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Impacts of restoration of forestry-drained peatlands on nutrient and organic carbon exports and methane dynamics

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Academic dissertation

To be presented, with the permission of the Faculty of Agriculture and Forestry of the University of Helsinki, for public criticism in lecture room B2, B-building (Viikki Campus, Latokartanonkaari 7, Helsinki) on Jan 20th 2017, 12 o'clock noon.

Title of dissertation: Impacts of restoration of forestry-drained peatlands on nutrient and organic carbon exports and methane dynamics

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Dissertationes Forestales 232

http://dx.doi.org/10.14214/df.232

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ISSN 1795-7389 (online)
ISBN 978-951-651-554-3 (pdf)

ISSN 2323-9220 (print)
ISBN 978-951-651-555-0 (paperback)

Publishers:
Finnish Society of Forest Science
Faculty of Agriculture and Forestry of the University of Helsinki
School of Forest Sciences of the University of Eastern Finland

Editorial office:
Finnish Society of Forest Science
Viikinkaari 6, FI-00790 Helsinki, Finland
http://www.dissertationesforestales.fi

Koskinen M. (2016). Impacts of restoration of forestry-drained peatlands on nutrient and organic carbon exports and methane dynamics. Dissertationes Forestales 232, 36p.
http://dx.doi.org/10.14214/df.232

In this study, the effects of restoration of forestry-drained peatlands on the nutrient and organic carbon exports and methane dynamics of the restored sites are explored. The study consists of four sub-studies. Two of the sub-studies are concerned with the effects on water quality and export of elements of restoration and were conducted on a catchment scale. One of the studies was conducted in the laboratory, and assessed the release of elements from peat samples under anaerobic inundation simulating the effects of a rising water table after restoration or logging. The fourth study was again a field study, in which the differences in methane emissions between undrained, drained and restored spruce swamp forests were assessed. In all, 24 different pristine, drained and restored sites are featured in the study, one site being present in two of the sub-studies.

The results indicate potentially large effects of restoration especially on the nutrient rich spruce-dominated sites, which had the highest restoration-induced increases in organic carbon and nutrient exports in the catchment studies, and which also exhibited high methane emissions after restoration, higher than in the undrained or drained state. The results should prompt research into the techniques applied in restoration of such sites and into the processes which lie behind these large effects.

ACKNOWLEDGEMENTS

This work began already in 2009 when I got a working grant for a year from the Science foundation of the University of Helsinki, with which I processed the manuscript that became the first publication of this thesis. During the time, and already during the making of my Master's thesis in 2007–2008, I learned water sampling and interpretation of water quality and runoff data from Phil.Lic. Tapani Sallantaus from the Finnish Environment Institute and worked under the supervision of Prof. Harri Vasander from the Department of Forest Sciences. After that I had a period of several years of no chance to work on the PhD project, during which I was given the chance to work on greenhouse gas measurement methodology by Dr. Kari Minkkinen. Under his supervision I learned programming, gas measurements, statistics and scientific writing. I also worked with Dr. Paavo Ojanen from the department, as well as with Dr. Annalea Lohila from the Finnish Meteorological Institute. Great times were had. Thank you for tolerating my temperament during the field work, and Paavo especially for all the conversations over the years.

The good will of Prof. Eeva-Stiina Tuittila, now in the University of Eastern Finland, was instrumental for the methane study. I also want to thank our field workers, Jyri Mikkola, Mirkka Kotiaho, Janne Sormunen and Salli Uljas. Dr. Liisa Maanavilja provided insight on the sites. PhD candidate Maija Lampela assisted me with topographical measurements in the field.

When it seemed improbable that I could ever make a thesis on the water quality effects of restoration of forestry-drained peatlands, Dr. Mika Nieminen from the Natural Resources Institute (then METLA) got a grant from the Maj and Tor Nessling foundation to do just that, and asked me if I would be the N.N. for whose work the funds had been granted. I said yes, and haven't regretted it. He has taught me a lot on getting work published. I have also had the opportunity to work with PhD candidate Annu Kaila, whose painstaking laboratory work and theoretical analysis is featured in the laboratory incubation article; and Dr. Sakari Sarkkola, who helped me with statistical methodology.

I wish to thank the pre-examimners of the thesis, Dr. Tapio Lindholm and Dr. Dominik Zak, for their comments on the summary and the last manuscript.

I express my gratitude to Prof. Hans Joosten from the Greifswald University for accepting the invitation to be my opponent.

Thank you to the steering committee of my doctoral studies: Kaisu Aapala, Tuomas Haapalehto, Ari Laurén and Samuli Joensuu.

I also wish to thank my friends at the university, especially Jani Anttila, without whose comradery and sense of humour these years would have been much duller; and my friends outside the university, who make the world tolerable.

To my parents, thank you for supporting us and for raising me to have an appreciation for education and the stubbornness to do what I want to do. To my wife Aino, thank you for never complaining about the early mornings and late evenings during field work periods; and to our child Tuure, thank you for being your radiant self.

LIST OF ORIGINAL ARTICLES

This dissertation is based on the following three published articles (I, III-IV) and one manuscript (II). In the summary, they are referred to using their roman numerals given below. The publications are reprinted here with the kind permission of the publishers.

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

II Koskinen M., Tahvanainen T., Sarkkola S., Menberu M., Laurén A., Sallantaus T., Marttila H., Ronkanen A.-K., Tolvanen A., Parviainen M., Koivusalo H. & Nieminen M. Restoration of fertile peatlands poses a risk for elevated exports of dissolved organic carbon, nitrogen, and phosphorus. Manuscript.

III Kaila A., Asam Z., Koskinen M., Uusitalo R., Smolander A., Kiikkilä O., Sarkkola S., O'Driscoll C., Kitunen V., Fritze H., Nousiainen H., Tervahauta A., Xiao L. & Nieminen M. (2016) Impact of re-wetting of forestry-drained peatlands on water quality–a laboratory approach assessing the release of P, N, Fe, and dissolved organic carbon. Water, Air, & Soil Pollution 227(8): 292.
doi: http://dx.doi.org/10.1007/s11270-016-2994-9

IV Koskinen M., Maanavilja L., Nieminen M., Minkkinen K. & Tuittila E.-S. (2016) High methane emissions from restored Norway spruce swamps in southern Finland over one growing season. Mires and Peat 17(2): 1–13.
doi: http://dx.doi.org/10.19189/MaP.2015.OMB.202

Markku Koskinen is fully responsible for the summary of this doctoral thesis.

I In the article, Markku Koskinen participated in the water sampling, was responsible for doing the calculations to produce the export and impact estimates using external simulated runoff data, the analysis and interpretation of the data and was the main author and reviser of the article.

II In the manuscript, Markku Koskinen combined the runoff and concentration data and calculated the exports. He did statistical analysis, modeling and interpretation of the data in co-operation with Sarkkola, Laurén and Nieminen. First draft of the manuscript was written co-operatively by Koskinen and Nieminen.

III In the article, Markku Koskinen was responsible for analysis and interpretation of dissolved organic carbon (DOC) and iron (Fe) in connection with DOC data. The first draft for the article was prepared co-operatively by Mika Nieminen, Annu Kaila and Markku Koskinen.

IV In the article, Markku Koskinen was responsible for planning the study design in co-operation with Tuittila, Minkkinen and Maanavilja, setting up the study plots, took main responsibility for the statistical analysis and interpretation of the CH_4 and water table depth data in co-operation with the other authors and served as the main author of the manuscript.

Contents

1 INTRODUCTION

1.1 Exploitation of peatlands

Peatlands are present in almost all parts of the world (Gore, 1983). In pristine state, they provide many ecological functions such as acting as filters for water (Nieminen et al., 2005), storing and sequestering carbon (C) (Turunen et al., 2002; Yu et al., 2010; Joosten et al., 2012) and maintaining biodiversity (Chapman et al., 2003). They have, however, been exploited for various goals, such as peat extraction, agriculture and forestry (Joosten & Clarke, 2002).

Drainage for forestry has been the most common form of peatland exploitation in Finland, where 55% of the 10 Mha peatland area has been drained for this purpose (Turunen, 2008). Overall, in the non-tropical world, 16% of peatlands have been drained, of which 30% for forestry (Joosten & Clarke, 2002).

Common to most forms of peatland exploitation is drainage, in order to lower the water table. This alters the functioning and surface structure of the peatland, increasing the aerated volume of the surface peat where rapid decomposition is possible (Freeman et al., 2001; Jaatinen et al., 2008) and causing subsidence of the soil first by physical compression, removing the supporting pressure of the water and then by the increased decomposition of the surface peat layers (Minkkinen & Laine, 1998; Jaatinen et al., 2008).

The ecological effects of peatland drainage for forestry include effects on receiving water courses, effects on the greenhouse gas (GHG) budget, and effects on biodiversity. The effects on receiving water courses include increase in suspended solids (SS) and dissolved elements particularly during the ditching and ditch maintenance (Joensuu et al., 2002), but also several years after the ditching operations (Sallantaus, 1992; Joensuu et al., 1999). Also forestry operations such as harvesting of the tree stand can cause considerable load of SS, dissolved organic carbon (DOC), nitrogen (N) and phosphorus (P) on the receiving water courses (Nieminen, 2003, 2004).

The GHG budget of a peatland drained for forestry is not straightforward and depends on the fertility of the drained site. On ombrotrophic and weakly oligotrophic sites, the increase in amount and/or changes in the quality of litter production can compensate the possibly increased rate of decomposition in the soil, whereas on more fertile sites the increased decomposition can cause significant loss of carbon from the soil (e.g. Silvola et al., 1996; Blodau & Moore, 2003; Ojanen et al., 2010, 2013). At the same time, the increased aeration of the top peat layer reduces the activity of methanogens (Blodau & Moore, 2003) and, at least initially, increases the activity of methanotrophs in the peat (Kettunen et al., 1999). This can turn the peatlands from sources to sinks of methane (CH_4) in the short term (Nykänen et al., 1998; Arnold et al., 2005). A notable exception to this are the drainage ditches themselves, which can be large point sources of CH_4 (Roulet & Moore, 1995; Minkkinen et al., 1997; Minkkinen & Laine, 2006). The long-term effect of the possible increase in tree stock after drainage depends on how it is used; whether it is left standing or is harvested and made into short- or long-lasting products (Minkkinen et al., 2002).

Forestry drainage of peatlands has major effects on biodiversity on the landscape scale even without other forestry operations, such as logging. As a result of the lowering water table, mire species are replaced by species characteristic of forests on mineral soils (Laine et al., 1995; Minkkinen et al., 1999) and thus the peatland sites start to resemble mineral soil forest sites.

1.2 Restoration of forestry-drained peatlands

In order to restore the ecological functions degraded by drainage and other operations carried out on peatlands, the restoration of forestry-drained peatlands started in Finland in the 1990s. Most commonly the methods of restoration of forestry-drained peatlands include damming of or filling in the drainage ditches and in some cases removing the tree stand if it has been significantly affected by drainage (Komulainen et al., 1999; Similä et al., 2014). These actions aim to restore the hydrological regime and light conditions that existed on the site before it was drained and thus enable the resurgence of mire vegetation (Rochefort et al., 2003) and the ecological functions such as the C sink and water purification that occur on pristine mires (Komulainen et al., 1999; Lucchese et al., 2010; Similä et al., 2014).

In Finland, restoration operations have been conducted mostly in national parks and other protected areas, at a rate of between 1000 and 2000 hectares per year in the 2010s(Similä et al., 2014). It has been estimated that there are up to one million hectares of forestry-drained peatlands in Finland where the economical feasibility of forestry is compromised due to the soil having too low a nutrient status (Ministry of Agriculture and Forestry, 2011). These sites will probably be left aside from forestry and are therefore attractive sites for fulfilling the EU strategy to restore 15% of degraded ecosystems by 2020 (EC, 2011). On the other hand, fertile sites such as spruce swamp forests have been the most affected by forestry drainage as they have a high potential for timber production; consequently, 73% of the spruce swamp forests in Southern Finland have been drained, making them a threatened biotope (Raunio et al., 2008), and thus a prime target for restoration projects aiming to protect and increase biodiversity (Similä et al., 2014).

Restoration aims to change the conditions in the surface peat layers of the restored peatland, which have already been altered due to the effects of drainage (Minkkinen & Laine, 1998; Jaatinen et al., 2008). In minerotrophic sites, runoff from the mineral soil catchment surrounding the peatland is reintroduced into the peat, while on ombrotrophic sites the movement of water away from the peat is slowed down once again. Thus restoration potentially has effects on the quality of water that flows out of the peatland and consequently on the receiving water bodies. Detrimental effects have been observed, for example by Vasander et al. (2003), who reported increased export of PO_4 from a restored buffer zone, and Sallantaus (2004), who reported an increase in P concentration from 10 to 160 $\mu g\ l^{-1}$ in a lake whose catchment included 30% of restored peatlands. Nieminen et al. (2005) reported elevated concentration of DOC in runoff from a forestry-drained peatlands restored for a forestry buffer. Rewetting of peat from drained peatlands has also been found to cause significant release of DOC and nutrients, particulariry P, in laboratory incubation studies on agricultural (e.g. Zak & Gelbrecht, 2007) and forestry-drained (Urbanová et al., 2011) peatlands. On the other hand, the anoxic conditions present in rewetted peat may cause export of nitrate-nitrite N ($NO_{2\text{-}3}$-N) to cease altogether with the restored peatland actually retaining added NO_3-N (Silván et al., 2005).

Fe and Al content have been found to be crucial to the release of P from rewetted peat. P is released from Fe^{III} associations as the Fe^{III} is reduced into Fe^{II} under anoxic conditions.

This is supported by observations of simultaneously rising Fe and P concentrations in soil water under anoxic conditions (e.g. Forsmann & Kjaergaard, 2014).

The highest export of DOC from peatlands under rewetted conditions has been observed in fertile, Fe-rich peatlands. Grybos et al. (2009) argued that the process behind the release of DOC from rewetted peat is the rise in pH associated with the falling redox potential (Eh7) of the soil solution. The rise in pH occurs as the redox reactions of Fe^{III} consume protons and thus reduce the H^+ activity in the soil solution. This results in breakup of associations between organic molecules and Fe^{III} (R-Fe^{III}-R) and increased electronegativity of the organic moieties, which makes them less attracted to the soil matrix.

Studies on the effects of restoration on CH_4 dynamics on peatlands have found controversial results. In some cases, restored sites have had similar CH_4 dynamics as pristine sites (Tuittila et al., 2000; Wilson et al., 2009), whereas in other cases the emissions have been either much lower (Juottonen et al., 2012) or higher (Wilson et al., 2013; Vanselow-Algan et al., 2015) than on comparable pristine sites. The lower emission have been linked to the methanogen community not having revived from the decline caused by the drainage (Juottonen et al., 2012). The reason behind the higher emissions has been estimated to be a fluctuating water table in connection with input of easily degradable material (Vanselow-Algan et al., 2015; Wilson et al., 2013). Most of the sites in the aforementioned studies are bogs or treeless fens; only few studies have been made on either undrained or restored spruce swamps. In pristine swamps, small emissions and small consumption of CH_4 were been reported by Huttunen et al. (2003), while the only study on restored sites which included a spruce swamp forest used as a forestry buffer zone reported negligible CH_4 emissions from that site (Juottonen et al., 2012). Measurements were only conducted in the mid-strip area of the peatland in that study.

There is thus a dearth of published knowledge on the range of impacts restoration of forestry-drained peatlands can have on receiving waterbodies, which sites are most at risk of producing high post-restoration exports of carbon and nutrients; and on the CH_4 dynamics of spruce swamp forests, undrained or drained and restored.

The aims of this thesis are: 1. to improve the understanding of the effects of restoration of forestry-drained peatlands on the runoff water quality and nutrient and organic carbon load on the receiving water bodies (I, II); 2. to assess the effects of restoration on the CH_4 dynamics of spruce swamp forests (IV); and 3. to examine the processes and factors behind these effects (III). The studies incorporated in the thesis include three field studies, two of which focus on water at the catchment scale (I, II) and one of which focuses on CH_4 dynamics on a sampling plot scale (IV); and one laboratory experiment (III).

2 MATERIAL AND METHODS

2.1 The impact of restoration on nutrient and organic carbon exports (I, II)

2.1.1 Study sites and sampling

Twelve catchments in all were used to study the nutrient and organic carbon load of restoration of forestry-drained peatlands (I, II) (Table 1). In study I, catchment Mustakorpi comprised of three connected sub-catchments and the results from catchment Seitseminen were means of three separate catchments. Of the eight catchments in study II, three were pristine (C1, C2) and drained (C3) control catchments and one treatment catchment was a separate catchment with no control (T2). The fertility level of the peatlands in the catchments varied between ombrotrophic and mesotrophic. In study I, the restoration measures included removal of the tree stand on the nutrient-poor Seitseminen sites that had been treeless mires before drainage, in addition to damming and filling in the ditches. In Mustakorpi, the tree stand was left intact and the restoration measures included only damming of the ditches. In study II, the tree stands were left intact on all sites, and the ditches were first filled in and then shallow dams were built to ensure the water did not flow in the filled-in ditches. Measurement weirs were built into the outlet points of the catchments to enable water sampling (I, II) and continuous measurement of runoff with water level loggers installed in the weirs (II).

2.1.2 Calculations

In study I, the impact of restoration was estimated as the annual difference between element exports calculated using background concentration values and the measured concentration values. Element concentrations for the background export were calculated as flow-weighted

Table 1: Basic information on catchment characteristics in studies I and II. Lat=latitude, Lon=longitude(WGS84 grid). N.E. = not estimated. Area is total area of catchment in ha (CA), TSV is tree stand volume in m^3 ha^{-1}, Upland/Peat.

Site	Lat	Lon	Fertility	Area	Peat area % of CA	TSV	Study
Mustakorpi	60 18.0	24 27.0	meso	48.5	29	N.E./300	I
Seitseminen	61 56.0	23 26.0	oligo-ombro	60.0	36–44	N.E./50	I
T1	61 59.8	23 53.0	meso	9.1	14	107/158	II
C1	61 51.4	24 14.2	meso	5.7	28	276/235	II
T2	60 37.9	26 10.0	meso	15.3	33	180/171	II
T3	61 59.8	23 52.8	ombro	34.0	34	229/21	II
T4	62 01.7	23 55.4	ombro	23.5	41	52/20	II
C2	62 00.1	23 54.3	ombro	10.6	58	216/0	II
T5	61 59.7	23 56.5	oligo	34.8	38	123/170	II
C3	61 59.8	23 56.2	oligo	17.8	41	131/145	II

Table 2: Mean winter concentrations of DOC, N and P (December–April) in % of the mean summer concentrations (May–November) in pristine, drained and restored peat sites in Finland (study II). OC is organic carbon, TOC for pristine and restored sites, DOC for drained sites. For the different data sets, see manuscript II.

	OC	NH_4-N	$NO_{2\text{-}3}$-N	N_{tot}	PO_4-P	P_{tot}
Pristine	80	79	130	81	91	98
Drained	83	110	130	88	71	66
Restored	79	94	130	80	79	79

mean concentrations (Eq. 2 in I) during a calibration period. the length of the calibration period varied between 6 and 18 months between the different catchments. Separate means were calculated for spring (December-May) and autumn (June-November) to accommodate the changing hydrological conditions over the year. Yearly export was calculated using the measured and interpolated concentration data and daily simulated runoff data (Eq. 1 in I). The impact was then calculated as kg per restored area (ha^{-1}) by dividing the result with the proportion of restored peatlands in the catchment (Eq. 4 in I). An index for the annual impact on the export was calculated to take into account the different runoff in each year by dividing the excess export with the expected background export (Eq. 5 in I). There were seven post-treatment years available in the data for both sites, Mustakorpi and Seitseminen.

In study II, a treatment-control catchment setup was applied. For each catchment in the study, yearly runoff was partially measured with a water level logger in a measurement weir and partially simulated with the FEMMA 2-d process-based model (Koivusalo et al., 2008). The need for simulation arose from the fact that the sites were difficult to access during wintertime and the snow melt period. This prevented sapmling of water during wintertime. Due to the risk of instrument breakage through freezing the loggers were removed from the measurement weirs approximately at the end of November and installed again at the end of April each measurement year. The concentration values were interpolated for the missing days between the first and last sampling date every year.

To make the estimates of wintertime exports more robust in study II, external data was used to estimate wintertime concentrations (December-April) relative to the mean concentrations during the previous measurement season (May-November). Significant seasonality was found in N, P and organic carbon concentrations in all types of catchments (pristine, drained, restored) (Table 2). The winter concentrations were 81–98% of the summer concentrations, except for $NO_{2\text{-}3}$-N on all site types and NH_4-N on drained sites, where the winter concentrations were 30% and 10% higher than the summer concentrations, respectively. The annual winter concentrations were produced by multiplying the mean summer concentration in our catchments by the average winter/summer ratio in the external data (Table 2) using the corresponding drainage status (pristine, drained, restored) as in our treatment catchments during the calibration period (drained) and treatment (restored) periods, and as in our control catchments (C1 and C2 pristine; C3 drained).

The measured, interpolated and calculated concentrations were paired with the daily measured and simulated runoff data and summed to calculate yearly export of the elements. In the data, there were three calibration years and four post-treatment years available for all catchments excluding T2, for which there was no runoff data or control catchment available and thus no annual export or background export could be estimated.

To estimate the impact of restoration on DOC and nutrient export in study II, yearly background export of DOC, N and P without restoration was estimated for the post-treatment years for the treatment catchments. Three models per treatment catchment and element were created, one for each control catchment used in the study (Eq. 1)

$$C_e = a_{e_i} \times C_{e_i} \tag{1}$$

, where C_e is the export of element e from the treatment catchment during the calibration period, and C_{e_i} is the export of element e from the control catchment i during the calibration year, $i = 1 \ldots 3$. We did not include an intercept term in the model as a reasonable assumption is that when export from the control catchment approaches zero, the export from the treatment catchment should also be close to zero (Laurén et al., 2009). The model's slope term a_{e_i} was then used to predict the annual background exports for the treatment catchments during the treatment period (Eq. 2)

$$B_{e_i} = a_{e_i} \times L_{e_i} \tag{2}$$

, where B_{e_i} is the calculated background export of element e from the treatment catchment during the treatment period using control catchment i and L_{e_i} is the export of element e from the control catchment. The impact of restoration treatment in kg per restored area (ha) was calculated as

$$E_{e_i} = \frac{L_{e_i} - B_{e_i}}{(A_p/A_{tot})} \tag{3}$$

, where E_{e_i} and is the restoration-induced export of element e from the treatment catchment based on control catchment i, and A_p and A_{tot} are the peatland area and total area of the treatment catchment, respectively. Including A_p and A_{tot} in the equation means that the impacts of restoration are expressed against the restored peatland area rather than the total catchment area.

The agreement between these three models was then used for estimating the reliability of the impact of restoration. When all three models predicted treatment load for a year, the load was considered to exist; otherwise it was considered not significant.

2.2 Assessing the effect of peat properties on rewetting-induced release of DOC and nutrients from drained peat (III)

2.2.1 Study sites and sampling

Peat cores for study III were collected from three nutrient poor and three nutrient rich drained peatland sites, two of each in Finland and one of each in Ireland (Table 3). The Finnish sites were located in south-central Finland and the Irish sites in western Ireland. The sites had been in a drained sites in some cases for over 100 years (P_{F2} and R_{F2}) prior to the study.

Peat samples were collected from the six sites (Table 3) using PVC tubes to be incubated under two water-level regimes (WT), high and low. Four (Finland) or five (Ireland) replicates were made of each site and WT, summing up to 52 peat cores in all. In the high WT cores, the water level was kept at approximately the peat surface level (waterlogged or re-wetted conditions), while in the low WT cores, the water level was at 35 cm below the peat surface (aerobic conditions). The cores were kept at an average temperature of 18 C for the duration of the experiment, about 25 weeks.

Table 3: Basic information on the study sites used in study III. $_F$ in site code indicates site in Finland, $_I$ in Ireland. Location in WGS84 coordinates.

	Nutrient-poor			Nutrient-rich		
	P_{F1}	P_{F2}	P_I	R_{F1}	R_{F2}	R_I
Location	61 47N, 24 18E	62 04N, 24 34E	54 00N, 09 32W	61 47N, 24 18E	62 04N 23 34E	53 85N, 09 31W
Dominant tree species	*Pinus sylvestris*	*Pinus sylvestris*	*Pinus contorta*	*Betula pubescens*	*Pinus sylvestris*	*Picea sitchensis*
Stand volume, m^3 ha^{-1}	40	150	370	130	120	400
Year of drainage / afforestation	1961	1909	1970	1961	1909	1970
Dominant field layer vegetation	*Calluna vulgaris*	*Ledum palustre*	*Calluna vulgaris*	*Vaccinium vitis-idaea*	*Hylocomium splendens*	*Calluna vulgaris*
	Empetrum nigrum	*Vaccinium uliginosum*	*Molinia caerulea*	*V. myrtillus*	*Brachythecium spp.*	*Molinia caerulea*
	Vaccinium uliginosum	*Empetrum nigrum*	*Eriophorum angustifolium*	*V. uliginosum*		
			Melampyrum pratense			
			Trientalis europaea			
Peat type	*Sphagnum*	*Sphagnum*	*Sphagnum*	*Sphagnum-Carex*	*Carex-Phragmites*	*Sphagnum*

2.2.2 Analysis

Water samples of 20-30 mL were collected from the cores with suction samplers using a suction of approximately 100 kPa from 10-19 cm below the peat surface over the course of 1-2 days per sampling. Because it can take several weeks to establish anaerobic conditions in rewetted peat after raising the WT (Zak & Gelbrecht, 2007), the first water samples were collected 10 weeks after the experiment began, and then every 2-4 weeks, totalling to 8 samples per tube in Finland and 11 in Ireland. The sample volume and evaporation loss was compensated for by adding deionised water to the surface. After filtering first with filter paper (Schleicher and Schuell 589^2) and then with 0.45 μm^2 membrane filters (Gelman Supor–450, Pall Corp., Port Washington, NY, USA), the samples were then analysed for their pH and Eh7, soluble reactive P (SRP), DOC, Fe, NH_4-N and NO_3-N.

The peat in the cores was analysed for its C, N, P, Al, Fe and Ca concentrations. Peat samples in their original moisture content were analysed for their easily soluble NO_3-N and NH_4-N using KCl extraction; and for soluble P, redox-sensitive P (P_{BD}) and Fe (Fe_{BD}), acid-soluble P and alkali-soluble P with the method of Psenner et al. (1984), as modified by Zak et al. (2008). Dried and milled samples were analysed for oxalate-extractable Fe (Fe_{ox}) and Al (Al_{ox}), as in Nieminen & Jarva (1996).

Microbial biomass and N and C mineralisation potential in the peat samples were analysed, the mineralisation potential as described by Priha & Smolander (1997) and the biomass by fumigation-extraction with chloroform (Brookes et al., 1982; Vance et al., 1987; Priha & Smolander, 1997).

DOC extracted from the top 20 cm of the peat was fractionated into weak (phenolic) and strong (carboxylic) hydrophobic acids (WphoA and phoA, respectively), hydrophilic acids and bases (phiA and phiB, respectively) and hydrophilic neutrals (phiN), according to Qualls

Table 4: Location (Latitude and longitude, WGS84 grid) and volume of tree stand (m^3 ha^{-1}) on the sites of the CH_4 dynamics study (IV). The stand volume on site RE3 was measured in 2007, the other sites in 2010.

site	Lat	Lon	*Picea abies*	*Betula pubescens*	Total
PR1	61.86	24.24	256	3	259
PR2	61.24	25.06	261	19	280
DR1	61.80	24.30	278	22	300
DR2	61.38	25.11	258	62	319
RE1	61.23	25.07	181	1	181
RE2	60.67	23.87	0	29	29
RE3	60.30	24.45	126	59	185

& Haines (1991) and Kiikkilä et al. (2013).

2.3 Estimating the impact of restoration on CH_4 dynamics (IV)

2.3.1 Study sites

The study on CH_4 dynamics (IV) was conducted on seven peatland sites in Southern Finland (Table 4), two of which (PR1, PR2) were undrained spruce swamp forests, two were drained (DR1, DR2) and three were restored (RE1, RE2, RE3) after a period of drainage. Both the drained and the restored sites had been drained for several decades. The restoration measures on the restored sites had been conducted 11, 17 and 11 years prior to our measurement campaign, respectively. The measures included filling in and/or damming of the ditches, but not removal of the tree stand.

CH_4 measurements were made on four locations at each site, each location comprising two round sampling plots (diameter = 30 cm). On each plot, a 2-cm deep groove was carved into the soil for the measurement chamber (sheet metal, round chamber, diameter = 30 cm, height = 30 cm, with a small fan in the ceiling) to ensure an air-tight connection between chamber and soil. On the drained and restored sites, two of the locations were in the mid-strip area (MID), one was on the area beside the ditch (DS) and one was in the ditch (DI). On the pristine sites, the four locations were on a transect perpendicular to the mire edge, one location being on the mire edge (Fig. 1). Wooden platforms were constructed adjacent to the sampling plots on the pristine and restored sites during the previous summer before the measurement campaign.

2.3.2 Calculations

CH_4 emissions were calculated from manual opaque closed-chamber measurements with discrete gas samples drawn into glass vials 5, 15, 25 and 35 minutes after placing the chamber on the soil. The data for the study was collected during one growing season, in 2012, twice per month. The gas samples were analysed for their CH_4 concentration at the laboratory of the

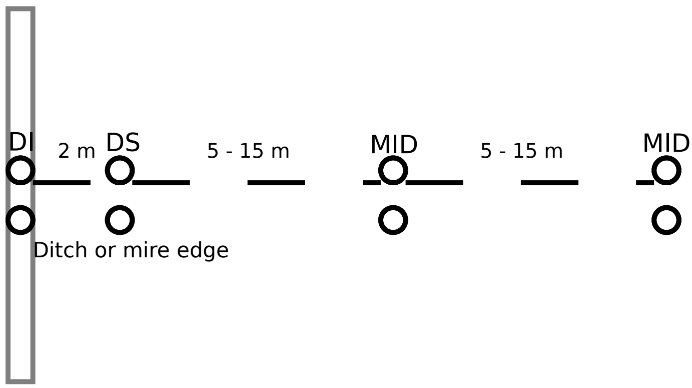

Figure 1: Measurement site sampling design of the CH_4 dynamics study (IV). Open circles represent measurement plots. Dashed line represents distance between measurement plot groups.

Finnish Forest Research Institute at Vantaa, Finland using a gas chromatograph fitted with an FI-detector for CH_4. The measurements were run and analysed with the Openlab CDS ChemStation program, Rev. C .01.03.

The concentration measurements were first checked visually and by fitting a linear function to the concentration values over time for ebullition or vial leakage. As there was no way to decide whether the ebullition was caused by the presence of the measurerer or by natural causes, all measurements with ebullition were rejected. 17% of the 290 measurements were rejected, mostly due to ebullition evident in the first three gas samples. In case of vial leakage, a measurement was considered valid if only one sample was discarded. After filtering the data, the change in CH_4 concentration during each measurement was estimated linearily from the accepted samples. The CH_4 flux (mg CH_4 m^{-2} d^{-1}) was then calculated using the slope of the linear function, the height of the chamber and the mean air temperature in the chamber during the measurement.

Water table levels were manually measured in each site during each measurement round. Each CH_4 measurement was associated with the WTL measured from the nearest measurement well.

The effect of treatment and measurement location on the CH_4 flux was estimated with a linear mixed effects model (Eq. 4)

$$F = \beta_0 PR + \beta_1 DR\text{-}DI + \beta_2 DR\text{-}DS + \beta_3 DR\text{-}MID + \beta_4 RE\text{-}DI + \beta_5 RE\text{-}DS + \beta_6 RE\text{-}MID + \varepsilon_{ij} \quad (4)$$

, where F is the CH_4 flux (mg CH_4 m^{-2} d^{-1}) and $\beta_{0...6}$ are the coefficients (parameters) that define the mean flux values over the growing season for pristine (*PR*), drained-ditch (*DR-DI*), drained-beside-ditch (*DR-DS*), drained-mid-strip (*DR-MID*), restored-ditch (*RE-DI*), restored-beside-ditch (*RE-DS*) and restored-mid-strip (*RE-MID*) management-plot pairs; and e_{ij} is the random effect of the measurement plot.

The effect of sampling location (DI, DS, MID) on CH_4 flux in the drained and restored sites was estimated by pairwise comparison between the appropriate management-location pairs. An average flux for the whole peatland area (mg CH_4 m^{-2} d^{-1}) was estimated assuming area proportions for the different locations of 3%, 6% and 91% for DI, DS, and MID, respectively. On pristine sites, 100% was allocated for location PR.

The effect of treatment on WTL was estimated with a linear mixed effects model (Eq. 5)

$$W = \beta_0 PR + \beta_1 DR + \beta_2 RE + e_{ij} \quad (5)$$

, where W represents the mean WTL over the measurement period; $\beta_{0...2}$ are the parameter values for pristine (*PR*), drained (*DR*) and restored (*RE*) sites, respectively; and e_{ij} is the random effect of the site and WTL measurement well. To get comparable results for each treatment, the WTL measurements from the ditches of the drained sites (DR1, DR2) were excluded from this estimation.

3 RESULTS

3.1 Leaching of nutrients and organic carbon (I, II)

Results from the two catchment-level studies were somewhat different. In study I, restoration of the fertile Mustakorpi catchments had higher impact of restoration on the exports of TOC and N whereas resotration of the poorer Seitseminen catchments had higher impact on the export of P (Table 5). In study II, the restoration of the fertile spruce-dominated catchment had a high impact on the exports of DOC, N_{tot}, NH_4-N, P_{tot} and PO_4 (Table 5). In the poorer catchments, much smaller impacts on DOC were observed in catchments T4 and T5, as well as impacts on N_{tot} and P in all poor catchments and on NH_4-N in catchments T3 and T4 (Table 5). The concentrations of elements in runoff from catchment T2 were also much higher post- than pre-restoration, which implicated high impact of restoration on export of DOC, N and P (Fig. 2).

The export of DOC from catchment T1 was highest during the first year after restoration, after which the impact was no longer significant according to the background export models (Fig. 3). The impact of restoration on exports of PO_4-P and P_{tot} was also highest in the first post-restoration year, but the it waned only gradually and was still significant in the last study year in catchment T1 and in the third post-restoration year in catchment T4 (Fig. 3). The impact on NH_4-N was largest in the third post-restoration year in catchment T1 and in the second post-restoration year in catchment T3. In contrast, in study I, the highest impact on PO_4-P and P_{tot} in the fertile Mustakorpi catchment were observed in the fourth post-restoration year, with the impacts gradually falling after that. The impacts on TOC and N followed roughly the same temporal pattern as in study II (Figs. 3, 6 and 8 in I; Fig. 3).

Table 5: Impacts of restoration on export of organic carbon (OC; TOC in study I, DOC in study II) and nutrients excluding catchment T2, for which no runoff data was available. Expressed as mean impact (kg restored ha^{-1} y^{-1}) over the study periods, 6 years in Mustakorpi and Seitseminen, 4 years in others. - means impact not significant in any post-treatment year.

		Average annual impact over 6 (I) or 4 (II) years					
Site	Study	OC	Ntot	NH4-N	NO23-N	Ptot	PO4
Mustakorpi	I	150	3.6	0.8	<0.1	0.3	0.2
Seitseminen	I	116	2.4	0.1	0	0.4	0.3
T1	II	327	16.1	2.4	<0.1	3.8	3.1
T3	II	-	0.4	0.3	<0.1	<0.1	<0.1
T4	II	15	1.4	0.1	<0.1	0.7	0.6
T5	II	13	1.55	-	<0.1	<0.1	<0.1

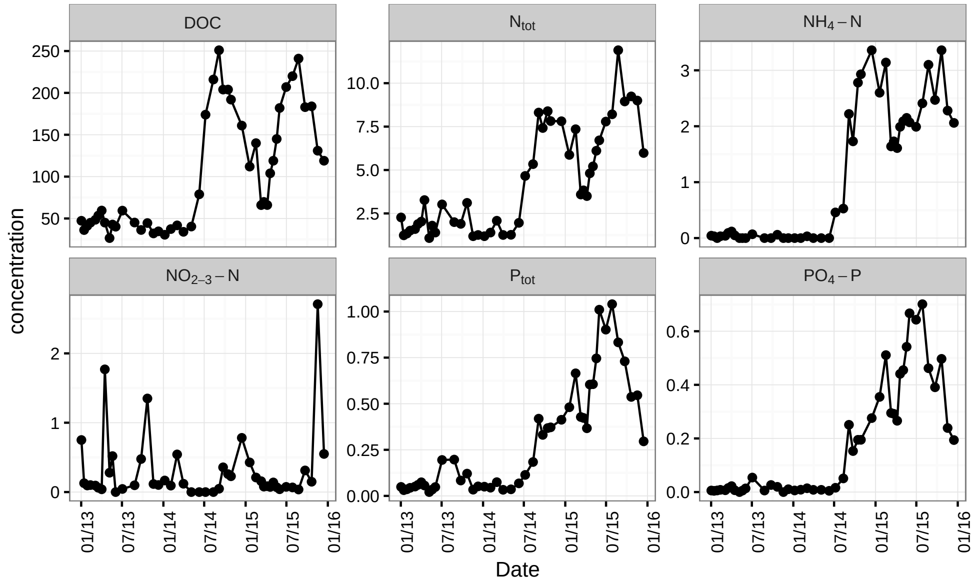

Figure 2: Concentrations of DOC, N and P (mg l^{-1}) in runoff from catchment T2 in study II. Restoration measures took place during summer 2014.

3.2 Factors affecting the release of DOC and nutrients from rewetted peat (III)

With regard to soil pH, all of the sites in study III were quite similar, with pH range 3.7–4.1 (Table 6). The total N contents were clearly lower in the Finnish nutrient-poor sites P_{F1} and P_{F2} (1.0–1.6%) than in the other sites (2.4–3.0%). The C/N ratios of peat varied widely between 50 and 18, with the highest ratios for P_{F1} and P_{F2},and the lowest for R_I. The total soil P concentrations of the nutrient-poor sites were 40–60%, the total Al concentration 20–80%, but the total Fe concentration only 3–7% of the corresponding concentrations in the nutrient-rich sites. The peat from the poor sites was generally more rich in easily soluble P, while in the rich sites more acid-soluble and NaOH-soluble P was found (Table 7). In redox-sensitive P (P_{BD}) content, no significant difference was found between the poor and rich sites (Table 7). Fe and Al were much more abundant in the rich sites than in the poor sites, Fe both in oxalate-extractable (Fe_{ox}) and redox-sensitive (Fe_{BD}) forms (Table 7).

The incubation experiment showed that the release of DOC, Fe and nutrients is in general much higher under anaerobic than under aerobic conditions. The variation between sites in the release of DOC, N and P under anaerobic incubation was high (Fig. 4). The release of Fe, DOC and NH_4-N was closely related to the decrease in Eh7 observed in the columns (Fig. 4). The low Eh7 reached in samples from site R_{F2} coincided with the highest concentrations (mmol l^{-1}) of Fe, DOC and NH_4-N observed in the study. In contrast, the highest concentrations of SRP were observed in the samples from sites R_I and P_{F2}, where the Eh7 did not fall below 200 mV, and almost no release of SRP was observed in the samples from the nutrient-rich sites in Finland (Fig. 4).

The release of DOC under anaerobic incubation (high WT) was most closely related to

Table 6: Peat properties in study III. For the site characteristics, see Table 3.

Chemical parameteres	P_{F1}	P_{F2}	R_{F1}	R_{F2}	P_I	R_I
Bulk density, g cm3	0.08	0.15	0.08	0.18	0.09	0.11
pH	3.9	3.7	4.1	3.8	3.9	3.7
C, %	50	55	55	54	53	53
N, %	1.0	1.6	2.6	2.4	2.4	3.0
P, mg kg^{-1}	520	530	870	880	420	1140
Al, mg kg^{-1}	410	690	1910	1150	820	1030
Fe, mg kg^{-1}	1210	780	13500	24050	730	11070
Ca, mg kg^{-1}	1650	3170	2170	1920	1010	320
NH_4–N_{KCl}, mg kg^{-1}	90	45	130	125	200	220
NO_3–N_{KCl}, mg kg^{-1}	0.3	0.1	0.1	19.5	0.4	3.1
Al:P-molar ratio	0.9	1.5	2.5	1.5	2.2	1.0
Fe:P-molar ratio	1.3	0.8	8.6	15.1	1.0	5.4

Table 7: Phosphorus and iron fractionation results (mg kg^{-1}) according to Psenner et al. (1984) modified by (Zak et al., 2008) in study III.

Profile	P_{NH_4Cl}	P_{BD}	Fe_{BD}	P_{HCl}	P_{NaOH}	Fe_{BD}:P_{BD}	Al_{ox}	Fe_{ox}
P_{F1}	21	48	99	15	171	1	230	1030
P_{F2}	24	45	23	15	189	0	370	550
R_{F1}	2	33	1144	67	315	19	1300	13200
R_{F2}	3	79	4660	69	389	33	860	21200
P_I	70	52	42	15	127	0	390	530
R_I	9	97	1120	186	411	6	630	11000

the peat Fe content (Fig. 5). No effect of microbial biomass, C mineralisation rate (Table 5 in III) or the DOC fractions (Table 6 in III) on the DOC release rate was discernible. The Fe and DOC concentrations also changed simultaneously in the same direction in the columns (Fig. 4).

P release was the highest in the columns from the sites with the smallest ratio of redox-sensitive Fe to redox-sensitive P in the peat (Table 6, Fig. 4), P_{F1} and P_{F2} and R_I. Comparably very little P was released from the iron-rich peats from sites R_{F1} and R_{F2}.

3.3 CH_4 dynamics (IV)

11 and 17 years after restoration, restored spruce swamp forests can be large sources of CH_4 into the atmosphere. Emissions from all sites of all management histories were highly variable (Fig. 6), and the distribution of emission rates was skewed to the right. However, the emissions from the mid strip measurement plots of the restored sites were 34 times higher than from the pristine sites (Table 8) and comparable to the emissions from the ditches in the drained sites.

The WTL was highest in the restored sites, although not significantly higher than in the

Table 8: Site management options (Management, PR = pristine, DR = drained, RE = restored), plot locations (Location, DI = ditch, DS = beside ditch, MID = mid-strip), parameter names for each management- location pair (Par.), parameter values (mg CH_4 m^{-2} d^{-1}) and standard errors (S.E.), significance (*p*) of parameter differences from pristine for Eq. (4)), percentage of area represented by each location (Area represented, %), and area-weighted fluxes per management category (flux per total area, mg CH_4 m^{-2} d^{-1}).

Management	Location	Par.	Par. value	S.E.	p	Area (%)	Area flux
PR		β_0	1.51	10.86		100	1.51
DR	DI	β_1	75.83	23.74	0.007	3	
	DS	β_2	-0.41	20.98	0.936	6	
	MID	β_3	-0.18	12.85	0.920	91	
	Total					100	2.09
RE	DI	β_4	52.04	19.28	0.027	3	
	DS	β_5	66.05	20.90	0.009	6	
	MID	β_6	51.99	14.70	0.009	91	
	Total					100	52.84

pristine sites. The variation in WTL was highest in the drained sites. (Fig. 7)

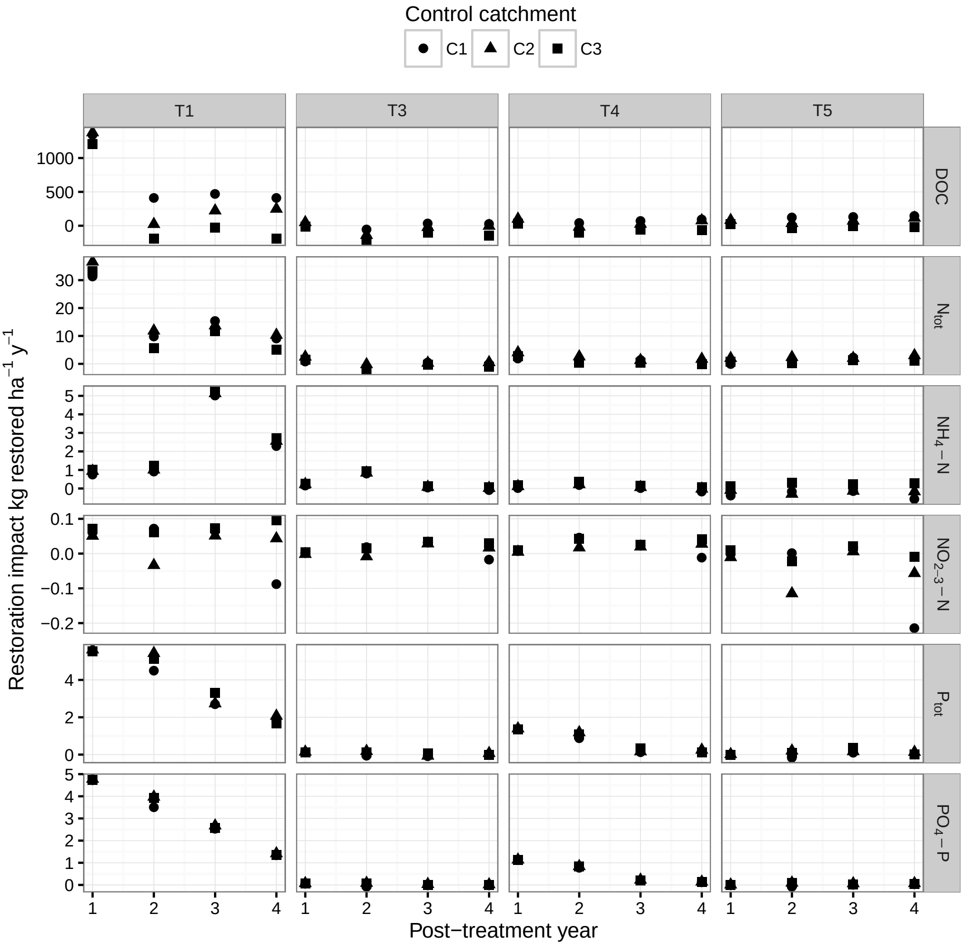

Figure 3: The annual impact of restoration on the exports of DOC, N and P from the treatment catchments in study II (kg restored ha^{-1} y^{-1}). Symbols indicate different control catchments used in background export calculation.

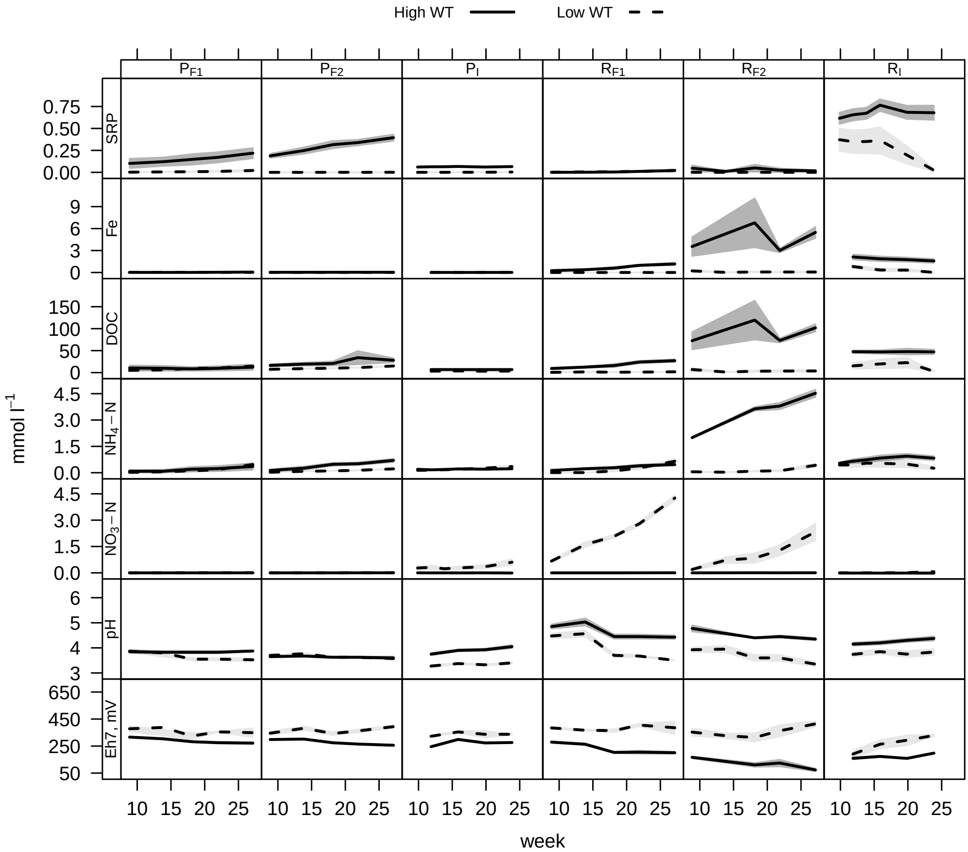

Figure 4: SRP, Fe, DOC, NH_4-N, NO_3-N (mmol l^{-1}), Eh7 (mV) and pH in pore water during incubation in study III. High WT columns: average (solid line), SE (dark grey area); low WT columns: average (dashed line), SE (light grey area).

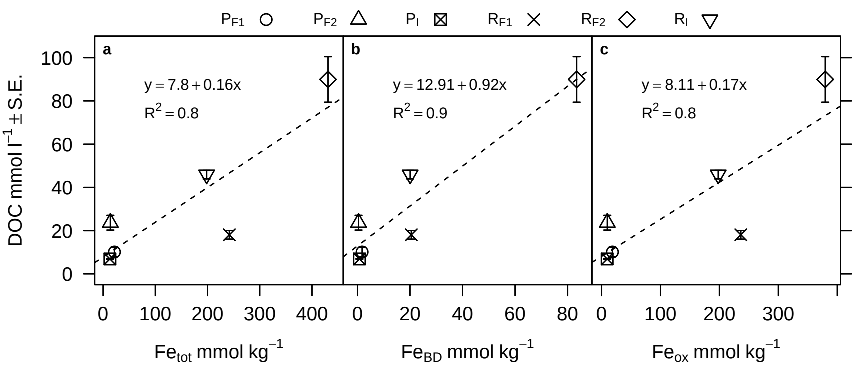

Figure 5: Average ± SE DOC content in pore water in high WT columns during incubation versus total Fe (**a**), Fe_{BD} (**b**) and Fe_{ox} (**c**) in peat in study III.

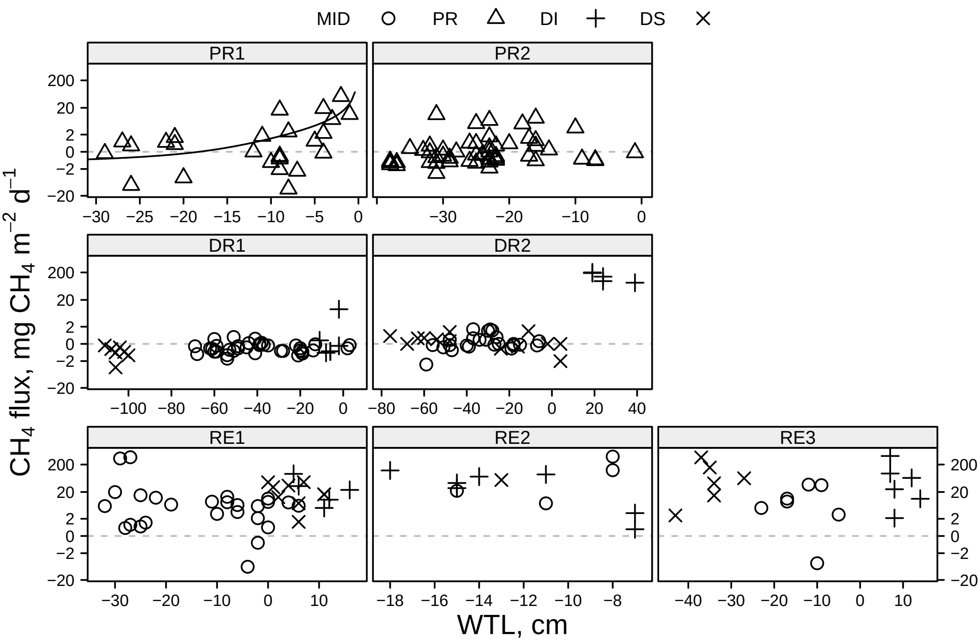

Figure 6: CH_4 fluxes (mg CH_4 m^{-2} d^{-1}) *versus* water table level (WTL, cm) in study IV grouped by site (site management codes: PR = pristine, DR = drained, RE = restored). Different colours of the points indicate different plot locations within the sites (MID = mid strip, PR = pristine, DI = ditch, DS = beside ditch). Regression curve (solid line) in PR1 shows the inverse relationship between WTL and CH_4 emissions ($F = -1.6 + (-30.3/WTL)$, p = 0.007). Note the hyperbolic arc-sine scale of the y-axis and the different x-axis scale in each panel. Y-axis values have been back-transformed to show real measured fluxes. Negative x-axis values indicate WTL below peat surface.

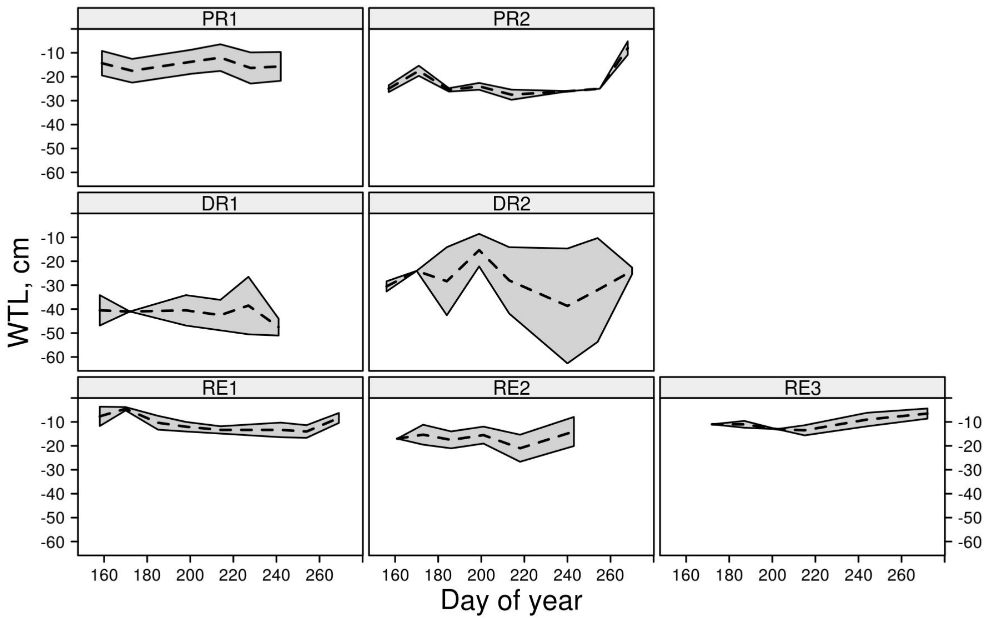

Figure 7: Time series of mean (dashed line) and sd (grey area) of water table level (WTL, cm) in the dip-wells at different measurement sites in study IV during the summer 2012 (day of year 160–270). Negative y-axis values indicate WTL below the peat surface. Ditch wells in the drained sites are excluded. Site codes: PR = pristine, DR = drained, RE = restored.

4 DISCUSSION

4.1 Does restoration decrease runoff water quality (I, II)

High impacts of rewetting on the biogeochemistry of forestry-drained peatlands were found in studies I and II. To put the impacts on OC and nutrient exports into context, a comparison to the impacts of other options available in managing drained peatland forests is in order. As most of the drained peatland forests in Finland have been drained already several decades ago, they are approaching the phase when the tree stocks will be harvested, the ditches in the sites will be consequently maintained and the sites will be prepared for regeneration. The major operations that have consequences for water quality in this chain are harvesting and ditch maintenance (e.g. Nieminen, 2004; Joensuu et al., 2002).

The paired catchment-approach suggested by Laurén et al. (2009) was applied in Palviainen et al. (2014) to assess the impacts of harvesting and site preparation on three northern forestry-drained peatlands. Of the substances studied here, they reported increases in N_{tot}, NO_3-N and PO_4-P in one catchment, decrease of NO_3-N in another and increase of PO_4-P in the third catchment. The impacts of the treatments lasted for over 10 years on all sites, adding up cumulatively per treated hectare to 1.2 kg N_{tot}, 0.008 kg and 0.011 kg PO_4-P and increase of 0.47 kg and decrease of 0.1 kg NO_3-N, respectively. The treatment impacts were thus much lower than what is reported here in studies I and II (Table 5). It should be noted, however, that the sites in Palviainen et al. (2014) were located in the north, where climatic conditions are cold and nutrient deposits low. Furthermore, the treatments in the study had peatland buffer zones before the water sampling points, which probably reduced the impacts somewhat. Earlier studies (Grip, 1982; Ahtiainen, 1990; Ahtiainen & Huttunen, 1999; Rosén et al., 1996; Haapanen et al., 2006; Mattsson et al., 2007; Löfgren et al., 2009) referred to by Palviainen et al. (2014, Table 4) reported annual impacts on N_{tot} export between 0.4 and 4.9 kg ha^{-1}; on NH_4-N export between 0.02 and 0.66 kg ha^{-1}; on P_{tot} export between 0.02 and 0.7 kg ha^{-1}; and on PO_4-P export between 0.01 and 0.5 kg ha^{-1}. These impacts were in the range of the annual impacts reported here excluding the NH_4-N impacts on spruce swamp sites Mustakorpi and T1 (Table 5), and N_{tot}, P_{tot} and PO_4-P impacts on site T1, which were much higher. Nieminen (2004) reported impacts of 80 and 184 kg DOC ha^{-1} over three years in two Norway spruce-peatland dominated catchments of which approximately 40% and 72%, respectively, were clear-cut. This would give an approximate annual impact of 66 and 85 kg DOC per harvested hectare, which is much less than the impact in the fertile spruce-dominated catchments in this study (Table 5). Respectively, the approximate annual impact on NH_4-N reported by Nieminen (2004) was 0.3 and 0.7 kg per harvested ha, which is more than restoration impact on NH_4-N in the nutrient-poor sites in this study (Table 5), but less than the impact in Mustakorpi and much less than in T1.

The very high impact of restoration on exports of DOC and P in catchment T1 in study II is supported by the changes in DOC and P concentrations in catchment T2 in the same study. Calculated into yearly export assuming the similar average pre- and post-restoration runoff values as in T1 and estimating the impact as roughly the difference between pre- and post-restoration yearly export, the impact in site T2 is perhaps even larger on DOC than in

T1, and slightly less than in T1 on P. High impact on P concentration has also been observed by Sallantaus (2014) in a catchment with 20% restored spruce swamp forests.

4.2 Factors affecting the release of DOC and nutrients from rewetted peat (III)

Regarding the processes behind the impacts of restoration on runoff water quality, study III supports the argument by Grybos et al. (2009) that redox reactions, mediated by microbes, are the main drivers behind DOC release, rather than microbial decomposition of organic substances as such. The simultaneous release of DOC and Fe (Fig. 4) supports the idea that the reducing of Fe^{III} to Fe^{II} breaking up the R-Fe^{III}-R-associations is a major source of DOC under reducing conditions. Higher release of DOC from peat with higher Fe content has also been reported by Zak & Gelbrecht (2007) and Urbanová et al. (2011). The main source of P release from nutrient-poor peats is apparently the easily soluble P pool, as the size of this pool was the main difference between the poor and rich sites (Table 7). This interpretation is supported by the high concentration in P in the soil solution of the cores from the nutrient-poor sites (labelled P in the tables and figures) already in the first samples in the study (Fig. 4). Redox conditions are also a major influence on the release of P, as the redox-sensitive pool of P (P_{BD}) is potentially released under reducing conditions, as happened in the samples from R_I in study III (Table 7, Fig. 4). This can be prevented, however, by a large enough pool of Fe or Al in the peat, as they can reabsorb the released P, the valence of Al being insensitive to reducing conditions (Darke & Walbridge, 2000). The low Fe_{BD}:P_{BD} ratio, the relatively high easily soluble P pool and the relatively small Al pool in the R_I peat (Tables 6, 7) could explain the much higher release of P from those cores than the cores from the other rich sites. The release of NH_4-N was highest in the columns from the site R_{F2}, where the peat content of NO_3-N prior to incubation was also the highest (Table 6, Fig. 4). In fact, no other peat property was found relevant to the release of NH_4-N.

4.3 Impact of restoration on CH_4 emissions (IV)

The impact of restoration on CH_4 fluxes (IV) was surprising, as the previous studies on restored forestry-drained peatlands had reported only low CH_4 emissions from spruce-dominated sites (Juottonen et al., 2012) and in the undrained state, spruce swamps are among the smallest sources of CH_4 of undrained peatlands (Huttunen et al., 2003). Peat samples taken from two of the three restored sites in this study as part of the study by Maanavilja (2015) revealed higher bulk density (BD) in the rewetted than the undrained sites. This could lead to slow movement of water in the rewetted surface peat layers, and consequently anoxic, reducing conditions close to the surface where the increased input of readily decomposable substrate (Maanavilja et al., 2015) provides good conditions for CH_4 production. Study IV was, however, limited in scope as there were only three restored sites and the study lasted only for one growing season. Therefore it cannot be ruled out that the emissions observed on the restored sites were somehow exceptional and not representative of the CH_4 dynamics of drained and restored spruce swamp forests in general.

High emissions from restored peatlands have also been reported in studies on restored cut-over peatlands (Wilson et al., 2009, 2013), when the water level was close to the soil surface.

Especially a fluctuating water table near the soil surface causing periods of inundation with intermittent periods of plant growth has been connected to high CH_4 emissions after restoration on different sites (Vanselow-Algan et al., 2015; Koh et al., 2009; Hahn-Schöfl et al., 2011). The higher WTL on the restored sites than on the undrained sites in study IV suggests that these conditions could be at least partially responsible for the high CH_4 emissions observed here.

4.4 Connections between the studies

According to the results of the studies presented here, peat properties reflected as the fertility level and affected by drainage and redox conditions after water level rise are key to the impact restoration of forestry-drained peatlands has on the runoff water quality. Low Eh7 is also a prerequisite for microbial production of CH_4. In study III, the lowest Eh7 was reached in the peat columns from the sites where the peat was the most rich in Fe, R_{F1} and R_{F2}. These were also the columns from which the most DOC, Fe and NH_4-N were released. That the risk of increased NH_4-N exports is the highest on nutrient-rich sites was evident in all three studies concerning release of nutrients: aside from study III, the nutrient-rich sites in studies I and II also had the highest exports of NH_4-N.

4.5 Recommendations for restoration of forestry-drained peatlands

The studies indicate that nutrient-poor (oligo-ombrotrophic) sites are often at risk of increased export of P after restoration, while on nutrient-rich sites there can be high release of any of the three elements (DOC, N, P) studied here. These impacts could at least initially be high enough to compromise the improvement in water purification processes, which is one of the expected positive outcomes of restoration of forestry-drained peatlands. In particular, the very high exports from some fertile sites (II) raise concern as these are among the most biodiverse wetland ecosystems and thus encounter specific restoration need. Even though these restored swamps will later turn from sources to nutrient sinks, it will plausibly take a very long time before the sinks fully compensate for the nutrients initially released due to restoration measures, as well as before the receiving water courses fully recover from the impacts of restoration. Furthermore, as wetlands (both pristine and restored) rather produce DOC into water courses that retain it from surrounding mineral soil areas, it may be, because of initial release of DOC and no later retention, that the overall effect of restoration is increased DOC input into receiving water courses.

Whether there is a risk of very high P export seems to depend on whether the peat Fe and Al suffice to re-adsorb the released easily-soluble and redox-sensitive P. The risk of high exports of DOC and P could be connected to the fact that spruce swamps usually have large mineral soil up-slope catchments, which supply them with substantial water input after the ditches bordering the peatland and the mineral soil catchment are filled in. This in connection with the decomposed and poorly water-conducting surface peat could create a situation where the water level rapidly rises, perhaps to higher level than in an undrained state, and anoxic conditions form in the surface layers of the restored swamp. Besides increasing the release of water-borne nutrients and DOC, these pools may increase the emissions of CH_4 (IV).

Small brooks or other discharge channels are often present in spruce swamps in their pristine state. These are often straightened and deepened in connection with the drainage operations and are treated as any other ditch in connection to the restoration operations. It should be investigated whether these could instead of being completely filled in and/or dammed, be restored with less intensive measures. These measures could allow some surface flow to occur and the very surface peat layers to remain partly aerobic after restoration, such as they plausibly are in respective pristine swamps. This could also prevent the high CH_4 emissions observed in study IV.

Redox conditions in peat have not thus far been connected *in situ* to exports of nutrients or CH_4 emissions in drained peatland forests restoration status notwithstanding; neither have the conditions present in restored forestry-drained peatlands been compared to those present in undrained sites. This is a definite gap in the understanding of processes connected to water quality in restored sites. In addition, active mitigation measures such as two-staged restoration where a small portion of the catchment is first restored to provide a wetland buffer zone for the main restoration operations, should be investigated.

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