Whole-tree harvesting and grass seeding as potential mitigation methods for phosphorus export in peatland catchments

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Abstract

Forest clearfelling is potentially a major environmental problem with respect to the degradation of water quality in receiving water courses due to phosphorus (P) release from soil and clearfelling residue stocks. Recent studies have highlighted the need to investigate the performance and benefits of potential mitigation methods such as whole tree harvesting (WTH) and grass seeding. In this study, fifteen plots (0.014 ha each) were constructed in a standing coniferous forest and P concentrations in plot runoff were monitored for one year prior to clearfelling. Following clearfelling three replicates of five forest harvesting management practices/treatments were applied to the plots: brash with grass seeding (Treatment 1), brash (Treatment 2), brash mat/ tree extraction route (Treatment 3), WTH (Treatment 4) and WTH with grass seeding (Treatment 5). These treatments were designed to comparatively assess the benefits of WTH and grass seeding practices on mitigating P released from forested peatlands following clearfelling and to determine the sources and sinks of P following clearfelling operations. Annual average total reactive phosphorus (TRP) concentrations in the plot runoff were $< 20 \mu\text{g L}^{-1}$ in all treatments before clearfelling, and increased to $79 \mu\text{g L}^{-1}$, $160 \mu\text{g L}^{-1}$, $335 \mu\text{g L}^{-1}$, $50 \mu\text{g L}^{-1}$ and $38 \mu\text{g L}^{-1}$ in Treatments 1, 2, 3, 4 and 5, respectively, after clearfelling. These results highlight that WTH and grass seeding can be used efficiently as methods to improve water quality, aiding in the protection of the biota residing in the aquatic systems draining peatland catchments.

Keywords: Blanket peat; Forest clearfelling; Phosphorus export; Whole tree harvesting; Grass seeding

1. Introduction

Peatland conversion to afforestation was frequently practiced in north-western Europe, Fennoscandia, the former USSR, and North America, during the late 20th century (Paavilainen and Päivänen, 1995) driven by a need to improve a recognised economically valuable natural resource. The economic viability and environmental ethics of afforesting peatland, however, has long since been questioned (MacMillan, 1993; Renou and Farrell, 2005; Anderson, 2010) and the future management of low production forests is the topic of much debate (Delaney, 2008; Tiernan, 2008; Renou- Wilson et al., 2011; Vanguelova et al., 2011). Peatland fed catchments contain oligotrophic headwaters, many of which contain IUCN (International Union for the Conservation of Nature) Red List species (e.g. salmonids and freshwater pearl mussels) which make them important biodiversity refuges (O'Driscoll et al., 2012). The European Union (EU) Water Framework Directive (WFD) legislation requires implementation of measures to maintain high status of water bodies where it exists and achieve good ecological status where it does not by 2015 (European Union, 2000). In early afforestation operations in Ireland and the UK, sites were cultivated by ploughing (Carling et al., 2001), a method which was shown to account for 40% more site disturbance than subsequent methods such as mounding (Worrell, 1996). Trees were planted on ribbons at 1.5 m intervals, and separated by furrows. Surface water flows along the furrows into collector drains which connect to the stream network. Fertiliser was typically applied at the afforestation stage on blanket peat in the UK and Ireland to increase timber production (Harriman, 1978; Rodgers et al., 2010). Presently, plantations reach their final felling age between 35 to 40 years old, at which time they are typically 'stem-only' clearfelled. The clearfelling residues (i.e. needles, twigs and branches) are collected together to form brash material mats for stem extraction protecting the peat surface during clearfelling operations. Following clearfelling operations brash mats are collected together to form windrows clearing the sites in preparation for subsequent rotation plantations.

Catchment scale studies have shown that clearfelling peatland forests can cause negative impacts on the receiving water quality with increased nutrient and suspended sediment release (Lebo and Hermann, 1994; Paavilainen and Päivänen, 1995; Ahtiainen and Huttunen, 1999; Ensign and Mallin, 2001; Nisbet, 2001; Cummins and Farrell, 2003; Nieminen, 2004; Rodgers et al., 2010, 2011). Phosphorus (P) is a key contributor to non-point surface water pollution causing nutrient enrichment of rivers and stimulating eutrophication (Reynolds, 1992). In three comparable studies (Nieminen, 2003; Cummins and Farrell, 2003; Rodgers et al., 2010) spanning national boundaries, similar P release patterns following a final harvesting event were observed. An immediate peak in P concentrations was measured in the runoff after harvesting, followed by a maximum peak the following summer with a long declining tail. The cause of this trend was not investigated in these catchment scale studies due to the complexity of P release and the many possible mechanisms controlling P concentrations (Cummins and Farrell, 2003). However, it has been suggested that the removal of the standing forest and associated lack of P uptake and mechanical site

disturbance could be the reason for the initial peak and mineralisation of organic P from decomposing clearfelling residues could be the cause of the second peak (Hyvönen et al., 2000; Ganjeguntea et al., 2004; Rodgers et al., 2010).

Current economically feasible opportunities for reducing P losses in forest harvested peatland catchments include: whole-tree harvesting; buffer zones and constructed wetlands; phased felling; and reductions in the amount of fertiliser applied. Whole-tree-harvesting (WTH) is achieved by removing the whole tree (i.e., all parts of the tree above the ground) from the site in a single operation (Nisbet et al., 1997). Needles and branches have much higher P concentrations than stem wood, with 46 kg P ha⁻¹ in needles and branches compared with 14 kg P ha⁻¹ in stem wood, reported for a Sitka spruce forest growing on shallow peat (Carey, 1980). In addition, higher soil P concentrations are associated with areas below windrows/brash material (Nieminen, 2004; Rodgers et al., 2010) in clearfelled peatland catchments indicating that WTH could be used as a means to decrease P export. Yanai (1998) reported negligible P loss to streams over 3 years from WTH on mineral soil at the Hubbard Brook Experimental forest in New Hampshire. However, mineral soil has been shown to have a much higher P adsorption capacity than blanket peat (Tamm et al., 1974). The WTH practice as a method for mitigating P release has not been examined for blanket peat soils with low iron and aluminium concentrations and low P adsorption properties such as the peat soils found in the Burrishoole catchment (Asam et al., 2012). The by products of WTH are gaining increasing value due to the rising demands for renewable energy (Vanguelova et al., 2010; Laudon et al., 2011) and concerns have been raised regarding decreased site productivity for subsequent forest rotations (Nisbet et al., 1997) necessitating scientific observations to underpin evidence based policy developments. Buffer zones and constructed wetlands are commonly used by forestry practitioners in management of freshwater aquatic systems (Newbold et al., 2010). They can control runoff by reducing the flow thus increasing deposition and interaction between incoming nutrients and soil matrices, and plant and microbial nutrient processes. However, Rodgers et al. (2010) highlighted that traditional buffers of 15–20 m may not be adequate to reduce P export as the majority of P released occurred during storm events when buffer zones can have low residence times. Constructed wetlands are costly to establish and many earlier afforested peat catchments in Ireland and the UK were established without riparian buffer areas and trees were planted to the stream edge (Ryder et al., 2011; O'Driscoll et al., 2014). O'Driscoll et al. (2014) found that a grassed peatland buffer zone removed only 18% of the total reactive phosphorus (TRP) released from an upstream clearfelled blanket peat site. O'Driscoll et al. (2011) demonstrated a novel method whereby native grass species were seeded on a site immediately after clearfelling and before reforestation, confirming that vegetation could immobilise nutrient movement from a recently clearfelled blanket peat catchment. O'Driscoll et al. (2011) suggested that the

observed uptake of P measured in the grass would render a corresponding reduction in the P exported by the stream after clearfelling.

Sources of non-point pollution are especially difficult to detect in catchment scale studies as they generally encompass large areas and involve complex biotic and abiotic interactions (Solbe, 1986). Catchment scale studies are very popular in observing human induced deterioration in water quality; however, they leave many unexplained questions and are difficult to replicate (van Es et al., 1998; Sliva and Williams, 2001; Townsend et al., 2004). Plot-scale studies are typically used to elucidate the mechanisms behind nutrient loss from soil to runoff (Buda et al., 2009); to evaluate different management practices which are applied as treatments; and to provide an experimental design that is statistically valid (van Es et al., 1998; Wainwright et al., 2000). Additionally, experiments using natural rain events tend to more accurately estimate P concentrations in runoff and runoff volumes than simulated rainfall experiments (Dougherty et al., 2004, 2008; Smith and Pappas, 2010). The sensitivity of clearfelling peatland catchments in the UK, Ireland and Fennoscandia has risen to prominence in recent years in terms of economic and conservational viability; however, sustainable protection methods are inadequately researched and insufficiently proven. While forest clearfelling impacts on runoff P concentrations in blanket peatlands have been addressed in various studies further research is needed to understand the mechanisms behind the increased export of P, and to assess the efficiencies of mitigation methods such as WTH and grass seeding in improving runoff water quality. Thus, this study employed a replicated treatment plot-scale experiment in a blanket peat forest catchment in order to (1) distinguish between the removal of the standing forest and mechanical site disturbance (2) assess the performance of WTH and (3) assess the novel practice – grass seeding immediately after clearfelling on P release control.

2. Materials and methods

2.1 Study sites

This study was primarily carried out in Srahrevagh, a sub-catchment of the Burrishoole Catchment, Co. Mayo (53°39' N, 9°32' W, 220 m a.s.l.) (Fig. 1). A 7.65 ha forested sub-catchment of Srahrevagh which was drained by a small first order stream, was partially (65%) felled during the study period (Fig. 1). The study catchment was afforested in 1971 with Lodgepole pine (*Pinus contorta*) using double-mouldboard ploughing, creating furrows and ribbons (overturned turf ridges) with a 2 m spacing, aligned down the main slope, together with several collector drains aligned close to the contour. Nursery grown transplants were planted on the ribbons at 1.5 m intervals, giving an approximate soil area of 3 m² per tree and rock phosphate fertiliser (at a rate of 13.2 kg P ha⁻¹) was manually applied. Thinning is not generally carried out on upland western blanket peats due to high risk of windthrow (Cummins and

Farrell, 2003). The average depth of the peat in the study site was about 2 m overlaying bedrock of quartzite, schist and basic volcanic rock. The peat typically has a gravimetric water content of greater than 80%. A more detailed description of soil characteristics can be found in Asam et al. (2012). The Burrishoole has a mean annual rainfall and air temperature of about 2000 mm and 11 °C, respectively. The first order stream draining the 7.65 ha sub-catchment was equipped with a monitoring station at a stable channel section, downstream of the study area (Fig. 1) in 2007 as part of an earlier study, see Rodgers et al. (2010, 2011). An H-flume, a water level recorder and a data logger were installed at the monitoring station, along with a tipping bucket rain gauge with readings recorded every 5 min.

2.2 Treatments

Fifteen plots (7 m x 20 m) were constructed in the 7.65 ha standing forest in January 2010 (Fig. 1). To prevent lateral inflow of water the peat plots were isolated from the surrounding peat by inserting corrugated plastic into the furrows (0.3 m deep). To prevent inflow of water from upslope, trenches were dug across the top of all fifteen plots (0.3 m deep). The end of each plot strip was isolated with corrugated plastic and connected into purposely designed outflow surface monitoring systems using white polyvinyl chloride piping. Outgoing surface flow rates from the plot strips were measured using tipping bucket flow gauges. In January 2011 5 ha of the 7.65 ha catchment and the 15 experimental plots within, were 'stem only' clearfelled. The volume of lodgepole pine clearfelled from the study area was 372 m³ ha⁻¹. A Valmet 941 Harvester was used to clearfell the site and a Valmet 840 Forwarder delivered the stems to truck collection points beside the forest road. Following best management practices (BMPs) the crown of the tree and associated brash material (branches, twigs and needles) were collected together to form brash mats, also known as extraction route mats, which were used for machine travel to improve soil bearing capacity against the heavy harvesting machinery and reduce erosion. Five treatments (Table 1) were applied to the plots immediately after clearfelling and each treatment had three replicates. Each of the three middle plots (Treatment 3) were clearfelled first and the brash from the associated trees was laid down the length of the middle plots (20 m) in accordance with BMPs. The quantity of brash in the brash mat plot (Treatment 3) was equivalent to the brash produced from 106 trees. The Valmet 941 Harvester then drove on Treatment 3 while clearfelling the four adjacent plots (2 on either side). The application for the brash with grass seeding (Treatment 1) and brash only (Treatment 2) plots were as follows: 9 m length of brash (equivalent to the brash produced from 53 trees) was collected by the forwarder and placed into the top half of the plots. The WTH plots (Treatments 4 and 5) received no extra brash material and any present was removed. The plots which included grass seeding (Treatment 1 and 5), were seeded at a rate of 36 kg ha⁻¹ as described by O'Driscoll et al. (2011).

2.3 Runoff, soil and vegetation sampling and analyses

Previous studies have reported that 70% of the P released from clearfelled peatlands occurred during storm events (Rodgers et al., 2010, 2011). In addition, the performance of buffer zones on P retention is significantly affected by hydrological conditions (Asam et al., 2012; O'Driscoll et al., 2014), therefore this study mainly focused on storm events. A storm event was defined as a period of rainfall that was preceded and followed by at least 12 h of no rainfall (Hotta et al., 2007). From March 2010 to February 2012, water samples were taken in storm events over 24 hourly periods at the outflow surface monitoring systems coming from the treatments. Water samples were taken monthly where possible (8 storms before felling and 14 storms after felling) from the treatments via multiple slave liquid acquisition instruments controlled by a purposely modified automated ISCO sampler. In addition, an ISCO sampler was used to collect simultaneous water samples from the first order stream draining the 7.65 ha catchment. Rainfall samples were also collected by placing an open and clean plastic container near the treatments during storm events for P analysis. Total reactive phosphorus (TRP) was analysed colorimetrically using a nutrient analyser (Konelab 20, Thermo Clinical Labsystems, Finland). Daily discharge weighted mean TRP concentrations and annual loading rates were calculated based on the calculations used in Rodgers et al. (2010).

Four replicate 300 mm deep soil cores were collected from between- the-furrows and four from in-the-furrows in each of the plots using a 30 mm diameter gouge auger, before (June 2010) and after (June 2011) clearfelling. Each sample was sectioned into three depths labelled 50 mm (the humic layer, 0–100 mm, 150 mm (100–200 mm) and 250 mm (200–300 mm). Composites were formed by placing the replicates in bags and mixing by hand until visually homogenised, such that 180 (15 plots x 2 furrow levels x 3 depths x 2 years) soil samples were available for further analyses. Subsamples of approximately 0.5 g (dry weight) were removed and extracted in 30 ml of deionised water on a reciprocating shaker at 250 rpm for 30 min. The supernatant was then filtered (0.45 µm) and analysed for water extractable phosphorus (WEP). Second subsamples of approximately 5 g (wet weight) were removed and dried to determine the gravimetric moisture content (Macrae et al., 2005). The dried subsamples were placed in a furnace at a temperature of 550 °C. After 24 h, 5 ml of 2 N HCl was added to the subsample and subsequently diluted to 50 ml with deionised water to extract the P. The solution was then analysed for total phosphorus (TP). The aboveground vegetation biomass in the plots was estimated according to Moore and Chapman (1986). Four 250 mm x 250 mm quadrats were randomly sampled in each seeded plot (Treatments 1 and 5) in August 2011. All vegetation lying within the quadrat was harvested to within 10 mm from ground level and dried on the day of collection at 80 °C in the for 48 h. Samples were then

weighed, and the biomass was calculated using Eq. (1). Dried subsamples were milled to pass a 2 mm sieve and TP content of the vegetation was measured in accordance with Ryan et al. (2001).

$$B_p = \frac{W_t}{S_t} \times 10000 \quad \text{Equation 1}$$

where B_p is the biomass production (kg ha^{-1}); W_t is the total dry weight of the samples (kg) and S_t is the total area (m^2).

2.4 Phosphorus source and sink estimations

Asam et al. (2014) estimated that biomass derived from the crown of the tree (branch diameter <40 mm) amounted to ~60,000 kg ha^{-1} dry weight for lodgepole pine of Yield Class 14 with an associated P content of ~36 kg P ha^{-1} . The P sources and sinks for the plot treatments were calculated based on these findings (Table 2). Stumps and root systems were not considered in these calculations as it was assumed that these parameters were similar across all treatments. It was estimated that there were 60,000 kg ha^{-1} of brash in Treatments 1 and 2 and 120,000 kg ha^{-1} in Treatment 3, giving an associated P content of 36.07 kg P ha^{-1} in Treatments 1 and 2 and 72.14 kg P ha^{-1} in Treatment 3. Therefore it was estimated that Treatments 1, 2 and 3 had potential to release ~36.07 kg P ha^{-1} in Treatments 1 and 2 and 72.14 kg P ha^{-1} , respectively from the brash. Estimations of the quantities of P released from brash decomposition were made using the litter bag method in Glennamong sub-catchment (see O'Driscoll et al. (2014) for a site description) located about 5 km from Srahrevagh. Brash from lodgepole pine clearfelling residues was collected from the site immediately after clearfelling. In the laboratory the brash was separated into three fractions i.e. needles, small branches (diameter < 10 mm) and bigger branches (diameter 10–40 mm) and air dried. After drying the small branches were cut into 50–100 mm and bigger branches into 150–200 mm long pieces and mixed thoroughly. After mixing a recorded mass of needles, small branches and bigger branches were each enclosed in litter bags made from nylon sheets with a mesh size of 1 x 1 mm. The litter bags were placed systematically at the top ($n = 5$), middle ($n = 5$) and bottom ($n = 5$) of brash windrows in November 2009 and were sampled after 1 year in November 2010. After collection, the litter samples were cleaned, dried at +60 °C to constant mass, and weighed for the remaining dry mass. For P analyses, the contents of 5 replicate litter bags, at each of three sampling locations in windrows, were combined and milled using an analytical mill (MF 10 IKA, with a 2 mm sieve) and analysed using an ICP/AES following dry digestion in HCl. Subsamples drawn for the initial concentration of P were analysed the same way and the concentrations of P were 1170, 369 and 180 mg kg^{-1} respectively for needles, small branches and bigger branches.

2.5 Statistical analyses

Comparisons of precipitation in the years before and after clearfelling were made using non-parametric Mann–Whitney Utests. Analysis of variance (ANOVA) models were used to find the subset of explanatory variables that best explained the variation treatment, position relative to the furrow (between-the or in-the) and soil depth (50 mm, 100 mm and 150 mm) were included as explanatory variables. The most satisfactory models were selected based on AIC (Akaike Information Criteria) and the exclusion of non-significant variables was determined by stepwise selection. The storm runoff data (TRP concentrations) was also analysed using ANOVA with treatment as an explanatory variable and Tukey's post hoc analyses were conducted for pair-wise comparisons between treatments. The data were tested for normality, homogeneity and independence to check whether the assumptions of linear modelling were met. Where conditions for normality were not fulfilled the data were log transformed. Significance was determined at the 95% level unless otherwise noted. Mean values for treatment replicates ($n = 3$) were used for the TP and WEP data to eliminate spatial autocorrelation (which appeared in autocorrelation function (ACF) graphs in preliminary analyses) to satisfy the assumption of independence.

4. Results

4.1 Precipitation, runoff and phosphorus concentration trends

Total precipitation from a nearby Met Eireann weather station (circa 12.1 km south of the study site) was 1398 mm before clearfelling (February 2010–January 2011) and 1981 mm after clearfelling (February 2011–January 2012) (Marine Institute, unpublished data), being significantly higher after clearfelling ($P < 0.0001$). The number of wet days (>1 mm of precipitation/ day (Hundecha and Bárdossy, 2005)) recorded was 174 before clearfelling and 244 after clearfelling. The average daily precipitation before clearfelling was 3.8 mm and 5.4 mm after clearfelling. Precipitation data indicates that there were 14 heavy precipitation (>20 mm/day (Ryder et al., 2011)) occurrences before clearfelling and 15 such occurrences after clearfelling (Fig. 2). While the different precipitation conditions before and after clearfelling may have an impact on the pre- and post-clear-felling comparisons, they are unlikely to impact the comparisons between treatments in the year after clear-felling. This is supported by the fact that the different treatments were hydrologically similar in the year after clear-felling; the relationship between the flows coming from the different treatments was very strong ($R^2 > 0.97$ for all pair-wise comparisons). Annual water yields in the 7.65 ha catchment ranged from 55% to 60% of the overall rainfall in the three years before harvesting (2007–2010) and increased to 75.8% after clearfelling (2011) (Fig. 3). Similarly, annual water yield from the treatments increased from 49% of the overall rainfall in 2010 to 73% in 2011.

The averaged TRP concentration measured in the rainfall over the study duration was $5.83 \pm 5 \mu\text{g L}^{-1}$. Measured TRP concentrations in the first order stream draining the 7.65 ha catchment were low

before and after clearfelling with average values before felling of $11.4 \pm 5 \mu\text{g L}^{-1}$ and $20.4 \pm 7 \mu\text{g L}^{-1}$ after felling. Measured daily TRP concentrations from the five treatments were low before clearfelling (Fig. 4). Four months after felling, the daily mean TRP started to increase, giving the 2011 annual discharge weighted mean TRP concentrations of $79 \pm 59 \mu\text{g L}^{-1}$, $160 \pm 116 \mu\text{g L}^{-1}$, $335 \pm 178 \mu\text{g L}^{-1}$, $50 \pm 32 \mu\text{g L}^{-1}$ and $38 \pm 22 \mu\text{g L}^{-1}$ in Treatments 1, 2, 3, 4 and 5, respectively. The highest daily average TRP concentration was observed coming from Treatment 3 (brush mat) runoff, with the value of $759 \mu\text{g L}^{-1}$. The TRP release load rates increased from $88 \pm 4.7 \text{ g ha}^{-1}\text{year}^{-1}$, $82 \pm 48 \text{ g ha}^{-1}\text{year}^{-1}$, $88 \pm 47 \text{ g ha}^{-1}\text{year}^{-1}$, $88 \pm 20 \text{ g ha}^{-1}\text{year}^{-1}$ and $113 \pm 23 \text{ g ha}^{-1}\text{year}^{-1}$ in Treatments 1, 2, 3, 4 and 5, respectively, in 2010, to $693 \pm 262 \text{ g ha}^{-1}\text{year}^{-1}$, $1403 \pm 724 \text{ g ha}^{-1}\text{year}^{-1}$, $2937 \pm 1324 \text{ g ha}^{-1}\text{year}^{-1}$, $438 \pm 149 \text{ g ha}^{-1}\text{year}^{-1}$ and $333 \pm 126 \text{ g ha}^{-1}\text{year}^{-1}$ in 2011.

The P release measured from the needles, small branches and bigger branches in the litter bag study was 18%, 33% and 37% of their initial content, respectively. This gave an approximate availability of $8.21 \text{ kg P ha}^{-1}$ release from the brush to the underlying soil in Treatments 1 and 2 and $16.42 \text{ kg P ha}^{-1}$ release in the case of Treatment 3 in the year after clearfelling.

4.2 Runoff total reactive phosphorus concentrations from the treatments during storm events

In one storm event (5th October 2011), TRP concentrations ranged from $300 \mu\text{g L}^{-1}$ to $650 \mu\text{g L}^{-1}$ in runoff collected from Treatment 3 and from $180 \mu\text{g L}^{-1}$ to $480 \mu\text{g L}^{-1}$ in runoff collected from Treatment 2 (Fig. 5). Runoff TRP concentrations from treatments 1, 4 and 5 were all lower than $100 \mu\text{g L}^{-1}$. In six consecutive storm events, TRP concentrations from Treatment 3 runoff were significantly higher ($p < 0.001$) than the other four treatments with an average value of $342.3 \pm 22 \mu\text{g L}^{-1}$. Treatment 2 was the next highest with an average value of $214.6 \pm 87 \mu\text{g L}^{-1}$. The third highest for P release was Treatment 1, followed by Treatments 4 and 5 with average values across six storm events of $141.3 \pm 58 \mu\text{g L}^{-1}$, $75.3 \pm 38 \mu\text{g L}^{-1}$ and $47.7 \pm 31 \mu\text{g L}^{-1}$. Treatment 5 was significantly lower than Treatment 1 (Tukey's post hoc test, $p < 0.001$) and Treatment 4 (Tukey's post hoc test, $p < 0.05$).

4.3 Soil water extractable and total phosphorus concentrations

Soil TP concentrations ranged from 0.22 to $0.25 \text{ mg TP g}^{-1}$ dry soil. The average TP in the soil was significantly different before and after felling ($p < 0.01$) (Table 3; Fig. 6). Values increased by an average of 0.09 mg P g^{-1} dry soil in the year after felling. Total phosphorus (TP) concentrations were significantly lower from samples taken in the furrows ($0.15 \pm 0.08 \text{ mg P g}^{-1}$ dry soil) compared to samples taken between the furrows ($0.22 \pm 0.07 \text{ mg P g}^{-1}$ dry soil) and TP concentrations were also observed to

increase with depth ($p < 0.01$) (Table 3; Fig. 6). The interaction term Year x Furrow was also significant ($p = 0.01$) (Table 3; Fig. 6).

Soil WEP concentrations ranged from 4 to 7 $\mu\text{g WEP g}^{-1}$ dry soil. The average soil WEP concentrations increased significantly in the year after clearfelling ($p = 0.01$) by 0.78 $\mu\text{g kg}^{-1}$ dry soil (Table 3; Fig. 6). The highest WEP value was obtained from Treatment 3 (Fig. 6). Water extractable phosphorus (WEP) concentrations were significantly lower ($p < 0.01$) between the furrows at 4.31 $\mu\text{g kg}^{-1}$ dry soil compared with 4.93 $\mu\text{g kg}^{-1}$ dry soil in the furrows (Table 3; Fig. 6). The term Treatment was also a significant factor in the year after clearfelling with WEP concentrations in the Treatment 3 (brash mat) increasing by 2.78 $\mu\text{g kg}^{-1}$ dry soil (Table 3; Fig. 6).

4.4 Biomass production and phosphorus concentration of the vegetation in the treatments after harvesting

No vegetation growth was observed in the treatments which did not receive grass seeding (i.e. Treatments 2, 3 and 4) in the year after clearfelling. Seeding of *Holcus lanatus* L. and *Agrostis capillaris* L. increased the aboveground vegetation biomass in both Treatments 1 and 5. Six months after grass seeding, Treatment 1 contained 100 kg ha^{-1} more aboveground biomass than Treatment 5 (Fig. 7a). The grass up take of P was 9.99 ± 4 and 5.60 ± 2 kg TP ha^{-1} , respectively, in Treatments 1 and 5 (Fig. 7b).

5. Discussion

5.1 Comparative assessment of treatments

The increased P concentrations observed in the plot runoff after clearfelling in this study are comparable with those previously reported in blanket peat catchments in Ireland (Cummins and Farrell, 2003; Rodgers et al., 2010). Phosphorus (P) released from the clearfelled peatland catchment to the receiving water depends on sources, sinks and transport pathways. Minimising sources and maximising sinks could effectively reduce the P release to runoff. In this study, P release from Treatment 2 (brash) runoff was ~3.2 times greater than Treatment 4 (without brash) runoff. These findings are supported by a laboratory flume experiment using soil samples from the same sub-catchment (Asam et al., 2014). While the laboratory P export values reported by Asam et al. (2014) were considerably higher than the values of P observed in the treatment runoff from this study, many studies agree that the clearfelling residues contain 50% of the total tree P store (Taylor, 1991; Titus and Malcolm, 1991; Stevens et al., 1995; Olsson et al., 1996; Hyvönen et al., 2000) and are a primary source of P to receiving water courses from clearfelled peatlands.

Higher soil TP concentrations were observed from treatment plots after clearfelling suggesting that peat soil acted as a P sink. Phosphorus (P) retained by soil/peat through adsorption to the peat/soil

surface depends on the P concentration in the solution with higher concentrations associated with increased adsorption and lower concentrations with desorption from the peat surface (Nieminen et al., 2005; Asam et al., 2012). Asam et al. (2014) found that the top 0.1 m of the humus layer had the capacity to absorb 4 kg P ha⁻¹. The upper soil layer (50 mm) had significantly higher TP concentrations than the deeper layers and it is likely that the upper soil layer acted as a sink for the P released from the brash material and from mineralisation of root and humus layers. The between- the-furrow samples contained more TP than the in-the-furrow samples consistent with P leaching from the soil in-the-furrows under reducing conditions and increased decomposition of the upper most aerobic soil humus layer in between-the-furrow areas (Holden et al., 2004). There was an overall increase in WEP in-the-furrows compared to between-the-furrows consistent with mobilisation of P into the furrow and increases in labile P in the furrow soils (Sapek et al., 2007). Similarly, in a nearby catchment with similar soil types, Rodgers et al. (2010) showed WEP concentrations were significantly higher (136%, 152.3%, 235% and 188.9%, in year 1, 2, 3 and 4 respectively), after clearfelling in the soil under the brash/windrow compared with the brash/windrow free soils. The increase in soil WEP concentrations can be attributed to the decomposition of the logging residues, organic horizon and organic matter in the surface soil and the reduced P uptake by the once standing forest (Palviainen et al., 2004). The findings of this study indicate that the humus layer (i.e. decomposing peat and root systems) can act as a source of P when the runoff has a low P concentration. In Treatment 4 (no brash), P release was higher after clearfelling (2011) than before clearfelling (2010), which suggests that the humus layer contributed to the observed increase in P runoff concentrations. However, this needs to be considered with caution as precipitation was greater in 2011 and with it higher release of redox-sensitive P is possible.

Lower P release was observed coming from Treatment 1 (brash and grass) than from Treatment 2 (brash and no grass). Similarly, Treatment 5 (WTH and grass) had lower P release than Treatment 4 (WTH and no grass). The P uptake (per ha) by vegetation in Treatment 1 with P release from brash was 2-fold the P uptake (per ha) by vegetation in Treatment 5 with no P input from brash, indicating that higher P concentrations in through-flow of the vegetated area stimulated vegetation growth and increase the P sink capacity in forest harvested peatland catchments. In Treatment 5, the presence of vegetation reduced the P release by 23% compared to Treatment 4, whereas, in Treatment 1 the presence of vegetation reduced the P release by 50% compared with Treatment 2. The vigorous growth of natural vegetation following harvesting has been identified as a significant factor in the retention of P (Silvan et al., 2004; Piirainen et al., 2007). In upland blanket peat catchments the natural re-growth process can take several years (Connaghan, 2007; O'Driscoll et al., 2011). The findings of this study are in agreement with O'Driscoll et al. (2011), demonstrating that vegetation could immobilise nutrient movement from a recently clearfelled blanket peat catchment. Additionally, this study has shown that the uptake of P

measured in the grass rendered a corresponding reduction in the P exported by the stream after clearfelling.

Soil compaction occurs during times of harvest and extraction machinery traffic (Nugent et al., 2003). The consequences of excessive soil compaction are a decrease in soil pore space and a reduction in surface water infiltration, enhancing conditions for surface run off; nutrient export and soil erosion (Tiernan et al., 2002). Brash mats composed of harvesting residues are used as a current approach to minimise soil disturbance (McDonald and Seixas, 1997; Tiernan et al., 2002). Following soil compaction with the harvesting and forwarder machinery, water may form pools beneath the brash mats as the upper humus layer becomes less permeable, increasing the release of labile and redox sensitive soil P. By comparing P loading rates from the “brash” Treatment 2 ($1403 \pm 724 \text{ g ha}^{-1}\text{year}^{-1}$) with the P released from the “extraction brash mat” Treatment 3 ($2937 \pm 1324 \text{ g ha}^{-1} - \text{Year}^{-1}$) and considering that Treatment 3 received twice the amount of brash that Treatment 2 received (i.e. eliminating the P released from brash from the P export of Treatment 3) it could be estimated that $131 \text{ g P ha}^{-1}\text{year}^{-1}$ was released from Treatment 3 in 2011 due to mechanical soil disturbance caused by forest harvesting machinery. Thus, the P release to runoff waters due to the mechanical soil disturbance by extraction machinery appears to be small compared with the larger contribution of the remaining clearfelling residues. However, the brash mat plot was only travelled on one occasion. The number of times travelled by machinery is an important factor in enhancing conditions for nutrient export. Further research is needed to determine the maximum number of times machinery should travel before the efficiency of brash mats in preventing soil disturbance is compromised.

5.2 Implications for best management practices of peatland forest harvesting

Two major concerns for forest harvesting BMPs on blanket peatland were highlighted by this study: (1) While WTH did reduce the P concentrations in runoff, the humus layer served as a source of P; and (2) while brash mats are employed to protect the peat and subsequently the water quality, the runoff draining these brash mats had the highest P concentrations after clearfelling. The P concentration in water draining pristine peatlands rarely exceeds $20 \mu\text{g L}^{-1}$ (Kenttämies, 1981), and is only marginally higher than the P concentration found in the associated rain water (Soveri and Peltonen, 1996; Rodgers et al., 2010). Many of these peatland fed rivers are ecologically sensitive containing IUCN Red List species which make them important biodiversity refuges (O’Driscoll et al., 2012). The Irish Phosphorus Regulations set the threshold of $<30 \mu\text{g L}^{-1}$ of Molybdate Reactive Phosphorus (MRP) for a stream to be of good quality. Best Management Practices (BMPs), such as clearfelling smaller coupes at one time have been shown to dilute the P concentration in the larger rivers (Rodgers et al., 2010; O’Driscoll et al.,

2013). This management strategy however, does not reduce the total P load leaving the clearfelled catchment (Rodgers et al., 2010) and draining into sensitive lakes (Drinan et al., 2013). Yanai (1998) reported negligible P loss to streams over 3 years from implementing WTH on mineral soils. This study's findings indicate that owing to the P release from the humus layer in peat soils the efficiency of the WTH method maybe limited. A conservative value of $>30 \mu\text{g L}^{-1}$ has been statistically linked with lower biological Q ratings (biological quality ratings) (EPA, 2005), phytoplankton production (Daniel et al., 1998) and increased algal growth in freshwaters (Haygarth et al., 2005). According to this study's findings, WTH may not be adequate in reducing P concentrations to this acceptable level. This study has shown that by combining WTH with grass seeding, P concentrations in the runoff were reduced to $38 \mu\text{g L}^{-1}$. In peatland forest catchments that are to receive a second rotation, the use of WTH in reducing P export is questionable, not only because of the limited efficiency to reduce P export as shown in this study, but also because WTH may deplete the nutrient pools and affect the growth of the second rotation of forestry (Kaarakka, 2013). A more viable solution in this instance could involve the arrangement of brash windrows across the gradient rather than with the gradient (Rodgers et al., 2010; Asam et al., 2012) and employing the grass seeding method (O'Driscoll et al., 2011) to maximise the potential of the humus layer and associated vegetation to retain P in situ and significantly reduce P loss from the site.

5.3 Scale Issues

While this novel experimental setup enabled the examination of blanket peat forest harvesting management practices on P sources and sinks in detail the results cannot be utilised to infer similar findings at the catchment level (Dougherty et al., 2004, 2008; Sharpley and Kleinman, 2003; Smith and Pappas, 2010; Hahn et al., 2012). Phosphorus (P) concentrations in the runoff from the plots were higher than in the sub-catchment both before and after harvesting. The release of nutrients is affected by transport pathways in addition to sources and sinks. Under natural conditions, i.e. were the plots not isolated, the furrows would have received runoff from upslope leading to dilution in the observed P export (Sharpley and Kleinman, 2003). Long contact times between the P sources/ sinks and runoff in the catchments can increase or decrease nutrient export (Sharpley and Kleinman, 2003; Dougherty et al., 2004). In this study, runoff contact times with P sources/sinks may not have been significantly different between the plots and the sub-catchment. After analysing 6 storm events, it was found that the time between the rise of the flow and the peak flow was similar in the plots and the sub-catchment. This has been reported previously in the literature and has been associated with the artificial drainage channels (furrows) and low hydraulic permeability of the peat (Müller, 2000; Holden, 2003). Water yields increased in both the sub-catchment and the runoff plots following clearfelling. The increase of water yield could be due to harvesting as this has been shown previously in the literature (Hornbeck et al., 1970;

Johnson, 1998; Robinson and Dupeyrat, 2005) however, the significantly higher precipitation in the 2011 period could have also contributed. In the hydrologically isolated plots, the recorded measurements reflect the surface and shallow sub-surface flow, whereas in the sub-catchment, measurements reflect deeper drainage systems (collector drains <1 m). Before harvesting water yields in the plots were similar to the subcatchment storm flow. After harvesting, however, water yields in the plots increased more than in the sub-catchment, which could be explained by the fact that the sub-catchment was not clearfelled completely, 35% of the forest remained and contributed to runoff retention by the tree stand evapotranspiration.

6. Conclusion

The findings of this study demonstrated that WTH reduced runoff P concentrations. Where WTH was implemented runoff P concentrations were higher than before clearfelling suggesting that the humus layer is a source of P. The highest runoff P concentrations were observed coming from the brash mat plot, where the amount of brash was twice that than in the brash treatment. Compared with direct P release to runoff from brash the release caused by mechanical soil and brash compaction in the brash mat plots was estimated to have little contribution to P export. The lowest runoff P concentrations were observed coming from the WTH and grass plot suggesting that by combining WTH with grass seeding the P concentration in runoff could be reduced to conservative levels (EPA, 2005). These findings have two practical applications in the context of mitigating P release on clearfelled blanket peat: (1) where the felled coupe is to receive a second forestry rotation and WTH is not seen as a feasible harvesting option due to the risk for depletion of site nutrient capitals the brash could be left on site, windrowed across the gradient and grass seeded to immobilise the P on site; (2) where the felled coupe is set aside from active forestry use WTH could be applied in addition to the grass seeding method to reduce the export of P from the soil layer and root decomposition.

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Table 1 the practices examined in the 5 plots

Plot	Treatment	Objectives
1	Brash with grass seeding	To investigate the impact of leaving brash on site and seeding the site with native grass species on P release
2	Brash only	To investigate the P export from the practice of leaving brash on the site
3	Brash mat/ extraction route	To investigate the impact of machine travelling on brash mat on P export
4	Whole tree harvesting (WTH)	To explore the whole tree harvesting method (where all brash material is removed from the site) on P release mitigation
5	WTH and grass seeding	To study the practice of whole tree harvesting plus grass seeding on Prelease mitigation

Table 2 Estimated sources and sinks of P in the plots

P sources and sinks	Treatment 1	Treatment 2	Treatment 3	Treatment 4	Treatment 5
Humus layer (ha)	0.014	0.014	0.014	0.014	0.014
Harvesting residue biomass* (kg)	840	840	1680	0	0
Clearfelling residue biomass P content* (g)	505	505	1010	0	0
Machinery disturbed humus layer area (ha)	0	0	0.014	0	0
Grass seeded area (ha)	0.0077	0	0	0	0.014

* These values have been calculated based on measurements taken by Asam et al. (2013).

Table 3 Results of the Analysis of Variance (ANOVA) determining significant sources of variation in total phosphorus and water extractable phosphorus (dependant variables) in the soils of five experimental plots. Three explanatory variables were included in the models as factors (Furrow, Depth and Year), and the table also includes significant interaction terms.

Total phosphorus (TP)	n	Df	F-Statistic	<i>p</i>
Furrow (Between, In)	60	1	49.8	<0.01
Depth (0, 15, 25 cm)	60	1	49.1	<0.01
Year (2010, 2011)	60	1	62.4	<0.01
Interaction (Year x Furrow)	60	1	7.6	0.01
Water extractable phosphorus (WEP)	n	Df	F-Statistic	<i>p</i>
Furrow (Between, In)	60	1	14.9	0.03
Treatment (1-5)	60	4	2.9	0.03
Year (2010, 2011)	60	1	7.7	0.01
Interaction (Year x Treatment)	60	4	3.2	0.02

Figure 1 Geographical location of the study site in the Burrishoole Catchment, Co. Mayo, Ireland and a schematic of the experimental runoff plots (Treatments 1- 5).

Figure 2 Daily precipitations recorded at the Met Eireann weather station (circa 12.1 km south of the study site) (a) before clearfelling, (b) after clearfelling and (c) the monthly periods. The grey line indicates the threshold for storm occurrence.

Figure 3 Water yield from the Srahrevagh sub-catchment and the mean water yield of the plots for 2007 – 2011. (The error bars indicate $\pm$ standard deviation).

Figure 4 Daily discharge-weighted mean total reactive phosphorus runoff concentrations from the Treatments (1 -5, n=3) before and after clearfelling and treatment establishment (March 2010 – February 2011). Dash lines indicate the clearfelling period.

Figure 5 Total reactive phosphorus concentrations during a storm event (5th of October 2011) after clearfelling

Figure 6 Soil water extractable (a) and total phosphorus (b) concentrations before and after clearfelling across all treatments; soil water extractable phosphorus (c) concentrations between- and in-the-furrows across all treatments; soil total phosphorus concentrations between- and in-the-furrows (d) and at 50 mm, 100 mm and 150 mm depths (e) across all treatments. (The error bars indicate $\pm$ standard deviation).

Figure 7 Biomass (a) and (b) P content of above ground vegetation in the study plots brash and grass and grass only. (The error bars indicate $\pm$ standard deviation).

Figure 1

Figure 2

Figure 3

Figure 4

Figure 5

(a)

(b)

(c)

(d)

(e)

Figure 6

Figure 7 a and b