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Civil Engineering
National University of Ireland, Galway

CYCLING AND TRANSPORT OF PHOSPHORUS AND NITROGEN
FROM HARVESTED PEATLAND FORESTS AND POSSIBLE
MITIGATION METHODS

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degree of Doctor of Philosophy

October 2012
Abstract

Drainage of peatlands has been part of forestry practice since the early 20th century. In Ireland and the UK, approximately 800,000 ha of peat catchments were drained and afforested between the 1950s and 1990s. A large number of these forests are now approaching maturity and the rate of forest harvesting will undergo a rapid increase in the near future. Nutrient release following harvesting of upland blanket peat forests in western Ireland has become a significant environmental concern, as these forests drain into ecologically valuable, oligotrophic, salmonid rivers. Therefore, the objectives of this study were to assess the mechanisms of release and transport of phosphorus (P) and nitrogen (N) from harvested peatland forests and the effectiveness of various mitigation methods to decrease nutrient export from them.

A litterbag study was used to investigate the decomposition and P and N release dynamics of harvest residue needles on different peat sites and a laboratory incubation study assessed the mobilization of P and N from peat due to a rise in water table. Laboratory flume and mini-catchment experiments investigated the effect of harvest residue brash windrows on nutrient export from harvested blanket peatlands and nutrient retention efficiency of blanket peatland buffer areas.

The results suggest that P is easily released from decaying harvest residue needles, whereas N is mostly retained or accumulated during the initial stages of decomposition. Therefore, the release of N from harvest residue needles is not a likely source of the increased N export following clear-cutting, but harvest residues may contribute to P losses. High water tables produced anoxic conditions in upper peat layers and resulted in P mobilization from moderately decomposed, nutrient-poor, Irish blanket peat and highly decomposed, nutrient-rich, Irish blanket peat. In peats from Finland, P mobilization occurred only from nutrient-poor, ombrotrophic peats, and not from nutrient-rich, minerotrophic peats. A high water table also caused higher mobilization of ammonium-N (NH$_4^+$-N) from nutrient-rich peats compared to nutrient-poor peats. These results suggest that water-logged conditions may cause different patterns of nutrient release from different types of peats depending on their biogeochemical characteristics.
The release of P from brash windrows in blanket peat catchments is a significant source of P to receiving waters. Although brash windrows do not release N during the early stages of their decomposition, the decomposition of brash may enhance N leaching and export, probably by enhancing microbial activity and soil N mineralization below brash. The removal of harvest residues and whole-tree harvesting could be an efficient means of decreasing N and P release and transport to water courses from blanket peat catchments. Conventionally-designed buffer areas, with and without seeded grasses, may have limited effect on P retention from through-flow waters in blanket peat catchments, as frequent high flow episodes, typical in blanket peat areas, are disadvantageous for effective P retention. Alternatively, a mini-buffers method, comprising brash mats and windrows positioned perpendicular to the slope, as opposed to along the main slope, and with the strips between the brash windrows seeded with native grasses, is proposed. In contrast to conventional buffer areas, a grassed or non-grassed mini-buffers method might be a feasible means of decreasing P export to water courses from harvested blanket peat sites.
Declaration

I hereby declare that this thesis is my own work and that it has not been submitted for another degree, either at NUI Galway or elsewhere. Where other sources of information have been used, they have been acknowledged.

Zaki-ul-Zaman Asam

Signature: ………………………………………..

Date: …………………………………………..
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Thanks to my parents, in-laws, family and friends for their prayers throughout these years. Finally, warmest and sincere thanks go to my wife for her endless sacrifices, support, love and prayers throughout these years. I would never have been able to achieve this goal without her so I dedicate this thesis to my dear wife, Afshan Sana, and our darling daughter, Amina.
Abbreviations

AI  Aluminium
ANOVA Analysis of variance
a.s.l. Above sea level
B  Boron
BF  Between the drainage furrows
BMP  Best management practice
C  Carbon
Ca  Calcium
Cd  Cadmium
Cu  Copper
DOC  Dissolved organic carbon
DRP  Dissolved reactive phosphorus
d.w.  Dry weight
EC  Electrical conductivity
EEA  European Environment Agency
EPA  Environmental Protection Agency
EU  European Union
Fe  Iron
ha  Hectare
hr  Hour
IF  Inside the drainage furrows
K  Potassium
Mg  Magnesium
Mn  Manganese
N  Nitrogen
NH$_4^+$-N Ammonium nitrogen
Ni  Nickel
NO$_3^-$-N Nitrate nitrogen
N$_2$O Nitrous oxide
OM  Organic matter
O₂  Oxygen
P   Phosphorus
Pb  Lead
PO₄-P  Ortho-phosphate
RM  Remaining mass
rpm  revolutions per minute
TRP  Total reactive phosphorus
WEP  Water extractable phosphorus
WFD  Water framework directive
WT  Watertable
WTH  Whole tree harvesting
yr  Year
Zn  Zinc
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Chapter 1

Introduction

1.1 Background

Drainage of peatlands has been part of forestry practices since the early 20th century (Paavilainen and Päivänen, 1995). In Ireland and the UK, approximately 800,000 ha of peat catchments were drained and afforested between the 1950s and 1990s (Farrell, 1990; Hargreaves et al., 2003; EEA, 2004). A large number of these forests are now approaching maturity and the rate of forest harvesting on drained blanket peatlands will undergo a rapid increase in the near future. One of the concerns raised about the increased harvesting of drained peatland forests is the deterioration of downstream water quality due to enhanced nutrient leaching. Hydrological losses of nutrients from forested peatland catchments are generally low, but several studies in Scandinavia (Rosén and Lundmark-Thelin, 1987; Lundin, 1998, 1999, 2000; Ahtiainen and Huttunen, 1999; Nieminen, 1998, 2003, 2004), and in Ireland and the UK (Nisbet et al., 1997; Cummins and Farrell, 2003a, 2003b; Rodgers et al., 2010) have shown that these losses can increase when forests are harvested.

Increased inputs of nutrients, especially phosphorus (P) and nitrogen (N), lead to eutrophication of inland surface waters (McGarrigle et al., 1993; Foy et al., 1995; Champ, 1998; Lucey et al., 1999; Irvine et al., 2001), and has been identified as one of the most serious pollution problems in Europe (European Union, 2000; Pietiläinen and Räike, 1999; Toner et al., 2005; Ministry of the Environment, 2007). Eutrophication can lead to a suite of problems including oxygen depletion, pH variability, shifts in species composition, food-chain changes, increases in toxic algal blooms and collapse of populations of sensitive fish species (Champ, 1998; Smith et al., 1999; Reynolds and Petersen, 2000). The Water Framework Directive (WFD) requires EU Member States to achieve ‘good ecological status’ for all water bodies by 2015 (European Union, 2000). Therefore, the reduction in nutrient release to water bodies has become a priority issue in river basin management in Europe.

This study attempts to identify the mechanisms of nutrient release and transport from harvested peatland forests with a main focus on blanket peatlands. Nutrient release from harvested upland blanket peat forests in the west of Ireland has become a significant environmental concern, as these forests drain into ecologically valuable oligotrophic salmonid rivers and lakes. Forestry activities are seen as one of the major threats to these
pollution-sensitive water courses. To prevent a possible negative impact of peatland forest activities on the receiving watercourses, one of the recommended and widely used methods is to direct the outflow over a riparian buffer area (Silvan et al., 2004; Nieminen et al., 2005, Väänänen et al., 2008). However, many of the earlier afforested upland blanket peat catchments in Ireland were established without any riparian buffer areas, and trees were planted to the stream edge (Ryder et al., 2010). Most of the research on functioning of buffer areas in forestry is from abroad and their knowledge in Irish conditions is limited. Conventionally-designed buffer areas (1% of upstream catchment area, (Nieminen et al., 2005)) may have limited effect on nutrient retention from through-flow waters in blanket peat catchments, as frequent high flow episodes, typical in blanket peat areas, are disadvantageous for effective nutrient retention. Assessment of the mechanisms of nutrient cycling and transport, and the efficiency of different mitigation options is needed to help in the development of management and harvesting guidelines for environmentally sustainable forestry in blanket peat catchments.

1.2 Processes contributing to N and P release from harvested peat forests

Forest harvesting disrupts nutrient cycling by reducing the uptake of nutrients by vegetation, and simultaneously increasing nutrient-rich litter material input (harvest residues) into the forest floor, which results in increased soil labile nutrients pools in the harvested area (Walbridge and Lockaby, 1994; Herz, 1996). The input of nutrients, especially inorganic N, to the forest floor in harvested sites may also increase because of the absence of atmospheric N retention in the forest canopy (Helmisaari and Mälkönen, 1989; Piirainen et al., 2002). Alternatively, the capture of dry deposition N may decrease after removing the forest canopy, sometimes resulting in decreased bulk N deposition to the forest floor, especially in the high N deposition areas (Nieminen, 1998).

In conventional stem-only harvesting on peatlands the harvest residues (e.g. needles, branches, roots) left onsite is seen as a potential high source of nutrients to receiving water bodies (Rodgers et al., 2010). Several studies have investigated the decomposition of harvest residues and related nutrient dynamics in Scandinavian and North-American conditions (Nilsson, 1972; Berg and Staaf, 1980; Rustad and Cronan, 1988; Laskowski and Berg, 1993; Rustad, 1994; Laskowski et al., 1995; Lundmark-Thelin and Johansson, 1997; Vesterdal, 1999; Palviainen et al., 2004a, 2004b; Moore et al., 2005; Lehto et al., 2010), whereas there is no data on harvest residue decomposition and nutrient dynamics from drained blanket peat
forests. Decomposition and nutrient release from harvest residues on blanket peatlands could be of significant importance because the amount of cutting residues in clear-felled blanket peat forests may be many-fold higher than, for example, peatland forests in Scandinavia. One reason is that the stands in blanket peat catchments approaching clear-felling phase are dense (>2800 stems ha\(^{-1}\), Rodger et al. (2010)), compared to respective peatland stands in, for example, Scandinavia (<1000 stems ha\(^{-1}\)), as no clearings or thinning operations are performed during stand development. The volume of the tree stand in final cutting phase is also significantly larger for blanket peat forests (>400 m\(^3\) ha\(^{-1}\)) than peatlands in the more northern areas (150–300 m\(^3\) ha\(^{-1}\)). In addition, the blanket peat sites were often afforested with Lodgepole pine (\textit{Pinus contorta}), with little self-pruning of branches during stand development. Apart from the differences in the amounts of harvest residues, the decomposition of cutting residues in blanket peat catchments may be different from drained peatlands in more northern areas also because of the differences in climatic conditions. The blanket peats locate in areas of relatively mild climate with no distinct snow and frozen soil period. The precipitation (>2000 mm year\(^{-1}\)) in blanket peat areas may be two to three-fold higher compared with peatland dominated areas in, for example, Scandinavia.

In operational forestry in blanket peat catchments, harvest residues are utilized as brash mats to improve the soil-carrying capacity against the heavy harvesting machinery. In these brash mats, the harvest residues are broken down into smaller fractions, which may increase their rate of decomposition. After the harvesting operations, the harvest residues are collected together to form brash windrows, between which the trees are planted. These windrows (each with a width about 4 m) run along the main slope in parallel rows at intervals of about 12 m down to about 20 m from the stream edge (Rodgers et al., 2010). The variation of soil temperatures and moisture below these brash windrows is likely to be smaller than in brash-free areas, which leads to an increased mineralization of nutrients in the humus layer below windrows (Rosén and Lundmark-Thelin, 1987). No vegetation establishes in brash windrows and there will be no root uptake to prevent nutrient leaching. Furthermore, the vegetation in brash-free areas may not capture the nutrients released from brash, as most of the water flow occurs downslope under the brash windrows (lateral movement of waters from brash windrows to windrow-free areas is negligible in the steep slopes typical of blanket peat catchments). Any lateral movement of water is likely to enter into closely spaced drainage furrows, which are at intervals of 3–5 m along the slope as the windrows. Present harvesting and regeneration practice with brash windrows along the main slope is likely to significantly
contribute to nutrient export from blanket peat catchments. Peatland forest harvesting activities create some mechanical disturbance on the ground surface, which could lead to release of suspended sediments (SS) as well and nutrients in undissolved form to the river systems (Rodgers et al., 2011), however, Rodgers et al. (2010), in their study, found that majority of P release (>80%) after harvesting on blanket peat catchments was in dissolved form.

The mineralization of N and P from the decomposition of organic layer/surface peat can also be accelerated after harvesting in brash-free sites. The soil temperatures are increased as a result of the reduced shading effect of the trees, as are the soil moistures because of the cessation of the tree canopy water interception and transpiration (Lundin, 1979; Jurgensen et al., 1997; Kubin and Kemppianen, 1991; Smolander et al., 2001; Martin et al., 2002). Changes in hydrological conditions result in increased annual and seasonal runoffs, as well as peak runoff rates (Beschta et al., 2000; Müller, 2000; Rosa and Larocque, 2008; Grayson et al., 2010), and a rise in the water table (Lundin, 1979; Huttunen et al., 2003). A rise in the water table following harvesting leads to anoxic conditions in previously aerobic peat. Under reduced conditions, microbially induced redox reactions are important for the mobilization of dissolved substances (Lamers et al., 2002; Lucassen et al., 2004; Shenker et al., 2005). Peat properties can have a decisive role in nutrient mobilization as a result of a rise in water table. Phosphorus leaching can increase under anaerobic conditions, particularly if a significant proportion of P in peat before harvesting was adsorbed by iron (Fe) hydroxides or oxides. In addition, the nutrient export may increase from the root system and dying mychorriza associations as a result of rising water levels.

The effect of a rise in water level on N and P export is particularly important in areas where active forestry is no longer continued after the clear-felling phase, but the site is rewetted and restored back to natural-like peatland. There are multiple reasons to restore blanket peats and other peatland sites, such as restoring the natural carbon cycling, conservation of wetland biodiversity, as well as re-establishing the other ecosystem service characteristics of natural peatlands. However, peatland restoration can have detrimental effects on the quality of runoff and receiving water courses. Koskinen at al. (2011) reported higher P leaching after restoration of a nutrient-poor, ombrotrophic peat site previously drained for forestry and lower P leaching from a nutrient-rich, minerotrophic peat site. Ammonium-N (NH₄⁺-N) and dissolved organic carbon (DOC) leaching per restored hectare was higher from the nutrient-rich site compared to the nutrient-poor site. This suggests that nutrients and DOC leaching
after restoration of peatlands drained for forestry can be different from different types of peatlands depending on the peat properties. Most of the research carried out on restoration in Europe has been concentrated on nutrient-rich peatlands originally drained for agricultural use (Niedermeier and Robinson, 2009; Kalbitz et al., 2002; Zak and Gelbrecht, 2007; Meissner et al., 2008), or on peatlands utilized for peat extraction (Lamers et al., 2002; Waddington et al., 2008). However, knowledge about the restoration of peatlands drained for forestry is scarce (Koskinen et al., 2011).

1.3 Mitigation of N and P export to water bodies

1.3.1 Phased felling

The current recommended practice to prevent the negative impact of harvesting on receiving salmonid sensitive aquatic systems in Ireland (Forest Service, 2000) and the UK (Forestry Commission, 1988) is phased felling. Harvesting appropriately sized coupes in a catchment, rather than the whole catchment, can minimize the nutrient concentrations in the main rivers (Rodgers et al., 2010) mainly due to the dilution effect by the waters from the non-harvested area within a catchment. Cummins and Farrell (2003a) reported higher P concentrations in the drains that covered a higher proportion of the harvested area in a blanket peat. Rodgers et al. (2010) found high P concentrations in the drains downstream from a harvested blanket peat catchment, but the concentrations in a larger receiving stream remained low. Phased felling may help keep the nutrient concentrations lower than the critical values for eutrophication of water bodies, but this management strategy does not reduce the total nutrient loads leaving the harvested catchment and entering the water bodies.

1.3.2 Whole tree harvesting and collection of harvest residues

Whole tree harvesting (WTH), which involves the removal of all above ground parts of the tree stand, may also reduce nutrient export to water bodies (Nisbet et al., 1997). Above ground harvest residues can contain 70–80% of the standing tree biomass pools of N and P (Nykvist, 1971; Kubin, 1977). Studies on mineral soil forests have shown that WTH can decrease nutrient export from harvested sites (Yanai, 1998; Wall et al., 2008). However, the effect of harvest residues removal on nutrient export from blanket peatlands has not yet been assessed. Whole tree harvesting or collection of harvest residues could be a particularly efficient mitigation method for blanket peat forests because of their poor soil nutrient retention capacity (Tamm et al., 1974) and because the amount of cutting residues in clean-
felled blanket peat forests is large. The collection of harvest residues on the other hand may result in erosion due to peat soil disturbance on blanket peatlands and may also increase N release as the harvest residues are reported to be a net sink of N and can accumulate N even from soil (Hyvönen et al., 2000; Palviainen et al., 2004b).

1.3.3 Buffer areas

One of the widely used means of managing water quality in blanket peat forests is to construct a buffer area between forestry land and receiving water bodies. Studies from Scandinavia indicate that buffer areas can significantly reduce particle transport (Nieminen et al., 2005). However, the efficiency of buffer areas in reducing P varies considerably from complete (100%) (Kubin et al., 2000; Väänänen et al., 2008) to partial P retention (Sallantaus et al., 1998; Nieminen et al., 2005; Silvan et al., 2005), to even increased leaching of P (Liljaniemi et al., 2003; Vasander et al., 2003). The creation of a riparian buffer area prior to clear-felling the main plantation in an upland blanket peat catchment has been proposed previously. However, to date, only a few studies have focused on the design of blanket peat buffer areas and assessment of their performance in the mitigation of released nutrients (Ryder et al., 2010).

Lack of interest in constructing buffer areas in blanket peat forests may partly be because the construction of a well-functioning buffer area in many blanket peat catchments is difficult. First, the peat in blanket peat buffer areas may not be a significant sink for P due to its low adsorption capacity (Tamm et al., 1974). Secondly, blanket peats often slope to the stream edge, have low water permeability (hydraulic conductivity) (Cummins and Farrell, 2003a; Lewis et al., 2012), and are located in areas of high precipitation of greater than 2000 mm year\(^{-1}\), which can result in frequent high hydraulic loadings to buffer areas. High hydraulic loading significantly decreases buffer P and N retention efficiency (Väänänen et al., 2008; Vikman et al., 2010), mainly by leading to the formation of continuous flow channels across the buffer area. In such channels, flow velocity is high and water residence time low; both these factors are disadvantageous for an efficient retention of nutrients (Koskiaho et al., 2003). High hydraulic loadings and resulting water-logged conditions in the buffer areas also decrease P retention because the reduction and redistribution of Fe leads to decreased soil P adsorption capacity (Jensen et al., 1999). Under low flow conditions, the contact time between buffer through-flow water and soil and vegetation of the buffer would be longer, and
the vegetative uptake and soil sorption of nutrients thereby greater than for high flow episodes (Väänänen et al., 2008; Vikman et al., 2010).

It is also difficult to construct a well-functioning buffer area in blanket peat catchments because the recovery of natural vegetation following buffer construction takes for several years (Connaghan, 2007; O’Driscoll et al., 2011). Therefore, unless the buffer area is constructed long before harvesting the main plantation, it may be inefficient in retaining the nutrient export, which mainly occurs during the first 3 years after harvesting (Cummins and Farrell, 2003a; Nieminen, 2003; Rodgers et al., 2010). In order to stimulate the establishment of the vegetation, the seeding of appropriate native grass species has been proposed as a solution. O’Driscoll et al. (2011) showed that *H. lanatus* and *A. capillaris* could be well established in blanket peat buffer areas two months after harvesting and had the potential to immobilize the P that would otherwise be available for leaching.

The nutrient retention capacity of buffer area is also strongly related to inflowing nutrient concentration. The effect of inflowing nutrient concentration on retention efficiency in forested catchments is seen in that the retention is generally more efficient under high than low inflowing nutrient concentrations. Negative P (Nieminen et al. 2005) and N (Hynninen et al., 2011) retention efficiencies have been reported for peatland buffer areas when the inflowing concentrations are not significantly higher than the background levels of unpolluted areas and when some N or P is released from the buffer area itself. Negative retention may also occur where persistent high nutrient loadings eventually lead to the saturation of the vegetative and soil nutrient sinks of the buffer (Drizo et al., 2002).

1.4 Aims of the study

The overall aim of this study was to investigate the mechanisms of P and N release from harvested peat forests and to assess the effectiveness of different mitigation methods.

The key objectives were to:

1. Assess the nutrient release from the decomposition of harvest residue.
2. Assess the effect of rise in the water table on nutrient release.
3. Assess the contribution of brash windrows to N and P export to receiving water courses.
4. Assess the P retention efficiency of blanket peat buffer areas.
1.5 Structure of the thesis

Chapters 2 and 3 addressed the first objective of this study. Chapter 2 presents the nutrient release from the decomposition of harvest residue needles on forested and harvested peatlands in Finland and Chapter 3 presents the decomposition and nutrient release from harvest residue needles on blanket peatlands in the west of Ireland. Chapter 4, addressing the second aim of this study, investigates the effect of a rise in water table on nutrient release from different types of peats from Ireland and Finland using a laboratory microcosm study. Chapter 5, addressing the third aim of this study, quantifies the nutrient release from the decomposition of brash windrows, and their effect on nutrient export from harvested blanket peat sites, using laboratory flumes and a field-scale mini-catchment study. Chapter 6, addressing the fourth aim of this study, quantifies the P retention efficiency of blanket peat buffers using laboratory flumes and also presents a novel mini-buffer method to decrease nutrient export from harvested blanket peatlands, resulting from the decomposition of brash windrows. Finally, Chapter 7 details the conclusions of the research and gives recommendations for further work.
1.6 Research output

Peer reviewed journal papers


Conferences papers/oral presentations


References


Zak, D., Gelbrecht, J., 2007. The mobilisation of phosphorus, organic carbon and ammonium in the initial stage of fen rewetting (a case study from NE Germany). Biogeochemistry 85, 141–151.
Chapter 2: Decomposition of harvest residue needles on peatlands drained for forestry- implications for nutrient and heavy metal dynamics

This paper has been published in Forest Ecology and Management (2012; 277, 141–149). Zaki Asam synthesized the experimental data and prepared the first draft of the manuscript. He contributed approximately 50% of the text to the final published paper and Annu Kaila contributed the other 50%. Mika Nieminen was responsible for carrying out the field work between 1994 to 1996 on peatland sites in Finland, and Sakari Sarkkola helped with the statistical analysis. The rest of the co-authors provided guidance and advice in writing the paper. Studies on harvest residue decomposition on harvested peatlands are scarce, so this study is used to compare harvest residue decomposition and nutrient release in Finland with similar studies on blanket peatlands in the west of Ireland.
2. Decomposition of harvest residue needles on peatlands drained for forestry - implications for nutrient and heavy metal dynamics

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Abstract

In boreal and temperate zones, about 15 million hectares of peatlands and wetlands have been drained for forestry purposes and a large number of these forests are now approaching their commercial thinning or regeneration age. One of the major concerns raised in connection with an increased harvesting of drained peatland forests is the deterioration of the downstream water quality due to an enhanced nutrient transport. Harvest residues left on the site are a potential high source of nutrients to recipient water bodies, and both increased nitrogen (N) and phosphorus (P) exports have been reported after conventional stem-only clear-cutting. The decomposition of \textit{Picea abies} and \textit{Pinus sylvestris} harvest residue needles at two clear-cut areas and two uncut forested areas on drained peatlands at two locations in southern Finland were studied. The results indicated that P is easily released from harvest residue needles. After the first three growing seasons, spruce and pine needles had lost approximately 31\% and 47\% of their initial P contents, respectively. There was no clear gain or loss of N. Most of the heavy metals accumulated in the needles as the decomposition process proceeded, especially at the study location near the heavily industrialized Helsinki capital area with high atmospheric deposition. Harvest residue needles are not a likely source of an increased N export that has been observed to occur from peat soils soon after clear-cutting, but P release from harvest residues may be a cause for the reported high P losses.

\textit{Keywords:} Harvesting, decomposition, litter bag experiment, needles, element release
2.1 Introduction

Since the early 20th century, drainage of waterlogged peatlands has been part of the normal forestry practices in Fennoscandia, the Baltic countries, the UK, Ireland, and some parts of Russia (Paavilainen and Päivänen, 1995). A large number of these forests are approaching their commercial thinning or regeneration age, and the rate of forest harvesting on drained peatlands will undergo a rapid increase in the near future. One of the major concerns raised in connection with an increased harvesting of drained peatland forests is the deterioration of the downstream water quality due to an enhanced nutrient export.

The export of nutrients after forest harvesting is generally higher from peatland-dominated catchments than for mineral soil forests (Lundin, 1998; Ahtiainen and Huttunen, 1999; Cummins and Farrell, 2003; Nieminen, 2003; 2004; Rodgers et al., 2010). The reason for higher nitrogen (N) export from peatland-dominated catchments may be because the N reserves in organic soils are much larger. Similarly, the export of phosphorus (P) is more from drained peatlands than for mineral soils, most likely because of the very low P adsorption capacity of most peat soils (Kaila, 1959; Cuttle, 1983; Nieminen and Jarva, 1996).

Harvest residues left on site after conventional stem-only clear-cutting are a potentially high source of nutrients to watercourses (Rodgers et al., 2010). For instance, Norway spruce harvest residues may contain 25–31 kg ha\(^{-1}\) of P and 245–320 kg ha\(^{-1}\) of N (Hyvönen et al., 2000). Rodgers et al. (2010) found that the water extractable phosphorus (WEP) contents in the soil were significantly higher below harvest residue material than in harvest residue-free areas. In a short-term perspective, the most important source of N and P are the needles and twigs, as the larger harvest residue components may release nutrients for more than a few decades (Hyvönen et al., 2000). A litter bag study conducted by Palviainen et al. (2004b) on a mineral soil forest in eastern Finland showed that conifer needles had a potential of releasing about 6 kg of P ha\(^{-1}\) during 3 years.

Decomposition of litter and harvest residues is a complex phenomenon influenced by the activity and nutrient demand of heterotrophic decomposers. The activity is regulated by environmental conditions such as soil temperature, nutrient availability and moisture conditions (Gosz et al., 1973; Prescott, 2005; Laiho, 2006). There are a number of nutrient release studies on conifer needle litter (e.g. Nilsson, 1972; Berg and Staaf, 1980; Rustad and Cronan, 1988; Laskowski and Berg, 1993; Rustad, 1994; Laskowski et al., 1995; Vesterdal,
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1999; Lehto et al., 2010) and harvest residues (e.g. Lundmark-Thelin and Johansson, 1997; Palviainen et al., 2004a; 2004b) from mineral soil forests, whereas knowledge on the element release dynamics from litter, and especially from harvest residues, on drained peatlands is scarce. As the environmental conditions for peat soils are different than for mineral soil sites, the nutrient release pattern from harvest residues and litter may also be different. However, Coulson and Butterfield (1978) and Moore et al. (2005) have reported that there were no consistent differences in mass loss rates of several litter materials between upland and nearby peatland sites. However, in the same study, Moore et al. (2005) found that Douglas-fir needles decomposed significantly faster in peatlands than in upland sites.

A common assumption is that decomposition is faster in clear-cuts than in undisturbed forests. However, decomposition rates for clear-cut sites have been reported to be faster, slower and similar compared to uncut control sites, most probably depending on the extent of harvest-induced change in soil moisture and regional microclimate (e.g. Yin et al., 1989; Hendrickson et al., 1985; Palviainen et al., 2004b; Palviainen, 2005). The effect of clear-cutting on harvest residue decomposition depends largely on the extent of change in the fungal and microbial activity and the processes influencing their activity (Lundmark-Thelin and Johansson, 1997; Prescott, 2005). On peatlands, clear-cuttings cause a significant rise in the water table level and increase peat temperature (Huttunen et al., 2003).

In this study, the mass loss and the dynamics of nutrients and metals in decomposing Norway spruce and Scots pine harvest residue needles on a clear-cut and an uncut forested peatland at two locations in southern Finland, Ruotsinkylä and Vesijako, were investigated. Although the chemical and physical conditions on peatlands are different than on mineral soils, the general trends in nutrient and heavy metal dynamics were expected to be similar to mineral soils. Therefore, it was expected that P would be released from the harvest residue needles in the early phases of decomposition, whereas N would be mostly immobilized (Palviainen et al., 2004b). The base cations largely present in the needle cell solution, e.g. potassium (K) and magnesium (Mg), were also expected to be released rapidly (e.g. Rustad and Cronan, 1988; Laskowski et al., 1995; Palviainen et al., 2004b), whereas calcium (Ca), mainly present in needle cell wall structures, was expected to be released slower (Palviainen et al., 2004a). Knowledge of aluminium (Al) and heavy metal dynamics during the decomposing processes of harvest residues on drained peatlands is scarce, but based on earlier studies with needle litter or harvest residue needles on mineral soils (Laskowski and Berg, 1993; Rustad and
Cronan, 1988; Laskowski et al., 1995; Palviainen et al., 2004a), the metals should accumulate or immobilize during the first few years of decomposition. The Ruotsinkylä experimental site is situated in a heavily industrialized area near the Helsinki capital region, where heavy metal deposition is significantly higher than in the rural Vesijako area (Poikolainen et al., 2004). It has been shown that in heavily polluted areas, needle litters, after losing their protective epiderm and cuticule, act like mosses and lichens by absorbing heavy metals from the air (Tyler, 1972). Therefore, it was also anticipated that the accumulation of heavy metals during needle decomposition was larger at Ruotsinkylä than in the Vesijako area.

2.2 Materials and methods

2.2.1 Site description, field work and laboratory analysis

The study was conducted on two nutrient-rich, old peatland drainage areas in Finland, at Ruotsinkylä (60°21′N, 25°03′E, 49 m a.s.l.) and Vesijako (61°23′N, 25°03′E, 125 m a.s.l.). The study areas are presented in detail in Huttunen et al. (2003) and Nieminen (2004), and only a brief outline of the sites is presented here (Table 2.1). The long-term (1961–1990) mean annual precipitation is 650 mm at Ruotsinkylä and 620 mm at Vesijako (Finnish Meteorological Institute, 1991). The mean annual temperature at Ruotsinkylä is +4.5 °C, with means of −6.8 °C in February and +16.6 °C in July. At Vesijako, the corresponding temperatures are +3.6, −8.3, and +16.0 °C. The average duration of the growing season, defined as the number of days with a mean temperature > +5 °C, is 172 days at Ruotsinkylä and 166 days at Vesijako. The mean temperature sum (threshold value +5 °C) is about 1350 and 1250-degree days.

At both locations, a clear-cut site and an adjacent uncut control site were selected (Table 2.1). All four sites were nutrient-rich, minerotrophic site types and had been drained for forestry purposes in the early 20th century. Clear-cutting at Ruotsinkylä and Vesijako was carried out in January–February 1994 using conventional stem-only harvesting. The mean depth of the ground water table varied between approximately 15–45 cm from the soil surface in the clear-cut sites and 30–80 cm on the uncut control sites during the three growing seasons after clear-cutting (Huttunen et al., 2003). The peat temperature during summer (June–August) at a depth of 16 cm was 1–4 °C higher at the clear-cut site than at the control site at Ruotsinkylä, but only about 1 °C higher at Vesijako (Huttunen et al., 2003).
Table 2.1. Characteristics of the study sites. The sites include both clear-cut and untreated stands dominated by Norway spruce. Site types for drained peatlands: Rhtkg = Herb-rich type, Mtkg = Vaccinium myrtillus type (Vasander and Laine, 2008).

<table>
<thead>
<tr>
<th>Study area</th>
<th>Ruotsinkylä</th>
<th>Vesijako</th>
</tr>
</thead>
<tbody>
<tr>
<td>Treatment</td>
<td>Clear-cut</td>
<td>Control</td>
</tr>
<tr>
<td>Site type</td>
<td>Rhtkg</td>
<td>Mtkg</td>
</tr>
<tr>
<td>Peat depth, m</td>
<td>1.0</td>
<td>0.5</td>
</tr>
<tr>
<td>Drainage year</td>
<td>1927</td>
<td>1932</td>
</tr>
<tr>
<td>Stand volume (m$^3$ ha$^{-1}$)$^a$</td>
<td>234</td>
<td>250</td>
</tr>
<tr>
<td>Tree species (% of volume)</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Pinus sylvestris L.</td>
<td>0</td>
<td>0</td>
</tr>
<tr>
<td>Picea abies Karst.</td>
<td>100</td>
<td>100</td>
</tr>
<tr>
<td>Betula spp.</td>
<td>0</td>
<td>0</td>
</tr>
<tr>
<td>Peat element content (0–20cm)</td>
<td></td>
<td></td>
</tr>
<tr>
<td>N (%)$^b$</td>
<td>1.9</td>
<td>1.4</td>
</tr>
<tr>
<td>P (mg kg$^{-1}$)$^c$</td>
<td>948</td>
<td>624</td>
</tr>
<tr>
<td>K (mg kg$^{-1}$)$^c$</td>
<td>350</td>
<td>400</td>
</tr>
<tr>
<td>Ca (mg kg$^{-1}$)$^c$</td>
<td>4848</td>
<td>4251</td>
</tr>
<tr>
<td>Mg (mg kg$^{-1}$)$^c$</td>
<td>510</td>
<td>440</td>
</tr>
<tr>
<td>Fe$^{tot}$(mg kg$^{-1}$)$^c$</td>
<td>5927</td>
<td>2353</td>
</tr>
<tr>
<td>Al$^{tot}$(mg kg$^{-1}$)$^c$</td>
<td>3340</td>
<td>1878</td>
</tr>
</tbody>
</table>

$^a$ Measured 1 year before clear-cut in 1993.
$^b$ LECO CHN-1000 analyzer
$^c$ Dry ashing + digestion in HCl (Halonen et al. 1983) and ICP/AES

To study the decomposition of harvest residue needles, current year Norway spruce and Scots pine needles were used. The Norway spruce needles were collected from the Vesijako and the Ruotsinkylä clear-cut sites soon after clear-cutting. The Scots pine needles were collected from a mineral soil forest in the Evo area, approximately 100 km from Vesijako. The needles were air-dried and mixed thoroughly. One gram dry weight of foliage litter was enclosed in polyester litterbags measuring 10 × 5 cm and with a mesh size of 0.5 × 0.5 mm. The bags were placed systematically in 10 sampling spots at each of the four areas (Ruotsinkylä clear-cut, Ruotsinkylä control, Vesijako clear-cut, Vesijako control), with five local (Ruotsinkylä or Vesijako) spruce needle bags and five pine needle bags from Evo in each spot. The bags were placed directly below the living moss layer in May 1994. The incubation study lasted for 3 years (1994–1996) and the bags were sampled each year either in September (1994, 1996) or in May and September (1995). On each sampling occasion, one bag with pine needles and one with local spruce needles was systematically collected from each of the 10 sampling positions at each site. After collection, the litter samples were cleansed, dried at +40 °C until dry, and weighted. From each site, the contents of 10 bags were combined for laboratory analyses and ground using an A 10 IKA-analytical mill. The N and carbon (C)
concentrations were analyzed by a LECO CHN-analyser and Ca, K, Mg, P, boron (B), iron (Fe), Al, manganese (Mn), zinc (Zn), lead (Pb), nickel (Ni), and cadmium (Cd) concentrations by inductively coupled plasma emission spectrophotometer (ICP/AES, ARL 3580) after dry ashing (quartz dishes) and digestion in HCl (plastic dishes).

The initial concentrations of the harvest residue needles are shown in Table 2.2. The changes in mass loss and element contents (concentration, mg g\(^{-1}\) \times\) remaining mass, g) in the decomposing needle harvest residues during three growing seasons were investigated. The changes in nutrient and metal contents during decomposition are expressed as percentage values from the initial content remaining on each sampling occasion.

**Table 2.2.** Initial element concentrations, and C/P, C/N and N/P-ratios of the harvest residue needles.

<table>
<thead>
<tr>
<th>Element</th>
<th>Ruotsinkylä</th>
<th>Vesijako</th>
<th>Evo</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>Spruce needle</td>
<td>Spruce needle</td>
<td>Pine needle</td>
</tr>
<tr>
<td>C (%)</td>
<td>53.3</td>
<td>53.2</td>
<td>53.8</td>
</tr>
<tr>
<td>N (%)</td>
<td>1.4</td>
<td>1.0</td>
<td>1.3</td>
</tr>
<tr>
<td>P (mg kg(^{-1}))</td>
<td>698</td>
<td>722</td>
<td>1140</td>
</tr>
<tr>
<td>B (mg kg(^{-1}))</td>
<td>18.1</td>
<td>10.9</td>
<td>9.2</td>
</tr>
<tr>
<td>K (mg kg(^{-1}))</td>
<td>2935</td>
<td>3615</td>
<td>4020</td>
</tr>
<tr>
<td>Al (mg kg(^{-1}))</td>
<td>27.7</td>
<td>20.4</td>
<td>363</td>
</tr>
<tr>
<td>Fe (mg kg(^{-1}))</td>
<td>47.1</td>
<td>32.9</td>
<td>62.8</td>
</tr>
<tr>
<td>Pb (mg kg(^{-1}))</td>
<td>0.7</td>
<td>0.8</td>
<td>0.9</td>
</tr>
<tr>
<td>Ni (mg kg(^{-1}))</td>
<td>2.0</td>
<td>1.8</td>
<td>2.6</td>
</tr>
<tr>
<td>Cd (mg kg(^{-1}))</td>
<td>0.1</td>
<td>0.1</td>
<td>0.1</td>
</tr>
<tr>
<td>Mn (mg kg(^{-1}))</td>
<td>880</td>
<td>735</td>
<td>660</td>
</tr>
<tr>
<td>Mg (mg kg(^{-1}))</td>
<td>780</td>
<td>573</td>
<td>644</td>
</tr>
<tr>
<td>Ca (mg kg(^{-1}))</td>
<td>7985</td>
<td>7295</td>
<td>4495</td>
</tr>
<tr>
<td>Zn (mg kg(^{-1}))</td>
<td>23.25</td>
<td>20.5</td>
<td>43.8</td>
</tr>
<tr>
<td>N/P</td>
<td>19.7</td>
<td>14.1</td>
<td>11.1</td>
</tr>
<tr>
<td>C/P</td>
<td>764</td>
<td>737</td>
<td>471</td>
</tr>
<tr>
<td>C/N</td>
<td>38.8</td>
<td>51.3</td>
<td>44.4</td>
</tr>
</tbody>
</table>

2.2.2 Statistical analysis

A mixed regression analysis was used to identify and test the effects of the explanatory variables on the remaining mass of the needles (RM%) and the content of C and elements (C%, N%, P%, B%, Mg%, Zn%, Al%, Fe%, Mn%, Cu%, Cd%), and the C/N, C/P and N/P ratios in the cutting residues during the decomposition process. The mixed model approach was used in order to take into consideration the possible autocorrelation between the successive measurements within the sites (see Searle, 1972). In the datasets, two hierarchical levels of variation were identified: variation between the sites (Ruotsinkylä clear-cut,
Ruotsinkylä control, Vesijako clear-cut, Vesijako control) and variation between the five sampling occasions (May 1994, September 1994, May 1995, September 1995, September 1996). The two hierarchical levels were used as random variables in the models. The tested fixed explanatory variables were: time since the beginning of the incubation (months), site dummy (whether the experiment was located in Ruotsinkylä or in Vesijako), tree species dummy (whether the needles were from pine or spruce), treatment dummy (clear-cut or non-managed control), and interactions between time and the above-mentioned dummy variables. If the relationship between the explanatory variable (time or its interactions) and the dependent variable was nonlinear, the explanatory variable was linearized with exponent or natural logarithm.

The mixed model was constructed as:

\[ Y_{ij} = a_{ij} + b_1x_{1ij} + b_2x_{2ij} + \ldots + b_nx_{nij} + u_j + e_{ij} \]  

Equation 2.1

where \( Y_{ij} \) is the remaining mass or the element content in the harvest residue needles, sample occasion \( i \) on the site \( j \), \( a_{ij} \) is the intercept, \( b_1 \ldots b_n \) are the model parameters, \( x_{1ij} \ldots x_{nij} \) are the explanatory variables, \( u_j \) is the random effect of the sites \( j \), and \( e_{ij} \) is the random error that accounts for the variation among the sample occasion \( i \) within the sites. The random effects were assumed to be independent and to follow a normal distribution, with the mean 0 and constant variances and covariances at each level.

MLwiN statistical software (Rasbash et al., 2001) was used for the analyses. The fixed and the random parameters were estimated simultaneously with the restricted iterative generalized least-square (RIGLS) method recommended for small samples. A parameter was determined to be significant if its absolute value was more than twice the size of the standard error. The value of \(-2(\text{log-likelihood})\) was used to compare the overall goodness-of-fit of the models of increasing number of explanatory variables. The model was constructed by adding the explanatory variables one after another premising that the added variable must maximize the decrease of the log-likelihood value (e.g. Sarkkola et al., 2009). Variables were added to the model until there was no significant improvement in the likelihood measure, or one or more of the explanatory variables become non-significant. Akaike Information Criterion (AIC) was used to compare the significance of the model improvement after adding the explanatory variable and finding the best fitting model. The order of the added variables did not affect the model predictions. The model reliability and accuracy were evaluated by calculating the
systematic error (absolute bias, $Ba$), relative systematic error ($Br$), and the variation in the data explained by the model (EV%) as follows:

$$Ba = \frac{1}{m} \sum_{j=1}^{m} \left[ \frac{1}{n_j} \sum_{i=1}^{n_j} (y_{ij} - \hat{y}_{ij}) \right]$$  \hspace{1cm} \text{Equation 2.2}$$

$$Br = \frac{1}{m} \sum_{j=1}^{m} \left[ \frac{1}{n_j} \sum_{i=1}^{n_j} \left( \frac{y_{ij} - \hat{y}_{ij}}{\hat{y}_{ij}} \right) \right]$$  \hspace{1cm} \text{Equation 2.3}$$

$$EV\% = \frac{\hat{u}_j + \hat{e}_{ij}}{u_j + e_{ij}} \times 100$$  \hspace{1cm} \text{Equation 2.4}$$

where $m$ is the number of sites, $n_j$ is the number of sampling occasions for mass loss or the element content in the harvest residue needles in site $j$, and $\hat{y}_{ij}$ is the predicted value of mass loss or element content. Terms $u_j$ and $e_{ij}$ are the variances of the random effects before the parameter estimation of the explanatory variables in the model, and $\hat{u}_j$ and $\hat{e}_{ij}$ are the variances after parameter estimation of the explanatory variables.

2.3 Results

During the three growing seasons of the study, the pine needles decomposed significantly faster than the spruce needles. The average mass loss of pine needles was 61%, and for spruce needles, 48% (Fig. 2.1). The release of C was strongly correlated with the mass loss of the needles ($r = 0.999$, $p < 0.001$).

![Fig. 2.1. Observed changes in mass and carbon content during incubation as % of the initial mass or element remaining. R =Ruotsinkylä, V=Vesijako.](image)
None of the tested explanatory dummy variables (tree species, study area, or treatment i.e. clear-cut vs. control) were statistically significant in explaining the variation in the needle N contents (Table 2.3). However, the effect of time on N release was significant, although the mean N release over all sites and needle types was only about 5% during 3 years (Fig. 2.2). The release of P from the needles was considerable and the differences between the tree species were significant (Table 2.3). After 3 years, the spruce and pine needles had released approximately 31% and 47% of their initial P content, respectively (Fig. 2.2). The average release of B during 3 years was 49% (Fig. 2.2), and there were no significant differences either between tree species, treatments or location (Table 2.3).

Fig. 2.2. Observed changes in N, P, B and K contents during incubation as % of the initial nutrient remaining. R =Ruotsinkylä, V=Vesijako.
Table 2.3. Model parameters and performance values of the mixed models constructed for the remaining mass (RM) (%) and the element content remaining (%) during three years since the litter bag installation on drained peatland sites in Finland. Only the statistically significant (p<0.05) parameters are presented. $\alpha = $ Intercept, Site= a site dummy (0/1), Tree = a tree species dummy (0/1); Time= months since the beginning of the incubation with different transformations; $u$ and $e$ are the random effects of the sites and the variation among the measurement time points. Bias= absolute bias; Biasr= the relative bias; EVr=explained of the total variance. For random part, the non-significant parameters ($u$, $e$) are marked with "ns".

<table>
<thead>
<tr>
<th>Variable</th>
<th>$\alpha$</th>
<th>Site</th>
<th>Tree</th>
<th>Time</th>
<th>Ln(Time)</th>
<th>(Time)$^1$</th>
<th>(Time)$^2$</th>
<th>(Time)$^3$</th>
<th>(Time)$^2$</th>
<th>Time</th>
<th>(Time)$^2$</th>
<th>(Time)$^3$</th>
<th>Time</th>
<th>(Time)$^2$</th>
<th>(Time)$^2$</th>
<th>LN(Time)</th>
<th>u</th>
<th>e</th>
<th>Bias</th>
<th>Biasr</th>
<th>EVr</th>
</tr>
</thead>
<tbody>
<tr>
<td>RM</td>
<td>46.1</td>
<td>-10.7</td>
<td></td>
<td>296.7</td>
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<td></td>
<td></td>
<td>1.5&lt;sup&gt;ns&lt;/sup&gt;</td>
<td>26.7</td>
<td>-0.00</td>
<td>-0.009</td>
<td>0.94</td>
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</tr>
<tr>
<td>C</td>
<td>46.8</td>
<td>-9.3</td>
<td></td>
<td>290.0</td>
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<td>0.8&lt;sup&gt;ns&lt;/sup&gt;</td>
<td>22.7</td>
<td>0.006</td>
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<td></td>
<td></td>
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<tr>
<td>P</td>
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<td></td>
<td>219.3</td>
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<td></td>
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<td></td>
<td></td>
<td></td>
<td>8.7&lt;sup&gt;ns&lt;/sup&gt;</td>
<td>33.2</td>
<td>-0.024</td>
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<tr>
<td>B</td>
<td>137.0</td>
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<td></td>
<td>11067.4</td>
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<td></td>
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<td></td>
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</tr>
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<td>96.0</td>
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<td>18.8</td>
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<td></td>
<td></td>
<td></td>
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<td>5865.5&lt;sup&gt;ns&lt;/sup&gt;</td>
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<td>-0.068</td>
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</tr>
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<td>-0.009</td>
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<sup>a</sup>Ruotsinkylä site = 1, Vesijako site = 0; Scots pine = 1, Norway spruce = 0
<sup>b</sup>Ruotsinkylä site = 0, Vesijako site = 1; Scots pine = 0, Norway spruce = 1
During the first year, 80–90% of K (Fig. 2.2) and 50–70% of Mg (Fig. 2.3) were released from the needles. The releases were significantly faster from the pine than from the spruce needles (Table 2.3). The net release of Ca did not start until after one year. The effect of the tree species was significant (Table 2.3), and after three growing seasons, the pine needles had released 43% and the spruce needles 27% of their initial Ca contents (Fig. 2.3).

![Fig. 2.3. Observed changes in Ca, Mg, Zn, and Mn contents during incubation as % of the initial element remaining. R =Ruotsinkylä, V=Vesijako.](image)

Among the different heavy metals, Mn was released from harvest residue needle decomposition (Fig. 2.3). The difference between the tree species was significant, with mean Mn
releases of 65% and 44% of their initial content for pine and spruce needles, respectively. Zinc was also released, but only from the pine needles (Fig. 2.3).

![Graphs showing observed changes in Al, Fe, Pb, Ni and Cd contents during incubation as % of the initial element remaining. R = Ruotsinkylä, V = Vesijako.](image)

**Fig. 2.4.** Observed changes in Al, Fe, Pb, Ni and Cd contents during incubation as % of the initial element remaining. R = Ruotsinkylä, V = Vesijako.

The effect of tree species was also significant in explaining the variation in needle Al contents during decomposition, with a clear accumulation in the spruce needles, particularly at
Ruotsinkylä, but with a slight release from the pine needles (Fig. 2.4). The difference between the two study locations was significant (Table 2.3).

At Ruotsinkylä, there was also a significant accumulation of Fe, Pb, Ni and Cd in the needles (Fig. 2.4). During 3 years, the average accumulation in Fe, Pb, Ni, and Cd contents at Ruotsinkylä were 557%, 447%, 160%, and 40%, respectively. At Vesijako, the accumulation rate of Fe and Pb was more moderate and there was no clear trend in Cd (Fig. 2.4). Nickel was released in Vesijako with net losses of 30% and 47% for spruce and pine, respectively.

2.4 Discussion

2.4.1 Element dynamics during needle decomposition

During the study period, there was no clear gain or loss of N from the needles. Although the time variable was significant in explaining the changes in N contents, the net release was small (approximately 5%). A negligible N release in the early stages of decomposition is in accordance with earlier studies (Rustad and Cronan, 1988; Lundmark-Thelin and Johansson, 1997; Hyvönen et al., 2000; Palviainen et al., 2004b). It has been suggested that the release of N does not start until the needle lignin starts to be mineralized (Berg and McClaugherty, 1989; Lundmark-Thelin and Johansson, 1997). It has also been proposed that polyphenolics, which are important constituents of conifer needles, form resistant complexes with N. These complexes slow down the decomposition and N release (Palm and Sanchez, 1991; Kainulainen and Holopainen, 2002).

In contrast to N, the P release from the harvest residue needles was considerable. The high initial loss of P probably reflects the tendency for some of the P in the living needles to be stored as inorganic P, which could be rapidly leached from the litter early in the decay process (Prescott, 2005). The high initial release of P from fresh harvest residue needles has been shown in studies on mineral soils (Hyvönen et al., 2000; Palviainen et al., 2004b), and also with senescent needle litter on mineral soils and peatlands (Rustad, 1994; Moore et al., 2005). However, Lundmark-Thelin and Johansson (1997) found that during the first 2 years, P was accumulated in the brown needle litter and released from the fresh harvest residue needles. The pattern of release or accumulation of P is suggested to largely depend on the initial P concentration of the needles (Lundmark-Thelin and Johansson, 1997; Vesterdal, 1999). This is because the decomposition-limiting nutrients occurring in suboptimal amounts are likely to be accumulated during the initial
stages of litter decomposition, while the nutrients exceeding the needs of the decomposers are released (Laskowski et al., 1995).

Earlier studies show enhanced N and P export to watercourses from peatland forests soon after clear-cuttings (Nieminen, 2003; Nieminen, 2004; Rodgers et al., 2010). The negligible N release from the decomposing needles in the present study indicates that the harvest residue needles not are a likely source of that increased N export. However, owing to the high release of P, the harvest residues may have contributed to P losses, especially because the chemical adsorption of the released P by peat may be negligible for peat soils (Nieminen and Jarva, 1996; Nieminen, 2003).

The initial N/P ratio was 11 for the pine needles, 14 for the Vesijako spruce needles, and 20 for the Ruotsinkylä spruce needles (Table 2.2). The N/P ratio is an important determinant for plant litter decomposition and nutrient dynamics, and it has importance for the relative proportions of fungi and bacteria in litter-associated microbial communities (Güsewell and Gessner, 2009). The optimal N/P ratio for heterotrophic decomposers has been reported to be 10 (Vogt et al., 1986). It has been suggested that decomposition is P-limited with high N/P ratios and N limited at low N/P ratios (Lockaby and Conner, 1999; Prescott, 2005; Güsewell and Gessner, 2009). Prescott (2005) suggested that litters with an initial N/P ratio of <15 are likely to retain N and release P, while the litters with an initial ratio >15 have the opposite effect. In the present study, however, the N content remained almost unchanged and P was released in all of the cases regardless of the initial N/P ratio of the needles.

Boron was released rapidly after the incubation started, although there appeared to be repeated phases of release and accumulation. There are no earlier studies on B dynamics in harvest residue needles on drained peatlands, but it has been shown that B was released rapidly from spruce needle litter on mineral soils (Lehto et al., 2010). However, with B-poor litter material, accumulation of B may also occur, probably due to transportation of B by fungi into the litter (Lehto et al., 2010).

The base cations with high concentrations in the needle cell solution, like K and Mg, are generally released in the early phases of litter decomposition, as was also the case in the present study. The results of the present study did not support the findings by Palviainen et al. (2004a),
who found that the release rate of K was faster from foliage incubated at a clear-cut plot than at a forest plot. Significant Ca release occurred only after 1 year of decomposition, which is consistent with the findings by Palviainen et al. (2004a). In the study by Lundmark-Thelin and Johansson (1997), Ca was released from brown Norway spruce litter during the first year, but accumulated in fresh harvest residue needles. Calcium is a structural component of plant litter and therefore, the release of Ca is more dependent on the microbial decomposition than on leaching (Blair, 1988), which may explain why Ca is not released rapidly from fresh litter.

2.4.2 Impacts of clear-cutting and study location

The mass losses of the needles in the present study were comparable to those reported for Norway spruce and Scots pine needles in earlier studies on mineral soil sites (Lundmark-Thelin and Johansson, 1997; Hyvönen et al., 2000; Palviainen et al., 2004b; Lehto et al., 2010). No significant differences in the mass loss, nutrient, or metal dynamics of the harvest residue needles were found between the clear-cut and control sites.

After clear-cutting, the water table during the growing season at the clear-cut sites varied between 15–45 cm below ground level, and was 15–35 cm higher than the uncut forested sites. The peat temperature at a depth of 16 cm during summer (June–August) was 1–4 °C higher at the clear-cut sites than at the uncut control sites. Apparently, these changes, together with possible alterations in wind velocity and microclimate, were not sufficient to decelerate or accelerate the decomposition of the needles significantly. In addition, the hydraulic conductivity of a dry soil is low (Laurén and Heiskanen, 1997; Laurén and Mannerkoski, 2001). Therefore, if the surface soil layers dry out during the summer months, the litter bags easily lose their capillary connection to the ground water, and the moisture conditions may then be similar between the clear-cut sites and the controls regardless of the level of the water table.

Most of the heavy metals accumulated in the needles as decomposition proceeded. The faster accumulation of Fe, Pb, and Ni in the both needle types, and Al and Cd in the spruce needles at Ruotsinkylä, was probably due to heavier atmospheric heavy metal deposition at Ruotsinkylä compared to Vesijakko. It has been demonstrated that as the decomposition proceeds, the needles lose their protective epiderm and cuticule. The plant material becomes more permeable and the needles act in a manner more like humus and mosses absorbing heavy metals from the air (Tyler,
It is evident that litter and mor layers, together with mosses and lichens, act as an efficient filter for heavy metal deposition (Tyler, 1972). The accumulation is governed by the equilibrium of ion exchange, and continues until equilibrium is attained between heavy metal concentrations in the litter and the metals in the water passing through the litter (Tyler, 1972).

2.4.3 Impact of tree species

The pine needles decomposed faster than the spruce needles, which is consistent with the findings by Palviainen et al. (2004b). Palviainen et al. (2004b) suggested that the slower decomposition of spruce needles was due to higher lignin content and lower initial N concentration, resulting in lower microbial activity. In the present study, however, there was no difference in the initial N concentrations between the tree species (Table 2.2). In addition, the initial C/N ratio, which is considered to be a good indicator of decay rates (Tyler et al., 1989; Prescott, 2005), did not correlate with the decomposition rate. Therefore, the slower spruce needle decomposition could mostly be related to the fact that spruce needles have higher lignin concentration (Johansson, 1995).

In the present study, P was released significantly faster from the pine needles than from the spruce needles. The remaining P content after 3 years was 53% and 69% for pine and spruce, respectively. These results contradict to the findings by Palviainen (2005) with harvest residue needles on mineral soils, where the remaining P content in pine needles after 3 years was 48%, and for spruce, 36%. These differing results may be related to the initial P concentrations, which were 40% higher for pine than for spruce in the present study (Table 2.2), but about 20% lower for pine than spruce in the studies by Palviainen et al. (2004b, 2005).

The initial concentration also appeared to affect Zn release, as the initial Zn concentration in the pine needles (44 mg kg\(^{-1}\)) was twice as high as for the spruce needles (22 mg kg\(^{-1}\)). Zinc was released from the pine needles but was retained in the spruce needles. In earlier studies with pine needle litter on mineral soils, Zn was accumulated into needles (Laskowski et al., 1995). Gosz et al. (1973) suggested that accumulation and release of Zn during the decomposition reflects the Zn requirement of the decomposing heterotrophs.
Though Al accumulated in the spruce needles, it was released from the pine needles. The net accumulation in the spruce needles was approximately 333% after 3 years, and the net release from the pine needles was approximately 19%. On mineral soils, accumulation of Al into Norway spruce needles (Lehto et al., 2010) and some other conifer litter types (Rustad and Cronan, 1988; Rustad, 1994) has been previously confirmed. The release of Al from pine needles is consistent with a study by Palviainen et al. (2004a), where the net release from pine harvest residue needles over 3 years was approximately 26%. In the present study, the pattern of release or accumulation of Al could be related to the initial Al concentrations, as the concentrations in pine needles were >10-fold higher than for the spruce needles (Table 2.2). The significantly higher initial Al concentration in the pine needles may be due to the fact that the pine needles were collected from a mineral soil site, most probably with much higher Al contents in the soil compared with the peat soils. Since Al is highly toxic to many microorganisms, it has been suggested that Al accumulation in litter is primarily an abiotic process in which Al is strongly adsorbed onto litter exchange sites, and the amount and stability of the complexes increase as the humification process proceeds (Rustad and Cronan, 1988; Rustad, 1994; Laskowski et al., 1995). Another explanation for the larger accumulation of Al in the spruce needles could be that the exchange sites in spruce needles are presumably more abundant than in pine needles because of their greater surface area.

2.5 Conclusions

Phosphorus is easily released from harvest residue needles, especially from pine needles, whereas there is a negligible release of N during the first few years of decomposition. In this study, most of the heavy metals accumulated in the needles as the decomposition proceeded, especially at the study location near the heavily industrialized Helsinki capital area, with a high atmospheric heavy metal deposition. Harvest residue needles are not a likely source of the increased N export that has been observed to occur from peat soils soon after clear-cutting. However, P release from harvest residues may be a cause for the reported high P losses in runoff.
Acknowledgements

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References


Chapter 3: Nutrient and heavy metal dynamics in decaying harvest residue needles on drained blanket peat forests

This paper has been submitted to Forest Ecology and Management. Zaki Asam designed the experiment. He manufactured and prepared all the litter bags used for this study and was fully responsible for positioning them onsite and collecting them over a 2-year period. He also carried out all the laboratory work, and analysed and synthesised the experimental data. He prepared the first draft of the paper and is the primary author. Mark O’Connor and Connie O’Driscoll helped with field work, and Annu Kaila and Sakari Sarkkola helped with the statistical analysis. The rest of the co-authors provided guidance and advice in writing the paper.
3. Nutrient and heavy metal dynamics in decaying harvest residue needles on drained blanket peat forests

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Abstract

The amount of cutting residues in stem-only harvested blanket peat forests may be significantly larger than for drained peatlands in other parts of Europe. Therefore, their decomposition can significantly contribute to nutrient and heavy metal export to receiving water courses. Nutrient and heavy metal dynamics were studied in decaying Sitka spruce and Lodgepole pine harvest residue needles inside and between furrows in uncut forest and clear-cut sites at two blanket peat areas in west of Ireland. The harvest residue needles decomposed faster (20–30\% higher mass loss over two years) at the uncut forest than the clear-cut sites. Phosphorus release from decaying harvest residue needles during two years was on average 20\% faster at the uncut forest sites, while nitrogen was released only at uncut forested sites. Boron, potassium, and magnesium release was faster from needles than reported earlier for peatlands in, for example, Scandinavia, possibly reflecting the influence of climatic factors on leaching from needle cell solution. Aluminium, nickel, cadmium, and zinc were released less or accumulated more in the spruce than pine needles. However, calcium and magnesium were released faster from spruce needles. Manganese was released from three sites with a total release of over 90\% during two years, but accumulated in harvest residue needles in a location, where manganese contents in peat was high. Aluminium accumulated in spruce needles and iron both in spruce and pine needles in the furrows of the site, where their contents in peat were high. Owing to their fast release from needles, the decomposition of harvesting residue needles on blanket peats may be a significant source of P to receiving water courses, but harvest residue needles are not a likely source of N export from blanket peatlands after harvesting.

Keywords: Blanket peatlands, harvest residue, litter bag experiment, decomposition, element release.
3.1 Introduction

Large areas of upland blanket peat catchments were afforested between 1950s and 1990s in Ireland and the UK. These forests are now approaching harvestable age and there will be a rapid increase in their harvesting in the near future. An increase in nutrient leaching from harvested peatland forests has been reported by several studies (Rosén and Lundmark-Thelin, 1987; Nisbet et al., 1997; Lundin, 1998, 1999, 2000; Nieminen, 1998, 2003, 2004; Ahtiainen and Huttunen, 1999; Cummins and Farrell 2003a, 2003b; Rodgers et al., 2010). Increased input of nutrients to inland surface waters leads to eutrophication, which is one of the most serious pollution problems in Europe (European Union, 2000; Pietiläinen and Räike, 1999; Toner et al., 2005; Ministry of the Environment, 2007).

In the conventional stem-only harvesting method the harvest residues (above ground e.g. needles, twigs, branches etc and below ground e.g. roots) are left on site. Decomposition of harvest residues is seen as a potential high source of nutrients to receiving water bodies from drained peatlands (Rodgers et al., 2010; Kaila et al., 2012), but may be a particularly important source of nutrients from blanket peat catchments. The blanket peat sites were often afforested with Lodgepole pine (*Pinus contorta*) with little self-pruning of branches compared with many other tree species. The stands in blanket peat catchments are dense (>2800 trees ha⁻¹, Rodgers et al., 2010) compared with those in, for example, Scandinavia, as no clearings or thinning cuttings are performed during stand development. The stand volume in sites approaching clear-felling phase is also significantly larger for blanket peat forests (>400 m³ ha⁻¹) compared with respective peatland stands in more northern areas (150–300 m³ ha⁻¹). All these facts mean that the amount of cutting residues in clear-felled blanket peat forests are significantly larger than, for example, Scots pine (*Pinus sylvestris*) dominated peatlands in Scandinavia. Asam (unpublished data) estimated the above ground harvest residues of Lodgepole pine in blanket peat forests in west of Ireland to amount over 80,000 kg (dry weight (d.w.)) ha⁻¹, while the harvest residues of Scots pine in Fennoscandia amounts to only about 14,000 kg (d.w.) ha⁻¹ in the clear-felling phase (Palviainen and Finér, 2012). Mitchell and Hankin (1993) reported the amount of above ground harvest residues in Ireland to vary from 50,000 to 100,000 kg (d.w.) ha⁻¹, depending on the tree species.
Similarly, Asam (unpublished data) estimated the fresh above ground harvest residues of Lodgepole pine in west of Ireland to contain 25–33 kg ha\(^{-1}\) of P and 330–430 kg ha\(^{-1}\) of N, the needles accounting for nearly 50% of the total stores of N and P in above ground harvest residues. These figures for N and P in harvest residues are significantly more than those for Scots pine-dominated sites in Scandinavia, which were reported to be between 8–15 kg ha\(^{-1}\) for P, and 77–135 kg ha\(^{-1}\) for N (Hyvönen et al., 2000; Palviainen and Finér, 2012).

Forested blanket peats differ from other forestry-drained peatlands also in that they were drained for forestry by closely spaced drainage furrows (about 1 m wide and at 3–5 m intervals) that may cover as much as 25–30% of the area (Ledger and Harper, 1987; Anderson et al, 2000; Rodgers et al., 2010). In standing forests, much of the litter is deposited and accumulated in these furrows where most of the water flow occurs before leaving the area. After harvesting much of the excessive cutting residues accumulate in the furrows. Reducing conditions in the furrows may result in different release and accumulation pattern of nutrients and heavy metals in harvest residue needles compared with more aerobic conditions in the areas between furrows.

Several studies have investigated the decomposition of conifer needle litter and harvest residues in mineral soils (Nilsson, 1972; Berg and Staaf, 1980; Rustad and Cronan, 1988; Laskowski and Berg, 1993; Rustad, 1994; Laskowski et al., 1995; Lundmark-Thelin and Johansson, 1997; Vesterdal, 1999; Palviainen et al., 2004a,b; Lehto et al., 2010) and also peatlands in Scandinavian and North-American conditions (Moore et al., 2005; Kaila et al., 2012), whereas there is no data on harvest residue decomposition and nutrient dynamics from drained blanket peat forests. Apart from the differences in the amounts of harvest residues and operational forestry practices, the decomposition of harvest residues in blanket peat catchments may be different from drained peatlands in more northern areas also because of the differences in climatic conditions. The blanket peats locate in areas of relatively mild climate with no distinct snow and frozen soil period. The precipitation (>2000 mm year\(^{-1}\)) in blanket peat areas may be two to three-fold higher compared with peatland dominated areas in, for example, Scandinavia.

In this study the decomposition of harvest residue needles of two dominant tree species on blanket peatlands, i.e. Lodgepole pine (\textit{Pinus contorta}) and Sitka spruce (\textit{Picea sitchensis}), were investigated on two contrasting blanket peatland sites in west of Ireland. Although the general trends of decomposition and nutrient release may be similar, as reported earlier for mineral soils
and peatlands (Rustad and Cronan, 1988; Laskowski et al., 1995; Palviainen et al., 2004a, 2004b; Kaila et al., 2012), the rate of decomposition of harvest residue needles and resulting nutrient release was expected to be faster compared with earlier studies from more northern peatland sites. It was also expected that the nutrient and heavy metal release and accumulation patterns in the drainage furrows (IF) to be different from the areas between the furrows (BF), and the decomposition patterns also to differ between the harvest residue needles of spruce and pine (Kaila et al., 2012).

3.2 Materials and methods

3.2.1 Site description

The experiment was conducted on two afforested blanket peat-dominated catchments in the west of Ireland, the Glennamong (53°58’N, -9°37’E, 69 m a.s.l.) and Skerdagh (53°56’N, -9°31’E, 125 m a.s.l.), located about 5 km from each other. Both catchments were drained with furrows and afforested in 1970s. The dominant tree species were Lodgepole pine (Pinus contorta) at Glennamong and Sitka spruce (Picea sitchensis) at Skerdagh. The four sites used for this study are: Glennamong clear-cut, Glennamong uncut, Skerdagh clear-cut, and Skerdagh uncut. The Glennamong clear-cut site was harvested in August 2009 and the Skerdagh site in January 2009, using conventional stem-only harvesting, where the harvest residue was left on site. The clear-cut site and uncut site are about 300 m from each other at Glennamong, and about 150 m at Skerdagh.

The two catchments have similar hydrological conditions and receive an average precipitation of approximately 2,000 mm year\(^{-1}\) with average rain days in excess of 225 per year. On the basis of measurements at the weather station operated by the Marine Institute at about 5 km from the study sites, the 2009–2011 mean annual temperature was 9.8°C, with means of 4.4°C in January and 14.4°C in July. The average slope at Glennamong was 5–10° and was 2–5° at the Skerdagh site.

The peat soil at Glennamong is highly fibric with a depth of over 1 m and overlies mainly quartzite and schist bedrock. At Skerdagh, layers of mineral soil have been deposited between the peat layers and there were more of those layers in the peat of the Skerdagh uncut site compared to the clear-cut site. The higher mineral content in Skerdagh is reflected as lower
organic matter and C contents, as well as higher Al and Fe contents in peat compared to the Glennamong sites (Table 3.1). However, the most drastic difference between the four sites was that the Skerdagh uncut site had exceptionally high contents of calcium, iron, aluminium, and manganese compared with all other peat soils presented in this study (Chapter 2; Chapter 4; Chapter 6).

Table 3.1. Peat properties of the study sites at 0–20 cm peat layer.

<table>
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<th>Sites</th>
<th>Glennamong clear-cut</th>
<th>Glennamong uncut</th>
<th>Skerdagh clear-cut</th>
<th>Skerdagh uncut</th>
</tr>
</thead>
<tbody>
<tr>
<td>Organic Matter (%)</td>
<td>95.5</td>
<td>95.7</td>
<td>67.4</td>
<td>64.7</td>
</tr>
<tr>
<td>C (%)a</td>
<td>53.9</td>
<td>53.1</td>
<td>38.5</td>
<td>37.6</td>
</tr>
<tr>
<td>N (%)a</td>
<td>2.4</td>
<td>2.3</td>
<td>1.7</td>
<td>2.2</td>
</tr>
<tr>
<td>P (mg kg⁻¹)b</td>
<td>364</td>
<td>348</td>
<td>398</td>
<td>619</td>
</tr>
<tr>
<td>K (mg kg⁻¹)b</td>
<td>274</td>
<td>228</td>
<td>597</td>
<td>366</td>
</tr>
<tr>
<td>Ca (mg kg⁻¹)b</td>
<td>791</td>
<td>896</td>
<td>600</td>
<td>4850</td>
</tr>
<tr>
<td>Mg (mg kg⁻¹)b</td>
<td>1000</td>
<td>930</td>
<td>869</td>
<td>498</td>
</tr>
<tr>
<td>Fe (mg kg⁻¹)b</td>
<td>1080</td>
<td>1310</td>
<td>3110</td>
<td>72100</td>
</tr>
<tr>
<td>Al (mg kg⁻¹)b</td>
<td>1010</td>
<td>1340</td>
<td>1720</td>
<td>3550</td>
</tr>
<tr>
<td>B (mg kg⁻¹)b</td>
<td>3.52</td>
<td>3.49</td>
<td>1.73</td>
<td>0.026</td>
</tr>
<tr>
<td>Zn (mg kg⁻¹)b</td>
<td>36.0</td>
<td>4.33</td>
<td>4.65</td>
<td>16.3</td>
</tr>
<tr>
<td>Mn (mg kg⁻¹)b</td>
<td>10.9</td>
<td>4.59</td>
<td>10.3</td>
<td>2350</td>
</tr>
<tr>
<td>Pb (mg kg⁻¹)b</td>
<td>16.6</td>
<td>15.8</td>
<td>12.2</td>
<td>11</td>
</tr>
<tr>
<td>Ni (mg kg⁻¹)b</td>
<td>1.89</td>
<td>3.01</td>
<td>6.38</td>
<td>1.94</td>
</tr>
<tr>
<td>Cd (mg kg⁻¹)b</td>
<td>0.14</td>
<td>0.13</td>
<td>0.08</td>
<td>0.026</td>
</tr>
<tr>
<td>Cu (mg kg⁻¹)b</td>
<td>1.66</td>
<td>1.95</td>
<td>0.859</td>
<td>0.150</td>
</tr>
</tbody>
</table>

a) LECO CHN-1000 analyzer.
b) Dry ashing + digestion in HCl and analysed using ICP/AES.

3.2.2 Decomposition experiment and analysis

The needles for both species were collected from the Glennamong clear-cut area a few hours after harvesting. The Glennamong catchment is dominated by Lodgepole pine, but there were also a sufficient number of Sitka spruce trees to allow us to collect the needles for both species from the same site. The needles were air dried in the laboratory and hand mixed thoroughly.
Subsamples were drawn for the determination of the initial nutrient and heavy metal concentrations of the needles.

Litter bags of size 10 × 10 cm were made from nylon sheets with a mesh size of 1 × 1 mm. 1.5–2.0 g (d.w.) of harvest residue needles was enclosed in the litter bags, and all the bags were equipped with plastic tags showing the initial needle material weight. The litter bags were placed systematically in 5 sampling spots (at a distance of about 10 m from each other) between the furrows (BF) and 5 sampling spots inside the furrows (IF) at each of the four areas, with four pine needle and four spruce needle bags in each spot. The bags were placed directly on top of the litter layer and were held in place by plastic pins. In the clear-cut sites, the bags were installed in the harvest residue free areas. The incubation study lasted two years (from November 2009 to November 2011) and the litter bags were sampled at 0.5, 1.0, and 2.0 years from the start of experiment. On each sampling occasion, one bag for pine and one for spruce was systematically collected from each of the 10 sampling spots (5 for BF and 5 for IF) from each of the four sites. The remaining one spruce needle and one pine needle bag left in each sampling spot will be collected later.

After collection, the litter samples were cleansed of any foreign material, dried at +60ºC to constant mass, and weighed for the remaining dry mass. For chemical analyses, the contents of 5 litter replicate bags were combined and ground using MF 10 IKA-analytical mill to pass a 2 mm sieve. Samples were analysed for N and C concentrations using LECO CHN 1000 analyser, and for P, B, K, Ca, Mg, Fe, Al, Mn, Zn, Pb, Ni, Cu, and Cd concentrations using ICP/AES following dry digestion in HCl. Subsamples drawn for the initial concentrations of elements were analysed the same way and Table 3.2 shows the initial concentrations in harvest residue pine and spruce needles. We investigated the changes in mass loss and element contents (concentration, mg g⁻¹ × remaining mass, g) in the decaying needles during two years. The changes in mass loss and nutrient contents during decomposition are expressed as percentage values from the initial content remaining at each sampling occasion.

The effect of tree species (Lodgepole pine / Sitka spruce), litter bag placement (inside furrows / between furrows), and stand treatment (uncut / clear-cut) on nutrient and heavy metal dynamics in decaying harvest residue needle litter were tested with repeated analysis of variance (ANOVA). The within-subject variable was sampling time, with three levels (0.5, 1.0, 2.0 years
after litter bag installation). Analyses were performed separately for the two sites (Glennamong and Skerdagh) due to their different peat chemical characteristics (Table 3.1), possibly contributing to nutrient and heavy metal dynamics in harvest residue needles. The basic multivariate test was used and the Wilks' Lambda significance value is reported. However, if the assumption of sphericity was violated (\( p < 0.05 \) in Mauchly’s test of sphericity.), the Huynh-Feldt (\( \varepsilon > 0.75 \)) or Greenhouse-Gesser (\( \varepsilon < 0.75 \)) correction were applied to the degrees of freedom. Statistical tests were performed with the SPSS software package version 17 (SPSS/PASW Inc. Chicago).

**Table 3.2.** Initial element concentrations and ratios of C/P, C/N and N/P in harvest residue needles.

<table>
<thead>
<tr>
<th>Element</th>
<th>Pine needles</th>
<th>Spruce needles</th>
</tr>
</thead>
<tbody>
<tr>
<td>C (%)</td>
<td>51.8</td>
<td>52.1</td>
</tr>
<tr>
<td>N (%)</td>
<td>1.42</td>
<td>1.08</td>
</tr>
<tr>
<td>P (mg kg(^{-1}))</td>
<td>1170</td>
<td>1100</td>
</tr>
<tr>
<td>K (mg kg(^{-1}))</td>
<td>5890</td>
<td>5900</td>
</tr>
<tr>
<td>Ca (mg kg(^{-1}))</td>
<td>1220</td>
<td>3950</td>
</tr>
<tr>
<td>Mg (mg kg(^{-1}))</td>
<td>1010</td>
<td>1270</td>
</tr>
<tr>
<td>Fe (mg kg(^{-1}))</td>
<td>27</td>
<td>21.1</td>
</tr>
<tr>
<td>Al (mg kg(^{-1}))</td>
<td>255</td>
<td>30.2</td>
</tr>
<tr>
<td>B (mg kg(^{-1}))</td>
<td>15.2</td>
<td>19.5</td>
</tr>
<tr>
<td>Zn (mg kg(^{-1}))</td>
<td>22.2</td>
<td>6.45</td>
</tr>
<tr>
<td>Mn (mg kg(^{-1}))</td>
<td>177</td>
<td>511</td>
</tr>
<tr>
<td>Pb (mg kg(^{-1}))</td>
<td>0.172</td>
<td>0.147</td>
</tr>
<tr>
<td>Ni (mg kg(^{-1}))</td>
<td>0.351</td>
<td>0.144</td>
</tr>
<tr>
<td>Cd (mg kg(^{-1}))</td>
<td>0.091</td>
<td>0.021</td>
</tr>
<tr>
<td>Cu (mg kg(^{-1}))</td>
<td>0.705</td>
<td>1.48</td>
</tr>
<tr>
<td>C/N</td>
<td>36.5</td>
<td>48.2</td>
</tr>
<tr>
<td>C/P</td>
<td>442.7</td>
<td>473.6</td>
</tr>
<tr>
<td>N/P</td>
<td>12.1</td>
<td>9.8</td>
</tr>
</tbody>
</table>
3.3 Results

During the two-years study period harvest residue needles decomposed significantly ($p<0.001$) faster in the uncut than in clear-cut sites both at Glennamong and Skerdagh; the average mass loss from both sites and both species after two years was 65% at the uncut and 50% at clear-cut sites with no significant difference between pine and spruce needles (Fig. 3.1). The effect of litterbag placement was significant in explaining the variation in needle mass loss rates at Skerdagh ($p<0.01$), but the difference between the litter bags placed inside and between the furrows was small. The release of C correlated strongly with the mass loss (Fig. 3.1) of the needles ($r^2 = 0.998$, $p<0.001$).

Fig. 3.1. Changes in mass of litter and carbon content during 2 years incubation as % of initial mass remaining. G = Glennamong, S = Skerdagh, BF = Between the drainage furrows, IF = Inside the drainage furrows, H = Harvested (clear-cut) plot, F = Forest (uncut) plot.
There was a significant difference in both N and P dynamics between the clear-cut and uncut sites both at Glennamong (p<0.001) and Skerdagh (p<0.01). During two years, the average net release of N was 37% at the Glennamong uncut site and 32% at the Skerdagh uncut site, whereas no release occurred at the clear-cut sites (Fig. 3.2). Nitrogen release did not differ significantly between pine and spruce needles. The litterbag placement was significant in explaining the differences in N dynamics only at Skerdagh (p<0.05), where the average release was slightly more from the sampling positions between furrows than inside furrows.

**Fig. 3.2.** Observed changes in N and P contents during 2 years incubation as % of initial mass remaining. G = Glennamong, S = Skerdagh, BF = Between the drainage furrows, IF = Inside the drainage furrows, H = Harvested (clear-cut) plot, F = Forest (uncut) plot.

The release of P from decaying needles was considerable from all sites and litterbag placement except that there was some initial accumulation for both spruce and pine needles inside the
furrows of the clear-cut Skerdagh site. The interaction of litterbag placement and stand treatment was significant in explaining the differences in P dynamics at Skerdagh \((p<0.05)\). There was no significant difference in P release between the two tree species. The average release of P after 2 years was 53% at Glennamong clear-cut site and 38% at the Skerdagh clear-cut site. The release from uncut sites was on average 20% faster than from the clear-cut sites (Fig. 3.2).

B and K were released fast from needles. During the first six months, on average (from all sites and needle types) 80–90% of K and 50–75% of B were released from the needles (Fig. 3.3). The net release of Ca and Mg were faster from spruce than from the pine needles. In two years, the net release of Ca at the Glennamong clear-cut site was 37% for pine and 71% for spruce, and at the Glennamong uncut site it was 45% and 82%, respectively (Fig. 3.3). At the Skerdagh clear-cut site the release of Ca was almost the same as for the Glennamong clear-cut site, whereas at the Skerdagh uncut site pine needles accumulated Ca, especially inside the furrows, while spruce needles released Ca (Fig. 3.3). Similarly, the net release of Mg at the Glennamong clear-cut site was 45% for pine and 58% for spruce, and at the Glennamong uncut site it was 63 and 72%, respectively (Fig. 3.3). At the Skerdagh clear-cut site the release of Mg was 55% from pine and 66% from spruce needles, whereas at the Skerdagh uncut site it was almost the same for spruce and pine (Fig. 3.3).

The forest treatment factor (uncut vs. clear-cut) was significant in explaining Ca \((p<0.01)\), Mg \((p<0.001)\), as well as Mn \((p<0.05)\) release dynamics at Glennamong, and also Mn \((p<0.001)\) dynamics at Skerdagh. There was no difference in Mn release between sites, needle types or litter bag placement at Glennamong and net release on average was 92% during two years. At the Skerdagh clear-cut site, Mn release from needles was similar to Glennamong sites, but at uncut site Mn was accumulated after an initial release period, especially in the pine needles (Fig. 3.4). Zinc was retained or accumulated in the spruce needles but was released from the pine needles. The difference between the tree species was significant both at Skerdagh \((p<0.01)\) and Glennamong \((p<0.01)\) (Fig. 3.4).
Fig. 3.3. Observed changes in B, K, Ca and Mg contents during 2 years incubation as % of initial mass remaining. G = Glennamong, S = Skerdagh, BF = Between the drainage furrows, IF = Inside the drainage furrows, H = Harvested (clear-cut) plot, F = Forest (uncut) plot.
Fig. 3.4. Observed changes in Zn and Mn contents during 2 years incubation as % of initial mass remaining. G = Glennamong, S = Skerdagh, BF = Between the drainage furrows, IF = Inside the drainage furrows, H = Harvested (clear-cut) plot, F = Forest (uncut) plot.

At Skerdagh, Al accumulated by three- to eight-fold in the spruce needles that were placed inside the furrows compared with initial Al contents (Fig. 3.5). However, the most remarkable effect of litter bag placement was the substantial accumulation of Fe in the needles placed inside the furrows. Iron was accumulated significantly (p<0.001) in the needles of both pine and spruce in the furrows of Skerdagh, the Fe contents after two years were over 100-fold higher than the initial Fe contents of the needles (Fig. 3.5).
Fig. 3.5. Observed changes in Al and Fe contents during 2 years incubation as % of initial mass remaining. G = Glennamong, S = Skerdagh, BF = Between the drainage furrows, IF = Inside the drainage furrows, H = Harvested (clear-cut) plot, F = Forest (uncut) plot. Note that logarithmic scale is used for Skerdagh sites.

Lead was accumulated in both pine and spruce needles, especially inside the furrows of the Skerdagh uncut site, where its contents in the end of study period were 10- to 30-fold higher compared with initial contents (Fig. 3.6). Copper was accumulated in pine needles, especially in the furrows of the Skerdagh uncut site. Copper was released from spruce needles, except inside the furrows of the Skerdagh uncut site, where it accumulated in spruce needles (Fig. 3.6).
Fig. 3.6. Observed changes in Pb, Cu, Ni and Cd contents during 2 years incubation as % of initial mass remaining. G = Glennamong, S = Skerdagh, BF = Between the drainage furrows, IF = Inside the drainage furrows, H = Harvested (clear-cut) plot, F = Forest (uncut) plot. Note that for Pb and Cu, logarithmic scale is used for Skerdagh sites.
At Glennamong, the litterbag placement was not significant in explaining the variation in heavy metal dynamics. However, the effect of forest stand treatment (uncut vs. clear-cut) was a significant factor in explaining the variation in needle Ni \((p<0.001)\) and Pb \((p<0.01)\) contents. Lead accumulation was significantly faster at the clear-cut site compared with the uncut, forest site (Fig. 3.6). Over two years, Ni was released by about 20–30% faster from the pine needles and accumulated by about 50% slower in the spruce needles at the Glennamong uncut site than the Glennamong clear-cut site (Fig. 3.6). The Cd dynamics in decaying needles followed a pattern of repeated phases of release and accumulation. However, after two years, Cd had been released from pine needles and accumulated in spruce needles. The average net release from spruce needles across all sites and sampling positions was approximately 50%, and the net accumulation in the pine needles was 70%.

### 3.4 Discussion

Clear-cutting resulted in lower decomposition of harvest residue needles: over the two-year study period the harvest residue needles decomposed 20–30% faster at the uncut than the clear-cut sites. During two years N was not clearly released or was slightly accumulated in the needles at clear-cut sites, which is consistent with the results of many earlier studies (Hyvönen et al., 2000; Palviainen et al., 2004b; Kaila et al., 2012). However, considerable N release took place from the uncut sites (32–37% of initial N content), which has not been reported earlier on peatlands. This high net release of N from the needles of the uncut sites is directly linked to their higher mass loss compared with the needles from the clear-cut sites, as the N concentrations (data not shown) did not indicate differences between uncut and clear-cut sites. Kaila et al. (2012) showed contrasting results from drained minerotrophic peatlands in Finland, where clear-cutting did not significantly affect the decomposition of spruce and pine harvest residue needles. Palviainen et al. (2004b) in their study on a mineral soil forest reported that clear-cutting decreased the decomposition of spruce needles. However, there was no difference in the decomposition of pine needles between clear-cut and uncut plots (Palviainen et al., 2004b).

There could be a number of reasons for the lower mass loss of needles in the clear-cut sites compared to the uncut sites. Decomposition of litter and harvest residues is a complex phenomenon influenced by the activity and nutrient demand of heterotrophic decomposers. The activity is regulated by environmental conditions such as soil temperature, nutrient availability
and moisture conditions (Gosz et al., 1973; Prescott, 2005; Laiho, 2006). Rise in water table level after harvesting may lead to anoxic conditions in surface peat, especially in high precipitation areas of blanket peat dominated soils. Anaerobic conditions could decelerate the aerobic decomposition of harvest residues. However, Laiho et al. (2004) found no difference in pine litter decomposition in drained (with low water table) and undrained (with high water table) peatland forest sites in southern Finland. Domisch et al. (2000) also observed that the mass loss of pine needles was not clearly affected by site drainage. Despite the rise in water level after harvesting, however, higher air (Kubin and Kemppainen, 1991) and soil (Huttunen et al., 2003) temperatures and wind velocity in the clear cut sites can lead to drying out of the surface peat layers, especially in the dry summer months, thus inducing moisture stress among microbial communities. Peat soils generally have a capillary fringe to move water to the surface in dry periods. However, if the surface peat eventually dries out, the harvest residues can loose their capillary connection to the subsurface water (Laurén and Heiskanen, 1997; Laurén and Mannerkoski, 2001). In contrast, the variation in temperatures and moisture in the surface soil under forest canopy of uncut sites is smaller than for clear-cut sites, thus maintaining more optimal moisture conditions for heterotrophic decomposers.

Apart from the initial accumulation in the needles inside the furrows of the Skerdagh clear-cut site, P was clearly released (38–53% of initial P content) from all sites and sampling positions. This study therefore supports the hypothesis presented in earlier decomposition studies (Palviainen et al., 2004b; Kaila et al. 2012) that P release from decaying harvest residues can be a significant source of P to receiving water courses, especially in the case of blanket peats and other peat soils with low P adsorption capacity (Tamm, 1974; Nieminen and Jarva, 1996; Asam et al., 2012). The initial accumulation of P inside the furrows of the Skerdagh clear-cut site could partly be explained by the fact that the harvest residue needles inside the furrows were also accumulating Fe and Al, which can act as sorption sinks for the P in furrow waters.

In contrast, negligible release of N from harvest residue needles on the harvested sites indicates that N release from harvest residues may not be an explanation for the enhanced N export that has been reported to occur from peat soils soon after harvesting (Cummins and Farrell, 2003b; Nieminen, 2004). Thus, increased mineralisation of N from peat could be a more probable cause for the reported high N exports. It should be noted, however, that despite of negligible N release
from harvest residues during the early stages of decomposition, the harvest residue piles can increase N export in case that they enhance N mineralisation and release in underlying soil (Rosén and Lundmark-Thelin 1988).

High release of easily soluble nutrients, for instance B, K, and Mg, in the early phases of decomposition is in accordance with earlier studies (Palviainen et al., 2004a; Kaila et al., 2012). However, the release in the present study was faster, possibly reflecting the effect of climatic conditions on nutrient leaching from harvest residues. Calcium started to be released only after about half a year. Slower release compared to K and Mg is because it is a structural component of needle litter and hence its release depends on microbial activity rather than direct leaching from needle cell solution (Blair, 1988). Other studies on mineral soil forests (Palviainen et al., 2004a) and forestry-drained peatlands (Kaila et al., 2012) have reported the Ca release to start after a lag period of approximately one year.

Among the different heavy metals, Mn was clearly released from harvest residue needles from the three sites, with a net release of >90% of the initial Mn content during two years, but it was accumulated at the Skerdagh uncut site, especially in the pine needles. Needle litters have been shown to mostly release Mn (Laskowski and Berg, 1993; Kaila et al., 2012). Accumulation or retention of Mn in the needles of the uncut Skerdagh site is most likely because of the high Mn contents in peat. Manganese content was over 200 times higher at the Skerdagh uncut site compared with the other three study sites and Mn accumulation and absorption from soil could explain the high Mn accumulation in harvest residue needles (Table 3.1). Similarly higher accumulation of Al in spruce needles at the Skerdagh uncut site compared with the other three sites and significantly higher accumulation of Fe in the furrows of both Skerdagh sites compared with the Glennamong sites could be due to high contents of these elements in the soils of respective sites (Table 3.1). However, the mechanisms by which litters accumulate Fe, Al, and Mn from soil need further studies.

Copper also clearly accumulated in the needles inside the furrows of the uncut Skerdagh site, although Cu contents in soil were very low (Table 3.1). However, inside the furrows, in particular, the accumulation of elements may be related more to soil characteristics upstream from the study site than the in situ soil characteristics. The litters decaying inside the furrows are in close contact with the water discharging from the upstream blanket peat area and after losing
their protective epiderm and cuticule the litters may easily absorb metals and other elements from this water (Tyler, 1972). Other than the differences in soil characteristics, one reason for the different accumulation and release behaviour of elements between Glennamong and Skerdagh could be their different topography. Less sloping terrain in Skerdagh can increase water residence time and hence the contact time between needle litter and elements in through-flowing water.

Zinc, Ni, and Cd were mostly accumulated in the spruce needles, but released from the pine needles, which is in accordance with the results by Kaila et al. (2012) from nutrient-rich minerotrophic peatlands in Finland. Among the different heavy metals, Pb was the only one showing any difference between the uncut and clear-cut sites (Fig. 3.6). Higher accumulation of Pb in the clear-cut sites is difficult to explain, but could be due to more atmospheric Pb deposition reaching the soil floor at clear-cut sites compared to uncut sites. High tree density in the study sites (>2800 tree ha\(^{-1}\), Rodgers et al., 2010) may not enable much of the Pb deposition reaching the forest floor inside the forest.

3.5 Conclusions

The nutrient and heavy metal dynamics were studied in decaying Sitka spruce and Lodgepole pine harvest residue needles inside and between furrows in forested and clear-cut sites at two blanket peat areas in west of Ireland. The main findings are:

- The harvest residue needles decomposed faster at uncut than at clear-cut sites
- Phosphorus was released from decaying harvest residue needles from all sites and faster from uncut sites, while N was released only from uncut sites
- B, K, and Mg release from decaying harvest residue needles was faster than reported earlier, possibly reflecting the influence of climatic factors on leaching from needle cell solution
- Aluminium, Ni, Cd, and Zn were released less or accumulated more in the spruce than pine needles, whereas Ca and Mg were released faster from spruce needles
Manganese was released from three sites and had a total release of over 90% during two years, but accumulated in harvest residue needles in a site, where Mn contents in peat were exceptionally high.

- Aluminum accumulated in spruce needles and Fe in both spruce and pine needles inside the furrows of the site, where Al and Fe contents in peat were high.

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Chapter 4: The effect of water table rise on nutrient and dissolved organic carbon (DOC) release from re-wetted peat profiles

This paper has been presented at the International Peat Congress 2012. This chapter is an extended version of the conference paper. Zaki Asam designed and manufactured the columns and sampler used in this experiment. He also collected the samples from the field and designed the laboratory experiment. He conducted the majority of the laboratory analytical work. He also collected, analysed and synthesized the experimental data, and is the primary author. Mark O’Connor helped with field work and Annu Kaila conducted part of this experiment in Finland.
4. The effect of water table rise on nutrient and dissolved organic carbon (DOC) release from re-wetted peat profiles

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Abstract

Large areas of pristine European peatlands have been drained for forestry or other purposes. Ecosystem services, protection of key biotopes, and failure to grow economically sustainable forest on peatlands have created needs to restore drained peatlands to natural mire ecosystems. However, peatland restoration can have detrimental effects on the quality of runoff water and receiving water courses. During the restoration process, harvesting of tree stands, the filling-in or blocking of ditches and the creation of dams, leads to significant rises in the water table (WT), which results in anoxic conditions in previously aerobic peat. This study was carried out to assess the effect of WT rise on phosphorus (P), iron (Fe), ammonium-N (NH₄⁺-N), nitrate-N (NO₃⁻-N), and dissolved organic carbon (DOC) mobilization in a range of peat types with differing physico-chemical characteristics using a laboratory microcosm incubation experiment. Peats samples were collected from seven contrasting sites in Ireland and Finland, and were incubated under water logged (high WT) or field moisture (low WT) conditions. It was hypothesised that a high WT will cause higher mobilization of P from nutrient-poor, ombrotrophic peats compared to nutrient-rich, minerotrophic peats and higher mobilization of nitrogen (N) and DOC from more nutrient-rich peats. The results showed P mobilization from nutrient-poor, ombrotrophic peats. Mobilization of P from nutrient-rich peat only occurred in peats with a low Fe:P ratio. There was higher mobilization of Fe, NH₄⁺-N and DOC from nutrient-rich, minertrophic peats compared to nutrient poor peats. There was no mobilization of NO₃⁻-N under high WTs. These results suggest water-logged conditions to cause different release behaviour of nutrients and DOC from different types of peat soils depending on their biogeochemical characteristics.

Keywords: Nutrient-poor ombrotrophic peat, nutrient-rich minerotrophic peat, water table rise.
4.1 Introduction

Up to 60% of pristine European peatlands have been drained for forestry or other purposes, and in countries such as Great Britain, France and Germany, this figure is up to 90%. Human-induced changes in peatlands comprise about 50% for agriculture, 30% for forestry and 10% for peat extraction (Joosten, 1997). Today, ecosystem services, protection of key biotopes, and failure to grow economically sustainable forests on peatlands, have created needs to restore drained peatlands to natural mire ecosystems. Currently, there are international efforts to protect peatlands or to rehabilitate their functions in a landscape through restoration measures (Erwin, 2009). However, peatland restoration can have detrimental effects on the quality of runoff water and receiving water courses. The fact that headwaters near these new restoration areas have already been affected by peatland drainage for decades stresses the importance of avoiding further loading (Koskinen et al., 2011).

The main hydrological threat of the restoration of drained peatlands is the increase in leaching of phosphorus (P) (Sallantaus 2004; Nieminen et al., 2005a; Zak et al., 2010; Koskinen et al., 2011), although minor or negligible increases in P leaching have also been reported (Vasander et al., 1998; Urbanová et al., 2010). In addition, active restoration work such as filling-in of ditches and cutting of tree stands have been shown to initially lead to increased nitrogen (N) and dissolved organic carbon (DOC) exports (Sallantaus, 2004; Zak et al., 2010; Urbanová et al., 2010; Koskinen et al., 2011). Nutrients and DOC leaching after restoration of peatlands drained for forestry are different for different types of peatlands. Koskinen et al. (2011) reported high P leaching after restoration of a nutrient-poor, ombrotrophic peat site and lower P leaching from a nutrient-rich, minerotrophic peat site. They also found that ammonium-N (NH₄⁺-N) and DOC leaching per restored hectare was higher from the nutrient-rich site compared to the nutrient-poor site. Most of the research carried out on restoration of peatlands has focused on nutrient-rich peatlands originally drained for agricultural use (Kalbitz et al., 2002; Zak and Gelbrecht, 2007; Meissner et al., 2008; Niedermeier and Robinson, 2009) or on peatlands utilized for peat extraction (Lamers et al., 2002; Waddington et al., 2008). Information on the restoration of peatlands drained for forestry is scarce (Koskinen et al., 2011).

During the restoration process, a significant rise in the water table (WT) results in anoxic conditions in previously aerobic peat. Under reduced conditions, microbiologically-induced redox
reactions are important for the mobilization of dissolved substances (Lamers et al., 2002; Lucassen et al., 2004; Shenker et al., 2005). Peat properties can be important in determining the mobilization of nutrients as a result of a rise in WT. Phosphorus leaching can increase under anaerobic conditions, particularly if a significant proportion of P in peat before water table rise was adsorbed by iron (Fe) hydroxides or oxides. The WT rise may also increase P leaching from undissolved fertilizer if the site was previously fertilized with a low release P fertilizer. In addition, nutrient export may increase from the root system and dying mychorriza associations as a result of rising WT.

In this study, the effect of a rise of WT on P, Fe, NH$_4^+$-N, nitrate-N (NO$_3^-$-N) and DOC mobilization across a range of peat types with differing physico-chemical characteristics was investigated using a laboratory microcosm incubation experiment. It was hypothesised that there would be a higher mobilization of P from nutrient-poor, ombrotrophic peats compared to nutrient-rich, minerotrophic peats, owing to their poor P adsorption capacity and lower ratios of Fe- and Al-to-P. It was also hypothesised that there would be higher mobilization of N and DOC from nutrient-rich peats, as there will be more recently dead and easily releasable organic material in the top-soil compared with nutrient-poor peats.

The study focused on peatlands drained for forestry in Finland and Ireland. Almost one-third of European peatlands are in Finland (Montanarella et al., 2006) and more than half of it has been drained, mostly for forestry. Ireland possesses significant portion of the world's blanket bogs and is one of the most important countries in Europe for this type of habitat (Sottocornola and Kiely, 2005). However, there is currently only 20% of the original area remaining in a relatively intact condition (Foss and O’Connell, 1998).

4.2 Material and Methods

4.2.1 Sample collection and incubation study

Peat samples were collected from seven contrasting peatland sites located in Finland and Ireland. In Finland, the peat samples were collected from three nutrient-poor, ombrotrophic sites: ombrotrophic, slightly decomposed peat (O); ombrotrophic, highly decomposed peat (O$_{HUM}$); ombrotrophic, ash-fertilized peat (O$_{ASH}$); and two minerotrophic sites: oligotrophic peat from moderately productive site (M$_{OLIG}$) and highly nutrient-rich, decomposed peat from a highly
productive site (M_{MESO}). In Ireland, peat samples were collected from a low productive, moderately decomposed blanket bog (B) and a moderately productive, highly decomposed blanket bog (B_{HUM}) (Table 4.1).

Table 4.1. General background information about the sample collection sites.

<table>
<thead>
<tr>
<th>Peatland name</th>
<th>Country</th>
<th>Site ID</th>
<th>Location</th>
<th>General description of the sampling area</th>
</tr>
</thead>
<tbody>
<tr>
<td>Lakkasuo</td>
<td>Finland</td>
<td>O</td>
<td>61° 47'N, 24° 18'E 162 m a.s.l</td>
<td>Ombrotrophic site. Relatively intact (near-pristine) peatland</td>
</tr>
<tr>
<td>Jaakkoinsuo</td>
<td>Finland</td>
<td>O_{HUM}</td>
<td>62° 3'N, 24° 29'E 120 m a.s.l</td>
<td>Ombrotrophic site. Old drainage area forest with pine trees</td>
</tr>
<tr>
<td>Taavetti</td>
<td>Finland</td>
<td>O_{ASH}</td>
<td>60° 55'N, 27° 33'E 85 m a.s.l</td>
<td>Ombrotrophic site, drained and ash fertilized. Clearfelled before sampling</td>
</tr>
<tr>
<td>Srahrevagh</td>
<td>Ireland</td>
<td>B</td>
<td>54°00'N, -9°32'E, 220 m a.s.l</td>
<td>Low productive blanket peatland site, drained and forested with lodgepole pine</td>
</tr>
<tr>
<td>Teevaloughan</td>
<td>Ireland</td>
<td>B_{HUM}</td>
<td>53°58'N, -9°31'E, 245 m a.s.l</td>
<td>Moderately productive blanket peatland site, drained and forested with Sitka spruce</td>
</tr>
<tr>
<td>Lakkasuo</td>
<td>Finland</td>
<td>M_{OLIG}</td>
<td>61° 47'N, 24° 18'E 162 m a.s.l</td>
<td>Oligotrophic site, dominated with pine forests</td>
</tr>
<tr>
<td>Jaakkoinsuo</td>
<td>Finland</td>
<td>M_{MESO}</td>
<td>62° 3'N, 24° 29'E 120 m a.s.l</td>
<td>Minerotrophic site. Old drainage area and dominated with spruce forests</td>
</tr>
</tbody>
</table>

To collect peat samples, a small (2 m x 2 m) plot was identified on each site. To obtain information on basic peat properties of the sample collection area, four replicates of peat samples were collected from a depth of 0–20 cm using a 10-cm-diameter core sampler. For the incubation study, the peat samples were collected using 10-cm-diameter and 65-cm-long PVC tubes. The PVC tubes were pushed into the peat down to a depth of 55–60 cm and the PVC tube with peat inside was dug around and the peat column was pulled up. Using this procedure, eight peat column samples were collected from each site and were taken to the laboratory. The samples for basic peat properties were stored in the laboratory at -18°C until further analysis. The peat columns for the incubation study were sealed from the bottom with a water-tight lid and water adjustment pipes were installed as shown in Fig. 4.1.
Four samples per site were incubated under anaerobic, waterlogged conditions by adjusting the water level to about 4 cm above the peat surface (high WT) and the other four were incubated at approximately field moisture content by adjusting the water level 40–50 cm below the peat surface (low WT). The experiment was carried out concurrently in both countries. After WT adjustment, the peat columns were incubated at 18°C for about 40 weeks and the water levels were kept constant by adding deionized water. Pore water samples were taken fortnightly or once a month at an approximate depth of 10–20 cm using suction samplers (Rhizon soil moisture filters, pore size 0.15–0.2 µm) with a polymeric tip of 9 cm in length and 4.5 mm in diameter, attached to a removable syringe (Fig. 4.1). After collection, the pore water samples were stored at 4°C (for a maximum period of 2 weeks) until further analysis was conducted.

**Fig. 4.1.** Incubation experiment setup in the laboratory.
4.2.2 Analysis of peat and water samples

Peat samples for the basic soil properties were dried at 60°C to a constant mass, weighed, milled to pass a 2 mm sieve and analysed with ICP/AES for Al, Fe, Ca and P following dry digestion in HCl, and for N and C using a Leco CHN 1000 analyser. Bulk density was calculated using the dry mass of the sample and the volume of core sampler (10 cm–diameter, 20 cm–length). Immediately after collection, the redox potential and pH of pore water samples were measured using an oxidation-reduction potential (ORP) probe and a pH meter, respectively. The pore water samples for the two Irish sites were analyzed for ortho-phosphorus (PO₄-P), Fe, NH₄⁺-N and NO₃⁻-N, using a Konelab 20 Analyser (Konelab Ltd., Finland) and dissolved organic carbon (DOC) using a Sievers 5310C Total Organic Carbon (TOC) analyser. The pore water samples from simultaneous experiment in Finland were analyzed for NH₄⁺-N and NO₃⁻-N using Tecaton FIA-analyzer, Fe was analyzed with ICP/AES, ARL-3580 PO₄-P colorimetrically with molybdenum blue method, and DOC with Shimadzu carbon analyzer.

The difference between the pore water concentrations of nutrients and DOC in aerobic and anaerobic conditions and between nutrient-poor and nutrient-rich peats were analysed using paired samples t-test at the 95% significance level (p = 0.05) using SPSS statistical tool (http://www.spss.com).

4.3 Results

4.3.1 Peat properties

The ash fertilized ombrotrophic site from Finland (OASH) had a similar bulk density (120 kg m⁻³) to the nutrient-rich, minerotrophic sites, but the other nutrient-poor sites had a lower bulk density (53–96 kg m⁻³). Peat pH varied from 3.7 to 4.3 and carbon content was between 50 and 55% (Table 4.2). The N content of the Finnish ombrotrophic sites (O, OHUM, OASH) was lower (1.0–1.6%) compared with the nutrient-rich, oligotrophic (MOLIG) and minerotrophic (MMESO) sites (2.4–2.6%). The N content of the moderately decomposed blanket bog (B) in Ireland was almost the same (2.4%) as the nutrient-rich, minerotrophic sites in Finland, whereas the highly decomposed blanket bog (BHUM) had a higher N content (3.0%) than the other six sites.
Table 4.2. Peat properties of selected sites at 0–20 cm layer. Values are an average of four replicates and ± indicates the standard error of mean.

<table>
<thead>
<tr>
<th>Site ID</th>
<th>BD kg m⁻³</th>
<th>C&lt;sub&gt;t&lt;/sub&gt; %</th>
<th>N&lt;sub&gt;t&lt;/sub&gt; %</th>
<th>P&lt;sub&gt;t&lt;/sub&gt; kg ha⁻¹</th>
<th>Fe&lt;sub&gt;t&lt;/sub&gt; kg ha⁻¹</th>
<th>Al&lt;sub&gt;t&lt;/sub&gt; kg ha⁻¹</th>
<th>Ca&lt;sub&gt;t&lt;/sub&gt; kg ha⁻¹</th>
<th>F&lt;sub&gt;e&lt;/sub&gt;:P&lt;sub&gt;t&lt;/sub&gt; mole mole⁻¹</th>
<th>A&lt;sub&gt;l&lt;/sub&gt;:P&lt;sub&gt;t&lt;/sub&gt; mole mole⁻¹</th>
<th>C&lt;sub&gt;a&lt;/sub&gt;:P&lt;sub&gt;t&lt;/sub&gt; mole mole⁻¹</th>
</tr>
</thead>
<tbody>
<tr>
<td>O</td>
<td>53 ± 3.0</td>
<td>49.6 ± 0.0</td>
<td>1.0 ± 0.0</td>
<td>55 ± 2</td>
<td>128 ± 3</td>
<td>43 ± 1</td>
<td>174 ± 1</td>
<td>1.3</td>
<td>0.9</td>
<td>2.4</td>
</tr>
<tr>
<td>O&lt;sub&gt;HUM&lt;/sub&gt;</td>
<td>96 ± 2.5</td>
<td>55.5 ± 0.0</td>
<td>1.6 ± 0.0</td>
<td>102 ± 1</td>
<td>150 ± 4</td>
<td>134 ± 2</td>
<td>610 ± 10</td>
<td>0.8</td>
<td>1.5</td>
<td>4.6</td>
</tr>
<tr>
<td>O&lt;sub&gt;ASH&lt;/sub&gt;</td>
<td>120 ± 4.6</td>
<td>52.6 ± 0.0</td>
<td>1.5 ± 0.0</td>
<td>182 ± 6</td>
<td>315 ± 10</td>
<td>768 ± 26</td>
<td>2421 ± 99</td>
<td>1.0</td>
<td>4.9</td>
<td>10.3</td>
</tr>
<tr>
<td>B</td>
<td>91 ± 1.0</td>
<td>53.4 ± 0.0</td>
<td>2.4 ± 0.0</td>
<td>77 ± 1</td>
<td>133 ± 6</td>
<td>148 ± 5</td>
<td>183 ± 11</td>
<td>1.0</td>
<td>2.2</td>
<td>1.8</td>
</tr>
<tr>
<td>B&lt;sub&gt;HUM&lt;/sub&gt;</td>
<td>110 ± 1.0</td>
<td>53.0 ± 0.0</td>
<td>3.0 ± 0.0</td>
<td>252 ± 7</td>
<td>2444 ± 157</td>
<td>226 ± 10</td>
<td>70 ± 7</td>
<td>5.4</td>
<td>1.0</td>
<td>0.2</td>
</tr>
<tr>
<td>M&lt;sub&gt;OLIG&lt;/sub&gt;</td>
<td>119 ± 2.3</td>
<td>54.7 ± 0.0</td>
<td>2.6 ± 0.0</td>
<td>206 ± 3</td>
<td>3211 ± 103</td>
<td>453 ± 3</td>
<td>515 ± 6</td>
<td>8.6</td>
<td>2.5</td>
<td>1.9</td>
</tr>
<tr>
<td>M&lt;sub&gt;MESO&lt;/sub&gt;</td>
<td>120 ± 4.0</td>
<td>53.8 ± 0.0</td>
<td>2.4 ± 0.0</td>
<td>211 ± 1</td>
<td>5768 ± 85</td>
<td>275 ± 3</td>
<td>461 ± 9</td>
<td>15.1</td>
<td>1.5</td>
<td>1.7</td>
</tr>
</tbody>
</table>

The total P and Fe content in peat in the nutrient-poor sites (O, O<sub>HUM</sub>, O<sub>ASH</sub>, B) was lower than for the nutrient-rich sites. The B<sub>HUM</sub> site had the highest P content (252 kg ha⁻¹) and M<sub>MESO</sub> had the highest Fe content (5768 kg ha⁻¹). The Al content was higher for the nutrient-rich peats compared to the nutrient-poor peats, except that the ombrotrophic ash fertilized peat, which had higher Al content (768 kg ha⁻¹), most probably because of the Al derived from ash (Nieminen et al., 2005b). The Ca content in B<sub>HUM</sub> was lower compared with the other six sites and Ca content was highest for O<sub>ASH</sub>, also because of the Ca derived from ash (Table 4.2). The molar ratio of Fe:to-P was lower for the nutrient-poor sites (0.8–1.3) compared to the nutrient-rich sites (5.4–15.1). The Al:P molar ratio for the nutrient-poor peats and nutrient-rich peats were almost the same and in the range of 0.9–2.5, except that the ombrotrophic ash-fertilized (O<sub>ASH</sub>) peat had a higher ratio of 4.9. The molar Ca:P ratio of nutrient-poor sites were higher (1.8–10.3) compared to the nutrient-rich sites (0.2–1.9). B<sub>HUM</sub> peat had the lowest Ca:P ratio and O<sub>ASH</sub> the highest (Table 4.2).

4.3.2 Mobilization of P and Fe from aerobic and anaerobic samples

The redox potential in pore water was significantly (p<0.05) lower for the high WT samples compared to the low WT samples for all seven sites. Under water logged conditions, the nutrient-rich peats (M<sub>MESO</sub>, M<sub>OLIG</sub>, B<sub>HUM</sub>) had lower redox values (100–190 mV) compared to nutrient-poor peats (O, O<sub>HUM</sub>, O<sub>ASH</sub>, B), which had values in the range of 222–244 mV (Fig. 4.2).
Fig. 4.2. The redox potential, pH value, and PO₄-P and Fe concentrations in the pore water samples of the different peats at a depth of 10–20 cm. Values are averages (+ standard error of mean) of all pore water samples taken over a period of 40 weeks from four replicates of peat samples.

In general, the pH was slightly higher for anaerobic samples compared to aerobic samples and was higher for Molig and Meso compared to the other sites. Under aerobic conditions (low WT), the pore water P concentrations were low and in the range of 0.01–0.24 mg L⁻¹, except that the Bhum site had slightly higher concentration of about 1.36 mg L⁻¹. The pore water P concentrations for anaerobic (high WT) samples of nutrient-poor peats were significantly (p<0.05) higher (2.0–8.1 mg L⁻¹) compared to aerobic samples (Fig 4.2). Among the nutrient-rich peats, the Bhum had the high pore water P concentration (17.2 mg L⁻¹), whereas P release from Molig and Meso peats was negligible. Iron concentrations in pore water of anaerobic samples were significantly (p<0.05) higher for the nutrient-rich peats (33–327 mg L⁻¹) compared to
nutrient-poor peats (0.4–1.6 mg L\(^{-1}\)) (Fig. 4.2). The P in pore water did not correlate with Fe in pore water or with total P or total Fe in peat. There was also no correlation of pore water P with molar ratios of Fe:P, Al:P, or Ca:P. Molar ratios of Fe:PO\(_4\):P in anaerobic pore water for nutrient-poor peats were lower compared to the nutrient-rich peats (Table 4.3).

Table 4.3. Pore water Fe:PO\(_4\):P ratios of anaerobic columns

<table>
<thead>
<tr>
<th>Site ID</th>
<th>Pore water Fe:PO(_4):P (mole mole(^{-1}))</th>
</tr>
</thead>
<tbody>
<tr>
<td>O</td>
<td>0.1</td>
</tr>
<tr>
<td>O(_{HUM})</td>
<td>0.1</td>
</tr>
<tr>
<td>O(_{ASH})</td>
<td>0.2</td>
</tr>
<tr>
<td>B</td>
<td>0.1</td>
</tr>
<tr>
<td>B(_{HUM})</td>
<td>3.0</td>
</tr>
<tr>
<td>M(_{OLIG})</td>
<td>95.5</td>
</tr>
<tr>
<td>M(_{MESO})</td>
<td>186.6</td>
</tr>
</tbody>
</table>

4.3.3 Mobilization of N and DOC from aerobic and anaerobic samples

During incubation, the concentrations of pore water NH\(_4\)\(^+\):N and DOC increased in anaerobic samples, but did not increase as much in the aerobic samples (Fig. 4.3). The increase in pore water NH\(_4\)\(^+\):N concentration in anaerobic samples was significantly (\(p<0.05\)) higher for the nutrient-rich peats (4.2–49.9 mg L\(^{-1}\)) compared to nutrient-poor peats (2.9–5.8 mg L\(^{-1}\)). Similarly, the increase in DOC was significantly (\(p<0.05\)) more for nutrient-rich peats (217–1080 mg L\(^{-1}\)) compared to nutrient-poor peats (83–285 mg L\(^{-1}\)). NO\(_3\)\(^-\):N increase was only seen for the aerobic nutrient-rich peats and moderately decomposed blanket bog (B), and there was no NO\(_3\)\(^-\):N increase in pore water for the ombrotrophic peats from Finland (Fig. 4.3). Pore water concentrations of NH\(_4\):N and DOC correlated positively with pore water Fe concentrations.
The observed higher release of PO₄-P from all rewetted nutrient poor, ombrotrophic peats was consistent with Koskinen et al. (2011), who reported a higher release of P from a restored ombrotrophic site compared to a restored minerotrophic site. The high release of P was most likely due to ombrotrophic peat soils having low concentrations of Fe and Al hydroxides or oxides, and therefore low P adsorption capacity (Kaila, 1959; Nieminen and Jarva, 1996). Nieminen and Penttila (2004) reported higher easily soluble inorganic P fractions for nutrient-poor than fertile peatland sites. Easily soluble P fractions are susceptible to leaching, which may
also explain why all ombrotrophic re-wetted peats released P into pore water. The high release of P from unhumified peat (O), reported in this study, contradicts the results of Zak and Gelbrecht (2007), who found negligible P export from slightly humified peat following re-wetting. However, their peat was not as mineral-poor as those used in the present study, indicating that there was more P sorption potential.

A lower redox potential and higher mobilization of Fe and NH$_4^+$-N in nutrient-rich peat types (Fig. 4.2 and 4.3) compared to nutrient-poor peats, suggests that there could be potential for P release also from nutrient-rich peats following re-wetting (Canfield et al., 1993; Qualls and Richardson, 2000; Zak and Gelbrecht, 2007). However, anaerobic conditions resulted in high P mobilization only from BHUM, whereas MOLIG and MESO indicated negligible or no P release (Fig. 4.2). Other studies also found differences in P mobilization between different nutrient-rich peats. Zak et al. (2010) found a range of P release rates from 0.1 to 52.3 mg m$^{-2}$ d$^{-1}$ in peat columns with highly decomposed fen peat. Urbanova et al. (2010) did not find any release of P following rewetting of drained minerotrophic fens that had been used as extensive pastures.

Productive and nutrient-rich peats have a higher fraction of inorganic P bound to Fe (Nieminen and Penttila, 2004) and have higher Fe:P ratios compared to nutrient-poor sites, as is also observed in this study (Table 4.2). The molar Fe:P ratio has been shown to correlate negatively with P mobilization from re-wetted nutrient-rich and highly decomposed peats (Zak et al., 2010). A high mobilization of P from highly decomposed blanket bog peat (BHUM), and not from MOLIG and MESO, may be because of higher P content and lower Fe:P, Al:P and Ca:P ratios (Table 4.2), indicating that there were fewer sorption sites for the P mobilized from peat due to re-wetting. Owing to the lower Ca and Al contents, it is also possible that proportionally more P in BHUM before re-wetting was adsorbed by Fe(III), from where P is easily released under anaerobic conditions.

Several studies have reported that P movement from pore water to surface water increases when the Fe:PO$_4$-P molar ratio of pore water is less than 1–3.5 (Lofgren and Bostrom, 1989; Smolders et al., 2001; Zak et al., 2004; Geurts et al., 2008; Geurts et al., 2010). In the present study, the pore water Fe:PO$_4$-P molar ratios for nutrient-poor peats were less than 0.2 (Table 4.3), suggesting a high risk of P release to surface waters after rewetting of these sites. There is also a
risk of P release from nutrient-rich, highly decomposed blanket bog (B\textsubscript{HUM}), which had a Fe:PO\textsubscript{4}-P molar ratio of 3. The other two nutrient-rich sites had Fe:PO\textsubscript{4}-P molar ratios of 96–187, suggesting no risk of P release as a result of rewetting.

A high release of NO\textsubscript{3}--N from the minerotrophic peats under aerobic conditions, and not in anaerobic conditions (Fig. 4.3), indicates low nitrification potential after re-wetting. Urbanová et al. (2010) suggested that in case of low nitrification potential, denitrification will become an unimportant process in N cycling in restored peatlands. Most of N in restored sites will be present either in the form of NH\textsubscript{4}+-N or organic forms. The observed higher NO\textsubscript{3}--N release from minerotrophic peats under aerobic conditions was consistent with earlier studies showing enhanced nitrification after lowering of the WT in nutrient-rich, minerotrophic peats but not in nutrient-poor, ombrotrophic peats (Regina et al., 1996). Nutrient poor blanket bog peat (B), unlike other nutrient poor peats (Fig. 4.3), surprisingly showed some NO\textsubscript{3}--N release under aerobic incubation. This may be because of its significantly higher N content compared with the other nutrient poor peats (Table 4.2).

The higher release of Fe and NH\textsubscript{4}+-N observed from M\textsubscript{MESO} and B\textsubscript{HUM} peats indicated a high microbial metabolism (Zak et al., 2010). The increase in DOC concentrations was also highest for these peats. It has been shown that the leaching of DOC increases for some time after restoration, when the waters reach the surface peat layers with recently dead organic material (Guggenberger et al., 1994). The results of the present study were consistent with those obtained by Koskinen et al. (2011) on restored peatlands previously drained for forestry. They found higher NH\textsubscript{4}+-N and TOC release from a nutrient-rich spruce mire area compared with a nutrient-poor area after restoration. An increase in NH\textsubscript{4}+-N and DOC from highly decomposed, nutrient-rich peat has also been reported by Zak and Gelbrecht (2007) and Urbanova et al. (2010).

In addition to the release of P from different inorganic fractions, the microbial control of P cycling is altered by rewetting (Baum et al., 2003), which may contribute to the release of P from organic forms. To explain the differences in P, DOC, Fe, and N release due to water-logging, further analysis is being conducted on the different biogeochemical and microbiologically-bound forms of P, N and C, and Al, Fe and Ca-bound P in the peats in the columns.
4.5 Conclusions

A high water table resulted in P mobilization from all nutrient-poor, ombrotrophic peats suggesting a risk of P leaching following the restoration of nutrient-poor sites. Phosphorus mobilization in case of nutrient-rich peats only occurred in the peat with low Fe:P, Al:P and Ca:P ratios. A high water table also caused higher mobilization of Fe, NH$_4^+$-N and DOC from nutrient-rich peats compared to nutrient-poor peats. There was no mobilization of NO$_3^-$-N under high WTs in any peat types. These results suggest water-logged conditions to cause different release behaviour of nutrients and DOC from different types of peat soils depending on their biogeochemical characteristics.

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Chapter 5: Nutrient release from brash windrows of Lodgepole pine (*Pinus contorta*) as a factor for P and N export from harvested blanket peat forests

This paper has been submitted to Agriculture, Ecosystems and the Environment. Zaki Asam designed the laboratory rainfall simulator experiment, prepared all the laboratory flumes used in this study and constructed the experimental plots (mini-catchments) in the field. Over a period of almost 3 years, he carried out majority of the relevant laboratory and field work, and the analytical work on water and peat soil samples. He also collected, analysed and synthesized the experimental data, and is the primary author of the paper. Mark O’Connor and Connie O’Driscoll helped with field work. The rest of the co-authors provided guidance and advice in writing the paper.
5. Nutrient release from brash windrows of Lodgepole pine (*Pinus contorta*) as a factor for P and N export from harvested blanket peat forests

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Abstract

The amount of cutting residues in clear-felled blanket peat forests dominated by Lodgepole pine may be many-fold larger than for peatland forests in other parts of Europe. In operational forestry on blanket peat catchments, the excessive cutting residues are utilized as brash mats to improve the soil carrying capacity against heavy harvesting machinery, before being collected to form brash windrows, between which the trees are planted. Three types of experiments were conducted to assess the impact of brash windrows, derived from Lodgepole pine forests, on phosphorus (P) and nitrogen (N) release to water courses from blanket peat catchments. First, a field survey was conducted in 6 areas clear-felled between 2005 and 2011 to study the easily extractable P and N contents in soil below brash windrows and respective brash free areas. The dynamics of N and P release when water flows from a brash windrow-covered area through a brash-free area was assessed using laboratory flume studies. Finally, the role of brash windrows in N and P export to water courses under field conditions was assessed using artificially isolated mini-catchments with and without brash material. The study showed that P release from brash windrows contributes significantly to soil P content and the exports of P from blanket peat catchments. The brash windrows may act as a sink rather than a source for N during the early stages of their decomposition. However, more N is exported from brash-covered than brash-free areas, the probable reason being that microbial activity and soil N mineralisation is increased under brash. The removal of cutting residues and whole-tree harvesting would be efficient means of decreasing P and N transport from blanket peat catchments.

Keywords: Blanket peat, Brash windrow, Laboratory flumes, mini-catchments
5.1 Introduction

Approximately 800,000 hectares of blanket peat catchments were afforested between the 1950s and 1990s in Ireland and the UK (Farrell, 1990; Hargreaves et al., 2003; EEA, 2004) and most of these forests are now approaching clear-felling and regeneration phase. Enhanced nutrient export from blanket peats and other peat soils after forest harvesting has been reported by several studies. Phosphorus (P) export increases especially from nutrient-poor, ombrotrophic sites with low contents of aluminium (Al) and iron (Fe) hydroxides and oxides in peat (Nieminen, 2003; Cummins and Farrell, 2003; Rodgers et al., 2010), and nitrogen (N) and dissolved organic carbon (DOC) from nutrient-rich, minerotrophic sites (Lundin, 1999; Nieminen, 2004). In conventional stem-only harvesting the harvest residue is left onsite. The amount of harvest residue largely depends on the tree species and the developmental stage of the tree stand (Hakkila, 1991; Hynynen, 2001).

Decomposition of cutting residue is seen as a potentially high source of nutrients to receiving water courses from drained peatlands (Rodgers et al., 2010, Kaila et al., 2012), but may have a particularly important role in N and P export from forested blanket peat catchments. Blanket peat sites were often afforested with Lodgepole pine (*Pinus contorta*) with little self-pruning of branches compared with many other tree species. The stands in blanket peat catchments are dense (>2800 stems ha\(^{-1}\), Rodgers et al., 2010) compared with those approaching clear-felling phase in countries such as Scandinavia (<1000 stems ha\(^{-1}\)). The volume of the tree stand in the final cutting phase is also significantly larger for blanket peat forests (>400 m\(^3\) ha\(^{-1}\)) than peatlands in more northern areas of Europe (150–300 m\(^3\) ha\(^{-1}\)). Therefore, the amount of cutting residues in clear-felled blanket peat forests may be many-fold larger than, for example, Scots pine (*Pinus sylvestris*) dominated peatlands in Scandinavia.

Asam (unpublished data) estimated that the cutting residues of Lodgepole pine amounts to over 80,000 kg (dry weight (d.w.)) ha\(^{-1}\), while the cutting residues of Scots pine in Fennoscandia amounts to only about 14,000 kg (d.w.) ha\(^{-1}\) (Palviainen and Finér, 2012). Owing to the milder climate, the decomposition of cutting residues and consequent nutrient release may also be significantly faster for blanket peat catchments in Ireland and the UK than for the more northern peatlands in Scandinavia, Russia, or Canada.
In operational forestry on blanket peat catchments, the harvest residues are utilized as brash mats to improve the poor soil-carrying capacity against the heavy harvesting machinery. Under the weight of harvesting machinery, the cutting residues are broken down into smaller fractions, which may also increase their decomposition. As cutting residues make the planting of trees difficult, they are collected together to form brash windrows, between which the trees are planted. These windrows, each with a width of about 4 m, run along the main slope in parallel rows at intervals of about 12 m down to about 20 m from the stream edge (Fig. 5.1).

The variation in soil temperatures and moisture below brash windrows is likely to be less than in brash-free areas, leading to an increased mineralization of nutrients in the soil below windrows (Rosén and Lundmark-Thelin, 1987). No vegetation establishes in brash windrows and there is no root uptake to prevent nutrient leaching. Furthermore, the vegetation in brash-free areas may not capture the nutrients released from brash, as lateral movement of waters from brash windrows to windrow-free areas is negligible in the steep slopes typical of blanket peat catchments, and any lateral movement of water is likely to enter into drainage furrows, which are at 3–5 m intervals along the same direction as the slope (Fig. 5.1). Therefore, the present harvesting and regeneration practice with brash windrows along the main slope is likely to significantly contribute to nutrient export from blanket peat catchments.

**Fig. 5.1.** Photo of a clear-felled blanket peat area in Srahrevagh river catchment.
Because of the low sorption capacity of blanket peat (Tamm et al., 1974; Asam et al., 2012), the P released from the brash windrows may be easily exported to downstream water bodies. However, the steep slopes in blanket peat catchments, low hydraulic conductivity of blanket peat (Cummins and Farrell, 2003; Lewis et al., 2012), and high precipitation of >2000 mm year\(^{-1}\) in areas of upland blanket peats means that the soil sorption and vegetation uptake of the other nutrients released from brash may also be low. The frequent occurrence of storm flow episodes under these conditions means that the contact time between water and vegetative and soil sinks of nutrients is too short for an efficient uptake and sorption of nutrients. However, the litter bag studies from peat soils (Kaila et al., 2012) suggest that cutting residues may only release P during the initial stages of their decomposition, and not act as a source of N. A number of studies indicate that cutting residues can even accumulate N from underlying soil during the initial stages of decay (Fahey et al., 1988; Hyvönen et al., 2000; Ouro et al., 2001; Palviainen et al., 2004; Kaila et al., 2012); the probable mechanism behind that is that micro-organisms (especially fungi) transport N from soil into litter, where decomposition is N-limited.

Three types of experiments were carried out to examine the role of brash windrows in P and N release and export to receiving water courses. (1) First a field survey was conducted in 6 areas clear-felled between 2005 and 2011 to study the easily extractable P and N contents in soil below brash windrows and brash-free areas. (2) The dynamics of N and P release when water flows from a brash windrow-covered area through a brash-free area was then assessed using a laboratory flume study (Mulqueen et al., 2006; Asam et al., 2012). (3) Finally, the role of brash windrows in N and P export to water courses was assessed under field conditions using artificially isolated mini-catchments with and without brash material. It was hypothesized that brash windrows are a significant source of P to receiving water courses, but may not contribute to the increased export of N, at least during the first few years after clear-felling, because brash windrows are more likely to act as a sink, rather than a source, of N during the early stages of litter decomposition (Kaila et al., 2012).

**5.2 Materials and methods**

**5.2.1 Field survey of brash impact on soil N and P**

Six sites ranging from 0.5 to 7 years since clear-felling were selected. The six sites are located within a 50 km\(^2\) area in the west of Ireland and are sub-sites of the Srahrevagh river catchment.
(54°00'N, -9°32'E, 220 m a.s.l.), the Glennamong river catchment (53°58'N, -9°37'E, 69 m a.s.l.) and the Skerdagh catchment (53°56'N, -9°31'E, 125 m a.s.l.). The sites were clear-felled between 2005 and 2011 (Table 5.1) using the stem-only harvesting method. Harvest residue was first used as brash mats and later collected together to form brash windrows. The dominant tree species on the selected sites was Lodgepole pine (*Pinus Contorta*), except on the Skerdagh site where it was mainly Sitka spruce (*Picea sitchensis*) with a mixture of Lodgepole pine. The peat soil is highly fibric with a peat layer of over 1 m and overlies mainly quartzite and schist bedrock. The sites have similar hydrological conditions and receive an average precipitation of approximately 2,000 mm year⁻¹. For more detail on the sites, please refer to O’Driscoll et al. (2011) and Chapter 6.

**Table 5.1.** General information on the study sites.

<table>
<thead>
<tr>
<th>Name of the catchment</th>
<th>Name of sub-site for sample collection</th>
<th>Site ID</th>
<th>Year of clear-felling</th>
<th>Year of soil sample collection</th>
<th>Time since clear-felling (year)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Srahrevagh river catchment</td>
<td>Srahrevagh-1</td>
<td>SR1 1st</td>
<td>Jan, 2011</td>
<td>2011</td>
<td>0.50</td>
</tr>
<tr>
<td></td>
<td></td>
<td>SR1 2nd</td>
<td></td>
<td>2012</td>
<td>1.50</td>
</tr>
<tr>
<td></td>
<td>Srahrevagh-2</td>
<td>SR2</td>
<td>Sept, 2010</td>
<td>2012</td>
<td>1.83</td>
</tr>
<tr>
<td></td>
<td>Srahrevagh-3</td>
<td>SR3 1st</td>
<td>Sept, 2005</td>
<td>2011</td>
<td>5.83</td>
</tr>
<tr>
<td></td>
<td></td>
<td>SR3 2nd</td>
<td></td>
<td>2012</td>
<td>6.84</td>
</tr>
<tr>
<td>Glennamong river catchment</td>
<td>Glennamong-1</td>
<td>GL1</td>
<td>Feb, 2011</td>
<td>2012</td>
<td>1.41</td>
</tr>
<tr>
<td></td>
<td>Glennamong-2</td>
<td>GL2 1st</td>
<td>Aug, 2009</td>
<td>2011</td>
<td>1.92</td>
</tr>
<tr>
<td></td>
<td></td>
<td>GL2 2nd</td>
<td></td>
<td>2012</td>
<td>2.92</td>
</tr>
<tr>
<td>Skerdagh catchment</td>
<td>Skerdagh</td>
<td>SK1 1st</td>
<td>Jan, 2009</td>
<td>2011</td>
<td>2.50</td>
</tr>
<tr>
<td></td>
<td></td>
<td>SK1 2nd</td>
<td></td>
<td>2012</td>
<td>3.50</td>
</tr>
</tbody>
</table>

To assess the effect of P and N release from decomposition of brash windrows on the underlying peat, peat samples were collected from below brash windrows and brash-free areas from six sites in July 2011 and July 2012 (Table 5.1). The sites were sampled either once in 2012 (Srahrevagh-2 and Glennamong-1) or during both 2011 and 2012. On each of the six sites, 3 plots, each with dimensions of 4 m x 4 m, in a brash-free area and 3 plots, of the same size, under brash windrows were established. The plots for brash-free areas were selected in locations where there was always a furrow separating a brash-free area from an area with brash windrows, thereby eliminating the risk of N or P from brash windrows entering the brash-free area. In each plot, 5 soil samples were collected to a depth of 0–10 cm using a 5.5 cm-diameter gouge auger, and the
samples were mixed to produce one composite sample for each of the three plots. The samples were hand-mixed and homogenized as moist, and were used to determine water extractable phosphorus (WEP) and KCl extractable ammonium-nitrogen (NH₄⁺-N) and nitrate-nitrogen (NO₃⁻-N) concentrations. For P determination, a sub-sample of moist soil (equivalent to 1 g of dry soil) was extracted with 30 mL of deionised water on a reciprocating shaker at 250 rpm for 30 min. For NH₄⁺-N and NO₃⁻-N concentrations, the extraction was with 30 ml of 2M KCl at 250 rpm for 60 min. The solutions were then filtered (0.45 µm) and stored at -18°C until further analyses.

5.2.2 Laboratory flume study

Peat soil samples for the laboratory flume study were collected within 1 month of forest clear-felling from the Srahrevagh-1 site (Table 5.1). Slabs of surface peat soil (0.1 m-deep) were collected using a 1-m-long, 0.225-m-wide and 0.1-m-deep rectangular flume sampler made of galvanized steel. The flume sampler was pushed upside down into the peat and then cut from the peat layer using a wide, flat spade. The spade was cleaned prior to taking every new peat sample. The peat inside the sampler was then transferred into 2-m-long, 0.225-m-wide and 0.13-m-deep flumes so that two slabs of peat were placed end-to-end and tight against each other to form a continuous peat surface (Mulqueen et al., 2006; Asam et al., 2012). Using the above method, a total of four flumes were prepared.

Brash windrows were created on two of the four flumes using Lodgepole pine tree harvest residue and the other two flumes were brash-free study controls. The brash windrows were prepared using 2.63 kg (5.84 kg m⁻²) dry mass of fresh harvest residue comprising 36% of needles, 34% of small diameter (<10 mm) branches and 30% of larger (15–40 mm) branches. This simulated the approximate harvest residue proportions of Lodgepole pine, and was based on three biomass sample trees in the Srahrevagh-1 site (unpublished data). Branches (small and large) were cut to less than 10 cm in length before constructing the brash windrows. Table 5.2 presents the initial mass and element concentrations of the harvest residue fractions. As brash windrows normally cover 20–25% of the area of clear-felled blanket peatlands (Rodgers et al., 2010), the brash windrows were created on the 50-cm upslope length of the flumes (covering 25% of the flume area; Fig. 5.2).
Table 5.2. Initial mass and elemental concentrations of different harvest residue fractions in brash windrows of the flume study.

<table>
<thead>
<tr>
<th>Cutting residue</th>
<th>Initial mass (kg m⁻²)</th>
<th>Initial elemental concentrations</th>
<th></th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td></td>
<td>Ca (%) (mg kg⁻¹)</td>
<td>N (%) (mg kg⁻¹)</td>
</tr>
<tr>
<td>Needles</td>
<td>2.08</td>
<td>51.8</td>
<td>1.42</td>
</tr>
<tr>
<td>Small diameter branches</td>
<td>1.98</td>
<td>51.5</td>
<td>0.47</td>
</tr>
<tr>
<td>Bigger diameter branches</td>
<td>1.78</td>
<td>52.2</td>
<td>0.27</td>
</tr>
</tbody>
</table>

* Measured with LECO CHN-1000 analyzer.
* Dry ashing + digestion in HCl and analysed using ICP/AES.

Fig. 5.2. Photo of laboratory flumes.

To study the N and P dynamics in runoff water when water flows from brash windrows through brash-free areas and into receiving streams, the flumes were irrigated with simulated rainfall on a weekly or bi-weekly basis. During the study period the flumes were stored in a glass house that was open to local weather conditions. Simulated rainfall was applied using a rainfall simulator consisting of a single 1/4HHSS14SQW nozzle (Spraying Systems Co., Wheaton, IL) attached to a 4.5-m-high metal frame, and calibrated to achieve an intensity of 30±1 mm h⁻¹ at 85%
uniformity (Regan et al., 2010; Asam et al., 2012). The source water for rainfall was potable tap water with an electrical conductivity of 0.435 dS m⁻¹, a pH of 7.7 and maximum orthophosphorus (PO₄-P), NH₄⁺-N and NO₃⁻-N concentrations of 0.005, 0.026 and 0.12 mg L⁻¹, respectively. The slope of the flumes was 5°, simulating the average hill slope gradient of the Srahrevagh catchment. In each irrigation event, simulated rainfall was first applied until the runoff started from the downslope overfall weir and irrigation then continued for one hour. The runoff water flowed from the flume into the runoff collection bucket (Fig. 5.2). At the end of each rainfall event, the runoff water was collected, weighed, and a subsample was filtered (0.45µm) and stored at -18°C until further analyses.

To study the P and N dynamics in soil water below brash windrows and at different distances downslope from the windrows, soil pore water samples were also collected. Pore water samples from the flumes were taken using suction samplers with a polymeric tip of 9 cm in length and 4.5 mm in diameter (Rhizon MOM 10 cm moisture filters), which was attached to a removable syringe capable of generating a suction of approximately 100 kPa. The filters were inserted at a 45° angle to a depth of about 5 cm from the soil surface and at 20, 40, 60, 100, 140 and 180-cm-distances from the upslope end of the flume. As the brash windrow on the flumes with brash covered the first 50 cm from the upslope end of the flume, the filters at 20 and 40-cm-distances were under brash windrows. Pore water samples were collected from 2 to 12 hours before each irrigation event and stored at -18°C until further analyses. At the end of the experiment, the cutting residue was collected from the two flumes, and separated into needles, small diameter branches, and larger diameter branches, and stored at room temperature until further analysis.

5.2.3 Field mini-catchment study

The mini-catchment study was conducted on the same Srahrevagh-1 site where the samples for the flume experiment were collected (Table 5.1). Six 7-m-wide and 20-m-long mini-catchments were hydrologically isolated using 120-cm × 60-cm corrugated plastic panels inserted in the peat so that 30 cm of each panel was below and 30 cm was above the soil surface. To insert the panels, a 30-cm-deep crack was first cut into the peat with a spade, and then the panels were inserted and attached to each other with aluminium H-shaped strips with a waterproof rubber sealer. Finally, the peat was pressed back and tightly against the panels. This depth of 30 cm was seen sufficient to isolate the catchments, as most of the water flow in blanket peat areas occurs as
surface flow and soil hydraulic conductivity decreases abruptly from the surface 0-10 cm-deep humus layer to deeper layers. 30-cm-deep trenches were also dug outside the mini-catchments to prevent any inflow of water from the upslope of the catchments. The waters from the isolated mini-catchments were diverted through PVC collection pipes into a runoff monitoring station, where water samples were collected for chemical analyses, and runoff was measured with a data logger and a tipping bucket.

The six experimental mini-catchments were constructed in January 2010 before clear-felling commenced in January 2011. The six mini-catchments were clear-felled by a harvester by removing all trees and transferring them as whole-trees so as no brash remained in the areas. Three of the mini-catchments were left as windrow-free controls and three were overlain with windrows. To create the mini-catchments with brash windrows, a 9-m-long brash mat, which was equivalent to brash from about 53 trees, was collected by the forwarder and placed into the upslope part of the mini-catchment, covering 20–25% of its area. Water samples were taken from March 2010 to August 2012 on a bi-weekly or monthly basis, the sampling rate depending on the occurrence of storms, during which most of the nutrient export may occur (Rodgers et al., 2010). After collection, the water samples were stored at -18°C until further analysis.

5.2.4 Laboratory analysis and calculations

The outflow and pore water samples from the flumes, as well as the soil water extracts, were analysed for dissolved (0.45 µm filter) reactive P (DRP), dissolved NH$_4^+$-N and NO$_3^-$-N using a Konelab 20 Analyser (Konelab Ltd., Finland). The outflow samples from the mini-catchments were analysed similarly, but for total (unfiltered) reactive P (TRP), total NH$_4^+$-N and NO$_3^-$-N.

The cutting residue needles, small diameter branches, and larger diameter branches, collected after the flume experiment, were dried to a constant mass at 60°C, weighed, milled to pass a 2 mm sieve, and analysed for N using a Leco CHN 1000 analyser and for P using dry digestion in HCl and ICP/AES.

In the flume study, the total loads (mg m$^{-2}$) of DRP, NH$_4^+$-N, and NO$_3^-$-N were calculated by multiplying the water volume in the bucket at the end of each irrigation event with the respective concentration, and then summing up the 25 events to calculate the total export load for the 15-month flume study period. The total loads of TRP, NH$_4^+$-N and NO$_3^-$-N from the mini-
catchments (kg ha\(^{-1}\)) were calculated by multiplying monthly runoffs with monthly concentrations and then summing up the monthly loads for 1.5 years of the mini-catchment study. The missing monthly concentration values were extrapolated from the closest available values.

The difference between the concentrations and release of nutrients from flumes with and without brash in flume study, and plots with and without brash in mini-catchment study, were analysed using paired samples \(t\)-test at the 95\% significance level \((p = 0.05)\) using SPSS statistical tool (http://www.spss.com).

5.3 Results

5.3.1 Field survey on P and N in peat under brash windrows

Water extractable P concentrations in the clear-felled sites were significantly higher under brash (6–63 mg kg\(^{-1}\)) compared to brash-free areas (2–15 mg kg\(^{-1}\)), except for the Srahrevagh-3 site, which was sampled about 6 years after harvesting. The differences in peat P between brash windrows and brash-free areas were particularly high for the samples collected one-to-two years after harvesting (Fig. 5.3).

Concentrations of extractable NH\(_4\)^+-N were higher under brash compared to brash-free areas on all sites except the Srahrevagh-1 site at the first sampling (SR1\(^{1st}\)) and the Shragrevagh-2 site (SR2). The differences between brash free and brash covered areas were not particularly high for the samples collected one-to-two years after harvesting as in the case of P, but were high for the samples collected two-to-three years after harvesting. Compared with P and NH\(_4\)^+-N, the concentrations of NO\(_3\)^-N were low in brash-free areas as well as under brash windrows (Fig. 5.3).
Fig. 5.3. Extractable DRP, NH$_4^+$-N and NO$_3^-$-N in peat in brash free area and under brash windrows over a time period from 0.5 to about 7 years after clear-felling. Each column is an average of 3 samples and bars indicate standard error of mean. SR1$^{1\text{st}}$ and SR1$^{2\text{nd}}$ indicate the first sampling in 2011 and the second sampling in 2012 for the Srahrevagh-1 (SR1) site, respectively. GL2$^{1\text{st}}$ and GL2$^{2\text{nd}}$, SK1$^{1\text{st}}$ and SK1$^{2\text{nd}}$ and SR3$^{1\text{st}}$ and SR3$^{2\text{nd}}$ are respective abbreviations for Glennamong-2 (GL2), Skerdagh (SK1) and Srahrevagh-3 (SR3). Glennamong-1 (GL1) site and Srahrevagh-2 (SR2) site were sampled once only in 2012.
5.3.2 Laboratory flume study

During the 15-month flume study, the flumes with brash released significantly ($p<0.05$) more DRP than the flumes without brash (Fig. 5.4). In the first three months of the flume experiment, there was no significant difference in DRP release between the flumes with and without brash, but following this, the DRP concentrations from the brash flumes started to increase, and peaked at about eight months after the start of the experiment (1.62 mg DRP L$^{-1}$), and again at the end of experiment. The peak concentration from the flumes without brash was 0.42 mg L$^{-1}$. The export of DRP from the flumes with and without brash over the study duration was 594±36 and 173±73 mg m$^{-2}$, respectively.

Ammonium-N release was also significantly ($p<0.05$) higher from the flumes with brash (0.04–1.56 mg L$^{-1}$) compared to the flumes without brash (0.04–1.16 mg L$^{-1}$). Concentrations of NO$_3^-$-N were in the range of 0–0.28 mg L$^{-1}$ from the flumes with brash and 0–0.48 mg L$^{-1}$ from the flumes without brash (Fig. 5.4). The export of NH$_4^+$-N from the flumes with and without brash was 642±111 and 564±66 mg m$^{-2}$, respectively, and for NO$_3^-$-N it was 107±6 and 169±18 mg m$^{-2}$.

The pore water had significantly ($p<0.05$) higher DRP concentrations from the flumes with brash compared with the brash-free flumes (Fig. 5.5). There was also higher pore water DRP concentrations under brash in the brash flumes compared to brash-free area. The concentrations of DRP increased under brash from about 2 mg DRP L$^{-1}$ during the first month of the study to 8 mg L$^{-1}$ 11–15 months after the start of the experiment. The pore water concentrations of NO$_3^-$-N and NH$_4^+$-N showed no systematic difference between the brash and non-brash flumes, and also no difference between brash covered and brash-free areas in the flumes (Fig. 5.5).
Fig 5.4. Concentrations of DRP, NH₄⁺-N and NO₃⁻-N released in runoff from the flumes with and without brash windrows. Values are an average of two flumes. Bars indicate standard error of mean.
**Fig. 5.5.** Changes in pore water concentration of P and N over time at different positions along the length of the flumes with and without brash windrows. Each column is an average of values for two flumes <1 months and between 1–5 months, 6–10 months, and 11–15 months after starting the experiment. Bars indicate standard error of mean.
Over the study duration, the average mass loss for needles, small diameter branches and larger diameter branches in the flumes with brash was 34%, 18%, and 9%, respectively (Table 5.3). Phosphorus release was faster from needles than branches (Table 5.3). The total P released from all brash fractions was 994 mg m⁻², out of which 84% was from harvest residue needles. The N released only from harvest residue needles and the branch fractions accumulated more N than the needles released (Table 5.3).

**Table 5.3.** Mass loss and nutrient release from decomposition of brash windrows during 15 months of the flume study. '-' sign indicates net accumulation. ± indicates the standard error of mean.

<table>
<thead>
<tr>
<th>Cutting residue</th>
<th>Mass loss (%)</th>
<th>P release (%)</th>
<th>N release (mg m⁻²)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Needles</td>
<td>34.4 ± 1.8</td>
<td>34.1 ± 1.0</td>
<td>830 ± 23.1</td>
</tr>
<tr>
<td>Small diameter branches</td>
<td>17.9 ± 0.3</td>
<td>13.6 ± 0.6</td>
<td>99.8 ± 4.0</td>
</tr>
<tr>
<td>Bigger diameter branches</td>
<td>9.1 ± 1.0</td>
<td>19.9 ± 3.7</td>
<td>63.8 ± 11.9</td>
</tr>
<tr>
<td>All fractions together</td>
<td></td>
<td>994 ± 31.0</td>
<td>-1459 ± 282</td>
</tr>
</tbody>
</table>

5.3.3 Mini-catchment scale study

The mean TRP, NH₃⁺-N and NO₃⁻-N concentrations from the mini-catchments before harvesting were 0.017, 0.024 and 0.088 mg L⁻¹, respectively (Fig. 5.6). After clear-felling, the TRP concentrations from the mini-catchments with brash increased to a peak concentration of 0.38 mg L⁻¹ after about nine months, and then decreased and increased again to 0.35 mg L⁻¹ after 18 months. The peak TRP concentration from the catchments without brash was about 0.15 mg L⁻¹. The total load of TRP from the mini-catchments with brash one-and-a-half years after harvesting was 3.57±0.6 kg ha⁻¹, and from the catchments without brash was 1.12±0.04 kg ha⁻¹.

Ammonium-N concentrations in out-flowing waters from the mini-catchments with brash were higher than from the catchments without brash in the first year after harvesting. The concentrations of NH₄⁺-N released during first year were in the range of 0.06–0.33 mg L⁻¹ from the plots with brash and 0.02–0.22 mg L⁻¹ from the plots without brash (Fig. 5.6). After about 1 year, the NH₄⁺-N concentrations started to decrease and there was no longer any difference between the mini-catchments with or without brash. The NO₃⁻-N concentrations in the out-flowing waters did not differ much between the brash and non-brash catchments (Fig. 5.6). The
total load of NH$_4^+$-N from the mini-catchments with brash 1.5 years after harvesting was 3.37±0.3 kg ha$^{-1}$, and for the catchments without brash, 1.83±0.23 kg ha$^{-1}$. The respective loads of NO$_3^-$-N from the mini-catchments with and without brash were 2.1±0.2 kg ha$^{-1}$ and 2.3±0.2 kg ha$^{-1}$.

Fig. 5.6. Mean monthly concentrations of TRP, NH$_4^+$-N and NO$_3^-$-N released in runoff from the mini-catchments with and without brash. Values are an average of three replicates and bars indicate standard error of mean. Dashed line shows the time of harvesting the plots and creating the brash windrows.
5.4 Discussion

This study supported the hypothesis that the brash windrows on harvested blanket peat catchments are a significant source of P to receiving water courses. The P release from brash windrows under field conditions was highest during the second year (Figs. 5.3 and 5.5), which may explain why the P exports from harvested peatlands generally peak during the second year (Nieminen, 2003; Rodgers et al., 2010). Recent studies from Finland (Kaila et al., 2012) suggest that whole-tree harvesting would be an efficient means of decreasing P export from soils with low P adsorption capacity, such as blanket peats (Asam et al., 2012). This present study supports these findings, as the DRP export from the flumes with brash was almost 3.4 times more than the export from non-brashed flumes. Similarly, the release of P from the mini-catchments with brash was 3.2 times more than from the non-brash controls.

The total P released from brash windrows in the flume study was 994 mg m\(^{-2}\) and the P exported from the brash flumes was 594 mg m\(^{-2}\), which suggested that the peat in the flumes had some adsorption capacity against leaching. According to a previous study conducted on a similar low-sorptive blanket peat site, the P sorption maxima for the peat in the flumes may be about 500 mg m\(^{-2}\) (Chapter 6).

Harvest residue is not considered to be a major source of N to receiving water courses during first few years after harvesting peatland forests (Fahey et al, 1988; Hyvönen et al., 2000; Ouro et al., 2001; Palviainen et al., 2004; Kaila et al., 2012), as the harvest residue acts as a sink rather than a source of N. This finding is supported by the present flume study, in which the branches in the brash windrows accumulated so much N that, despite of slight N release (6% of initial N) from needles, the brash windrows were a net sink for N.

Given that the brash windrows are a net sink for N during the early stages of decomposition, the higher discharge \(\text{NH}_4^+\)-N concentrations in the brash treatment both in the flume and the mini-catchment study during the first year may seem surprising. As the re-vegetation of blanket peats after harvesting takes several years (O’Driscoll et al., 2011), the limited vegetation establishment and N uptake under brash windrows compared with the non-brash controls is an unlikely explanation for the high N release. It is hypothesised that the higher N release is due to enhanced N leaching from the soil under brash. Indeed, many factors could contribute to higher N release.
from soil under brash windrows compared with brash-free areas. Although not providing N, the decomposition of brash can provide other nutrients and energy to heterotrophic decomposers, thereby enhancing soil N mineralisation under brash. A small variation in soil temperatures and moisture under brash windrows may also lead to increased microbial activity and soil N mineralization compared with brash-free areas (Fontaine et al., 2004, 2007). Higher NH$_4^+$-N concentrations in percolating soil water under piles of cutting residues than respective residue-free areas have also been reported by Rosén and Lundmark-Thelin (1987), Nieminen (1998) and Wall et al. (2008). The higher N$_2$O emissions from the piles of cutting residues compared with respective residue free areas also suggest that soil N mineralization is increased under cutting residue piles (Mäkiranta et al., 2012).

The extractable NO$_3^-$-N concentrations in soil (Fig. 5.3) and NO$_3^-$-N concentrations in outflowing waters from the flumes (Fig. 5.5) were considerably lower than the respective NH$_4^+$-N concentrations, whereas the mini-catchment study had similar NO$_3^-$-N and NH$_4^+$-N concentrations (Fig. 5.6). Along with the general nutrient-poor and acidic nature of blanket peats, a possible explanation for low NO$_3^-$-N concentrations in the flumes is that conditions during irrigation were saturated for almost all of the flume area except the upslope part, and therefore too anaerobic for nitrification to occur (Regina et al., 1996). Under thick brash windrows in the field, temperature and oxygen conditions may be unfavourable for nitrifying bacteria. On the other hand, when the waters from brash covered areas under field conditions enter brash-free areas, some nitrification may occur, as indicated by in the mini-catchment study. However, the possibility also exists that isolating the mini-catchments by preventing any water flow from upslope increases the depth of the aerobic zone in peat and thereby enhances nitrification compared with true conditions in blanket peat catchments. The possible contribution of this isolation-effect to soil oxygen conditions and nutrient cycling should be assessed before introducing the mini-catchment approach in blanket peat nutrient cycling research at a larger scale.

In both the flume and mini-catchment experiments, the brash was piled in the upslope part of the plots, thus enabling N and P sorption in the peat in the down-slope brash-free area. Under normal conditions in operational forestry in blanket peat catchments, any sorption in the brash-free area may be less than here, as the only larger brash-free area below brash windrows would be a 15–
20-m-wide riparian buffer zone along the recipient stream (Asam et al., 2012; Finnegan et al., 2012). Due to significant sorption of P in the down-slope brash free area (Fig. 5.5), the results of the present study are likely to underestimate the true export loads under field conditions, as well as the differences in export loads between brash covered and brash-free areas.

5.5 Conclusions

The results from all three experiments suggest that the release of P from brash windrows in blanket peat catchments is a significant source of P to receiving waters. The brash windrows may not release N during the early stages of their decomposition. Nevertheless, the decomposition of brash may enhance N leaching and export, probably by enhancing microbial activity and soil N mineralisation below brash. The removal of cutting residues and whole-tree harvesting could be an efficient means of decreasing N and P release and transport to water courses from blanket peat catchments. The excessive cutting residues also make the blanket peats attractive sources of cutting residues for bioenergy purposes. Given that cutting residues are to be collected and used increasingly for bioenergy purposes and active forestry is continued in blanket peats after clear-felling, the effect of brash removal on site nutrient capitals and long-term sustainability of forestry needs to be assessed.

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References


Chapter 6: Assessment of phosphorus retention efficiency of blanket peat buffer areas using a laboratory flume approach

This paper has been published in Ecological Engineering (2012; 49, 160–169). Zaki Asam designed the laboratory rainfall simulator experiment, and prepared all the laboratory flumes used in this study. He carried out majority of the relevant laboratory and field work, and analytical work on water, vegetation and peat soil samples. He also collected, analysed and synthesised the experimental data, and is the primary author of the paper. Mark O’Connor and Connie O’Driscoll helped with field work, and Sakari Sarkkola and Annu Kaila helped with the statistical analysis. The rest of the co-authors provided guidance and advice in writing the paper.
6. Assessment of phosphorus retention efficiency of blanket peat buffer areas using a laboratory flume approach

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Abstract

Large areas of upland blanket peat were afforested in Ireland and the UK before the importance of the riparian buffer areas was realized. These forests are now reaching harvestable age. In order to reduce the possible negative impact of harvesting activities on receiving water bodies, the creation of buffer areas along receiving water courses prior to the clear-felling the main plantation has been proposed. In this study, laboratory flumes were used to assess the dissolved reactive phosphorus (DRP) retention efficiency of the vegetated and non-vegetated blanket peatland buffer areas under different hydraulic loadings. In addition, a mini-buffer approach is proposed and tested with flumes. The idea with mini-buffers is to keep P on site and reduce P transportation from harvest residues. In the grassed flumes, DRP removals decreased from 80% to 40% and 30% to 18% when the hydraulic loadings increased from 49 to 123 mm hr\textsuperscript{-1} and 256 to 444 mm hr\textsuperscript{-1}, respectively. In the non-grassed flumes, the DRP removal was about the same as for the grassed flumes when the hydraulic loading was 49 mm hr\textsuperscript{-1}, but was significantly lower when the hydraulic loadings increased to 123, 256 and 444 mm hr\textsuperscript{-1}. The mini-buffer method could retain 88–96% (with seeded grasses) and 54–66% (without seeded grasses) of added DRP under different inflow DRP concentrations (between 0.5 and 3 mg DRP L\textsuperscript{-1}). After ending the overland flow experiment, the water extractable phosphorus (WEP) and pore water P concentrations in the non-grassed flumes were significantly higher than the grassed flumes, indicating a higher risk of subsequent P release from non-grassed buffer areas. The results indicated that the overall P retention capacity of the normal buffer areas in blanket peat catchments is low due to the frequent occurrence of high flow events that significantly reduce buffer P retention efficiency. In contrast, the mini-buffer method could be a feasible means of increasing onsite P retention. This study is of particular interest currently as large areas of upland blanket peat forests without buffer areas in the UK and Ireland are being harvested or are scheduled to be harvested in the near future.
Keywords: Blanket peat forest buffer area, Dissolved reactive phosphorus (DRP), Hydraulic loading, Laboratory Flumes, Simulated overland flow, Mini-buffer

6.1 Introduction

Eutrophication of inland surface waters has been identified as one of the most serious pollution problems in Europe (European Union, 2000; Toner et al., 2005). This deterioration in water quality is attributed to increases in the availability of nutrients, especially phosphorus (P) (McGarrigle et al., 1993; Foy et al., 1995; Champ, 1998; Lucey et al., 1999; Irvine et al., 2001). The Water Framework Directive (WFD) requires European Union (EU) Member States to achieve ‘good ecological status’ for all water bodies by 2015 (EU, 2000), and the reduction in nutrient releases to water bodies has become a priority of river basin management in Europe.

Approximately 800,000 hectares of peat catchments were afforested between the 1950s and 1990s in Ireland and the UK (Farrell, 1990; Hargreaves et al., 2003; EEA, 2004) and most of these forests are now approaching maturity. Hydrological losses of P from forested peat catchments are generally low, but several research studies in Scandinavia (Lundin, 1998; Ahtiainen and Huttunen, 1999; Nieminen, 2003), Ireland (Cummins and Farrell, 2003; Rodgers et al., 2010), and in the UK (Nisbet et al., 1997) have shown that these losses can increase when forests are harvested. In a study carried out on a blanket peat catchment in the west of Ireland, Rodgers et al. (2010) found that in the first 3 years after harvesting, 5.15 kg ha$^{-1}$ of total reactive phosphorus (TRP) was released to surface water, with the highest release of 2.3 kg ha$^{-1}$ occurring during the second year.

To prevent possible negative impacts of forest activities on the receiving watercourses, it is recommended to direct the outflow over a buffer area. In Ireland and the UK, many of the earlier afforested upland blanket peat catchments were established without any riparian buffer areas, with trees planted to the stream edge (Ryder et al., 2010). The creation of a riparian buffer area prior to clear-felling the main plantation in upland blanket peat catchments to mitigate the effect of harvesting on receiving waters has been proposed (Ryder et al., 2010). Studies from Scandinavia indicate that buffer areas can significantly reduce particle transport (Nieminen et al., 2005). However, the efficiency of buffer areas in reducing the P load seems to vary from
complete (100%) retention (Kubin et al., 2000; Väänänen et al., 2008) to partial P retention (Ihme, 1994; Sallantaus et al., 1998; Nieminen et al., 2005; Silvan et al., 2005), to even increased leaching of phosphate (Liljaniemi et al., 2003; Vasander et al., 2003).

The P retention capacity of a buffer area is strongly related to inflowing P concentration and the rate of hydraulic loading. High hydraulic loading significantly decreases buffer P retention efficiency (Väänänen et al., 2008), as it leads to the formation of continuous flow channels over the buffer area. In such channels, flow velocity is high and water residence time low; both these factors are disadvantageous for an efficient retention of P (Koskiaho et al., 2003). High hydraulic loadings result in water-logged conditions in the buffer areas, which also decrease P retention. This is because the reduction and redistribution of iron (Fe) leads to decreased soil P adsorption capacity (Jensen et al., 1999). Under low flow conditions, the contact time between buffer through-flow water and soil and vegetation of the buffer is longer and the vegetative uptake and soil sorption of nutrients are thereby greater than for high flow episodes (Väänänen et al., 2008; Vikman et al., 2010). Phosphorus retention is generally more efficient under high than low inflowing P concentrations. Negative P retention efficiencies have been reported for peatland buffer areas, especially when the inflowing P concentrations are not significantly higher than the background levels of unpolluted areas (Nieminen et al., 2005). Negative retention may also occur when high P loadings eventually lead to the saturation of the vegetative and soil P sinks of the buffer (Drizo et al., 2002). However, negative or low nutrient retention capacity due to saturation of nutrient sinks is unlikely to be as important a factor in forested blanket peat catchments as in agricultural areas (Dorioz et al., 2006) and in buffers used for waster water treatment (Nichols, 1983; Ronkanen and Kløve, 2009).

To date, few studies have focused on the design of blanket peat buffer areas and assessed their performance in the mitigation of P. This may be because the construction of a well-functioning buffer area in many blanket peat catchments is difficult. First, the peat in blanket peat buffer areas may not be a significant sink for P due to its low P retention capacity (Tamm et al., 1974). Secondly, blanket peats are often sloping towards the stream edge, have low water permeability (hydraulic conductivity) (Cummins and Farrell, 2003; Lewis et al., 2012), and are located in areas with precipitation of greater than 2000 mm year\(^{-1}\), which can result in frequent high hydraulic loadings to buffer areas. Thirdly, it has been reported that the recovery of natural
vegetation following buffer construction by harvesting the tree stand takes several years (Connaghan, 2007; O’Driscoll et al., 2011). Therefore, unless the buffer area is constructed a sufficiently long time before harvesting the main plantation, it may be inefficient in retaining P export that mainly occurs in the first 3 years after harvesting (Cummins and Farrell, 2003; Nieminen, 2003; Rodgers et al., 2010). In order to stimulate the establishment of the vegetation, the seeding of appropriate native grass species has been proposed as a solution. O’Driscoll et al. (2011) showed that *Holcus lanatus* and *Agrostis capillaris* was well established in blanket peat buffer areas two months after harvesting and had the potential to immobilize the P that would otherwise be available for leaching.

Because of the difficulties in establishing well-functioning buffer areas in many blanket peat catchments (Ryder et al., 2010) and the uncertainties over their performance in dissolved nutrient retention, other water purification methods need to be developed. In operational forestry in blanket peat catchments, the cutting residues are first used as brash mats and then later collected together to form brash windrows (width about 4 m) that run along the main slope in parallel rows (Fig. 6.1 a and b) at intervals of about 12 m (Rodgers et al., 2010). The brash mats are used to improve the soil bearing capacity against the heavy harvesting machinery and trees are planted in the areas between the brash windrows. One method to decrease the export of nutrients may be to create the brash mats/windrows, wherever possible, no longer along the main slope, but against it and then seed the strips between brash mats/windrows with native grasses. In this way, the seeded strips would act as mini-buffer areas (Fig. 6.1c), which could immobilize the P released from the brash mats/windrows.

Most of the previous studies on the performance of buffer areas in forestry were carried out under field conditions. It is difficult to consider many factors concurrently and to study the processes in detail. However, this is possible in laboratory-scale experiments. Laboratory flumes are commonly used to study the soil processes and nutrient transport under simulated rain and overland flow (Ghadiri et al., 2001; Mulqueen et al., 2006; Hussein et al., 2007; Regan et al., 2010; Brennan et al., 2011). It is possible to predict the efficiency of mitigation methods for P retention under different conditions using laboratory flumes and hence important factors for efficient P retention can be determined before studying the details of their functioning under field conditions.
Fig. 6.1. Photo of a clear-felled Srahrevagh site (a), schematic presentation of a "normal" clear-felled area and buffer area (b), and proposed mini buffer approach (c).

The aim of this study was to estimate the roles of hydraulic loading, inflow P concentration, and seeded vegetation on dissolved P retention in blanket peat buffer areas using laboratory flumes. It was hypothesized that grass-seeded buffer areas would have higher dissolved reactive phosphorus (DRP) retention than non-grassed areas. It was also hypothesized that the frequent high hydraulic loadings significantly decrease buffer DRP retention efficiency in blanket peat catchments and that the mini-buffers between brash windrows receiving low hydraulic loading would be efficient in P retention.
6.1 Materials and methods

6.1.1 Experimental design

The study was carried out as two separate flume experiments. In the first experiment (Experiment 1), a simulated overland flow study investigated the efficiency of grassed and non-grassed "normal" buffer areas to retain DRP under different rates of high hydraulic loadings, but constant inflow P concentration. In the second experiment (Experiment 2), a similar overland flow study investigated if the grass-seeded mini-buffers between brash mats/windrows would be efficient in retaining the P released from brash. For this test, the hydraulic loading was kept constant and lower than in Experiment 1, but the flumes were subjected to higher inflow P concentrations than Experiment 1.

6.1.2 Site description and soil characteristics

Soil samples for the laboratory flumes were collected from the Glennamong River catchment (53°58'N, -9°37'E, 69 m a.s.l.) and Srahrevagh River catchment (54°00'N, -9°32'E, 220 m a.s.l.), both of which are typical blanket peatland forests in the west of Ireland. The peat soil is highly fibric. Both sites were afforested with Lodgepole pine (*Pinus contorta*) in 1971. Peat depth at the sites is over 1 m and overlies mainly quartzite and schist bedrock. The sites have similar hydrological conditions and receive an average precipitation of approximately 2000 mm year⁻¹. A more detailed description of the catchments is contained in Rodgers et al. (2010) and O’Driscoll et al. (2011).

To obtain information on the soil characteristics which control P retention, 3 plots (25 m x 25 m) were established at both sites and soil samples from depths of 0–20 cm and 20–40 cm were collected using a 5.5 cm-diameter gouge auger and grid sampling technique with sampling positions at a distance of 10 m from each other. A second set of samples were collected from the same sampling positions, but at depths of 0–5 cm (humus layer) and 5–20 cm (peat layer). Samples from each sampling position and plot were mixed to produce one composite sample for each depth and site. The first set of samples from the 0–20 and 20–40 cm layers were dried at 60°C to constant mass, weighed for dry mass, milled to pass a 2 mm sieve and analysed with ICP/AES for aluminium (Al), Fe and P following dry digestion in HCl, and for N and carbon (C) using a Leco CHN 1000 analyser. Bulk density was calculated using the dry mass of the sample.
and the volume of gouge auger (5.5 cm diameter, 20 cm length). Organic matter (OM) content was determined by loss on ignition at 550°C (BSI, 1990). The second set of samples were hand-mixed and homogenized when moist, and were used to determine P sorption–desorption isotherms (Nieminen and Jarva, 1996; Section 6.2.6). After mixing, subsamples (equivalent to 1 g dry weight) were added to bottles containing solutions (40 mL) with 0.0, 0.1, 0.2, 0.5, 1.0, 2.0, 5.0, 10 mg L⁻¹ of P. The bottles were then shaken on a reciprocating shaker at 250 rpm for 1 hr and were left to equilibrate for 23 hr, after which the suspensions were shaken again for 5 min at 200 rpm. The suspensions were then filtered (0.45 µm) and stored at -18°C until further analysis of DRP remaining in the filtrate using a Konelab 20 Analyser (Konelab Ltd., Finland).

6.2.3 Flume preparation

At the Glennamong River catchment, an area of about 1 ha was considered as a potential buffer area. The area was clear-felled in August 2009 and the harvest residues (i.e. needles, twigs and branches) were removed. One part of the buffer area was seeded with *H. lanatus* and *A. capillaris* (50:50) in March 2010 at a total seeding rate of 36 kg ha⁻¹ and another was left as a non-grassed control. In July 2010, intact slabs of peat soil (0.1 m-deep) were collected for Experiment 1 from both the grassed and non-grassed buffer areas using a 1-m-long, 0.225-m-wide and 0.1-m-deep rectangular flume sampler made of galvanized steel. The flume sampler was pushed upside down into the peat and then cut off from the peat layer using a wide, flat spade. The spade was cleaned prior to taking every new sample. The peat inside the sampler was then transferred into a 2-m-long, 0.225-m-wide and 0.13-m-deep flume so that two slabs of peat (surface layer up) were placed end-to-end and tight against each other to form a continuous peat surface after Mulqueen et al. (2006) (Fig. 6.2). Using the above method, two flumes with grass and two without grass were prepared.

For Experiment 2, peat slabs were collected from the forest floor (between the furrows) from the forested Srahrevagh River catchment, in June 2010, using the same sampling technique as for the Experiment 1, and six flumes were prepared. Three flumes were seeded with *H. lanatus* and *A. capillaris* (50:50) using the same seeding rate (36 kg ha⁻¹) as used in the buffer area in the Glennamong catchment. The other three flumes were left as non-grassed controls. The seeded vegetation was left to develop in a glass house, which was open to local weather conditions, until
fully established in June 2011. For irrigation, the flumes were transferred to a laboratory every 1 or 2 weeks during the grass establishment period and simulated rainfall was applied.

![Figure 6.2](image.png)

**Fig. 6.2.** Schematic presentation of overland flow experiment using laboratory flumes.

The rainfall simulator consisted of a single 1/4HHSS14SQW nozzle (Spraying Systems Co., Wheaton, IL) attached to a 4.5-m-high metal frame, and calibrated to achieve an intensity of 30 ± 1 mm h⁻¹ at 85% uniformity (Regan et al., 2010). The source water was potable tap water with a maximum DRP concentration of 0.005 mg L⁻¹, electrical conductivity of 0.435 dS m⁻¹, and pH of 7.7. In each irrigation event, simulated rainfall was applied for 1 hr.

### 6.2.4 Overland flow experiments

**Experiment 1: simulation of DRP removal by normal buffers under high flow conditions**

The buffer area flumes from the Glennamong River catchment (Experiment 1) were subjected to four different high overland flow rates of 0.37 L min⁻¹ (Q1), 0.92 L min⁻¹ (Q2), 1.92 L min⁻¹ (Q3) and 3.33 L min⁻¹ (Q4) using a water pump (Fig. 6.2), giving hydraulic loadings of 49 mm hr⁻¹, 123 mm hr⁻¹, 256 mm hr⁻¹, and 444 mm hr⁻¹, respectively, and the equivalent DRP loadings were 31, 62, 121, and 198 mg m⁻². These hydrological loadings were chosen so that they would correspond to the loadings from a 10-ha catchment to a 0.1-ha buffer area. A buffer area size of 1% in relation to the area of upstream catchment has been recommended in a number of studies (e.g. Nieminen et al., 2005). According to a 1 year study in a blanket peat-dominated catchment in west of Ireland, about 48% of the hydraulic loadings from a 10-ha catchment area to a 0.1-ha buffer area would be between 0 and 49 mm hr⁻¹, 22% would fall between 50 and 123 mm hr⁻¹,
21% between 125 and 256, and 9% would be between 257 and 444 mm hr\(^{-1}\) (unpublished data). Thus, the upper limits of these four flow classes were applied in the present study, representing typical high flows in western Ireland.

The slope of the buffer area flumes was adjusted to a 1°-angle, corresponding to an average slope of blanket peat catchments near the stream edge. The P-rich water was pumped from the feeding tank to a small chamber at the head of the flume and then over a weir onto the surface of the peat (Fig. 6.2). At the lower end, the water flowed over the peat surface, over a second weir into effluent collection chamber and subsequently into a container. To avoid water ponding in the flume, the outflow weir was designed so as its edge was only a few millimetres above the peat surface. The concentration of DRP in the feeding tank was maintained at 425 µg L\(^{-1}\) by dissolving KH\(_2\)PO\(_4\) in the same source water as used for the Experiment 2 flume irrigations. This concentration was selected based on the findings of Rodgers et al. (2010), who found that TRP concentrations during storm events was in the range of 100 to 450 µg L\(^{-1}\) during the first two years after clear-felling in a blanket peat catchment, with more than 80% of the TRP being in dissolved form.

The four buffer area flumes (n=2 grassed; n=2 non-grassed) were first tested four times for Q1 and then twice for Q3, twice for Q2, and finally once for Q4. Each overland flow testing started by initially adding overland flow until the flume was fully saturated, and then the testing commenced and continued for 1 hr. At the end of each flow event, the runoff volume was measured from the container, a runoff sample was collected and filtered (0.45µm) and then stored at -18°C until further analysis. There was a gap of 1 week between the different flow tests and, during this time, the flumes were stored in a glasshouse.

At the end of experiment (11 weeks), all the grass was harvested from the flumes and stored for further analysis. Soil samples were collected at 15 cm-intervals from the inlet towards the outlet of the flumes, and were combined to get one composite sample for each of the four flumes. These samples were then hand mixed until visually homogenized. The samples were tested for water extractable P (WEP) by shaking a sub-sample of moist soil (equivalent to 1 g of dry soil) with 30 mL of deionized water on a reciprocating shaker at 250 rpm for 30 min; then the solution was filtered (0.45 µm) and stored at -18°C until further analyses. Prior to the experiment, parallel soil samples were collected next to the flume sampling spots and also analysed for WEP.
Experiment 2: simulation of DRP removal by mini-buffers under low flow conditions

Experiment 2 overland flow testing was similar to Experiment 1, except that the hydraulic loading rate was kept lower and constant (30 mm h⁻¹). Four different concentrations of DRP were applied to both the seeded (n=3) and un-seeded (n=3) flumes, simulating the different rates of P release from brash mats into mini-buffers during different stages of brash decomposition. The slope of the flumes was set at 5°, simulating average hill slope gradient for the proposed blanket peat mini-buffers. The added DRP concentrations were: 0.5 (C1), 1.0 (C2), 2.0 (C3), and 3.0 (C4) mg L⁻¹ and equivalent DRP loadings were 15, 30, 60, and 90 mg m⁻², respectively. Each DRP concentration was added twice, and the order of additions was as follows: C1, C2, C3, C4, C4, C3, C2, and C1.

In addition to runoff samples, pore water samples were also collected from the flumes of Experiment 2. They were collected using suction samplers with a polymeric tip, 9 cm in length and 4.5 mm in diameter (Rhizon MOM 10 cm moisture filters) attached to a removable syringe, capable of generating a suction of approximately 100 kPa. The filters were inserted at a distance of 50, 150 and 175 cm from the inlet of the flume, and pore water samples were collected within 1 hr after each overland flow event. A second sample was taken 1 week after each overland flow event in order to study if the P in pore water was effectively taken up by the vegetation. At the end of overland flow testing, the vegetation in the flumes was harvested, dried, and stored as in Experiment 1. There was also some vegetation growth (mostly mosses) in the non-seeded flumes, and that vegetation was similarly harvested and stored.

6.2.5 Analysis of water and vegetation samples

The flume outflow samples, the pore water samples from Experiment 2, as well as the soil water extracts from Experiment 1, were analysed for DRP using the molybdenum blue method and a Konelab 20 Analyser (Konelab Ltd., Finland). The harvested vegetation was dried at 60°C until reaching a constant mass, weighed, milled to pass a 2-mm sieve, and then analysed for P using dry digestion in HCl and ICP/AES.
6.2.6 Calculations and statistical analysis

To characterize the peat P retention properties in the Glennamong and Srahrevagh catchments, a modified Langmuir equation (Eq. 6.1) (Hartikainen and Simojoki, 1997) and Freundlich equation (Eq. 6.2) (Fitter and Sutton, 1975) were fitted to the P sorption isotherm data. The threshold P concentration (EPC0) above which net sorption occurs was determined graphically.

\[ q = \frac{P_{max}KC}{1-KC} - q_{0L} \]  
\[ q = kc^n - q_{0F} \]

Equation 6.1  
Equation 6.2

In Eq. 6.1 and 6.2, \( q \) is P sorbed (mg g\(^{-1}\)), \( c \) is P in equilibrium solution (mg L\(^{-1}\)), \( P_{max} \) is a constant describing sorption maximum (mg g\(^{-1}\)), \( q_{0L} \) and \( q_{0F} \) are instantly labile P (mg g\(^{-1}\)), and \( K \), \( k \), \( n \) are constants.

The results from each overland flow testing are expressed as mg DRP m\(^{-2}\) and % DRP retained from the added DRP. Before starting the overland flow tests in Experiment 1, the flumes were subjected to 1-hr irrigations as explained for the vegetation establishment phase of Experiment 2 flumes (Section 6.2.3). The DRP released from the flumes into the collection containers during these priming irrigations was subtracted from the outflow P concentration of each overland flow testing to account for the background release that was not caused by the DRP addition.

The effects of vegetation (grassed vs. non-grassed), water flow rate, and DRP concentration on the total (DRP-ret\_tot; mg m\(^{-2}\)) and relative DRP (DRP-ret\%; % of the added DRP) retention was analysed by applying the mixed model regression method. This accounts for the possible autocorrelation, which may be involved in a time series of repeated observations (e.g. Goldstein, 1995; Searle, 1997). In the present study, three hierarchical levels were identified. For Experiment 1, the levels were flume, flow event (Q1-Q4) and DRP measurement occasion, and for Experiment 2: flume, input DRP concentration, and DRP measurement occasion. The mixed model had the following general form:

\[ y_{ijk} = \alpha_{ij} + \beta_1x_{1ijk} + \beta_2x_{2ijk} + ... + \beta_nx_{nijk} + v_k + u_{jk} + e_{ijk} \]  
Equation 6.3

where \( y_{ijk} \) denotes the value of the response variable (total DRP retention or relative DRP retention) for measurement \( i \) in flow event \( j \) in flume \( k \); \( \alpha \) denotes the mean intercept, \( x_{1ijk}...x_{nijk} \)
are the explanatory variables, \( v_k \) is the flume-level residual, \( u_{jk} \) is the flow event-level residual (Experiment 1) or input DRP concentration-level residual (Experiment 2), and \( e_{ijk} \) is the DRP measurement point-level residual. The underlying assumption is that the random variables \( v_k \), \( u_{jk} \) and \( e_{ijk} \) are uncorrelated and follow normal distributions with zero means, so that it is sufficient to estimate their variances, \( \sigma_k^2 \), \( \sigma_{jk}^2 \) and \( \sigma_{ijk}^2 \). These variances are the random parameters of the model, whereas \( \alpha \) and \( \beta \) are fixed parameters.

The estimation was done by using MLwiN software (Rasbash et al., 2001), which estimates the fixed and random parameters simultaneously. The models were constructed by adding the explanatory variables one by one and they were estimated with the restricted iterative generalized least-square (RIGLS) method recommended for small samples. The value of -2×log-likelihood was used to compare the overall goodness-of-fit of the models of increasing number of the explanatory variables, and the significance of the differences between the likelihood-values was tested by \( \chi^2 \)-test. Parameter standard error (s.e.) was used as a measure of parameter significance (value should be at least twice its s.e.). If necessary, appropriate transformations were applied to linearize the relationships between the dependent and explanatory variables, and to homogenize the variance.

To evaluate the model reliability and accuracy, systematic error \((B)\), and relative systematic error \((Br)\) were calculated:

\[
B = \frac{1}{m} \sum_{j=1}^{m} \left[ \frac{1}{n_j} \sum_{i=1}^{n_j} (y_{ij} - \hat{y}_{ij}) \right] \quad \text{Equation 6.4}
\]

\[
Br = \frac{1}{m} \sum_{j=1}^{m} \left[ \frac{1}{n_j} \sum_{i=1}^{n_j} \left( \frac{y_{ij} - \hat{y}_{ij}}{y_{ij}} \right) \right] \quad \text{Equation 6.5}
\]

where \( n \) is the number of observations in flume \( m \), \( y_{ij} \) the observed value of total or relative DRP retention, and \( \hat{y}_{ij} \) the predicted value of total or relative DRP retention.
6.3 Results

6.3.1 Peat properties in Glennamong and Srahrevagh

The OM content in the surface and subsurface peat of both Glennamong and Srahrevagh sites was over 91%. There was no significant difference in OM, C and N content in the surface (0–20 cm) peat between the two sites (Table 6.1). The P content was higher in Srahrevagh (0.47 g kg\(^{-1}\)) compared to Glennamong (0.35 g kg\(^{-1}\)), whereas Al and Fe contents in Glennamong were almost 2-fold higher compared with the contents in Srahrevagh.

Table 6.1. Soil properties in 0–20 and 20–40 cm peat layers in Glennamong and Srahrevagh.

<table>
<thead>
<tr>
<th>Study sites</th>
<th>Depth of sample cm</th>
<th>Bulk density g cm(^{-3})</th>
<th>Organic matter %</th>
<th>Carbon %</th>
<th>Nitrogen %</th>
<th>P g kg(^{-1})</th>
<th>Fe g kg(^{-1})</th>
<th>Al g kg(^{-1})</th>
</tr>
</thead>
<tbody>
<tr>
<td>Glennamong</td>
<td>0–20</td>
<td>0.10</td>
<td>95.7</td>
<td>53.1</td>
<td>2.3</td>
<td>0.35</td>
<td>1.31</td>
<td>1.34</td>
</tr>
<tr>
<td></td>
<td>20–40</td>
<td>0.15</td>
<td>91.7</td>
<td>52.9</td>
<td>1.8</td>
<td>0.19</td>
<td>0.94</td>
<td>1.67</td>
</tr>
<tr>
<td>Srahrevagh</td>
<td>0–20</td>
<td>0.10</td>
<td>97.0</td>
<td>53.8</td>
<td>2.3</td>
<td>0.47</td>
<td>0.74</td>
<td>0.77</td>
</tr>
<tr>
<td></td>
<td>20–40</td>
<td>0.10</td>
<td>98.5</td>
<td>56.3</td>
<td>2.5</td>
<td>0.27</td>
<td>0.26</td>
<td>0.66</td>
</tr>
</tbody>
</table>

The P adsorption increased with increasing P concentration in equilibrium solution in Glennamong, especially in the humus layer, but a clearly defined maximum adsorption could not be detected at the applied P concentration range (Fig. 6.3).

Fig. 6.3. Phosphorus sorption-desorption isotherms of humus and peat layer at the Glennamong and Srahrevagh sites.
In the Srahrevagh site, however, the sorption isotherms indicated that the peat and humus layers were about to reach their maximum P sorption capacity. Parameter values of the fitted Langmuir and Freundlich equation are presented in Table 6.2. In the humus layer, the threshold concentration (EPC₀) above which net sorption occurs was 28 µg L⁻¹ and 17 µg L⁻¹ for the Glennamong and Srahrevagh sites, respectively, and EPC₀ for the peat layer was 25 µg L⁻¹ and 13 µg L⁻¹ (Table 6.2). The maximum P (P_max) retention capacity of the humus layer at both sites was higher than that of the peat layer.

**Table 6.2.** Equilibrium phosphorus concentration (EPC₀) and parameter values of the fitted Langmuir (q₀L, P_max and K) and Freundlich equations (q₀F, k and n) for the humus and peat layers in Glennamong and Srahrevagh.

<table>
<thead>
<tr>
<th>Study site</th>
<th>Sample layer</th>
<th>EPC₀</th>
<th>q₀L</th>
<th>P_max</th>
<th>K</th>
<th>R²</th>
<th>q₀F</th>
<th>k</th>
<th>n</th>
<th>R²</th>
</tr>
</thead>
<tbody>
<tr>
<td>Glennamong</td>
<td>Humus</td>
<td>28</td>
<td>0.002</td>
<td>0.178</td>
<td>0.233</td>
<td>0.998</td>
<td>0.007</td>
<td>0.040</td>
<td>0.525</td>
<td>0.981</td>
</tr>
<tr>
<td></td>
<td>Peat</td>
<td>25</td>
<td>-0.001</td>
<td>0.077</td>
<td>0.080</td>
<td>0.998</td>
<td>0.001</td>
<td>0.007</td>
<td>0.684</td>
<td>0.999</td>
</tr>
<tr>
<td>Srahrevagh</td>
<td>Humus</td>
<td>17</td>
<td>0.001</td>
<td>0.079</td>
<td>0.919</td>
<td>0.998</td>
<td>0.005</td>
<td>0.037</td>
<td>0.337</td>
<td>0.957</td>
</tr>
<tr>
<td></td>
<td>Peat</td>
<td>13</td>
<td>-0.002</td>
<td>0.023</td>
<td>0.398</td>
<td>0.946</td>
<td>0.001</td>
<td>0.010</td>
<td>0.338</td>
<td>0.979</td>
</tr>
</tbody>
</table>

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6.3.2 Overland flow tests

Experiment 1: Simulation of DRP removal by normal buffers under high flow conditions

The constructed regression models having the best performance for the total and relative DRP retention by the buffer area flumes were:

\[
\text{DRP-rettot} = 7.778 + 4.811 \times \text{GRASSED}, \quad \text{Equation 6.6}
\]

\[
B = 41.0, \ B_r = 0.69
\]

\[
\text{DRP-ret}\% = -3.221 + 42.54 \times \text{FLOW} \times \text{GRASSED}^{-0.6964} + 27.707 \times \text{FLOW} \times \text{NON-GRASSED}^{1.0534}, \quad \text{Equation 6.7}
\]

\[
B = 0.36, \ B_r = -0.046
\]

where FLOW is the water flow rate to the flume (L min\(^{-1}\)) and GRASSED and NON-GRASSED are dummy variables (1/0). The random effects of flow event \((u_{ijk})\) and DRP retention measurements \((e_{ijk})\) were significant, whereas the random effect of flumes \((v_k)\) was not (not shown). According to the models, the differences in the total and relative DRP retentions between the grassed and non-grassed buffer area flumes were significant (Equations 6.6 and 6.7). Furthermore, the relative DRP retention was significantly negatively related to the water flow rate (Equation 6.7, Fig. 6.4). For the total DRP retention, only the GRASSED dummy was significant, and the accuracy of the model was poor.

In the grassed flumes, the DRP removal efficiencies gradually decreased from 80% to 40% and 30% to 18% when the hydraulic loadings increased from 49 mm hr\(^{-1}\) to 123 and 256 to 444 mm hr\(^{-1}\), respectively. In the non-grassed flumes, the DRP removal was about the same as for the grassed flumes when the hydraulic loading was 49 mm hr\(^{-1}\). However, when the hydraulic loadings increased to 123, 256 and 444 mm hr\(^{-1}\), significantly less P was retained compared to the grassed flumes (Table 6.3).
Fig. 6.4. (A) The relative retention of DRP (%) by the grassed and non-grassed flumes in relation to the water flow rate simulated by the regression model (Eq. 6.7). (B) The total retention of DRP (mg m\(^{-2}\)) by the grassed and non-grassed flumes in relation to the input DRP concentration simulated by the regression model (Eq. 6.8).
Over the study duration (11 weeks), total DRP retained by grassed and non-grassed flumes was 258 mg m\(^{-2}\) and 179 mg m\(^{-2}\), respectively. Total P in the above-ground grass biomass of the grassed flumes was 386 mg m\(^{-2}\) at the end of experiment. The WEP in peat before applying the P-rich overland flow was 0.8 mg (kg dry soil\(^{-1}\)) for grassed and 2 mg (kg dry soil\(^{-1}\)) for non-grassed flumes. At the end of the overland flow experiments, WEP was 1.1 mg (kg dry soil\(^{-1}\)) and 9.6 mg (kg dry soil\(^{-1}\)) from grassed and non-grassed flumes, respectively.

Table 6.3. DRP retention by grassed and non-grassed flumes in Experiment 1. Values are average of two flumes and all the flow events for each hydraulic loading. ± is the standard error of mean.

<table>
<thead>
<tr>
<th>Flow event</th>
<th>Hydraulic loading (mm hr(^{-1}))</th>
<th>Number of flows applied</th>
<th>Average DRP loading (mg m(^{-2}))</th>
<th>Grassed flumes</th>
<th>Non-grassed flumes</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td></td>
<td></td>
<td>Average DRP removal (%)</td>
<td>DRP retained (mg m(^{-2}))</td>
<td>Average DRP removal (%)</td>
</tr>
<tr>
<td>Q1</td>
<td>49</td>
<td>4</td>
<td>31</td>
<td>80 ± 1</td>
<td>25 ± 1</td>
</tr>
<tr>
<td>Q2</td>
<td>123</td>
<td>2</td>
<td>62</td>
<td>40 ± 3</td>
<td>25 ± 4</td>
</tr>
<tr>
<td>Q3</td>
<td>256</td>
<td>2</td>
<td>121</td>
<td>30 ± 2</td>
<td>36 ± 1</td>
</tr>
<tr>
<td>Q4</td>
<td>444</td>
<td>1</td>
<td>198</td>
<td>18 ± 1</td>
<td>36 ± 1</td>
</tr>
</tbody>
</table>

Experiment 2: Simulation of DRP removal by mini-buffers under low flow conditions

The regression models having the best performance for the total and relative DRP removal by mini-buffer flumes were:

\[
\text{DRP-rettot} = -7.925 + 14.447 \text{ (GRASSED)} + 22.876 \text{ (DRP_input)} \quad \text{Equation 6.8}
\]

\[
B = 0.001, B_r = -0.117
\]

\[
\text{DRP-ret\%} = 58.891 + 32.346 \text{ (GRASSED)} \quad \text{Equation 6.9}
\]

\[
B = -0.003, B_r = -0.181
\]
where GRASSED is dummy variable (1/0) and DRP_input is the concentration of the input DRP in each flow-event (mg L$^{-1}$). As in the models for Experiment 1, the differences in the total and relative DRP retentions between the grassed and non-grassed mini-buffer flumes were significant (Equations 6.8 and 6.9). The total DRP retention was also significantly related to the input DRP concentration (Equation 6.9). For the non-grassed mini-buffer flumes, the DRP retention was at a lower level, and the variation between flumes was larger than for the grassed flumes. However, the relationship between input DRP concentration and total DRP retention followed a similar positive linear pattern for both the grassed and non-grassed flumes (Fig. 6.4).

The grassed flumes reduced DRP input by 88–96%, depending on DRP inflow concentration (Table 6.4). The retention in non-grassed flumes was 54–66%. Average retention of DRP by grassed and non-grassed mini-buffer flumes over all overland flow tests was 90% and 60%, respectively.

**Table 6.4.** DRP retention by grassed and non-grassed flumes in Experiment 2. Values are average of three flumes and two flows for each inlet DRP concentration. ± is the standard error of mean.

<table>
<thead>
<tr>
<th>Flow events</th>
<th>Input DRP conc. (mg L$^{-1}$)</th>
<th>Number of flows applied</th>
<th>Average DRP loading (mg m$^{-2}$)</th>
<th>Grassed flumes</th>
<th>Non-grassed flumes</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td></td>
<td></td>
<td></td>
<td>DRP removal (%)</td>
<td>DRP retained (mg m$^{-2}$)</td>
</tr>
<tr>
<td>C1</td>
<td>0.5</td>
<td>2</td>
<td>15.0</td>
<td>96 ± 1</td>
<td>14 ± 0</td>
</tr>
<tr>
<td>C2</td>
<td>1.0</td>
<td>2</td>
<td>30.0</td>
<td>92 ± 2</td>
<td>28 ± 1</td>
</tr>
<tr>
<td>C3</td>
<td>2.0</td>
<td>2</td>
<td>60.0</td>
<td>89 ± 2</td>
<td>54 ± 1</td>
</tr>
<tr>
<td>C4</td>
<td>3.0</td>
<td>2</td>
<td>90.0</td>
<td>88 ± 2</td>
<td>79 ± 1</td>
</tr>
</tbody>
</table>

The pore water DRP concentrations were high in the samples taken within an hour after each overland flow testing for both the grassed (0.1–1.1 mg L$^{-1}$) and non-grassed (0.9–2.1 mg L$^{-1}$) flumes. After 1 week, the pore water DRP concentrations were still high (0.9–1.7 mg L$^{-1}$) in the non-grassed flumes, but had decreased to 0.01–0.05 mg L$^{-1}$ in the grassed flumes, indicating effective P uptake by vegetation (Fig. 6.5). The total DRP retained by grassed and non-grassed
flumes during Experiment 2 were 350 mg m\(^{-2}\) and 234 mg m\(^{-2}\), respectively. Total P in the above-ground vegetation biomass was 890 mg m\(^{-2}\) for grassed and 272 mg m\(^{-2}\) for non-grassed flumes.

![Graph showing DRP concentrations in grassed and non-grassed flumes](image)

**Fig. 6.5.** Pore water DRP concentrations within 1 hour and 1 week after application of DRP rich overland flow. Values are averages of three flumes and three sampling positions per flume. Bars indicate the standard error of mean.

### 6.4 Discussion

The sloping topography of blanket peats, high precipitation rates and low hydraulic conductivity of upland blanket peat can result in frequent high flow events in the west of Ireland. The results from Experiment 1 supported the hypothesis that "normal" buffer areas along streams receiving frequent high hydraulic loadings may not be efficient for P retention in blanket peat catchments. Rodgers et al. (2010) found about 70% of DRP export from harvested blanket peat catchments occurred during storm events, corresponding to overland flow testings of Q2-Q4 in the present study. Less than 28% and 14% of the added DRP was retained by grassed and non-grassed flumes, respectively, during these high hydraulic loadings. The low P retention was due to low water residence time (Koskiaho et al., 2003), and also because the P loading was high during high hydraulic loadings. Based on Experiment 1 only, it is not possible to assess which factor was more important for low retention efficiency: high hydraulic loading or high P loading. In Experiment 2, where the hydraulic loading was low (30 mm hr\(^{-1}\)), P retention efficiency was higher than for Experiment 1, even though the P loading was higher. Several other studies also showed efficient P retention during high P loading and low hydraulic loading (Väänänen et al., 2008). However, in the
case that the buffer area eventually becomes saturated with P due to high P loading, its P retention capacity may decrease significantly, as shown for the buffers used for waste water treatment (Drizo et al., 2002).

The results from Experiment 2 indicated that constructing mini-buffers between brash mats could be an efficient means for P retention due to significantly lower hydraulic loadings than "normal" buffer areas along streams. This would be the case even if the mini-buffers were not seeded with grasses, as the non-grassed flumes of Experiment 2 also retained P efficiently. However, the results from both experiments indicated that the retention by non-grassed buffers may only be temporary. In Experiment 1, the soil WEP concentrations increased from 2 mg kg\(^{-1}\) to 9.6 mg kg\(^{-1}\) in the non-grassed flumes, giving an increase in WEP of about 140 mg m\(^{-2}\) (78% of DRP retained during 11 weeks of overland flow testing). During the same time, the soil WEP concentrations in the grassed flumes increased from 0.8 mg kg\(^{-1}\) to 1.1 mg kg\(^{-1}\), giving an increase in soil WEP of about 17 mg m\(^{-2}\). These results indicate that most of the DRP removed by the non-grassed flumes was retained as WEP in soil. Soil WEP has a strong relationship with P concentration in runoff (Schindler et al., 2009) and has been proven to be a useful indicator of soluble P concentration in runoff water (Daly and Styles, 2005; Regan et al., 2010). The high increase in soil WEP concentration in non-grassed flumes therefore indicates a high risk for subsequent P release from non-vegetated buffer areas after the end of P loading. In Experiment 2, much of the added P in non-grassed flumes appeared to remain as DRP in pore water (Fig. 6.5), also susceptible to subsequent P leaching. Gradual saturation of the pore water with DRP was probably the reason why the non-grassed flumes of Experiment 2 appeared to lose much of their retention capacity during the course of the overland flow testing. This was suggested to be the case as the second overland flow testings for different concentration levels indicated much lower DRP retention (18–59% lower, data not shown) than the first testings. Similar decrease in retention efficiency was not seen for the grassed flumes.

In both experiments, the total P in above-ground grass biomass was more than the DRP retained by the grassed flumes according to water samples; a probable reason for this is that the grass was seeded before starting the overland flow tests and grass had been accumulating P from the peat prior to the tests. The flume experiments indicated that seeded vegetation may be an important sink for the P released from harvested blanket peat catchments. If P is not released from vegetation due
to wilting and decay at the end of a growing season, vegetation may act as a long-term P sink. Previous studies have also indicated that peatland vegetation can retain P effectively from through-flow water in peatland buffer areas (Silvan et al., 2004; O’Driscoll et al., 2011). In China and Australia, vetiver grass in buffer zones and wetlands has shown a high potential for removing P from wastewater and polluted water (Wagner et al., 2003). Vegetation has an important role in P retention, not only through P accumulation in the living biomass (Silvan et al., 2004), but also because the vegetation slows down the water flow in the buffer areas, reduces the formation of preferential flow paths, and increases the contact time between soil and water (Braskerud, 2001). Grasses also make the soil more porous, thus diverting more water to subsurface flow, where P can be retained by chemical adsorption or end up in the rhizosphere of peatland vegetation. In the short-term, as in these flume experiments, the effect of vegetation may actually be due to the vegetation’s ability to increase the temporary retention of labile P in soil. Subsequent to the end of a P-rich overland flow event, this labile P can be taken up by vegetation, as indicated by the pore water samples (Fig. 6.5) in Experiment 2.

It may at first seem surprising that in Experiment 1, grass did not affect DRP retention at the lowest hydraulic loading (Q1 in Table 6.3), whereas with a very similar hydraulic loading and DRP inflow concentration, the grassed flumes in Experiment 2 (C1 in Table 6.4) had significantly better DRP retention compared with the non-grassed flumes. A probable reason is that all the four overland flow tests for Q1 were repeated first before testing the effects of any higher hydraulic loadings. It may therefore be that the Glennamong soil was not yet fully saturated with P and the soil adsorption sites were a sufficient sink for added P. The total amount of P added during the four Q1 overland flow tests was 126 mg m$^{-2}$. Fitting the Langmuir equation to the sorption data in Fig. 6.3 indicated that the soil P adsorption capacity was higher (about 135 mg m$^{-2}$).

A major disadvantage in utilizing the P accumulation capacity of vegetation is that the recovery of blanket peat vegetation following forest harvesting usually takes several years (Connaghan, 2007; O’Driscoll et al., 2011). However, O’Driscoll et al. (2011) showed that \textit{H. lanatus} and \textit{A. capillaris} vegetation could be well established in blanket peat forest areas 2 months after harvesting, and it had the potential to immobilize P that would otherwise be available for leaching.
6.5 Conclusions

Based on the results of the present study, seeding "normal" buffer areas with grasses may have limited effect on P retention, as the overall retention capacity is low for both grassed and non-grassed buffer areas due to frequent high hydraulic loadings. In contrast, seeding mini-buffers between brash windrows aligned along the contour might be a feasible means of increasing onsite P retention. If the P taken up by vegetation ends up as an organic structural component of plants and eventually accumulates as one of the many peat components, sustainable P retention is possible. However, the sustainability of P retention needs to be assessed in long-term field studies. Changing the orientation of windrows also requires changes in present harvesting routines, and the machinery stability and safety has to be assessed before introducing the mini-buffer method in the large-scale in operational forestry.

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Chapter 7

Conclusions and Recommendations

Overview

The forest industry is continuously examining the cost effectiveness of forestry on peatlands in terms of timber quality, harvesting and road building costs, along with the requirement of management practices that have to be put in place to safeguard environmentally sensitive, ecologically valuable, oligotrophic, salmonid rivers and lakes in upland blanket peat catchments. Nutrient releases from forestry activities are seen as one of the major threats to these pollution-sensitive water courses. An understanding of the nutrient cycling processes and the effect of different mitigation measures is needed to help in the development of management and harvesting guidelines for environmentally sustainable forestry in blanket peat catchments.

This study examined the mechanisms of the release and transport of phosphorus and nitrogen from harvested peatland forests with main focus on blanket peatlands. This study also investigated the effectiveness of different mitigation methods to decrease nutrient export to water courses from harvested blanket peat forests.

7.1 Conclusions

The main conclusions from this study are:

1. Results of the harvest residues decomposition study carried out on peatland forests in Finland and Ireland suggests that phosphorus (P) is easily released from decaying harvest residue needles, whereas nitrogen (N) is mostly retained or accumulated during the initial stages of decomposition. Therefore, release of N from harvest residue needles is not a likely source of the increased N export that has been observed to occur from peat soil soon after clear-cutting, but P release from harvest residues may be a cause for reported high P losses.
2. High water tables, which cause anoxic conditions in upper peat layers, resulted in P mobilization from moderately decomposed nutrient-poor blanket peat and highly decomposed nutrient-rich blanket peat from sites in Ireland. In peats from Finland, P mobilization occurred only from nutrient-poor, ombrotrophic peats, and not from nutrient-rich, minerotrophic peats. A high water table also caused higher mobilization of ammonium-N (NH$_4^+$-N) from nutrient-rich peats compared to nutrient-poor peats. These results suggest that water-logged conditions may cause different pattern of nutrient release from different types of peats depending on their biogeochemical characteristics.

3. The release of P from brash windrows in blanket peat catchments is a significant source of P to receiving waters. The brash windrows may not release N during the early stages of decomposition. Nevertheless, the decomposition of brash may enhance N leaching and export, probably by enhancing microbial activity and soil N mineralization below brash. The removal of harvest residues and whole-tree harvesting could be an efficient means of decreasing N and P release and transport to water courses from blanket peat catchments.

4. Conventionally-designed buffer areas, with and without seeded grasses, may have limited effect on P retention from through-flow waters in blanket peat catchments, as frequent high flow episodes, typical in blanket peat areas, are disadvantageous for effective P retention. Alternatively, a mini-buffer method, comprising brash mats and windrows positioned perpendicular to the slope, as opposed to along the main slope, and with the strips between the brash windrows seeded with native grasses, is proposed. In contrast to conventional buffer areas, a grassed or non-grassed mini-buffer method might be a feasible means of decreasing P export to water courses from harvested blanket peat sites.

7.2 Recommendations and future research work

1. The removal of brash or harvest residue should be considered as a part of forestry practices on blanket peatlands. The excessive cutting residues also make blanket peats exceptionally attractive sources of cutting residues for bioenergy purposes. Given that cutting residues will
be collected and used increasingly for bioenergy purposes and active forestry is continued in
blanket peats after clear-felling, the effect of brash removal on site nutrient capitals and the
long-term sustainability of forestry needs to be assessed in catchment scale studies.

2. The feasibility and sustainability of mini-buffers method to decrease nutrient export from
harvested sites should be assessed in long-term, catchment scale, field studies before their
introduction in operational forestry.

3. To further explain the differences in nutrient release patterns from different types of peats
due to high water tables, the factors controlling their release under anaerobic conditions
need further investigation. Future work should include an investigation of the changes in
different biogeochemical (e.g., Al, Fe and Ca-bound) forms of P and microbiologically-
bound forms of P and N in peats as a result of anaerobic conditions.