample collection, to allow fresh pore-water recharge (Strack et al., 2008). Samples were collected into LDPE bottles (Nalgene) using a syringe connected to flexible PVC tubing pre-rinsed with 5 mL of sample. On some occasions (particularly during dry summer months pre-restoration), there was no water available in certain shallow or deep piezometers, as water tables were lower than the sampling zone.

Surface-water was sampled from main forestry drains in restoration and afforested control plots and from hill drains (blocked in 1996 and 2004) in open bog control plots. In general, one drain sample was collected from each plot, but, where more than one forestry drain crossed a piezometer transect, these were also sampled. In open bog and afforested controls, surface runoff was also sampled using crest tubes (Wallage et al., 2006). Surface-water was sampled from October 2013-December 2015, at the same time as pore-waters and in total there were seven surface-water sampling rounds. For both pore- and surface-water samples, temperature, pH and conductivity were measured in the field (using a Hanna HI 991300 multiparameter probe).

2.3 Sample preparation and analysis

Water samples were refrigerated at 4°C on return to the laboratory and then vacuum-filtered, normally within 24 hours of collection (and always within 36 hours). For dissolved organic carbon (DOC) and dissolved inorganic carbon (DIC) analysis, filtration was through pre-combusted 0.7 µm glass-fibre filters (Fisherbrand MF300) followed by high-temperature catalytic combustion (Shimadzu TOC-L; Sugimura and Suzuki, 1988). For nutrients (dissolved ammonium (NH_4^+), soluble reactive phosphate (PO_4^{3-}) and total oxidised nitrogen (TON; the sum of nitrate plus nitrite) and elements (Ca, Mg, Na, K, Fe, Mn, Al, S, and Zn), samples were vacuum-filtered through 0.45 µm cellulose acetate filters (Sartorius Stedim). Nutrient analysis was performed immediately using a Seal AQ2 discrete analyser, following standard methods adapted from ISO international water quality standards (http://www.seal-analytical.com/). For macro- and trace-element analysis, filtered samples were acidified (to 5% using trace metal grade concentrated nitric acid) and then analysed using inductively coupled plasma optical emission spectrophotometry (ICP-OES; Varian 720ES; Clesceri et al., 1998). Certified reference materials (CRMs; MERCK nitrate 200 mg L^{-1}, Fluka PO_4^{3-} and NH_4^+ 1000 mg L^{-1} and the multi-element environmental CRM MAURI-09 (river water) Lot #913, Environment Canada) were used to validate each method and analytical run, with recoveries of certified parameters in the range of 81% to 104%.

2.4 Statistical analyses

All statistical analyses were performed using RStudio (Version 0.98.501, R Core Team, 2016). Each plot was assigned to a treatment: open bog control (BOG), afforested control (FOR), restoration *standard harvesting* (STD) or restoration *enhanced harvesting* (ENH). There were six replicates of STD and ENH treatments (three blocks, each with two STD and two ENH plots) and three replicates of FOR and BOG control plots. Each plot contained four piezometer pairs, which were considered repeated measures in space, therefore, spatial replication was at the whole-plot level. For surface-water samples, results from drain and crest tube samples were combined. Additionally, the dataset was broken into pre-restoration (May 2013 to October 2014) and post-restoration periods (restoration commencement date to end of study; October 2014 to December 2015).

For water table level (WTL), the STD and ENH plots were compared with FOR and BOG controls, in a before-after-control-impact (BACI) design (Stewart-Oaten et al., 1986), using a linear mixed model (function *lme*, package *nlme*, Pinheiro et al., 2016). Treatment and period (i.e., pre- or post-restoration) were fixed factors, along with the treatment×period interaction. To account for temporal autocorrelation (from repeated measures) at the season level, “sampling season” (e.g., Spring 2014) was added as a random slope and “plot” was added as a random intercept, to account for correlation between the repeated measures within plots, i.e., within each piezometer transect. If the model interaction term was significant (i.e., if the single p-test for the treatment×period interaction was $p < 0.05$), a Tukey-adjusted post-hoc comparison (function *lsmeans*, package *lsmeans*, Lenth, 2016) was performed to determine if treatments were significantly different between the pre- and post-restoration periods. Appropriateness of model fit was checked visually by assessing normality and homoscedasticity of residuals (Zuur et al., 2011).

To look at overall changes in water chemistry (in a holistic way) by treatment over time, we used principal response curves (PRCs; package *vegan*, Oksanen et al., 2016). PRCs are a type of redundancy analysis, where the response variable (a combined measure of water quality, based on the major axes of variation in $\log_{10}$ transformed water chemistry data) for the treatments can be compared to a reference (control), producing a graphical output - the response curve (van den Brink and Ter Braak, 1998, 1999). Here, the open bog control sites (BOG) were the reference to which other treatments were compared. The PRC summarised the trajectory of combined water quality measures (time×treatment interaction), in relation to the reference, and displayed the strength of association of the various water chemistry variables with the overall response trajectory by giving each variable a score (van den Brink and Ter Braak, 1999). The significance of the differences among principal response curves was tested using Monte Carlo permutations (n = 999). Water chemistry variables with an absolute score value greater than 0.5 on the principle response (y-) axis were considered to influence the overall temporal trend (van den Brink and Ter Braak, 1999).

After carrying out PRC analysis, which is in effect a single hypothesis test of the treatment by time interaction across all water quality variables (for each depth; surface-water, shallow and deep pore-water), we then examined individual water quality variable trends graphically. We present, and discuss, any variables which showed an effect due to restoration, defined as a marked concentration change (increase or decrease) following restoration in the STD or ENH plots, outside the range of the FOR and BOG controls (Gaffney et al., 2018). Parameters which showed no change following restoration are presented in the Supplementary Information. We also calculated some simple descriptive statistics to quantify post-restoration concentration changes in the STD or ENH plots (Table S2, S3). The ratio of the mean post-restoration to mean pre-restoration concentration for each parameter (RE, or restoration effect), was calculated for BOG, FOR, STD and ENH individually. We then calculated a mean RE for the controls (i.e., mean RE of FOR and BOG, as the REc). Finally the RE for each of the restoration treatments (STD and ENH) was divided by the REc, which gave a concentration change factor following restoration in the STD and ENH treatments relative to the controls.

3. Results

3.1 Overall temporal patterns in water chemistry between treatments

Overall water chemistry in the afforested controls (FOR), enhanced harvesting sites (ENH) and standard harvesting sites (STD), changed significantly over time as compared to the open bog control sites (BOG), in terms of surface-water (F = 95.2, p = 0.001; Figure 2a), shallow pore-water (F = 59.1, p = 0.001; Figure 2b) and deep pore-water (F = 50.2, p = 0.001; Figure 2c). The surface- and shallow pore-water results indicated that water chemistry in restoration treatments (ENH and STD) diverged from that of the controls (BOG and FOR) post-restoration (Figure 2). In deep pore-water, there was comparatively little change pre- and post-restoration; treatments were different across the whole study (Figure 2). Additionally, the relative amounts of variation explained by the first axis of the PRC curve for the treatment×time interaction differed between the depths; 11% in deep pore-water against 20% in shallow pore-water and 28% in surface-water.

For surface-water, in the pre-restoration period, the ENH and STD plots were similar to the FOR plots, but different from the BOG reference (Figure 2a). Post-restoration, from June 2015, the ENH and STD treatments moved further away from the BOG reference (and diverged from the FOR control), as concentrations of certain parameters (particularly PO_4^{3-}, Al, K and DOC – the top scoring parameters) increased following restoration management. Towards the end of the monitoring period, concentrations relative to the BOG were decreasing again in the ENH treatment but still appeared to be rising in STD plots. Meanwhile, the water chemistry response curve of the FOR controls showed little change.

In shallow pore-water, the FOR, STD and ENH plots consistently showed higher PO_4^{3-}, NH_4^+, Al, Ca and DOC than the BOG reference (Figure 2b). However post-restoration, from September 2015 concentrations (of these parameters) increased in ENH and STD plots, remaining elevated until the end of the study. Meanwhile, as for surface water, the FOR controls showed little change.

In deep pore-water, water chemistry changes were less pronounced throughout the monitoring period. The FOR, ENH and STD treatments were all similar (albeit different to the BOG control), except in June 2015 (post-restoration period) when PO_4^{3-}, NH_4^+, S, Ca and Mg was elevated in STD and ENH plots compared to both FOR and BOG controls. After June 2015, FOR, STD and ENH curves converged again.

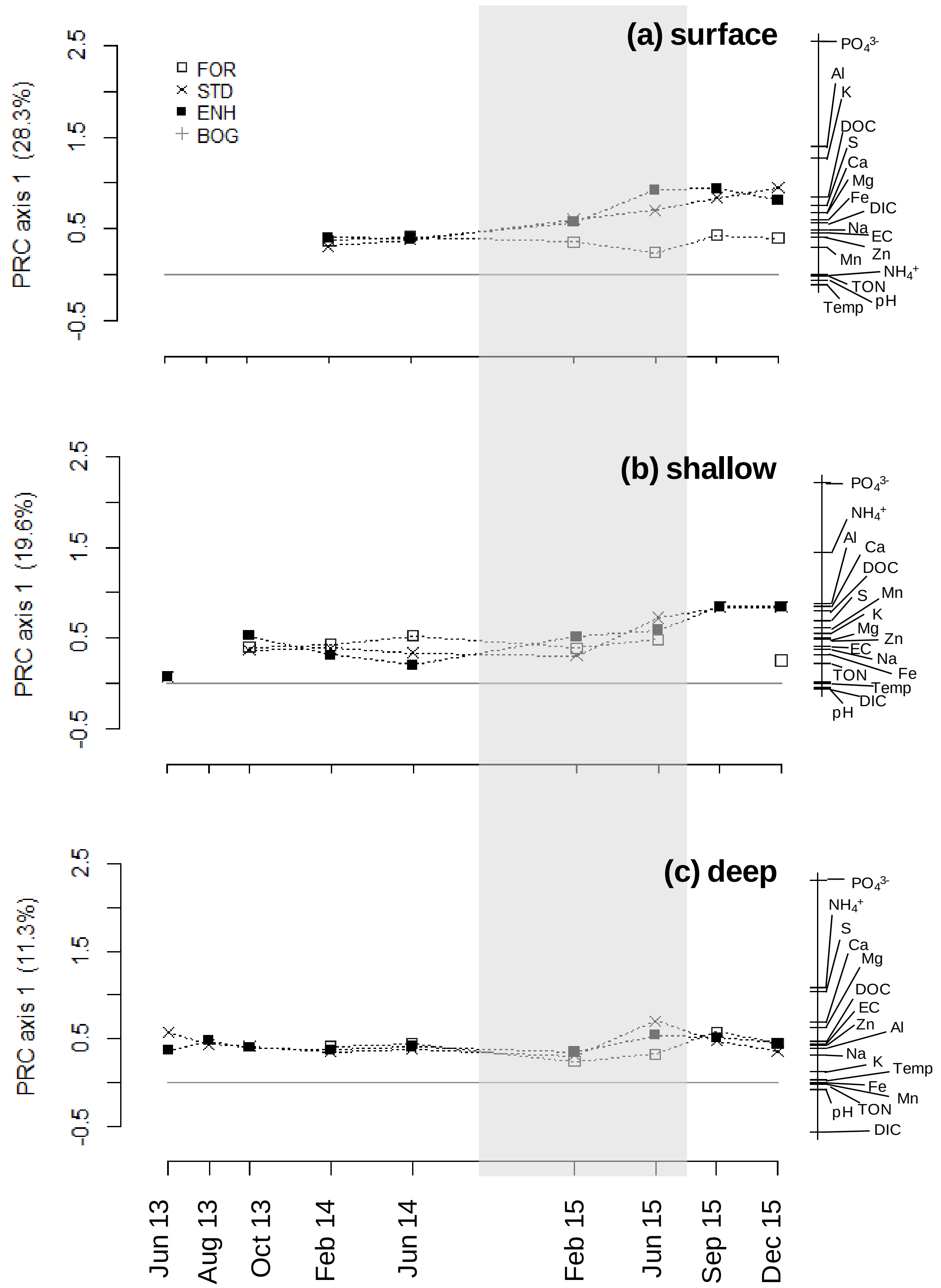

Figure 2: Principal response curves (PRCs) for water chemistry for (a) surface-water, (b) shallow pore-water and (c) deep pore-water. The time series on each panel represents overall deviation from the reference open bog control plots (BOG), for the other treatments (afforested control

plots (FOR), enhanced harvesting plots (ENH), and standard harvesting plots (STD)). This is expressed as a canonical coefficient on the first principal component axis (PC1), in comparison with the reference BOG control plots – represented by the zero line. The right hand axis of each panel shows canonical coefficients for all the chemical variables interpreted. A more positive coefficient shows a stronger relationship with the curve, while a more negative coefficient suggests the opposite trend to the curve. The grey panels show the period when forest-to-bog restoration management was occurring (October 2014 to August 2015). Some treatments have missing data points, where there was insufficient sample volume to measure all parameters.

3.2 Effects of forest-to-bog restoration on water table

There was a mean rise in WTL post-restoration of 10.2 cm in ENH and 9.8 cm in STD plots, compared to 5.0 cm in BOG controls, and 0.6 cm in FOR controls (Figure 3). There was a significant treatment×time interaction for WTL (F = 2.78, p = 0.041), whereby WTL was significantly shallower in ENH plots post-restoration, but this was not the case in the other treatments. Once felling and restoration management commenced, the restoration treatments tended to have shallower water tables than the FOR controls, contrasting with the pre-restoration-period; meanwhile, BOG controls generally had water tables tens of centimetres shallower than restoration plots or forest controls (Figure 3).

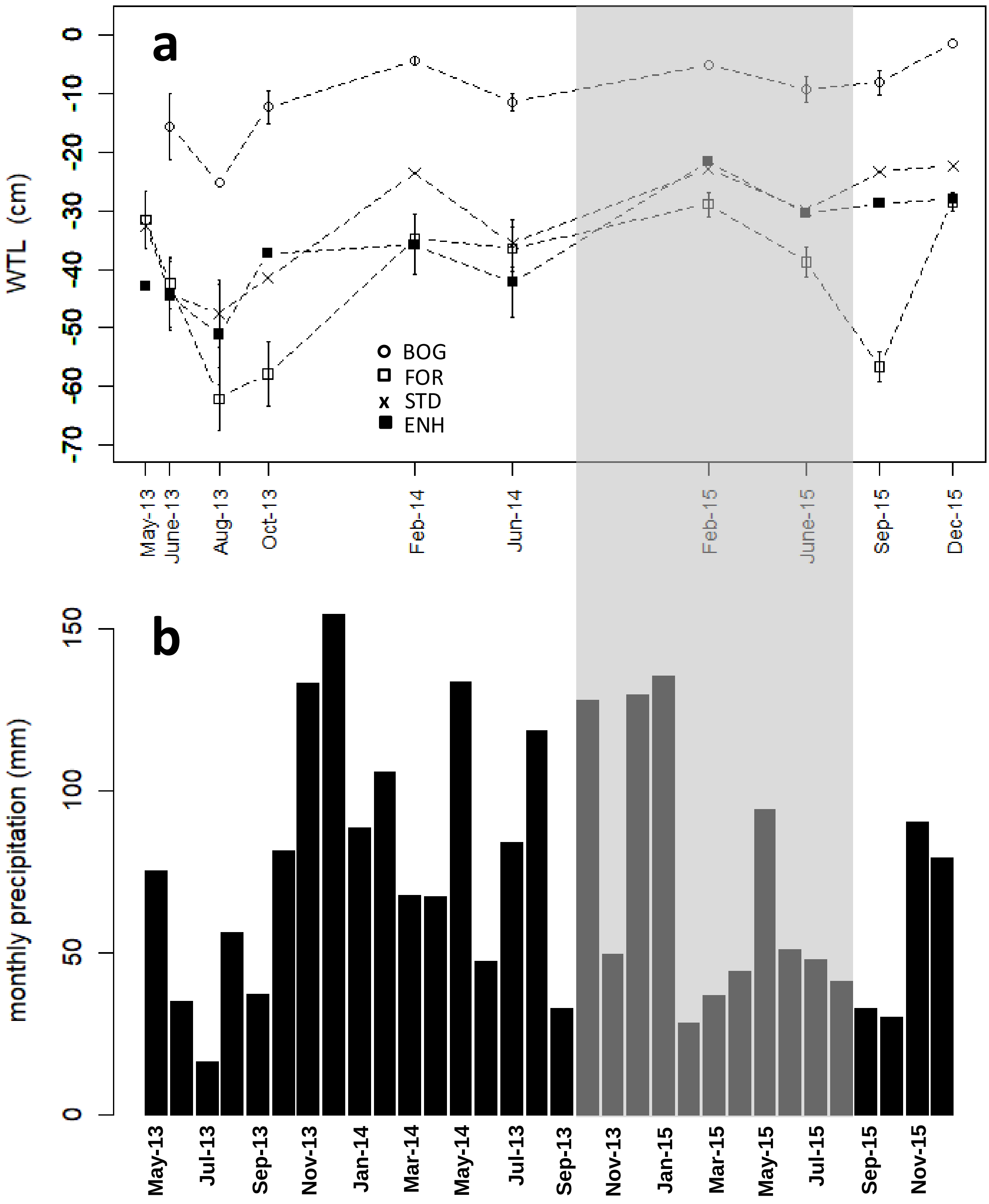

Figure 3: (a) Time series of mean water table level (WTL) ± SE from May 2013 to December 2015 (measured manually) for each treatment per sampling round. (b) Monthly precipitation totals from May 2013 to December 2015, measured close to site BOG2. Grey shading represents period of forest-to-bog restoration (Oct 14 to Aug 15).

3.3 Effects of forest-to-bog restoration on individual water chemistry variables

Concentrations of DOC, NH_4^+, PO_4^{3-}, K, Ca, Al, Fe and Mn showed changes due to restoration, with at least a doubling in concentration in STD and ENH plots, in comparison to any changes in FOR and BOG controls (section 2.4, Table S2, S3). In general, our results showed the greatest concentration increases in surface water, although DOC and NH_4^+, PO_4^{3-} and K, also exhibited strong increases in shallow pore-water, while deep pore-water showed smaller changes.

3.3.1 DOC, NH_4^+, PO_4^{3-} and K

DOC concentrations increased post-restoration, in both STD and ENH plots at all three water depths (Figure 4a). The DOC increases in surface- and shallow pore-waters (2.0-4.6 fold) occurred from June 2015, wherein mean concentrations peaked at ~150 mg L^{-1} in ENH plots and slightly less in STD plots in September 2015 (Figure 4a). By December 2015, mean DOC concentrations decreased slightly. In deep pore-water mean concentrations increased (1.9-fold) in restoration sites to ~75 mg L^{-1} post-restoration and remained at this level (Figure 4a).

NH_4^+ also increased in shallow pore-water and surface-water following restoration, relative to FOR and BOG controls (Figure 4b). In surface-water this increase was mainly in ENH plots (a 13-fold increase), while in shallow pore-water both ENH and STD plots increased (3.7- and 15-fold, respectively). In deep pore-water, NH_4^+ concentrations were higher than in shallow and surface-water, and FOR sites had more than double that of other treatments at times (Figure 4b). However, there were no obvious changes in deep pore-water NH_4^+ concentrations following restoration.

Following restoration, PO_4^{3-} concentrations increased markedly in surface-water and shallow pore-water in both ENH and STD treatments from June 2015 (Figure 5a). In surface-water PO_4^{3-} concentrations increased approximately 25-fold in STD plots and 100-fold in ENH plots post-restoration, with the highest mean concentrations here. In shallow pore-water, PO_4^{3-} concentrations increased by 7- and 40-fold in ENH and STD plots, respectively (Figure 5a). While STD plots had higher mean shallow pore-water PO_4^{3-} concentrations post-restoration, concentrations decreased here slightly by December 2015, while they continued to increase in ENH plots. In deep pore-water, concentrations increased following restoration but just 3-fold in ENH plots and 1.7-fold in STD plots.

For K, concentration increases were much less pronounced in pore-water compared to surface-water (Figure 5b). In surface water, mean concentrations increased 7- and 10-fold in STD and ENH plots respectively, while in shallow pore-water, concentrations increased 4-fold, with slightly higher concentrations in STD plots. There appeared to be no effects of restoration on deep pore-water K.

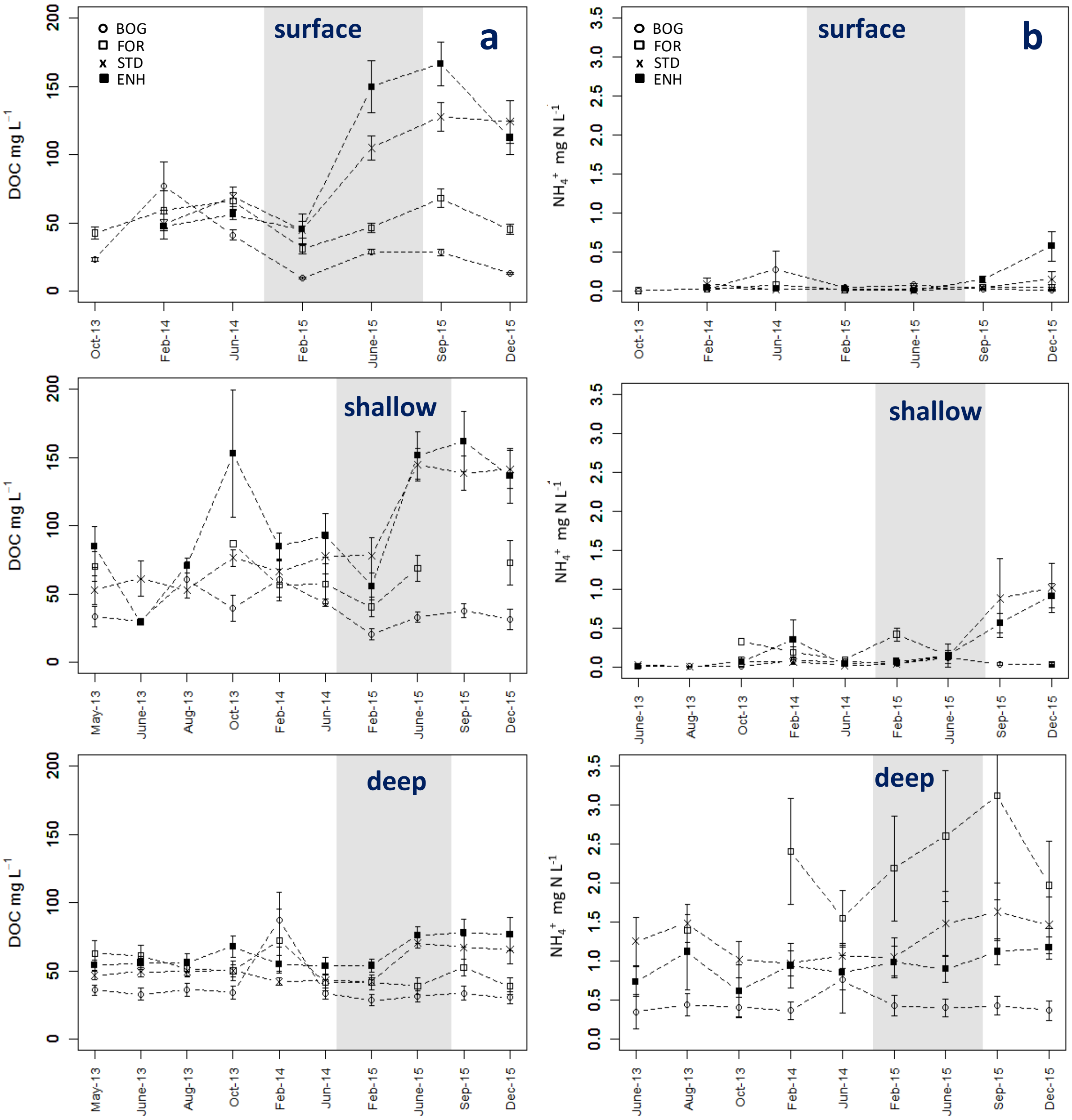

Figure 4: Time series of (a) dissolved organic carbon (DOC) and (b) ammonium (NH_4^+) in surface-water (top panels) from October 2013 to December 2015, shallow pore-water (middle) and deep pore-water (bottom) from May/June 2013 to December 2015. Plots are mean values ± SE for each treatment per sampling round. Grey shading represents period of forest-to-bog restoration (Oct 14 to Aug 15).

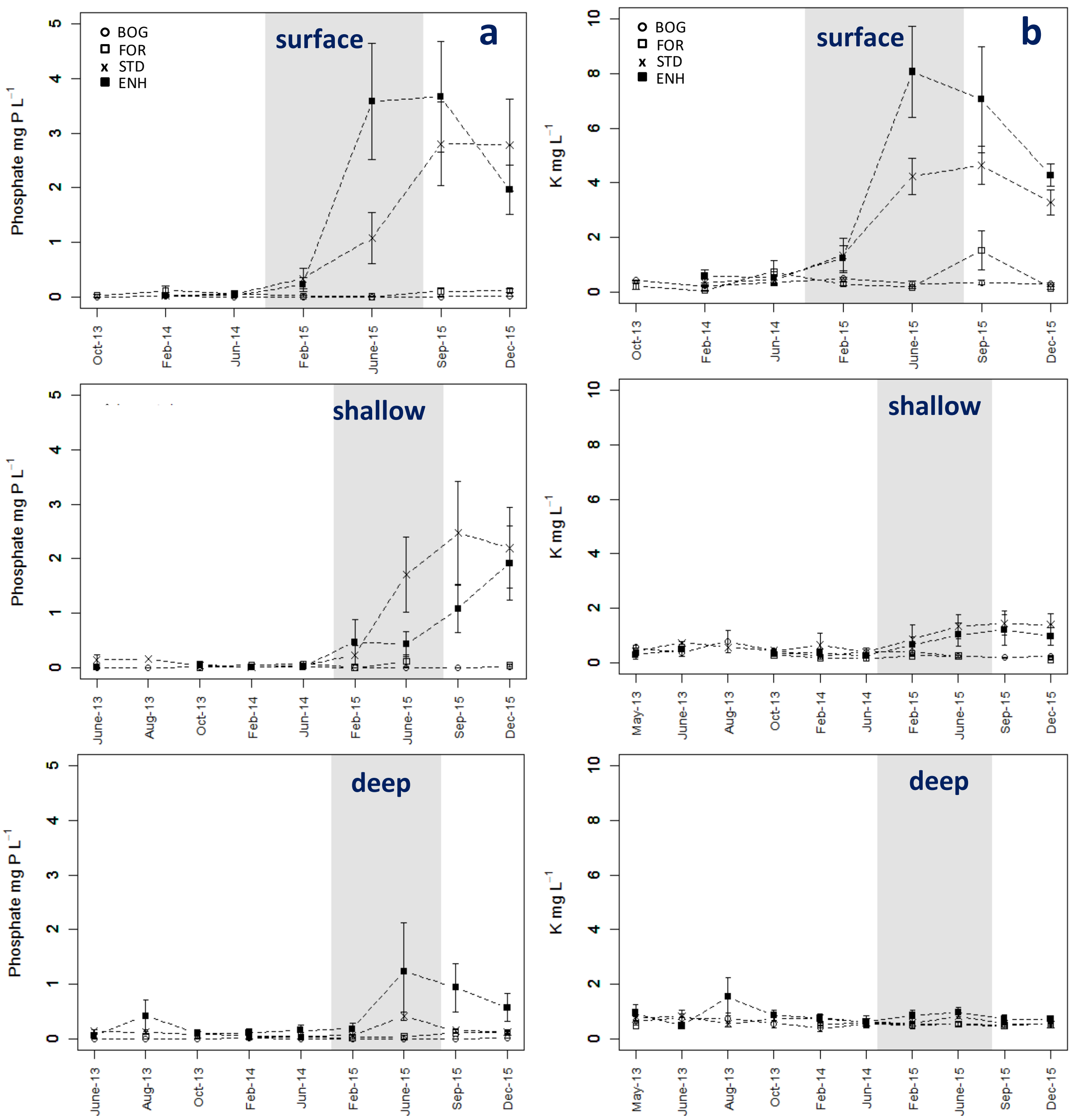

Figure 5: Time series of (a) phosphate (PO_4^{3-}) and (b) potassium (K) in surface-water (top panels) from October 2013 to December 2015, shallow pore-water (middle) and deep pore-water (bottom) from May/June 2013 to December 2015. Plots are mean values ± SE for each treatment per sampling round. Grey shading represents period of forest-to-bog restoration (Oct 14 to Aug 15).

3.2.2 Metals, pH and conductivity

In surface-water, Ca concentrations increased 1.7-fold post-restoration with the highest concentrations in ENH plots (Figure 6a). In shallow and deep pore-water there were smaller concentration changes in restoration plots (≤1.6-fold). For Al, concentrations increased post-restoration in STD and ENH in surface water (by 2.3 and 3-fold respectively), and in shallow-pore water (by 1.9 and 2-fold respectively; Figure 6b). Fe and Mn each showed concentration increases (up to 4- and 12-fold respectively) post restoration in surface water, which were greatest in ENH plots (Figure S10, S11). In shallow pore-water, Fe increased 2-fold in STD plots, while Mn increased 2-fold in ENH plots.

Meanwhile, pH and conductivity changes associated with restoration were relatively minor (Table S2, Figure S1, S2) and other measured parameters, including TON showed no notable effects of restoration in surface- or pore-water (see supplementary information).

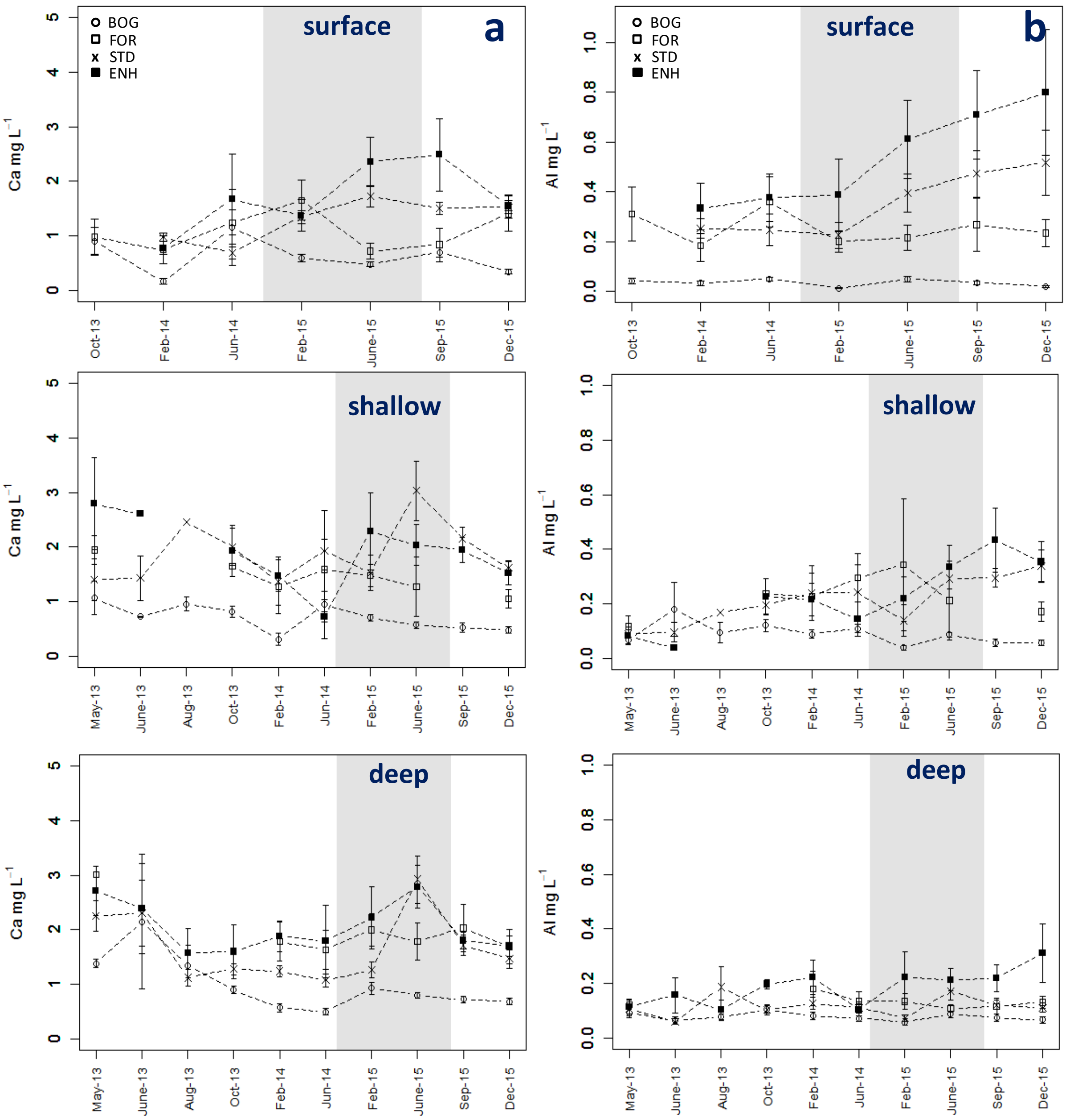

Figure 1: Time series of (a) calcium (Ca) and (b) aluminium (Al) in surface-water from October 2013 to December 2015 and shallow pore-water (middle) and deep pore-water (bottom) from May 2013 to December 2015. Plots are mean vales for each treatment ± SE. Grey shading represents period of forest-to-bog restoration (Oct 14 to Aug 15).

4 Discussion

The water chemistry in STD and ENH plots was significantly different to BOG controls over the monitoring period, and following forest-to-bog restoration, there was a concentration increase in several chemical parameters associated with both restoration treatments. We observed the largest effects in surface- and shallow pore-water post restoration, wherein DOC, PO_4^{3-}, NH_4^+ and K showed the greatest increases relative to BOG and FOR controls. Meanwhile, in deep pore-water, restoration resulted in comparatively little change in chemical parameters over time.

Post-restoration, in surface- and shallow pore-water, there was also a seasonal influence on the measured water quality restoration effects, which has also been found by others (Shah and Nisbet, 2019; Gaffney et al., 2021). In our study, this was especially evident in ENH plots, where post-restoration, concentrations of many parameters (e.g. DOC, $PO_4^{3-}$and K) reached a maximum in the June or September sampling rounds and then often subsequently decreased by the end of the growing season (in December). This coincides with higher summer air temperatures and thus decomposition rates (Davidson and Janssens, 2006), which would normally influence peatland water chemistry (Wieder, 1985; Proctor, 2006).

One of the key drivers influencing pore-water chemistry is water table level (Haapalehto et al., 2014). A rise in WTL will in particular affect DOC and associated metals such as Fe, along with PO_4^{3-} concentrations, by release from peat both through biotic (enzyme activity) and abiotic processes (desorption, increased solubility) (Fenner et al., 2001; Clark et al., 2012; Kaila et al., 2014). The (10 cm) rise in WTL observed here (following restoration) showed that tree removal and drain blocking was effective in retaining more water in the restoration areas, as found by other studies (Gaffney et al., 2020b). This WTL rise (in both ENH and STD plots) occurred from the first post-restoration sampling round (February 15), and thus potentially affected pore- and surface-water chemistry immediately. This rapid rewetting may have been favoured by a relatively average autumn and winter rainfall during the sampling period (470 mm of rainfall, Oct 2014-Feb 2015, 1981-2010 average at nearby station: 506mm). The same management occurring during drier periods may not have had the same rapid effect on WTL.

Here, DOC was one of the water chemistry variables most strongly affected by restoration, with increasing concentrations (ranging from 1.8 to 4.7-fold) at all depths sampled in both STD and ENH treatments (relative to controls). Pore-water DOC concentrations have been found to respond strongly to water table increases following peatland restoration, with some very rapid (~1 month) and large concentration increases occurring (>2-fold; Fenner et al., 2011; Clark et al., 2012). This may be related to a stimulation of microbial processes upon rewetting (of peat previously above the water table by tree transpiration and drainage) due to the interplay between phenol oxidase and hydrolase enzymes (Freeman et al., 2001; Fenner et al., 2011). However, others have attributed increases in DOC (upon re-wetting) to abiotic processes, such as decreased acidity and ionic strength causing enhanced DOC solubility (Chapman et al., 2005; Clark et al., 2012).

In addition to effects due to rewetting, decomposition of brash and needles left on-site (particularly in STD plots) could also contribute significantly to increased DOC concentrations in pore- and surface water (Gaffney, 2016; Gaffney et al., 2020a). Indeed, mass loss experiments have demonstrated that up to 60% of the initial carbon present in conifer needles, and 30% in conifer branches, can be lost within a three-year period post deposition (Palviainen et al., 2004). In our study, the largest increase in DOC concentrations was in ENH plots, although concentrations (in both surface-water and shallow pore-water) began to decrease by December 2015. The extra disturbance of the ground surface due to the additional machinery used in brash removal in ENH plots may have caused a sharper DOC increase (Kiikkilä et al., 2014), but the smaller mass of leftover brash will also mean that concentrations may decline sooner. In contrast, STD plots had lower, but longer lasting peak DOC concentrations, which persisted into December 2015, suggesting decomposition continued into autumn/winter 2015. Longer-term studies of forest-to-bog restoration have shown that peak DOC concentrations tend to occur in the first three years post-restoration, but it may take > 20 years for concentrations to fall within the range of intact open bog sites (Gaffney et al., 2018).

4.1 Effects of forest-to-bog restoration on PO_4^{3-}, K, NH_4^+ and Mn

Increases in PO_4^{3-} were particularly strong in surface-waters (up to 100-fold) and in shallow pore-waters (up to 40-fold) post-restoration treatment. Phosphate is known to be released from brash (Asam et al., 2014a) and in particular from decomposing conifer needles (which have been found to lose up to 47% of their initial phosphorus in the three years following deposition; Kaila et al., 2012). However, PO_4^{3-} can also be readily released from peat upon rewetting - particularly in nutrient-poor peatlands such as blanket bogs which have low Fe and Al and thus limited capacity for PO_4^{3-} sorption (Kaila et al., 2014, 2016).

Increased K concentration in surface-water (up to 10-fold) and in shallow pore-water (up to 4-fold) was also likely due to release from brash and needle decomposition (Palviainen et al., 2004; Kaila et al., 2012; Asam et al., 2014a), and this is a well-documented mechanism which affects watercourse concentrations following peatland conifer felling (Rosén et al., 1996; Cummins and Farrell, 2003; Muller et al., 2015).

For both PO_4^{3-} and K, the highest measured concentrations here were in surface water. After restoration brash was sometimes present in drains (potentially releasing nutrients directly into surface-water) and surface runoff could readily transfer nutrient inputs from surrounding areas into drains. However, surface-water PO_4^{3-} and K concentrations were higher in ENH plots, which had less brash remaining on-site (and a ~5cm deeper WTL); therefore, additional physical disturbance in ENH plots may again be a factor here (section 4.3). Conversely, in shallow pore-water, the highest PO_4^{3-} and K concentrations occurred in STD plots, which had more brash and a shallower WTL.

Post-restoration increases in NH_4^+ (up to 15-fold) observed in shallow pore-water and surface-water and increases in Mn (up to 12-fold) in surface-water, were again a likely product of brash and needle decomposition (Asam et al., 2014a). Increased NH_4^+ has been found previously in surface runoff from brash mats on felled areas, and also in peat porewater under brash mats (Asam et al., 2014b), while increased Mn is known to occur in peatland streams post conifer felling (Cummins and Farrell, 2003). As the highest NH_4^+ concentrations occurred here in shallow pore-water in STD plots (with a shallower WTL), this suggest that NH_4^+ release from peat following rewetting may also have been a contributing factor (Kaila et al., 2016). Notably, there were no effects of restoration on TON concentrations, as found in previous work (Gaffney et al., 2021), indicating that in these wet reducing conditions NH_4^+ was the dominant form of inorganic nitrogen present (Table S2).

The presence of excess plant nutrients in peatlands has been widely found to favour vascular plant growth and negatively impact upon *Sphagnum* species (Xing et al., 2011; Larmola et al., 2013). Recent work on forest-to-bog restoration sites has suggested that peat nutrient levels may be elevated for as long as 14 years after felling, slowing recovery of *Sphagnum* mosses (Hancock et al., 2018). Raised nutrient levels may result in plants more typical of acid grassland and heath, such as *Deschampsia flexuosa* (wavy hair grass) being favoured. This highlights the importance of undertaking restoration of drained afforested peatlands in a way that minimises initial nutrient release and longer-term nutrient levels, in order to assist long term habitat recovery (section 4.3).

4.2 Effects of forest-to-bog restoration on other water chemistry variables

Other changes observed here in pore- and surface-water chemistry variables (following forest-to-bog restoration), may be related to WTL rise and inputs or disturbance of mineral material. For example, Fe concentrations in peatland waters are well known to be closely related to water table and resultant redox conditions (Fenner et al., 2001; Muller et al., 2015). In shallow pore-waters, insoluble Fe (III) in peat would be reduced to soluble Fe (II) as peat is re-wetted (Küsel et al., 2008), which may explain some of the Fe increases seen here in surface- and shallow pore-water post-restoration in STD and ENH plots.

Increased Al (in surface- and shallow pore-water) and increased Ca (in surface-water) post-restoration in ENH and STD plots, may have been related to disturbance by felling/harvesting machinery, which may have exposed/introduced mineral material or related to release from the peat (Muller and Tankéré-Muller, 2012; Gaffney et al., 2018). Additionally, as ditches (containing surface water) are preferential flow paths (Haapalehto et al., 2014), they may collect mineral material or even sometimes cut into mineral layers beneath the prevailing peat, which may explain why post-restoration Al and Ca concentrations were higher than for pore-water. In this study, larger increases in Al were found in ENH plots than in STD plots, which may be due to some site-specific factors (i.e., ENH plots also had slightly higher Al pre-restoration).

The slight increase in pH in surface-waters in STD and ENH plots following restoration may have been related to increased concentrations of some base cations (K, Ca) and to a decrease in S concentrations following a WTL rise (Adamson et al., 2001; Table S2, S3). However, these changes may also have been influenced by prevailing weather conditions, e.g., drought periods during the pre-restoration summers (2013, 2014), which can be associated with acidification of pore-water (Clark et al., 2012), as pH in the BOG sites was also lower pre-restoration.

4.3 Conclusions and implications of management techniques and recovery to bog

Our short-term results indicated that in surface water, ENH plots often saw larger water chemistry parameter concentration increases than in STD plots. In ENH plots, whole tree harvest occurred, first by harvesting stems and then by removing brash. Thus, the larger observed effect (in ENH) may be partly due to the extra machine passes required (physical disturbance), as this treatment required the use of four different machines (whereas the STD treatment used only two). This effect was then measured for DOC (at all depths) and for many other variables in surface-waters (NH_4^+, PO_4^{3-}, Fe, K, Mn, Al, Ca). However, concentrations of many of these parameters (except for NH_4^+ and Al) decreased by December 2015, to below levels in STD plots, suggesting that such disturbance related impacts were shorter term, or, more closely related to seasonal cycles.

In STD plots, there were larger concentration increases in shallow pore-water, for NH_4^+, PO_4^{3}, Fe and K, than in ENH plots. Most of these parameters did not show concentration decreases by December 2015 (unlike in ENH plots) suggesting that in STD plots there may be a slower and more continuous release from leftover brash, which leaches into shallow pore-water. It would be expected that STD plots may release more nutrients over time as brash decomposes (Moore et al., 2011), given that all brash is left on site. Other studies have in fact found that concentrations of DOC and NH_4^+ may not recover to bog levels even after 20 years post-restoration when tree material remains on site (Gaffney et al., 2018). Additionally, the physical presence of brash mats on STD plots may retard bog vegetation recovery on the peat surface (Asam et al., 2014b). WTL is another relevant factor, which on average was shallower in STD plots than in ENH plots (despite both treatments seeing a similar significant rise post-restoration). Shallower newly rewetted zones in STD plots may also have stimulated the release/mineralisation of nutrients, contributing to increased concentrations in shallow-pore water (Borken and Matzner, 2009; Kaila et al., 2016). However, the shallower WTL in STD plots was closer to that of BOG controls, which is generally expected to aid recovery of typical bog vegetation and fauna.

From our results, brash is a clear likely source of many elements and compounds measured in this study and therefore has a key influence over pore- and surface-water chemistry. In the longer-term, removing brash from plots would appear to be a useful measure to help mitigate against high concentrations of nutrients in pore- and surface-water (Asam et al., 2014b; O'Driscoll et al., 2014). Some studies have found that removing brash did not reduce the fluvial export of nutrients (Kaila et al., 2014, 2015), while others have found significant reductions in nutrient export (Asam et al., 2014b; O'Driscoll et al., 2014). This may therefore depend upon local factors such as forestry practices, e.g., tree density (which will affect the volume of brash and depth of the needle layer) and climate (e.g., precipitation levels), between countries (Nieminen et al., 2017).

At the catchment scale, the best method for mitigating downstream water quality impacts may be to carry out phased restoration on small areas of catchments (Shah and Nisbet, 2019; Gaffney et al., 2021), or potentially testing mitigation measures to actively remove e.g. nutrients prior to their entry in freshwater aquatic ecosystems (O'Driscoll et al., 2014). Removing brash might also have other benefits in terms of management beyond water quality, including a reduced fuel load and associated fire risk and a more rapid return of specialist species across a range of taxa (Fernandez, 2018). Therefore, at the restoration site scale, where the emphasis is on the recovery of bog vegetation and function (Hancock et al., 2018), and where pore- and surface-water quality are important factors, removing brash and needles (as a primary source of nutrients) should be considered. Additional measures, such as modifying vegetation grazing levels by herbivores (e.g., deer), by removal fencing (originally used to protect trees during afforestation), may also be effective in redistributing nutrients (Bokdam, 2001). This may (over time) then further encourage re-colonisation by open bog vegetation.

Therefore, overall, while felling enhancements involving brash removal may have short-term impacts on water quality, we consider them more likely than standard approaches to allow faster convergence with bog-like conditions. Accordingly, long-term experimental work should seek to understand how long any water quality effects will last, and whether and how any of these immediate effects to adjacent water courses can be mitigated effectively.

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