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# Effect of clear-felling and harvest residue removal on nitrogen and phosphorus export from drained Norway spruce mires in southern Finland

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Calibration-period/control-area approach was used to study nitrogen and phosphorus export from drained and productive Norway-spruce-dominated peatland forests following conventional stem-only and whole-tree harvesting. The study indicated high nitrogen and particulate phosphorus exports and lack of significant differences between the harvest treatments during the first 3–4 years after harvesting. The high extra nitrogen exports, increasing to a maximum level of about 10 kg ha$^{-1}$ during the third year after harvesting, were partly caused by the higher nitrate export than in previous studies. The study has a practical outcome that management of harvest residues (i.e. left on site or harvested) may not be an efficient means of mitigation of nitrogen and phosphorus exports. The high exports following harvesting underline the importance of using the best available water protection methods, such as sufficiently large wetland buffer areas, to decrease nutrient exports to watercourses from productive Norway spruce dominated peatland catchments.

## Introduction

About 15 million hectares (ha) of peat soils have been drained for forestry in temperate and boreal zones (Paavilainen and Päivänen 1995). Forest drainage activity increased in Finland after the 1950s and 1960s, when ploughing and mechanized excavating techniques replaced manual ditching, and largely ended by the early 1990s. A large number of these forests are now approaching their regeneration age and the rate of forest clear-felling on drained peatlands will undergo a rapid increase in the near future. Increasing interest in harvest residues as a source of bioenergy has led to peatland forests also being more intensively harvested.

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Under Fennoscandian conditions, productive, Norway spruce (*Picea abies*) dominated sites are the most attractive peatland sites for bioenergy harvesting because of the large amount of branch and needle biomass left on site as harvest residues after clear-felling. While the amount of harvest residues may only be 15 000 kg ha$^{-1}$ at sites with low productivity, typically dominated by Scots pine (*Pinus sylvestris*), harvest residues in productive Norway-spruce-dominated peatlands may amount to 50 000 kg ha$^{-1}$ (Palviainen and Finér 2012). Harvest residues are a potentially high source of nutrients to watercourses (Rodgers *et al*. 2010, Kaila *et al*. 2012, Asam *et al*. 2014) and harvesting them for bioenergy could have an additional advantage of reducing water pollution. Hyvönen *et al*. (2000) estimated Norway spruce harvest residues to contain 25–31 kg ha$^{-1}$ of phosphorus (P) and as much as 245–320 kg ha$^{-1}$ of nitrogen (N).

Phosphorus (P) is released from harvest residues already during the early phases of their decomposition, while N is mostly retained and accumulated, especially in larger branches and stumps (Palviainen *et al*. 2004, Kaila *et al*. 2012, Asam *et al*. 2014). It is thus generally accepted that leaving harvest residues on site would only increase P leaching during the first years after clear-felling, and not much that of N. However, modern harvesting techniques leave harvest residues as distinct piles and these piles can be point-type sources of major N export. This may be a result of an increased mineralization of N from underlying soil due to more favorable temperature and moisture conditions than for the residue-free areas (Rosén and Lundmark-Thelin 1987, Nieminen 1998, Asam *et al*. 2014). Also, as no vegetation establishes in the pile sites there is no root uptake to reduce N leaching.

As compared with P export, N export following clear-felling can be a more serious problem at productive Norway spruce sites, because peat at such sites typically has high aluminium (Al) and iron (Fe) contents and a high phosphate adsorption capacity (Nieminen and Jarva 1996). Thus, Lundin (1999) and Nieminen (2004) reported negligible P export following harvesting of drained and productive spruce mires, whereas Nieminen (2003), Rodgers *et al*. (2010), and Kaila *et al*. (2014) found highly enhanced export from drained peatlands with low Al and Fe content and consequent low adsorption capacity.

The risk of high N losses following clear-felling increases substantially when there is nitrification and production of nitrate which is soluble and poorly adsorbed in soil (Vitousek *et al*. 1982). It is well established that nitrification rates are low in acid, anaerobic, and cold environments such as northern peatland sites. However, enhanced nitrate export to watercourses was reported following stem-only harvesting on productive spruce mires both in Sweden (Lundin 1999) and Finland (Nieminen 2004). Much of this nitrate export could have resulted form increased nitrification under harvest residue piles. The study by Mäkiranta *et al*. (2012) at a drained peatland site indicated over 3-fold higher $N_2O$ fluxes in areas covered with harvest residues piles than in respective residue-free areas. Knowing that nitrate production is a prerequisite for $N_2O$ formation, removal of harvest residues for biofuel could be beneficial not only through avoiding $N_2O$ emissions to atmosphere (Mäkiranta *et al*. 2012), but also through decreased nitrate export to watercourses. The study by Rósen and Lundmark-Thelin (1987), who investigated the quality of water percolating in mineral soil covered and uncovered with harvest residue piles, suggested that removal of harvest residues decreased both nitrate and ammonium leaching. The percolation water study by Nieminen (1998) in two clear-felled spruce mires suggested that harvest residue piles were more clearly a source of ammonium than nitrate export.

Compared with mineral soil forests, the water quality impacts of harvesting in drained peatlands has received minor attention (Nieminen 2004 and references therein). This study was carried out to quantify N and P exports from clear-felled Norway-spruce-dominated peatlands with a specific aim to study whether the sites clear-felled using conventional stem-only harvesting and whole-tree harvesting have distinctly different N and P release patterns. We hypothesize that P export following harvesting will be low from productive, Norway-spruce-dominated peatlands owing to the high phosphate adsorption capacity of peat, and N export after the harvesting will be high, and leaving the harvest residues on site will enhance N release to watercourses.

## Material and methods

### Site description

The study was conducted in Lapinjärvi, southern Finland (ETRS89 60°37.946´N, 26°11.809´E). The long-term mean annual precipitation in the Lapinjärvi region is 672 mm and the mean annual air temperature is about +5.3 °C, with monthly means of –7.2 °C in February and +14.6 °C in July.

The study was conducted in two peatland areas about 10 km apart. There was a control (Ctrl-WTH) and two whole-tree harvested (WTH 1 and WTH 2) catchments in one area, and a control (Ctrl-SOH) and two stem-only harvested (SOH 1 and SOH 2) in the other area (Table 1). Catchment sizes varied between 6 and 15 ha. The lowest parts of the catchments were productive, Norway-spruce-dominated peatlands and Norway spruce also dominated on the mineral soil in the surrounding uplands. The volume of standing forest in the catchments ranged from 171–301 $m^3 ha^{-1}$ and Norway spruce accounted for 64%–86% of the total volume. The proportion of drained peatlands from the total catchment areas varied between 37% and 86%. According to the classification by Vasander and Laine (2008), the peatland forests in each catchment belonged to the most fertile herb-rich type. The surrounding upland soils were of the *Vaccinium myrtillus* or *Oxalis–Myrtillus* type (Cajander 1926).

The peat chemical analysis revealed that Ctrl-WTH (Table 1) was less mineral rich than the other areas. Particularly, the concentrations of iron, aluminium, phosphorus, potassium, and magnesium were lower at Ctrl-WTH. The peat nitrogen content was lower at WTH 1 than in the other areas. Except for the peat layer (depth of about 1 m) at Ctrl-WTH, the sites were shallow-peated (depth < 0.30 m). The very fine-textured mineral soil in the bed of ditches and the mineral soil underlying peat was classified as clay.

Clear-felling was carried out in January–February 2009 at SOH 1, WTH 1, and WTH 2, and in January 2010 at SOH 2. SOH 1 and SOH 2 were harvested using conventional stem-only clear-felling in which only stems thicker than 7 cm were removed. At WTH 1 and WTH 2, harvest residues were collected using a mechanized harvesting technique, which is estimated to remove 60%–80% of all harvest residues in Norway spruce stands (Nurmi 2007). In total, 86% of the catchment area was harvested at SOH 1, and 75%, 55%, and 37% at SOH 2, WTH 1, and WTH 2, respectively. The clear-fellings and harvest residue removals were carried out during the frozen-soil periods, and there was thus no substantial damage to the soil caused by the heavy

**Table 1**. Characteristics of the studied catchments and their peat properties (0–10 cm). Peat samples are from 6–7 systematically located sampling positions in each area. SOH = stem only harvesting, WTH = whole tree harvesting.

| Site characteristics | SOH 1 | SOH 2 | Ctrl-SOH | WTH 1 | WTH 2 | Ctrl-WTH |
|---|---|---|---|---|---|---|
| Harvested area (HA)/Whole area (WA) (ha) | 5.3/6.0 | 4.9/6.6 | 9.2 | 6.0/11.0 | 3.4/9.3 | 15.3 |
| Stand volume HA/WA ($m^3 ha^{-1}$) | 251/236 | 238/246 | 233 | 301/274 | 293/219 | 171 |
| Spruce HA/WA (%) | 70/64 | 68/73 | 86 | 88/86 | 77/64 | 68 |
| Pine HA/WA (%) | 11/16 | 16/12 | 2 | 1/3 | 1/9 | 3 |
| Birch HA/WA (%) | 20/15 | 16/13 | 11 | 11/12 | 18/22 | 28 |
| Other HA/WA (%) | 0/6 | 1/2 | 1 | 0/0 | 4/5 | 1 |
| Bulk density (g $cm^{-3}$) | 0.18 | 0.16 | 0.16 | 0.21 | 0.20 | 0.18 |
| pH_$H_2O$ | 4.2 | 4.1 | 4.2 | 3.8 | 3.9 | 3.8 |
| C (%) | 40 | 43 | 46 | 37 | 43 | 53 |
| N (%) | 2.1 | 2.1 | 2.2 | 1.8 | 2.2 | 2.3 |
| P (mg $kg^{-1}$) | 1580 | 1390 | 1450 | 1590 | 1380 | 1130 |
| Al (mg $kg^{-1}$) | 18400 | 16100 | 13400 | 23600 | 20500 | 2920 |
| Fe (mg $kg^{-1}$) | 11600 | 12300 | 10200 | 12600 | 13600 | 5490 |
| Ca (mg $kg^{-1}$) | 3780 | 4320 | 5370 | 2600 | 4600 | 3620 |
| Mg (mg $kg^{-1}$) | 1780 | 1890 | 1560 | 1490 | 1560 | 542 |
| Mn (mg $kg^{-1}$) | 67 | 88 | 59 | 73 | 35 | 39 |
| K (mg $kg^{-1}$) | 1690 | 1900 | 1290 | 2330 | 2090 | 604 |

harvesting machinery. No soil preparation or other regeneration operations were performed at the treatment catchments after harvesting.

## Water sampling

Water sampling began in January 2007, i.e. two or three years before the harvest, and continued until the end of 2012. Outflow water was sampled from a discharge pipe (∅ = 20 cm) of a soil embankment in the main outflow ditch of each area. The sampling interval was once a week during the snowmelt period in spring and twice a week or once a month during other seasons. Altogether 1585 samples were collected from the six catchment areas during the six years (2007–2012). During each sampling, runoff was manually measured using a stopwatch and a 10-l bucket, or a larger 40-l container during high-flow events.

The samples were analyzed for total (unfiltered) N ($N_{tot}$) using flow injection analysis (Tecaton FIA) and for total P ($P_{tot}$) using ascorbic acid method after potassium peroxodisuphate digestion (National Board of Waters 1981). Then the samples were filtered through 0.45 $\mu$m fiberglass filters and analyzed for dissolved reactive P (DRP) using the molybdenum blue method and for dissolved ammonium-nitrogen ($NH_4$-N) and nitrate-nitrogen ($NO_3$-N) by flow injection analysis.

## Data analysis

### Reconstruction of runoff using FEMMA model

We were not able to equip our catchments with automatic and insulated water-level recorders to obtain continuous year-to-year runoff data, but the daily runoff from the catchments was simulated with an ecohydrological model FEMMA (model for Forestry Environmental Management). The FEMMA model consists of sub-models for snow accumulation and melt, interception and transpiration in the overstory and understory vegetation layers, and soil- and ground-water interactions, and runoff generation (Laurén *et al*. 2005, Koivusalo *et al*. 2008). In the model, a drained peatland area is described as a hydrological response unit, which is a vertical one-dimensional soil column that resides between the drainage ditch and the midpoint between two parallel ditches or the catchment boundary (Koivusalo *et al*. 2011). Canopy and snow sub-models are driven by daily standard meteorological input data and soil water movement and runoff generation processes are then simulated using potential transpiration and throughfall/snowmelt series available from the canopy and snow sub-models (Koivusalo *et al*. 2008). Within each harvested catchment the hydrological processes were simulated separately for the harvested areas and the surrounding mineral soils that were left unharvested.

The daily time series of air temperature, precipitation, relative humidity, wind speed, and downward short and long-wave radiation were the meteorological input data. The input data for our study locations were spatially interpolated according to Venäläinen *et al*. (2005) using the measurements from the nearby weather stations operated by the Finnish Meteorological Institute. The tree stand volume was measured with a spatially systematic relascope sample plot inventory with a distance of 25–50 m between plots and the stand data were used to parameterize the canopy model. Average slope and ditch network parameters (ditch depth and spacing) were estimated using the field measurements and terrain maps. In our nutrient-rich peatland sites the understory vegetation re-established already during the first year after clear-felling, and therefore we used a high initial value of ground vegetation biomass of 500 g $m^{-2}$ (Muukkonen and Mäkipää 2006). Finally, the model was calibrated against manually measured runoff by adjusting soil hydraulic properties such as peat water content, water potential and the depth of the surface soil layer with high hydraulic conductivity.

The performance of FEMMA was assessed graphically and using two statistical evaluation criteria reflecting the match between the manually measured and modeled runoff of the measurement days: the Nash-Sutcliffe efficiency coefficient ($E_{NS}$) and bias ($P_{BIAS}$, %). The $E_{NS}$ value indicates how well the plot of observed *versus* simulated values fits a 1:1 line. An $E_{NS}$ value

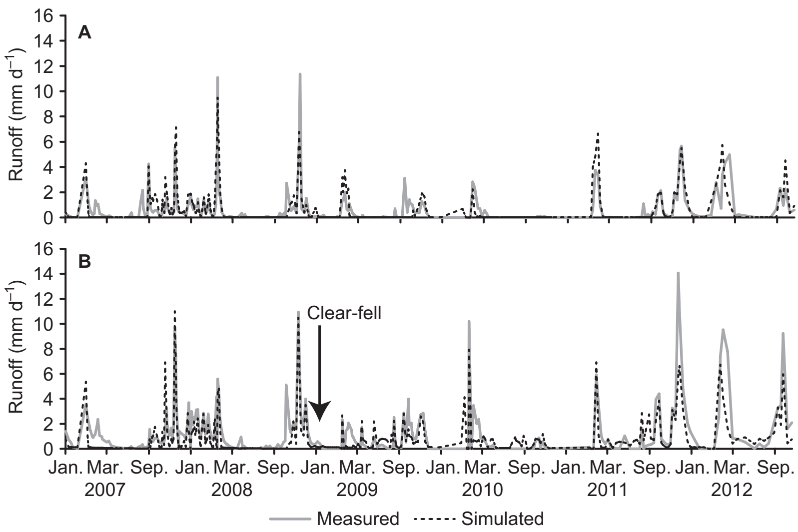


**Fig. 1**. Measured and simulated runoff from (**A**) Ctrl-SOH, and (**B**) WTH 2 during 2007–2012. Ctrl-SOH is an example of very good and WTH 2 of almost satisfactory model fit, according to Moriasi *et al.* (2007). Time of clear-felling at WTH 2 in 2009 is indicated by an arrow.

lower than zero indicates that the average value of the observed time series would be a better predictor than the model. $E_{NS}$ was calculated as follows:

$$E_{NS} = 1 - \sum_{i=1}^{n} \frac{\left(M_i - S_i\right)^2}{\left(M_i - M_{avg}\right)^2}, \quad (1)$$

where $M_i$ is the runoff (mm $d^{-1}$) measured on day $i$, $M_{avg}$ is the average of measured daily runoffs (mm $d^{-1}$), $S_i$ is the runoff (mm $d^{-1}$) simulated for day $i$, and $n$ is the total number of days with runoff measurements.

$P_{BIAS}$ measures the average tendency of the simulated data to be greater or smaller than their corresponding measured counterparts. $P_{BIAS}$ was calculated as follows:

$$P_{BIAS} = \frac{\sum_{i=1}^{n}\left(S_i - M_i\right)\times 100}{\sum_{i=1}^{n} M_i}. \quad (2)$$

Moriasi *et al*. (2007) suggested runoff model performance to be evaluated as satisfactory when $P_{BIAS} < \pm 25\%$ and $E_{NS} > 0.50$, although they discussed that many additional criteria, such as the quality and quantity of measured data, model calibration procedure, evaluation time step, and project scope and magnitude, are important in model performance evaluation.

The $E_{NS}$ values of our simulations with 151–297 measured and simulated runoff values ranged between 0.51 and 0.63 and the $P_{BIAS}$ values were –23.4% for Ctrl-WTH, +21.6% for WTH 1, –25.7% for WTH 2, +9.8% for Ctrl-SOH, +24.7% for SOH 1, and 13.7% for SOH 2, indicating very good ($P_{BIAS}$ for Ctrl-SOH, Fig. 1a), satisfactory, or almost satisfactory model fit ($P_{BIAS}$ for WTH 2, Fig. 1b).

## Calibration-period/control-area method

The effect of harvesting on annual runoff (mm $a^{-1}$) and N and P exports (kg $ha^{-1}$ $a^{-1}$) was studied with the calibration-period/control-area method (also called the paired catchment approach, *see* e.g., Laurén *et al*. 2009). In the calibration-period/control-area method, similar catchments are monitored during a pre-treatment period. During a post-treatment period, one of the catchments is left as an untreated control while the others are treated. Monitoring (runoff and nutrient export) is carried out in all areas. The relationship during the calibration period between the control area and the areas to be treated is then used to predict the behavior of the treated catchment during post-treatment period as if it had not been treated. The treatment effect can then be determined as the difference between the actual measured values and the predicted background values during the post-treatment period.

To produce the annual N and P exports, we first summed the simulated values of daily runoff in order to produce monthly runoff. Then, we calculated monthly export (g ha$^{-1}$) by multiplying monthly runoff by the measured monthly average N and P concentration. Concentration values for months with no observations were interpolated from the closest available monthly values. Values of monthly runoff and N and P export were then summed to calculate annual runoff and export.

The effect of harvesting on annual runoff and N and P export were analyzed separately for the SOH and WTH catchments by combining the two SOH or two WTH catchments and their respective control catchment in the same analysis using the following linear mixed model:

$$f_j^{-1}E_{ij} = a_1C_{ij}f_j^{-1} + b_1I_1 + b_2I_2 + b_3I_3 + b_4I_4 + u_{0j} + e_{ij}, \text{ for } i = 1\text{–}4 \quad (3)$$

where $E_{ij}$ is the annual runoff (mm a$^{-1}$) or N or P export (kg ha$^{-1}$ a$^{-1}$) from the WTH or SOH treatment catchments, $i$ is the year, $j$ is the treatment catchment, $f_j$ is the proportion of the treated area (Table 1) of the catchment, $a_1$ is the slope coefficient, $C_{ij}$ is the annual runoff (mm a$^{-1}$) or export (kg ha$^{-1}$ a$^{-1}$) from the control catchment, $b_1$, …, $b_4$ are the regression coefficients for the harvest induced increase in runoff or export for each post-harvest year (mm a$^{-1}$ or kg ha$^{-1}$ a$^{-1}$), $I_1$, …, $I_4$ are the dummy variables for the post-harvest years, $u_{0j}$ is the random error of the catchments $j$, and $e_{ij}$ is the random error that accounts for the variation among the years $i$ within the catchments $j$. In the model, the dummy variables $I_1$, …, $I_4$ were one at a time assigned with a value 1 to separately indicate each of the post-harvest years, else $I_1$, …, $I_4$ were 0 (i.e, if the first post-harvest year $I_1$ was 1, the other post-harvest years $I_2$–$I_4$ were assigned with a value 0). The post-harvest runoff and P and N export without harvest effect (predicted runoff and export) were calculated as $(a_1C_{ij}f_j^{-1})f_j$. Inclusion of $f_j^{-1}$ in the equation means that the different proportions of harvest areas (Table 1) in the catchments are accounted for, i.e. the impacts of harvesting on annual runoff and N and P export are expressed as mm or kg per harvested area, not per the whole catchment area. The constant was excluded from the model following the theoretical essence in the paired catchment approach that the control catchment and the treatment catchments should behave similarly during the pre-treatment period (Laurén *et al*. 2009), i.e. whenever runoff and export from the control catchment decreased during the calibration period, runoff and export from the treatment catchments also approached zero. The goodness of fit of the model was assessed using the likelihood-ratio test (results not shown).

## Results

### Concentration and export of N and P before and after harvesting

The $N_{tot}$, $NH_4$-N and $NO_3$-N concentrations were higher after clear-felling than during pre-harvest calibration period in the harvested catchments, whereas there were no clear differences in $P_{tot}$ and DRP concentrations (Figs. 2 and 3). Total phosphorus concentrations remained below approximately 0.20 mg l$^{-1}$ and DRP concentrations below 0.08 mg l$^{-1}$ throughout the study period.

The highest $N_{tot}$ concentrations occurred in 2011 when the concentrations peaked to the maximum values of 9.9 mg l$^{-1}$, 6.1 mg l$^{-1}$, 9.5 mg l$^{-1}$ and 6.5 mg l$^{-1}$ for SOH 1, SOH 2, WTH 1, and WTH 2, respectively. In the case of the WTH areas, there was another concentration peak in autumn 2010. The concentrations of $NO_3$-N peaked concurrently with $N_{tot}$ concentrations and reached nearly the same maximum values, indicating that the increased $N_{tot}$ during concentration peaks was mostly due to nitrification and leaching of $NO_3$-N. The increase in $NH_4$-N concentrations was clearly lower as compared with that in $NO_3$-N.

During the pre-harvest calibration period, the annual runoff varied between 174 mm and 356 mm, accounting for on average 35% of the precipitation and being in the treatment catchments about 14% higher than in the control catchments. After clear-felling, the annual runoff from the treatment areas varied between 208 mm and 560 mm, and from control areas between 136 mm and 318 mm, being about 70% higher in the treatment areas (Fig. 4).

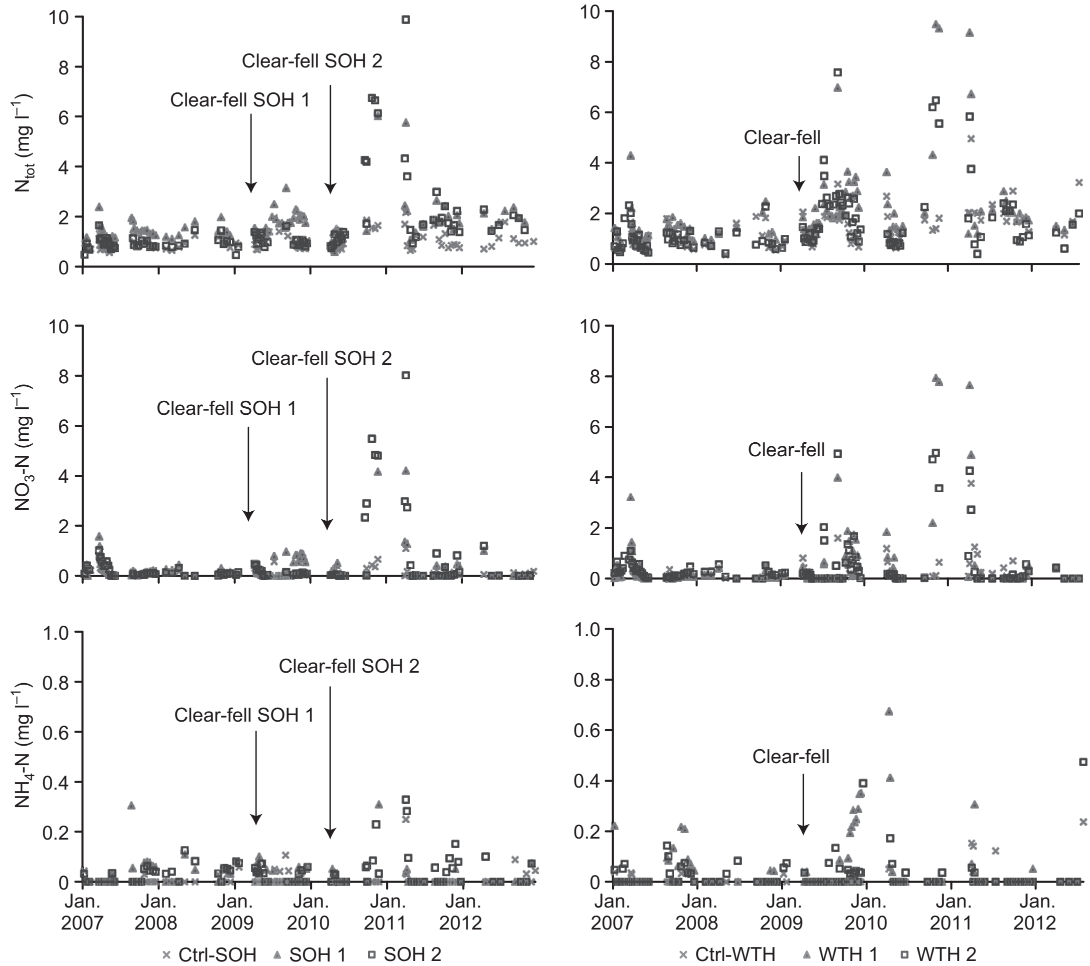


**Fig. 2**. $N_{tot}$, $NO_3$-N and $NH_4$-N concentrations before and after clear-felling. Clear-felling was carried out either in February 2009 or in January 2010 (SOH 2). SOH = stem-only harvesting, WTH = whole-tree harvesting, Ctrl = control. Note the different scale for $NH_4$-N.

Prior to clear-felling, the annual $N_{tot}$ export from the treatment catchments varied between 1.7 kg ha$^{-1}$ and 4.6 kg ha$^{-1}$, and after clear-felling between about 4.5 and 15 kg ha$^{-1}$ (Fig. 5). The annual $NO_3$-N loads from all areas during the pre-harvest calibration period were less than 1 kg ha$^{-1}$. After clear-felling, the annual $NO_3$-N exports were significantly higher from the harvested catchments; the highest exports occurred in 2011, with the exports of 3.6, 7.2, 6.3, and 4.5 kg ha$^{-1}$ for SOH 1, SOH 2, WTH 1, and WTH 2, respectively (Fig. 5). The annual $NH_4$-N exports were generally below 0.1–0.2 kg ha$^{-1}$ a$^{-1}$, but higher exports occurred from SOH 2 in 2011 and 2012 and from WTH 1 in 2010.

The $P_{tot}$ exports ranged between 88 and 260 g ha$^{-1}$ a$^{-1}$ during the pre-harvest calibration period, but the maximum annual exports were > 300–400 g ha$^{-1}$ during the post-harvest period (Fig. 6). The post-treatment loads from the harvested catchments were 2.5–4.5-fold higher than from the control catchments. The DRP exports were below 100 g ha$^{-1}$ a$^{-1}$, except for the exports from SOH 1 and SOH 2 which were slightly higher in 2011 and 2012.

## Impacts of harvesting

In the analysis of the effects of harvesting on

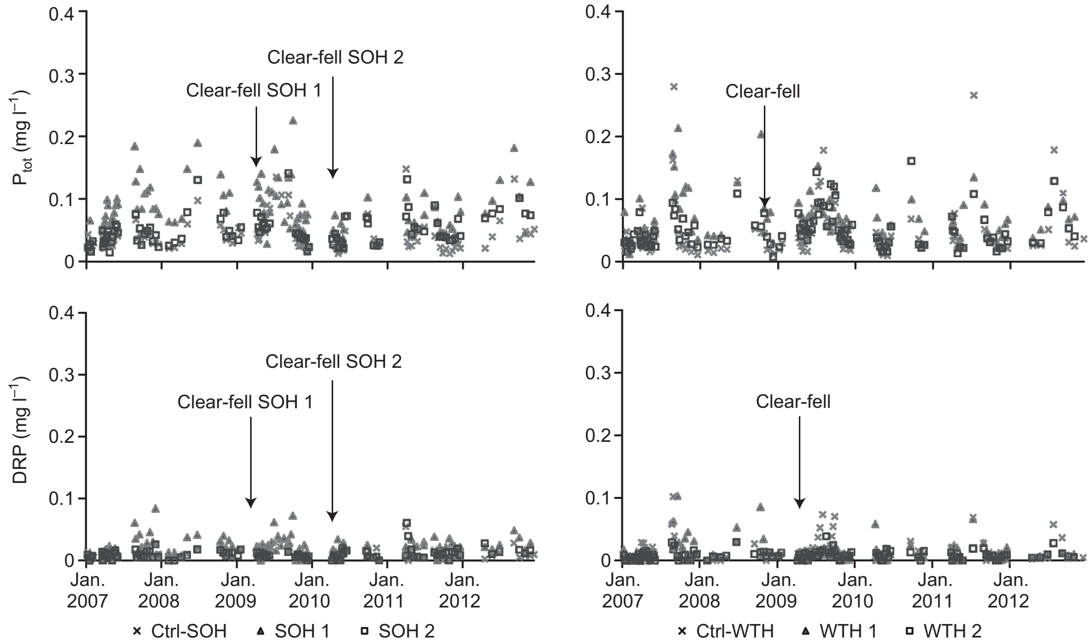


**Fig. 3**. $P_{tot}$ and DRP concentrations before and after clear-felling. For further details, *see* Fig. 2.

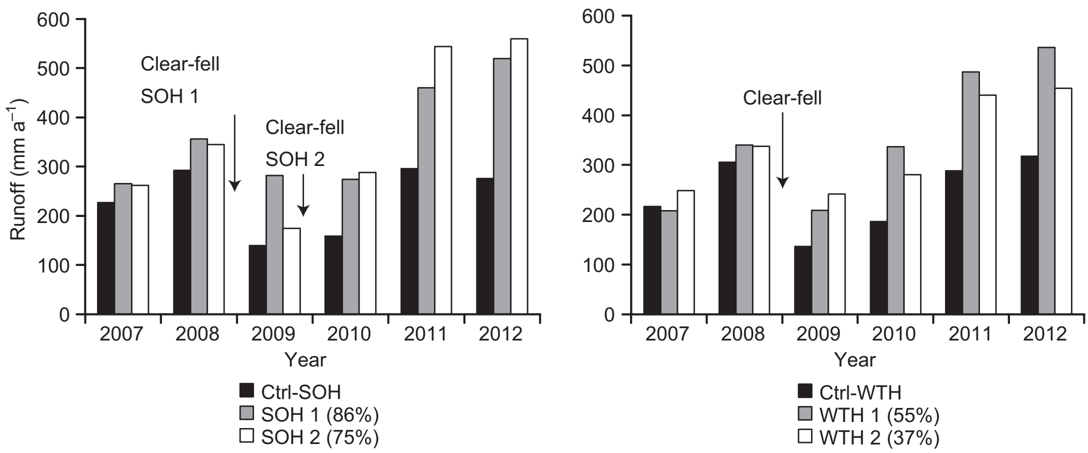


**Fig. 4**. Runoff before and after clear-felling. Harvest area proportions in parentheses. For further details, *see* Fig. 2.

annual runoff and N and P export using the linear mixed model (Eq. 3), the 95% confidence intervals for runoff and all N and P fractions at the WTH and SOH catchments overlapped, indicating that difference between the treatments is not significant. We, therefore, recalculated the impacts of harvesting by combining all four harvest catchments and two control catchments in the same analysis, again using the linear mixed model (Eq. 3). In the subsequent analysis, the average annual runoff from the four harvested catchments increased significantly due to harvesting. The increases varied between 150 mm $a^{-1}$ and 260 mm $a^{-1}$ per harvested area (Table 2), meaning that the runoff after harvesting was on average 1.8-fold higher than the predicted runoff without the harvest effect. The $N_{tot}$ exports increased by 6.0 to 9.9 kg $ha^{-1}$ $a^{-1}$ and were on average three to five-fold higher than the predicted exports without harvest effect. The

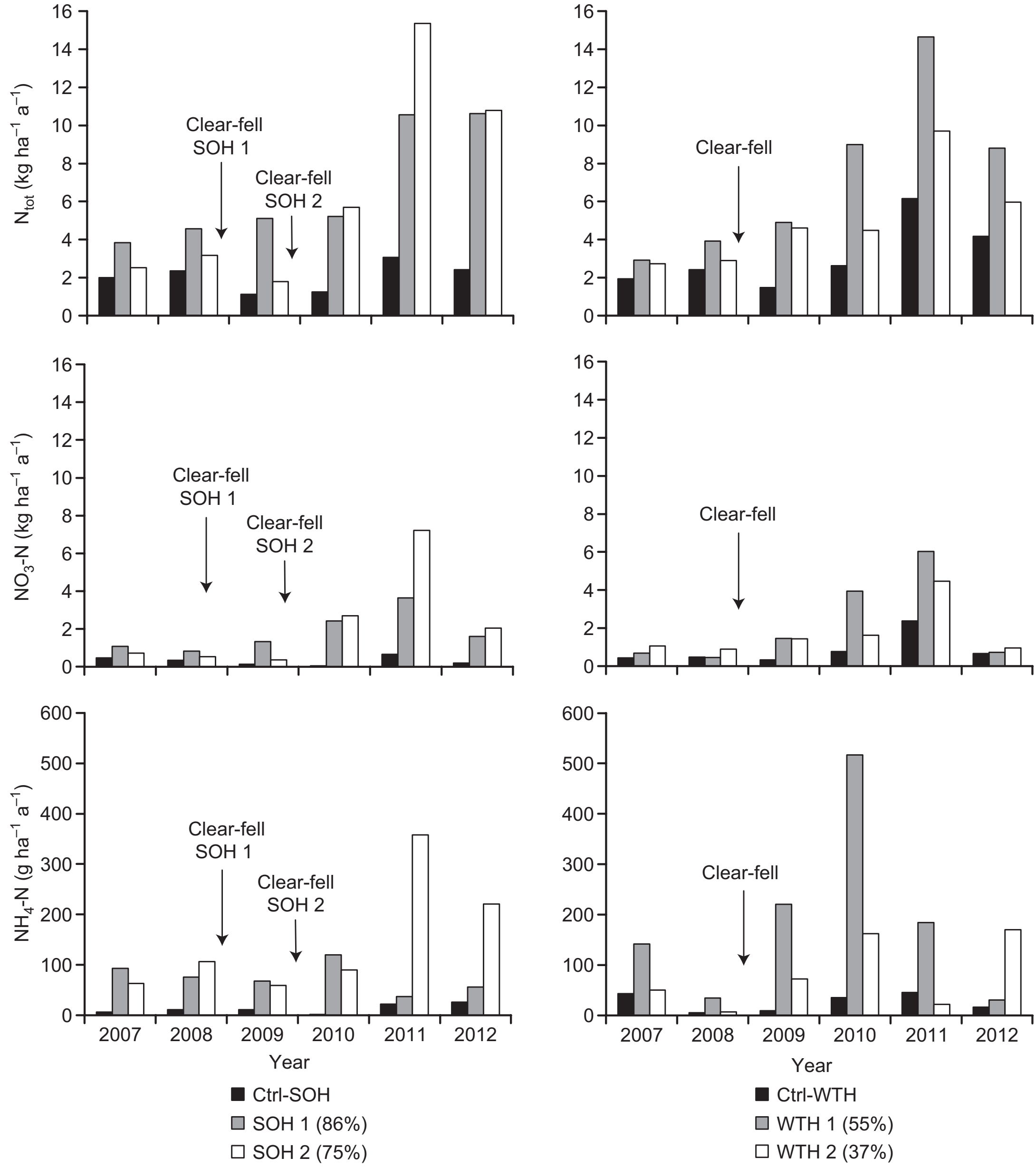


**Fig 5**. $N_{tot}$, $NO_3$-N, and $NH_4$-N exports before and after clear-felling. Harvest area proportions in parentheses. For further details, see Fig. 2. Note the different unit (g ha$^{-1}$ a$^{-1}$) for $NH_4$-N.

$NO_3$-N exports increased significantly during the first three years after clear-felling (Table 2). During this period, the exports increased by 1.4 to 3.1 kg ha$^{-1}$ a$^{-1}$, while the predicted exports without the harvest effect varied between 0.3 and 2.0 kg ha$^{-1}$ a$^{-1}$. The harvest-induced increases in $NO_3$-N exports were about 10-fold higher than the concurrent increases in $NH_4$-N exports. The increase in $NH_4$-N export was significant only during the second year after clear-felling.

Clear-felling increased the $P_{tot}$ exports significantly during all four years after clear-felling. The average annual increase varied between 150 and 315 g ha$^{-1}$ a$^{-1}$, being about 2.6 times higher than the predicted export without harvest. The DRP exports increased significantly during

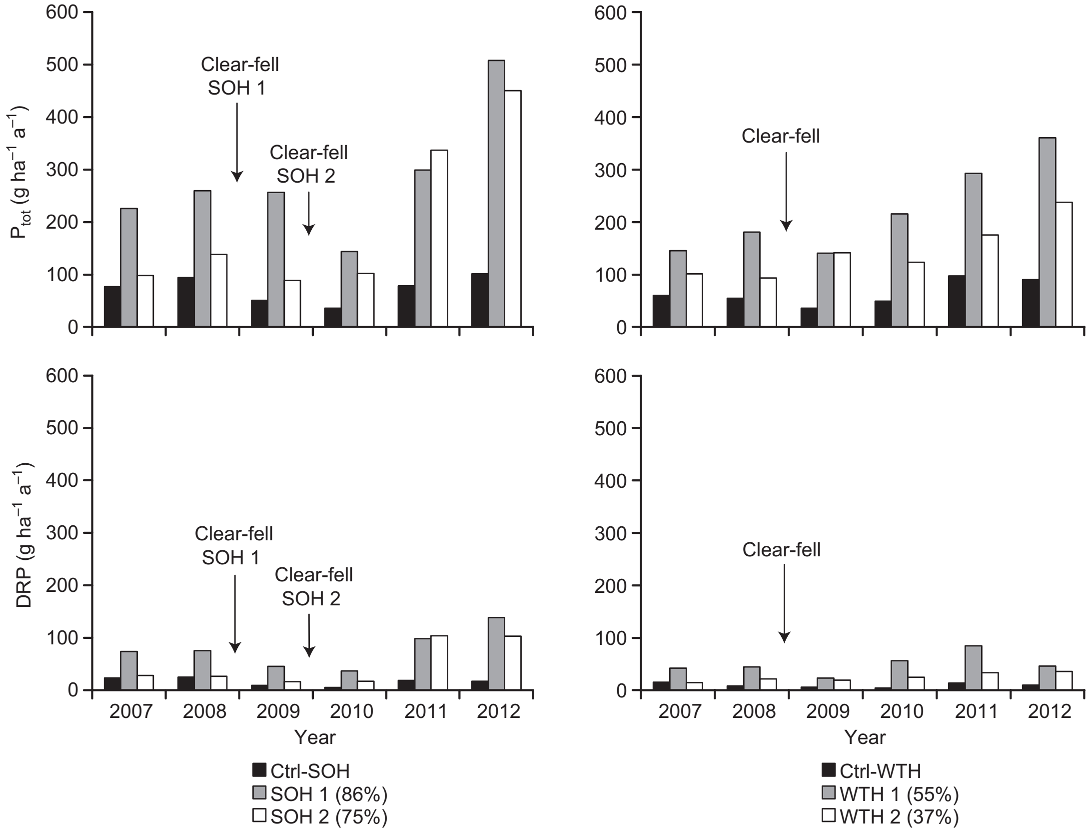


**Fig 6**. $P_{tot}$ and DRP loads before and after clear-felling. Harvest area proportions in parentheses. For further details, *see* Fig. 2.

**Table 2**. Harvest-induced increases in runoff and N and P exports with their 95%CIs, and predicted runoff and N and P exports without harvest effect during the post-treatment years $T_1$–$T_4$. The data from all the six catchments (2 controls, 2 WTHs, 2 SOHs) were combined in the analysis. Harvest-induced increases are the parameter estimates $b_1$, …, $b_4$ in Eq. 3. Parameter estimates set in boldface are statistically significant ($p < 0.05$). Predicted values without the harvest effect are calculated as $(a_1 C_{ij} f_j^{-1}) f_j$ (Eq. 3), where $f_j$ is the proportion of the treated area (Table 1) of the catchment area, $a_1$ is the slope coefficient, and $C_{ij}$ is the annual load from the control catchments.

| | $T_1$ | $T_2$ | $T_3$ | $T_4$ |
|---|---|---|---|---|
| Increase in runoff (mm a$^{-1}$) | **151 ± 61** | **194 ± 65** | **256 ± 72** | **260 ± 83** |
| Runoff without harvest effect (mm a$^{-1}$) | 210 | 277 | 247 | 275 |
| Increase in $N_{tot}$ (g ha$^{-1}$ a$^{-1}$) | **5959 ± 3182** | **8712 ± 3448** | **9858 ± 4399** | **6444 ± 4412** |
| $N_{tot}$ without harvest effect (g ha$^{-1}$ a$^{-1}$) | 1499 | 2251 | 5353 | 3821 |
| Increase in $NO_3$-N (g ha$^{-1}$ a$^{-1}$) | **1452 ± 1169** | **3051 ± 1267** | **2196 ± 1744** | 422 ± 1414 |
| $NO_3$-N without harvest effect (g ha$^{-1}$ a$^{-1}$) | 311 | 545 | 2008 | 570 |
| Increase in $NH_4$-N (g ha$^{-1}$ a$^{-1}$) | 166 ± 184 | **388 ± 212** | 26 ± 237 | 114 ± 224 |
| $NH_4$-N without harvest effect (g ha$^{-1}$ a$^{-1}$) | 24 | 43 | 79 | 49 |
| Increase in $P_{tot}$ (g ha$^{-1}$ a$^{-1}$) | **148 ± 107** | **170 ± 113** | **189 ± 138** | **315 ± 153** |
| $P_{tot}$ without harvest effect (g ha$^{-1}$ a$^{-1}$) | 78 | 77 | 158 | 172 |
| Increase in DRP (g ha$^{-1}$ a$^{-1}$) | 20 ± 34 | **61 ± 35** | **70 ± 40** | **71 ± 42** |
| DRP without harvest effect (g ha$^{-1}$ a$^{-1}$) | 15 | 10 | 33 | 27 |

the last three years after clear-felling. During these three years, the exports increased by 60–70 g ha$^{-1}$ a$^{-1}$.

## Discussion

The results of the calibration-period/control-area method applied to the studied peatland forests supported our hypothesis that there may be a substantial increase in N export from harvested, Norway-spruce-dominated peatland forests. Using the previous results from a number of experiments carried out in Finland, Finér *et al*. (2010) estimated that the average increase in total N export following harvesting of mineral-soil-dominated forest catchments varied from 0.8 to 1.0 kg ha$^{-1}$ a$^{-1}$ during the first four years after harvesting, whereas the average annual increase found in the present study was from 6.0 to 9.9 kg ha$^{-1}$ a$^{-1}$. Compared with previous results from similar Norway-spruce-dominated peatland catchments (Lundin 1999, Nieminen 2004), the total N exports in the present study were also higher. Transforming the load increases of $N_{tot}$ from the study by Lundin (1999) into harvest-area-specific exports by multiplying them by the catchment area/harvest area ratio, gives an annual increase of 2.7 to 7.4 kg ha$^{-1}$ a$^{-1}$. Similarly, taking into account the treatment area, the total dissolved N exports during the non-frost season (5–6 months) in the study by Nieminen (2004) varied between 1.4 and 3.4 kg ha$^{-1}$. Furthermore, the increases in $NO_3$-N exports in the study by Nieminen (2004) were smaller than harvest-induced increases in $NH_4$-N exports, but our results indicated about 10-fold higher increases in annual $NO_3$-N export (1.5–3.1 kg ha$^{-1}$) than in $NH_4$-N export (0.2–0.4 kg ha$^{-1}$). For comparison, in low-productive Scots pine (*Pinus sylvestris*) dominated peatlands, regeneration operations increased $NO_3$-N and $NH_4$-N concentrations in runoff only if the site was ditched and mounded after harvesting the tree stand (Nieminen 2003).

Higher $NO_3$-N export in the present study as compared with that measured by Nieminen (2004) in similar fertile, Norway-spruce-dominated catchments in southern Finland is difficult to explain, as the peat contents of N, for example, are not significantly different between our harvest catchments (1.8%–2.3%) and those of Nieminen (2004) (1.9%–2.1%). However, the concentrations of Fe and Al at our harvest sites were high (Fe 11 600–13 600 mg kg$^{-1}$, Al 16 100–23 600 mg kg$^{-1}$) as compared with those in the study by Nieminen (2004) (Fe 2500–4500 mg kg$^{-1}$, Al 1900–2200 mg kg$^{-1}$), and also compared with typical concentrations in peat soils in Finland (Nieminen and Jarva 1996). As high pH is one factor behind increased nitrification, one mechanism by which harvesting of our study sites increased nitrification and nitrate export could be the harvest-induced water-level rising and subsequent anoxic conditions in previously aerobic peat profiles. Reduction reactions of Fe and manganese in these anoxic peat profiles would consume protons, which may increase the pH of the soil solution (Stumm and Sulzberger 1992).

The results supported our hypothesis also in that the P exports following harvesting were significantly lower than from Al- and Fe-poor low-P-sorptive peatlands, where the annual exports of DRP or total reactive P may have exceeded several hundreds of grams to over 2 kg ha$^{-1}$ (Rodgers *et al*. 2010, Kaila *et al*. 2014, O'Driscoll *et al*. 2014). However, the impact of harvesting on $P_{tot}$ (unfiltered samples) exports in the present study was much higher than the effect on DRP exports, which is possible because of increased export of particulate P along with increased export of suspended solids. Although harvesting was carried out during the frozen soil period in winter and there was no significant damage to the soil caused by the heavy harvesting machinery, harvesting increased erosion and suspended sediment export (data not shown). As no ditching or soil preparation for planting was carried out either, the high particulate P export is plausible because erosion of the very fine-textured soils in the ditches of our study catchments increased along with increased runoff.

The results did not support the hypothesis that N export would be greater from SOH than WTH sites; the hypothesis was presented because the harvest residue piles left on site have been shown to have higher concentrations of N in underlying soil and percolate water collectors than the respective residue-free areas (Rosén and Lundmark-Thelin 1987, Nieminen 1998, Asam *et al*. 2014). A probable reason is that the harvest residue piles did not cover particularly large areas within the catchments and the N concentrations below the piles were diluted by the waters from the residue-free areas before entering the water sampling point. On blanket peat catchments in the west of Ireland, where the harvest residues may amount to over 80 000 kg ha$^{-1}$ and cover as much as 20% of the harvest area, significantly higher P export occurred from SOH than WTH areas (Asam *et al*. 2014), whereas Kaila *et al*. (2014) reported no significant differences in P export between SOH and WTH areas at similar low-soil P-sorptive sites in Finland, where the harvest residues likely amounted to less than 15 000 kg ha$^{-1}$ (Palviainen and Finér 2012). In the same study on blanket peat, Asam *et al*. (2014) reported higher $NH_4$-N export from their SOH than WTH areas, but no differences in $NO_3$-N export between the harvest treatments.

In the interpretation of our results, it should also be noted that, in operational forestry on drained peatland forests, ditch network maintenance (DNM) and soil preparation operations, such as ditch-mounding, are generally done after harvesting the tree stand. These operations may result in larger exports than in the present study, where the catchments were not treated with either DNM or soil preparation. Larger export is particularly true for particulate N and P, as ditching and ditch-mounding may both be significant sources of eroded particulate soil material (Joensuu *et al*. 1999, Nieminen 2003). DNM and ditch-mounding after harvesting may also results in extra dissolved N exports, particularly $NH_4$-N, compared with harvesting without soil preparation and DNM (Joensuu *et al*. 2002, Nieminen 2003).

The effect of harvesting on runoff in the present study falls within the range reported in previous studies carried out in similar site and climatic conditions (Rosén 1984, Lundin 1999, Nieminen 2004) although some of the runoff increases (> 70 mm a$^{-1}$) reported by Lundin (1999) are surprisingly high, given that the harvest areas accounted for only 15%–19% of the catchment area. However, it should be noted that we measured runoff only concurrently with water sampling at one- to four-week intervals and the daily runoff was simulated using the hydrological model FEMMA (Koivusalo *et al*. 2008). Although the correspondence between modeled and measured runoffs was relatively good for both periods (before and after clear-felling), the lack of measured continuous runoff data introduces uncertainty in our runoff and P and N export estimates. In our simulation of runoff, the lack of local weather data was among the major factors causing differences between modeled and measured runoff. Precipitation during summer storms, in particular, may occur locally and our precipitation estimates that were interpolated from the nearby operational weather station data may have missed such storm events.

In the calibration-period/control-area approach, the uncertainty in the relationship between the treatment catchment and control catchments during the calibration period introduces an error in the treatment effect (Laurén *et al*. 2009), but unlike in many other studies, our calculations took that uncertainty into account. The relatively large 95% confidence limits for runoff and N and P exports reflect that the calibration period was short, the degrees of freedom were low and the dynamics between control areas and treatment areas were different in different years. However, a much longer calibration period (5–8 years) (Palviainen *et al*. 2014) and larger data sets with more treatment/control pairs (Nieminen *et al*. 2010) do not necessarily result in smaller confidence limits. When combining many catchments in the same analysis, such as we did, it should be noted that the high variation in harvest impact may also be due to true differences between catchments. For example, the higher the pre-harvest stand volume, the stronger the harvesting impact on runoff (Heikurainen and Päivänen 1970, Seuna 1988). However, the differences in tree stand volumes in our harvested sites were relatively small, the stand volumes ranging from 238 to 301 m$^3$ ha$^{-1}$ (Table 1).

In conclusion, the study confirmed the results of the earlier studies by Lundin (1999) and Nieminen (2004) that the impacts of harvesting on N export may be higher for fertile, Norway-spruce-dominated peatlands than for mineral-soil-dominated catchments or low-productive-peatland catchments in the boreal zone (Nieminen 2003, Finér *et al*. 2010, Palviainen *et al*. 2014). The N exports in the present study were even higher than in the previous studies from similar site and climatic conditions (Lundin 1999, Nieminen 2004), mostly because nitrate export increased significantly more. The mechanisms behind increased nitrification and nitrate export in fertile peatlands forests following harvesting need further research. We present a hypothesis that at sites with high Fe contents the pH increase associated with Fe reduction could be a factor increasing nitrification and nitrate export in harvested peatlands. The $P_{tot}$ exports were also higher than in previous studies, possibly because of increased particulate P export along with increased erosion and suspended sediment export. The study has a practical outcome that the management of harvest residues (i.e. left on site or harvested) may not be efficient means of mitigation of N and P exports. High N exports following harvesting underline the importance of using the best available water protection methods, such as sufficiently large wetland buffer areas, to decrease nutrient exports to watercourses from productive, Norway-spruce dominated peatland catchments. Although sedimentation ponds are efficient in retaining the suspended solids from ditching operations (Joensuu *et al*. 1999), their use cannot be recommended for harvested sites due to their inefficiency in retaining dissolved nutrients.

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