Title: Soil and water nutrients in stem only and whole tree harvest treatments in restored boreal peatlands

Running Head: Tree harvest, nutrients and peatland restoration

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Abstract

In boreal peatlands felling and tree harvest are commonly carried out as part of peatland restoration. Stem only harvest is the principal harvest method and it leaves the live crown material (felling residue) containing most tree nutrients at the site. Whole tree harvest where felling residue is removed, is not favoured due to higher transport costs, although it might better promote the recovery of nutrient poor peatlands towards pristine conditions. We investigated whether initial differences in N mineralization and decomposition rates observed between tree harvest methods continued out to six years after restoration and whether the spatial variation in water table level (WT) and water nutrient concentrations parallels with the observed pattern in mineralization and decomposition rates. The study was done at 15 peatland sites in Natura 2000 protection areas in Finland during 2007-2013. Concentration of ammonium in soil water was higher in the stem only harvest treatment compared to that of the whole tree harvest treatment, whereas the previously observed differences in net N mineralization and decomposition rates had levelled out by the sixth year after restoration. The spatial variation

created by the ditch network still affected the hydrology and peatland functions so that the nutrient concentrations were higher near ditches than in other locations, implying potential risk for nutrient leaching. Based on this study, there is no reason to prefer either harvest method over the other in poor drained peatlands with low tree volumes, which constitute the majority of available peatland restoration area in Finland.

Key words: decomposition, ditch blocking, felling, N mineralization, rewetting, water table

Implications for Practice:

- The spatial variation created by the ditch network before restoration is reinforced by the disturbance caused by machinery work during restoration. The observation calls for method development regarding ditch filling to alleviate the artificial spatial variability, minimize the nutrient runoff, and promote a balanced recovery of peatland functions.
- Based on this study, there are no measurable reasons to prefer one of the harvest methods over the other, at least in nutrient poor peatlands with low tree volumes. Information on the impact of higher quantities of felling residue is however needed to help evaluate the impacts of tree harvest in a wider set of peatland habitats.

Introduction

Protection and restoration of peatlands has been stated as one of the core actions to protect biodiversity, mitigate climate change, and help achieve sustainable development goals (Joosten et al. 2012; Joosten 2015). Progressive and cost-efficient restoration of peatlands presumes knowledge of processes which draining has disrupted, as the aim is to reverse these processes. Concerning boreal forestry-drained peatlands, the principal methods of restoration are rewetting and partial or complete removal of the tree layer. Rewetting, i.e. blocking of drainage ditches is done by peeling the peat from around the ditches and filling it back into the ditches by excavators. After rewetting, water table level (WT) rises quickly back to the level of undrained peatlands

(Jauhiainen et al. 2002; Worrall et al. 2007; Haapalehto et al. 2011; Laine et al. 2011), which initiates the natural carbon cycle of peatlands (Kalbitz et al. 2000; Anderson et al. 2016; Laine et al. 2016, 2019; Purre et al. 2019) and slows down mineralization and decomposition processes to resemble more closely those occurring in undrained peatlands (Tarvainen et al. 2013). The spatial variation in the WT caused by the ditch network may, however, influence the degradation and recovery of the plant communities after rewetting (Haapalehto et al. 2017).

Felling and tree harvest are carried out as part of the restoration of boreal forestry-drained peatlands to counteract the direct and indirect impacts of drainage. Felling removes the excess tree biomass grown after drainage, decreases evapotranspiration (Komulainen et al. 1999; Jauhiainen et al. 2002), restores light conditions (Lamers et al. 2002) and reopens wooded peatland landscapes (Similä et al. 2014). The amount of felling depends on the pre-drainage tree cover. In Finland, stem only harvest has been the principal harvest method (Similä et al. 2014). Stem only harvest nevertheless leaves live crown material (felling residue) with a considerable amount of nutrients (Hyvönen et al. 2010) at the site. The decomposition of felling residue may cause twofold problems: i) it may counteract the recovery of drained peatlands back towards their nutrient-poor, pristine conditions, and ii) it may be a high source of nutrients to watercourses after restoration (Kaila et al. 2012; Asam et al. 2014). Whole tree harvest has higher transportation costs due to higher volume requirements (Tolvanen et al. 2013) and is therefore less favoured. Since the removal of live crown material is expected to remove most nutrients that were released from the felling residue after restoration, whole tree harvest may be a more efficient method to return the low nutrient status of poor peatlands.

To counteract the degradation of biodiversity, Finland launched in 2020 a nationwide Helmi habitats programme, in which peatland restoration is one of the core actions (Ministry of the Environment 2020). Large-scale ecosystem restoration calls for cost-effective methods to maximize the benefits and minimize risks arising from restoration. Harvest methods connected with peatland restoration have been only rarely studied

(Tarvainen et al. 2013; Muller et al. 2015). In Scotland Muller et al. (2015) compared a method where cut trees were sheared and compacted into plough furrows with a mulching method where trees were chipped and spread on the ground surface. The study showed higher leaching of potassium (K) and organic carbon (DOC) from areas impacted by mulching, but the work is not comparable to the practice carried out in Finland, since all tree biomass was left at the site in both treatments.

We have shown earlier that whole-tree harvest may be a more efficient restoration method than stem only harvest as it slows down the mineralization and decomposition rates more than does stem only harvest (Tarvainen et al. 2013). The study was carried out in nutrient poor drained peatlands with low tree volume. These poor peatland types cover 80% of the total 0.8 Mha low-productive drained peatland area in Finland (Tolvanen et al. 2018, Juutinen et al. 2020). Low-productive peatlands constitute the available peatland restoration area in Finland, since restoration is not generally carried out in commercially productive peatland forests. Short-term results lead us to investigate further whether the observed differences between tree harvest methods continued six years after restoration. Since spatial variation created by the ditch network was also observed in our earlier study, and this variation has been shown to influence the speed of recovery (Haapalehto et al. 2017), we also analysed whether the spatial pattern in WT and water nutrient concentrations was related with the pattern observed in mineralization and decomposition rates.

Methods

Research areas and restoration arrangements

The study was carried out in two Natura 2000 protection areas, Elimyssalo (64º 10'N 30º20'E) and Iso-Palonen (64º20'N 30º10'E) in eastern Finland. The region belongs to the middle boreal vegetation zone (Ahti et al. 1968). Long-term mean annual temperature was 2.3°C and mean annual precipitation 591 mm at the time of the study (Pirinen et al. 2012). Peatlands in the region, typically oligotrophic pine fens (Laine et al. 2011), were commonly drained for forestry during the 1970s and 1980s. The main impact of drainage was that forest species, such as trees (pines and birches) and shrubs had started to occupy space from mire species through secondary succession (Laine et al. 2011). After the establishment of the Natura 2000 protection areas, drained peatlands therein were destined to be restored.

In 2005, 5 undrained and 10 drained peatland sites of 0.6 - 4 ha were chosen for the study. The tree volume ranged between 0 and 20 m^3 / ha at the undrained sites and between 10 and 45 m^3 / ha at the drained sites. Sites next to each other were pooled into blocks that included one undrained and 1 - 2 neighbouring drained sites, except two blocks which consisted only of two neighbouring drained sites (Fig. 1). The neighbouring drained sites were later assigned to different tree harvest treatments. Within each protected area, the blocks were located 400 m - 1 km from each other, while the protection areas are located 10 km apart. The reason for an unbalanced experimental design was that the restoration was driven by the management needs of the protected areas.

Restoration was implemented in 2007. Two tree harvest treatments were applied: in the stem only harvest treatment (7 sites), only stems of trees over 10 cm in dbh were harvested and the slash was left at the site. In the whole tree harvest treatment (3 sites), both stems and slash were harvested and removed from the site. Felling was done in February - April and ditches were filled by excavator in August 2007.

Measurements in the field and laboratory

To measure changes in the WT, 20 - 36 permanent water table wells (altogether 346 wells) were placed at each peatland site (Fig. 1). The wells were hollow plastic tubes of 80 cm in length and 2 cm in diameter and were drilled with 10 holes in the lower end to allow water to enter them. They were placed at 10 cm above peat surface and covered with plastic caps to prevent the rainwater. At undrained sites, the wells were placed at even distances 20 - 30 m apart. In the harvest treatments wells were placed at even distances along transects, in the middle of the strips (location: strip) and 5 m from ditch (location: near ditch, Fig. 1). The wells were surveyed three times per growing season during spring flood in May, low water level in early August, and autumn flood in October. The measurements used in this study were done during the post-restoration period during 2007 - 2013.

To assess the concentrations of ammonium (NH_4-N), nitrate (NO_3-N) and phosphate (PO_4-P) in soil water, four sampling wells (plastic tubes of 80 cm depth, 6 cm in diameter, 10 holes drilled in the bottom, caps on the top) were placed at two of the undrained sites, and three wells per location (strip, near ditch) at two of the sites in stem only harvest and whole tree harvest treatments in 2008 (Fig. 1). Soil water was sampled from the bottom of the wells using a battery-powered pump three times per growing season along with the water table measurements. The samples were kept at +6°C until analysed.

Net N mineralization of peat was measured at 125 spots (5 at undrained sites and 5 per location in stem only harvested and whole tree harvested treatments) during 2012 – 2013 (Fig. 1). Net N mineralization rates were estimated in field incubations using intact soil cores placed into PVC tubes of 6.7 cm in diameter in the upper peat layer (10-15 cm). The tubes were kept in the soil from May 2012 to May 2013. Samples were analysed for exchangeable N using 0.5 M K_2SO_4-extraction method. The procedure of the measurements is explained in detail in Tarvainen et al. (2013).

Nutrient concentrations in soil water and K_2SO_4-extracts were analysed spectrophotometrically using standard laboratory methods. Concentration of ammonium was analysed by salicylate method (wavelength 655nm; detection limit 3.3 µg/L). Nitrate was analysed using wavelength 545 nm (detection limit 10 µg/L). The soil water samples were filtered through a 0.45 µm membrane before analyses. Concentration of phosphate was analysed after persulfate digestion (wavelength 880 nm; detection limit approximately 2 µg/L). In this study total dissolved N in the K_2SO_4-extracts was analysed using oxidative digestion with peroxodisulfate method by flow injection analysis (FIA8000), being a different method than in Tarvainen et al. (2013). Net ammonification and nitrification were calculated by subtracting the initial ammonium and nitrate concentrations in the reference samples from those concentrations in the field incubated samples, respectively. Net N mineralization was calculated as the sum of net ammonification and net nitrification. If concentrations were less than detection limit, a value half of the detection limit was used in further calculations.

Decomposition rate of birch leaves (*Betula pendula*), pine needles (*Pinus sylvestris*), *Sphagnum* moss (*Sphagnum angustifolium*), and cellulose strips in the peat were measured at the same 125 spots as the net N mineralization. The same plant species were used as in our previous work, since they are easily available standard material that represent different functional types and decomposition rates. Plant material of each species was placed separately in 7 x 7 cm^2 bags made from Sefar open mesh fabrics (500-μ mesh size) on the peat surface. Pre-weighed cellulose strips were placed in the open mesh fabrics (circa 1 mm mesh size) on the peat surface and at 10 cm and 30 cm depths. The decomposition rates were calculated using mass loss percentage. An exponential decomposition rate constant *k*-value was calculated as in Olson (1963, see Tarvainen et al. 2013); higher values reflect higher decomposition rates.

Statistical analyses

The statistical analyses were performed using R 3.6.0 (R Core Team, 2019). The effects of 'treatment' (undrained, stem only harvest, whole tree harvest) and 'location' (undrained or strip, near ditch) on the measured variables were analysed using linear mixed models (*lme4* package, Bates et al. 2014). Because the undrained sites did not have ditches, undrained 'location' was classified into the same class as strip. 'Block' and 'site within a block' were used as a random factor in the mixed models. The random factor takes into account the fact that a replicate of treatments was located within block, and a subset of locations (strip, near ditch) was located within each restored site. When the measurements were repeated in consecutive years and months, 'year' was used as repeated factor and 'month' as random factor in order to consider the independent impact of fixed factors. In the case of significant interactions, the analyses were made separately for both harvest treatments. If location (but not treatment) caused significant differences, the effects were analysed using the location as the sole factor.

The variables were analysed using linear models with Gaussian error distribution (*lmer* with restricted maximum likelihood; *lme4* package, Bates et al. 2014). When needed, cubic root or log-transformation was used to fulfil the model requirements (See Tables S1-S3). The normality and homogeneity of the model residuals were checked using diagnostic plots in order to get confident results. An analysis of variance table of type 3 errors was produced for the *lmer* models using the Satterthwaite approximation for degrees of freedom (*lmerTest* package, Kuznetsova et al. 2014). Multiple comparisons of means were analysed using Tukey contrasts (*glth* function from *multcomp* package; Hothorn et al. 2008). The estimated difference between two means was considered statistically significant ($p < 0.05$), when the absolute value of the *t*- or *z*-score in the mixed models was ≥2 (Crawley 2007). The *t*- and *z*-scores were obtained by dividing the model parameter estimates (coefficients) by the model standard errors.

The figures and tables present the original values averaged to each peatland site.

Results

Average WT of peatland sites varied considerably and annual variation was high (Fig 2a, Table S1). There was no difference in WT between treatments. However, the location affected WT which was slightly but significantly higher (i.e. closer to the surface) near ditch (range -56 cm below surface - +15 cm above surface) than in strip (range -60 cm - +7 cm), but at the same level as at undrained sites (range -57 cm - +14 cm) (z = 2.44 and -1.51, p = 0.031 and 0.255, respectively).

Based on multiple comparisons, the concentration of ammonium in soil water was higher in the stem only harvest compared to the whole tree harvest treatment (Fig. 2b, $z = 3.49$, $p = 0.001$). Concentration of nitrate did not show clear trends between treatments (Fig. 2c, Table S1). Even though the concentration of phosphate did not differ between treatments, it showed a linear decrease along the years in the stem only harvest treatment (Fig. 2d, $t= -2.20$, $p = 0.028$).

Location had a greater impact on soil water nutrient concentrations than had treatment. For example, the concentration of phosphate was on average higher near ditch location compared to in the strip and the undrained sites (Fig. 2d Table S1, $z = 4.48$ and 3.22, $p < 0.001$ and 0.003, respectively). A similar, almost significant difference was seen for the concentration of ammonium (Fig 2b, Table S1, $t = 1.86$, $p = 0.063$). For nitrate, there were significant treatment x location interactions. In the stem only harvest treatment, nitrate concentration was higher near ditch (SN) than in strip (SS; $t = 2.37$, $p = 0.018$).

Due to the high within-treatment variability, the net N mineralisation rate in the peat showed no differences between treatments or locations (Fig 3a, Table S2). Based on the *k* values, the decomposition rate of birch leaves was only slightly different between the treatments ($p = 0.065$, Table S2). According to the pairwise comparisons however, the rate was slightly faster in undrained compared with stem only harvest treatment (Fig. 3b, $z = 2.61$, $p = 0.024$). In the case of *Sphagnum* moss, the decomposition rate was faster in undrained compared to either the stem only or whole tree harvest treatments (Fig. 3d, $z = 5.37$ and 2.66, $p < 0.001$ and $p = 0.021$, respectively). Decomposition rate of pine needles was not different between treatments or locations (Fig. 3c, Table S2).

The decomposition rate of cellulose strips was influenced by incubation depth (Table S3) in that it was considerably lower at the 30 cm depth compared to either the 10 cm depth ($z = 4.92$, $p < 0.001$) or the peat surface ($z = 3.72$, 0.001, Fig. 4). Testing the three incubation depths separately, however, showed no consistent differences between treatments or locations in the surface or at 10 cm depth, while at 30 cm depth there were no significant differences at all.

Discussion

Our study shows few differences between whole tree harvest and stem only harvest treatments over the six-year observation period. Based on this study, there are no reasons to prefer one harvest method over the other, at least in nutrient poor peatlands with low tree volume. The most consistent difference was the higher concentration of ammonium in soil water in the stem only harvest treatment compared with the whole tree harvest treatment. This difference, although small, may suggest the potential for higher nutrient exports from stem only harvested sites at least initially after restoration. These results are similar to other observations after stem only harvest in drained peatland forests (Kaila et al. 2012; Asam et al. 2014). The decomposition of retained felling residues may enhance N leaching through enhanced microbial activity and soil N mineralization (Asam et al. 2014). The difference in N mineralization rate in the upper peat soil layer had levelled out by the sixth year after restoration in this study, contrasting the second year results where the N mineralization rate was significantly higher in the stem only harvest treatment near ditches (Tarvainen et al. 2013). A pulse of nutrient release from labile fractions in the felling residues likely occurred in short order after harvest, and were no longer detectable by the sixth year after restoration

In our previous study, decomposition rates were lower in whole tree harvest compared to the stem only harvest treatment (Tarvainen et al. 2013). This difference supported our suggestion that whole tree harvest may be a viable restoration method in returning peatland functions, such as the lower decomposition rates, towards those in undrained peatlands. In this study the differences in decomposition between the two harvest treatments had levelled out. The fact that undrained sites showed higher decomposition rates than stem only harvest or whole tree harvest treatments may have resulted from slightly moister conditions in the undrained treatment in 2012, indicated by the higher WT (see also Laine et al. 2011). Drought is known to limit decomposition in drained peatlands (Laiho et al. 2004).

The spatial variation created by the ditch network before restoration did affect the hydrology and peatland functions. A consistent pattern had the WT slightly but significantly higher (closer to the surface) near ditches than in strip or in undrained peatlands, reversing the pre-restoration pattern where the WT was generally lowest near ditches (Laine et al. 2011, Haahti et al. 2012; Haapalehto et al. 2014). This is an opposite result to an earlier study in which there were either no differences between the locations, or WT continued to stay lower near ditches than in the strips (Haapalehto et al. 2014). The rationale for our observation may be peat subsidence (shrinking and compression of peat due to drainage), which is more prevalent near ditches due to oxidation (Minkkinen & Laine 1998, Price et al. 2003). It should be understood that peat has to be peeled from 2-5 m width from the ditches to block them, which lowers the soil surface level and increases the WT in relation to the surface. As a result, the use of heavy machinery in rewetting enhances the spatial effect of ditches. It can, however, be expected that this difference will diminish over time along with the vegetation succession.

Concentrations of soil water nutrients were generally higher near ditches than in the strip or at undrained sites. The higher WT near old ditches may cause higher mobilization of nutrients due to an increase in anoxic conditions during rewetting (e.g. Kaila et al. 2016). Increased mobilization of nutrients may, in turn, enhance nutrient runoff from these restored peatlands, and the quantity and duration of the runoff would be expected to vary considerably depending on site fertility and minerotrophy (Koskinen et al. 2017; Nieminen et al. 2020; Gaffney et al. 2018). At the catchment level, the initial increase of phosphorus loading has been shown to stabilize within a few years after restoration (Koskinen et al. 2017), which is consistent with our results that showed decreasing phosphate concentrations over the six-year observation period. Gaffney et al. (2018) showed in blanket bogs that even though phosphate and ammonium concentrations started to decrease three years after restoration, the concentrations had not recovered to natural levels after 17 years. These results, therefore, suggest the potential long-term impacts on nutrient runoff, thereby calling for innovations to manage water quality in restored peatlands (Koskinen et al. 2017).

The data analysis supports that most of the initial differences, albeit small, in peatland functions between stem only harvest and whole tree harvest treatments, had converged by the sixth year after restoration. Our results apply for nutrient poor peatlands with low tree volumes, which, at least due to their high availability, constitute the majority of the potential peatland restoration area in Finland. Nevertheless, restoration is also applied in nutrient rich peatlands with higher tree volumes, where nutrient runoff has been observed to be more substantive (Koskinen et al. 2017). The amount of felling residue may largely determine how extensive and long-term the impact of the residue is on the functions and biodiversity of restored peatlands. This kind of information on residue quantity effects are not currently available, which calls for research that tests best harvesting methods at nutrient rich sites. Also, method development is needed to alleviate the nutrient runoff and spatial variation in hydrology and peatland functions after ditch filling.

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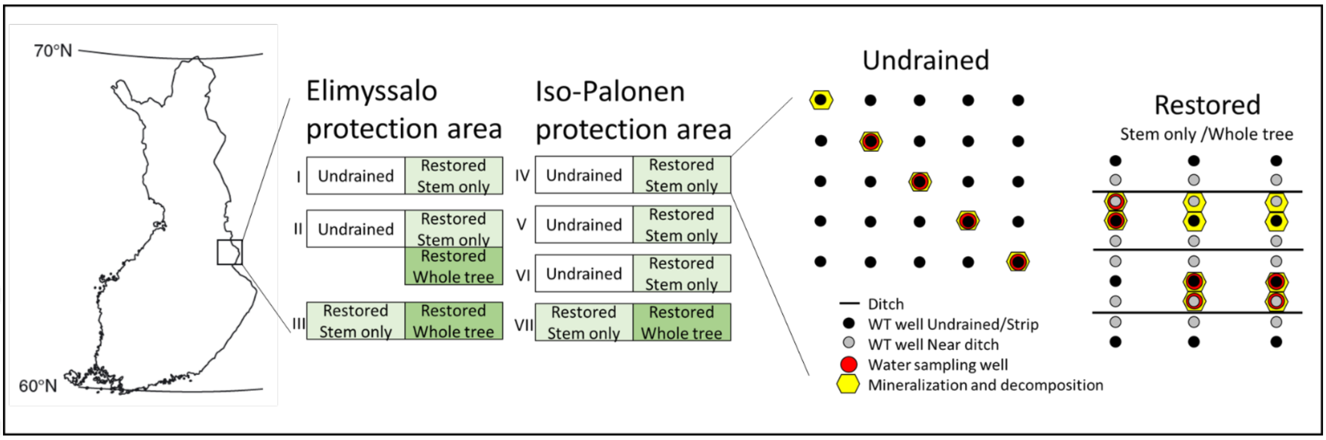

Figure 1. Location of the study area, experimental setup in the two protection areas, and monitoring setup at undrained and restored sites. Roman numbers indicate the seven blocks.

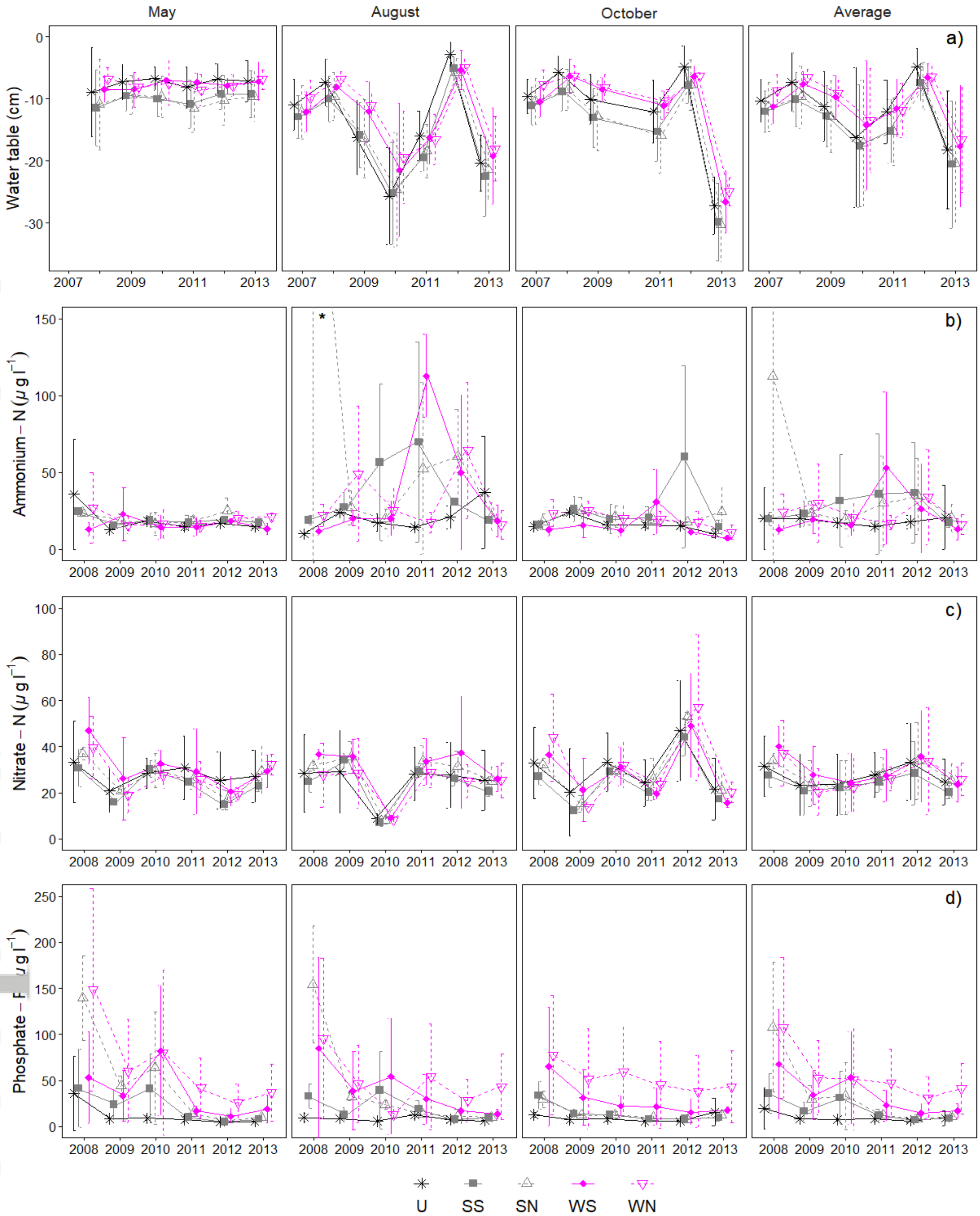

Figure 2. Mean monthly (±SD) and average annual (±SD) (a) WT, (b) ammonium, (c) nitrate and (d) phosphate concentration (μg l^{-1}) in soil water in undrained (U) and restored treatments after restoration in 2007/2008-

2013. SS = stem only harvest, strip; SN = stem only harvest, near ditch; WS = whole tree harvest, strip; WN = whole tree harvest, near ditch. Asterisk (*) refers to exceptionally high ±SD value for SN in Fig. 2b.

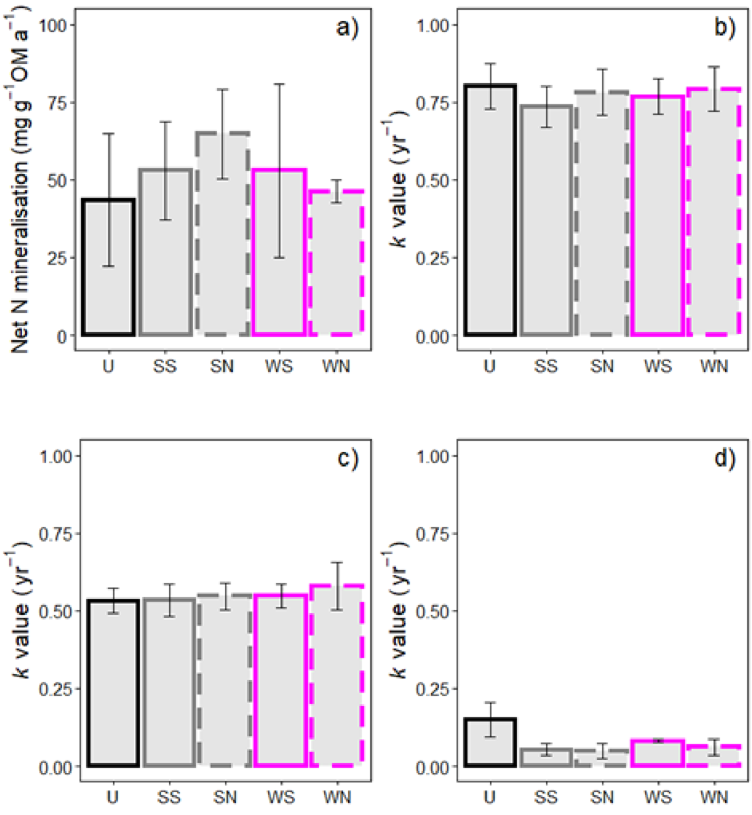

Figure 3. Average (±SD) net N mineralization rate of peat (mg g^{-1} OM a^{-1}) (a), and average (+SD) exponential decomposition rate values (*k* value yr^{-1}) of birch leaf (b), pine needle (c) and *Sphagnum* moss (d) incubated from 2012 to 2013. U = undrained; SS = stem only harvest, strip; SN = stem only harvest, near ditch; WS = whole tree harvest, strip; WN = whole tree harvest, near ditch.

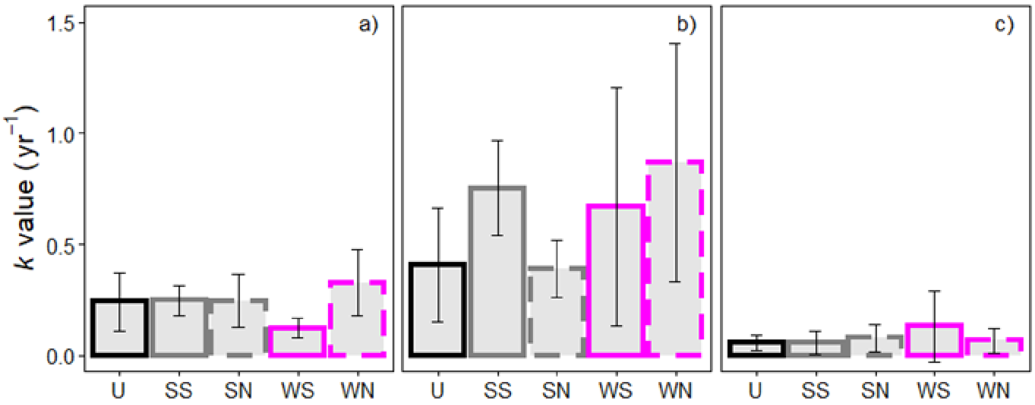

Figure 4. Average (+SD) exponential decomposition rate values (*k* value yr^{-1}) of cellulose strips incubated from 2012 to 2013 at tree depths: a) surface, b) 10 cm depth and c) 30 cm depth. U = undrained; SS = stem only harvest, strip; SN = stem only harvest, near ditch; WS = whole tree harvest, strip; WN = whole tree harvest, near ditch.