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Implications of applied best management practice for peatland forest harvesting

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ABSTRACT

Elevated levels of nutrients and suspended sediment (SS), and changes to other environmental parameters, are frequently associated with forestry harvesting (clearfelling) operations, and are indicative of the potentially complex changing environment associated with clearfelling. Current and future recommended best management practices (BMPs) for forestry clearfelling on upland peat catchments must provide for a healthy soil and good water quality. The aim of this study was to quantify the effects of implementation, or violation, of BMPs in the clearfelling of an upland peat conifer forest. Over periods of 12 months prior to clearfelling and 15 months after clearfelling, two peatland forests, comprising a study control (no clearfelling) and a study site (clearfelling), were monitored for the release of phosphorus (P) and nitrogen (N) species (dissolved reactive phosphorus (DRP), total phosphorus (TP), total oxidised nitrogen (TON) and ammonium nitrogen (NH_4^+-N)), SS, dissolved oxygen (DO), electrical conductivity (EC), pH and stream water temperature. Clearfelling was conducted during poor weather conditions and a watercourse, which drained the study site, was not protected. The maximum recorded concentration exported from the study site after clearfelling was 471 μg L^{-1} for DRP, 611 μg L^{-1} for TP, 1336 μg L^{-1} for NH_4^+-N, and 194 μg L^{-1} for TON. Concentrations of SS exiting the study site increased in one of the two samples taken during clearfelling (maximum release of 481 mg L^{-1}, with 68% of this organic) and returned to pre-clearfelling levels, or below, within 6 months of the commencement of clearfelling. Exports of TP and DRP from the study site were 0.9 and 0.4

kg ha^{-1} yr^{-1}, which were greater than the study control (0.6 and 0.2 kg ha^{-1} yr^{-1}, respectively). This indicated that the mitigation practices employed on site were not effective in phosphorus retention.

***Keywords**:* nutrients, forestry, brash mats, peat, clearfelling, harvesting

1. Introduction

Ireland's forest cover stands at 10% (698,000 ha) of the total surface area of the island and 59.6% of total afforestation is on peat (National Forest Inventory, 2007). Most of this forestry is now at harvestable age (Renou-Wilson et al., 2011) and peatland forests are particularly sensitive to soil erosion from clearfelling (the harvesting of all marketable trees in a stand at the end of a rotation) (Forest Service, 2000a). Clearfelling of this forest may cause elevated levels of nutrients (Cummins and Farrell, 2003a; Rodgers et al., 2010; Finnegan et al., 2012) and suspended sediment (SS) (Rodgers et al., 2011) in adjacent waterways for up to 4 years after harvest (Adamson and Hornung, 1990; Neal et al., 1999), which may have an impact on the ecosystem of the recipient watercource and soil quality in the forest (Vasconcellos et al., 2013). Therefore, current and future recommended best management practices (BMPs) for forestry clearfelling on upland peat catchments must consider soil and water quality (Collins et al., 2000).

Forestry operations on peatland throughout the world are now moving towards a 'progressive management approach' (Joosten and Clarke, 2002), which aims to reduce the potentially negative effects to the surrounding environment. Coillte, the Irish State's current forest management company, is certified under the Forest Stewardship Council (FSC) to enforce environmental, economic and social criteria for sustainable forest management (Coillte, 2012). These criteria include detailed planning (prior to the commencement of clearfelling) to provide protection to watercourses from drainage, fertilisation and afforestation, final harvest and regeneration (Owende et al., 2002). The 'Code of Best Forest Practice – Ireland' (Collins et al., 2000), and the associated guidance documents (Forest Service, 2000a,b,c,d,e,f), which are based on the principles of Sustainable Forest Management (SFM), contain BMPs for all forestry operations, including nursery practices, planting, thinning and transport of materials (Collins et al., 2000). Under present BMPs, management of final harvest needs to include consideration of felling coupe (an area of woodland that has been clearfelled or is planned for clear felling; Coillte, 2013) size and shape, road construction, soil type and sensitivities, local watercourses, extraction routes (the areas, overlain by brash material, on which harvested trees are transported from site) and landing areas (Collins et al., 2000) (Table 1). In particular, the practice of clearfelling in dry weather, the use of brash mats (logging residues used for machinery traffic) and ancillary structures such as silt traps, are recommended (Forest Service, 2000a). Harvest site restoration guidelines include provisions for drain and road repair, and water management on extraction routes (Forest Service, 2000a) in order to prevent, or reduce, excessive loss of nutrients and sediment to receiving watercourses.

Best management practices in Irish forestry are not based on quantifiable scientific data, but are based on empirical data arising from local knowledge as well as BMPs in existence elsewhere. They do, however, provide a conceptual framework, to which the adherence or non-adherence to BMPs may be compared. In fact, some evidence suggests that the implementation of BMPs may not be effective in reducing phosphorus (P), for example, as the major cause for enhanced P export may be P release from harvest residues (Palviainen et al., 2004; Kaila et al., 2012). Similarly, BMPs concerning the time of year and the weather during which felling is conducted, may not impact the export of nutrients from a harvested area (Rodgers et al., 2010). Although BMPs are of questionable merit, they do govern forestry practices in Ireland and elsewhere. Therefore, the hypothesis of this study is that adherence to BMPs means that export of nutrients and suspended solids (SS), arising from clearfelling of forested blanket peat, will be mitigated.

Nutrients such as nitrogen (N) and P are often applied to land at the afforestation stage to enhance and promote growth of selected species within ombrotrophic blanket peats (peats which have low nutrient concentrations and poor adsorption capacities) in the west of Ireland (Farrell and Boyle, 1990; Renou and Farrell, 2005). This, combined with N deposition from the atmosphere and ammonification within the peat layers, has led to N saturation, primarily present as ammonium (NH_4^+), in some upland peat catchments in the UK (Daniels et al., 2012). Ammonium can leach from the peat and be converted to nitrate (NO_3^-) by nitrification within streams (Daniels et al., 2012), leading to toxic environments for aquatic life forms (Stark and Richards, 2008). Similarly, concentrations of P (> 35 μg L^{-1} molydbate reactive phosphorus (MRP)) can have a negative impact on water quality (Bowman, 2009), leading to restrictions for fisheries, recreation, industry and drinking water (Sharpley, 2003; Elrashidi, 2011). Blanket peat has a poor P adsorption capacity (O'Driscoll et al., 2011) and during the forest operations of drainage, fertilisation and clearfelling, hydrological losses of P can increase (Cummins and Farrell, 2003a; Nieminen, 2003; Väänänen et al., 2008). Phosphorus release in the clearfelled area in the first 3 years after clearfelling of a mixed boreal forest was shown by Palviainen et al. (2004) to be mainly due to decomposition of foliage, which accounted for 70% of total P release from logging residues. Furthermore, Kaila et al. (2012) showed that P is easily released from harvest residue needles in clearfelled areas and concluded that this easy P release may be a cause for the reported high P losses from peat soils soon after clearfelling. Rodgers et al. (2010) reported high P losses from a clearfelled peat site and showed that more than 80% of P export occurred during storm events. However, they also showed that P levels in streams draining clearfelled areas can return to pre-clearfell levels within 4 years of clearfelling. Peat soils are also susceptible to damage by clearfelling, machinery traffic and subsequent rutting and compaction (Collins et al., 2000). After clearfelling, SS levels in receiving waters can increase due to soil disturbance, bank erosion and increased flow from the harvested areas, but these impacts are generally not long-term (Rodgers et al., 2011).

Other environmental parameters, such as dissolved oxygen (DO) (Ensign and Mallin, 2001), electrical conductivity (EC) (Cummins and Farrell, 2003b), pH (Neal et al., 1992) and stream water temperature (Stott and Marks, 2000), may be impacted by clearfelling, and are indicative of the potentially complex changing environment associated with forestry harvesting (Rodgers et al., 2008). An increase in biochemical oxygen demand (BOD) from increased organic material and algal blooms can decrease the DO levels in waterbodies downstream of clearfelled areas (Ensign and Mallin, 2001). By comparing lakes in catchments with different land uses, Drinan et al. (2012) were able to show that DO concentrations were lower in lakes located in catchments with clearfelling or mature plantations, than lakes located in catchments where only unplanted blanket bog was present.

Stream water temperature is seen as one of the best indicators of stream vitality, and can be affected by forestry operations such as afforestation and deforestation (Stott and Marks, 2000; Quinn and Wright-Stow, 2008). Studies in the UK have shown that a decrease in stream water temperature occurs after afforestation (Weatherley and

Ormerod, 1990), while an increase occurs after deforestation (Neal et al., 1992). A reduction in water temperature in spring and summer due to tree coverage of streams can lead to lower rates of development of invertebrates and fish (Weatherley and Ormerod, 1990). However, the impact of deforestation on ecology and the recovery of ecology are less clear, with either increases in invertebrates (Kirby et al., 1991) or no change in biological status being reported (Gee and Smith, 1997).

The upland peat catchments of the west of Ireland are classified as acid sensitive with the main pressures (such as acidification and nutrient and sediment addition) on rivers coming from forestry operations and peat degradation (O' Driscoll et al., 2012). The typical low pH values (approximately 4) of these catchment streams is assumed to result from the high runoff from low permeability, acidic soils, with little interaction with groundwater to neutralise the acidity, as seen in similar sites in the UK (Neal et al., 2004). Forests may exacerbate the existing acid conditions both indirectly, through canopy interception of atmospheric pollutants, and directly, by the uptake of base cations and nutrients during biomass growth and subsequent removal from site during clearfelling (Johnson et al., 2008). The net loss of base cations that accompanies the harvesting of stem wood, or any other form of biomass extraction, may affect the vitality and stability of the forest ecosystem (Hüttl and Schneider, 1998). Base cations are important for buffering against changes in soil and water acidity (Lucas et al., 2013). As the number of base cations decreases, there is an increase in the percentage of aluminium (Al^{3+}) and hydrogen (H^{+}) ions relative to base cations and therefore a reduction in soil pH. Little is known about the impact of clearfelling on stream water pH in upland peat forestry in Ireland.

To date, there are little published data on the effects of forest clearfelling on receiving waterbodies in Ireland (Rodgers et al., 2011). There is a need to quantify the effects of implementation of BMPs (or deviation from BMPs), in peatland forestry clearfelling operations, on nutrient and sediment release (Coillte, 2008). Therefore, the aim of the present study was to examine, in a paired catchment study including a study control (no clearfelling) and a study site (clearfelling), the impact of clearfelling of an upland peat conifer forest on the release of P, N and SS, expressed as concentrations and loads released, and the changes in DO, EC, pH and stream water temperature, after the implementation of BMPs.

2. Materials and Methods

2.1. Study Site Description

The study area was located in the Glennamong forest in the Burrishoole catchment in Co. Mayo, Ireland (ITM reference 494252, 803180) (Figure 1). Two adjacent sub-catchments were studied: (1) a control catchment (CC), in which no clearfelling or forestry operations took place and (2) a study catchment (SC), in which clearfelling of the catchment took place (identified as 'Control' and 'Study' in Figure 1). The CC and SC are each approximately 10 ha in area, and each is drained by a small ephemeral stream instrumented with sampling equipment (identified as 'Steams' and 'Sampling' in Figure 1). These streams flow into the Glennamong River, which is a fourth-order river at the point of entry of the streams (Strahler, 1964). The study area is situated at an approximate elevation of 95 m above ordnance datum (AOD) and there is a moderate climate, which is heavily influenced by the proximity of the Atlantic Ocean. The average air temperature is 13 °C in summer and 4 °C in winter, while the mean annual rainfall for the catchment is 2000 mm (Rodgers et al., 2011). The catchment has a low buffering capacity and has been classified as acid oligotrophic (O'Driscoll et al., 2012). Blanket peat of varying depth down to 1 m covers the site, which overlays an Anaffrin formation of quartzite and schist bedrock (McConnell and Gatley, 2006). The blanket peat is an *in situ* blanket mire with an average gravimetric water content of 85% and a dry bulk density of approximately 0.1 g cm^{-3}. The organic matter content is greater than 91%, and the P, iron (Fe) and Al content of the peat layers down to a depth of 40 cm below the surface varies between 0.19 and 0.35, 0.94 – 1.31 and1.34 – 1.67 g kg^{-1} dry peat, respectively (Asam et al., 2012). The threshold phosphorus concentration (EPCo) above which net sorption occurs, was estimated by Asam et al. (2012) to be 28 μg L^{-1} in the humus layer (0 – 5 cm depth below the surface).

The site was planted with lodgepole pine (*Pinus contorta*) in 1972. The CC is at the same topographical location as the SC, and has a similar slope and peat depth. Clearfelling of the SC commenced on February 8, 2011. Bole-only clearfelling, which involves the removal of only the merchantable timber from site, leaving the branches and logging residue (brash material) to degrade on site, was carried out with a harvester (Timberjack 1470D) and a forwarder (8-wheeled, Timberjack 1110). A total of 14.8 ha of forest was clearfelled, of which 9.4 ha drained into the small stream in the SC. This stream has a mineral bed, no buffer strip due to its size, occasionally goes underground and is not identified on ordinance survey (OS) maps. Therefore, very little care was afforded to the stream during clearfelling, and occasionally brash mats were laid over the stream and parallel to the path of the stream. Operations continued during heavy rainfall and resulted in deep rutting (up to 1.5 m) on the main extraction routes. Timber was removed from the site *via* extraction racks running parallel to the slope of the site, and was deposited at a timber landing area adjacent to the road. Harvesting finished at the end of March 2011 and forwarding continued until the middle of April 2011. Temporary silt traps were installed on completion of forwarding and extra brash was placed on the rutted extraction routes for water management control. Three permanent silt traps, preceded upslope by settling ponds, were constructed with filter stone and geotherm at the end of April 2011. No drain cleaning took place on site and, to date, no maintenance of silt traps has been conducted. Windrowing (arranging the brash mats into piles) and replanting took place on the site in late 2013.

2.2. Best management practice

A comparison of the actual clearfelling practice at the study site, in comparison to BMPs (Forest Service, 2000a, b), is shown in Table 1. As far as practicable, clearfelling at the study site was carried out in accordance with BMPs, with harvesting plans, coupe size, timber landing areas, use of brash mats, site restoration and machine servicing being conducted. Road planning and construction was not necessary due to the existing road on site. A number of deviations in the implementation of BMPs were encountered due to the specific site conditions, which included lack of dry weather and avoidance of watercourses. The study site received a total rainfall of 5250 mm distributed over 625 rain days during the duration of the present study (February 2010 to May 2012; 821 days in total). As such, there were only 196 days (24% of study duration) on which clearfelling could take place in the absence of rainfall. In 2011 alone, the rainfall recorded in the SC was 3037 mm. During the clearfelling operations, which lasted approximately 80 days, there were 59 days of rainfall (387 mm in total), of which 44 were classed as wet days (measured rainfall greater than 1 mm). Due to time constraints and availability of the machines, clearfelling was conducted during poor weather conditions.

The forestry and water quality guidelines (Forest Service, 2000b) and the Code of Best Forest Practice – Ireland (Collins et al., 2000) stipulate the establishment of buffer zones along all aquatic zones, within which ground preperation and other forest operations are curtailed. They define an aquatic zone 'as a permanent or seasonal river, stream, or lake shown on an Ordnance Survey 6 inch map'. The stream draining the SC was not on the Ordnance Survey 6 inch map, as it was little more than a drainage channel and occasionally went underground. Therefore, a buffer zone was not established adjacent to the SC stream prior to planting. Upland spate streams are very characteristic of peat catchments in the west of Ireland, particularly within the Burrishoole catchment (Allott et al., 2005) and during periods of high rainfall, the SC stream carried large volumes of water (relative to the normal flow within the SC stream) from the catchment to the receiving river. Due to the sensitive nature of peatland sites, these small streams, despite their lack of order number, should be protected during clearfelling operations. In the present study, temporary silt traps were installed at the end of clearfelling, but this may have been too late to prevent SS export during the clearfelling process (Section 3.1.3).

2.3. Measurement and Analysis

Installation of H-flumes (or open channel flow nozzles) and water level recorders for flow measurement (OTT SE200, Germany), data sondes (Hydrolab, USA) for continuous measurement (every 5 minutes) of environmental parameters (DO, EC, pH and temperature) and ISCO samplers (Teledyne ISCO, USA) for stream water collection in the two streams in the CC and SC (identified as 'Sampling' in Figure 1) began in February 2010. The ISCO samplers were set to collect samples every hour, but occasionally it was set to take samples every two hours. The upper flow limit of the H-flumes was 148 L s^{-1}. The sondes were removed for calibration every 8-10 weeks. For analysis, the SC was divided into pre-clearfell (pre-CF) and post-clearfell (post-CF) periods. Pre-CF data collection took place from February 2010 to February 2011 (12 months pre-CF data) and post-CF data collection took place from February 2011 to May 2012 (15 months post-CF data). The nutrient and sediment release during a total of 18 storm events (n=8 pre-CF, n=2 during CF and n=8 post-CF; and n=24 samples within each storm event), over 24- or 48-hour time periods, were monitored using the ISCO samplers in the CC and SC streams (Figure 2). A weather station (Vantage Pro 2, Davis, USA) was positioned at the study site.

After collection, all water samples were returned to the laboratory and SS and nutrient analysis was carried out within 24 hr. Occasionally, some water samples were frozen at -20^{o}C and analysed for nutrients within 2 to 3 days of collection. The water quality parameters measured were: (1) dissolved reactive phosphorus (DRP) (2) total phosphorus (TP) (3) NH_4^+-N (4) total oxidised nitrogen (TON = NO_3^- + nitrate (NO_2^-)) and (5) SS. The SS component was further classified into organic suspended sediment (OSS) and mineral suspended sediment (MSS). All water samples were tested in accordance with standard methods (APHA, 2005) using a nutrient analyser (Konelab 20; Thermo Clinical Labsystems, Finland). Suspended sediment testing was carried out by passing a known volume of water through a pre-dried and weighed 1.2 μm GF/C filter disc (Whatman, England) under suction. The filter and retained sediment were then dried at 105 °C for 24 hr and reweighed to give the total SS (APHA, 2005). The MSS was determined by loss on ignition (LOI) at 550 °C (BSI, 1990). The OSS was calculated as the difference between total SS and MSS.

In order to determine the flow-weighted mean concentration (FWMC) of each nutrient for each storm event, it was first necessary to calculate the mass of nutrient lost during each sampling period. This was done by multiplying the concentration (mg L^{-1}) of nutrient in a sample by the total flow volume (L) measured in the stream over the sampling period. The sum of the mass release over the 24 samples (collected during the storm) was then divided by the sum of the flow in the stream over the sampling duration to give the FWMC. This allowed for comparisons, independent of flow, between the SC and CC to be conducted.

Storm events in the control and study catchments were monitored for discharge, sediment and nutrient concentrations. A storm event was defined as the time when the flow begins to increase on the rising limb of an event to the time when flow on the falling limb intercepts the base flow with a slope of 0.0055 L s^{-1} ha^{-1} hr^{-1} (Yusop et al., 2006). Yields of TP, SS and DRP were related to cumulative water discharge during these events by (Rodgers et al., 2010):

$$Y = \alpha Q + \beta \qquad [1]$$

where Y represents the total yield of a parameter of interest (g; measured using autosamplers), Q is the cumulative discharge of a storm event (L; measured in the H-flumes of the streams in the control and study catchments), and α (g L^{-1}) and β (g L^{-1}) are obtained by the least squares method. Using Eq. [1], the sediment and nutrient loads in kg per harvested/control areas were estimated over the study duration. During base flow conditions, the discharge and parameter concentrations were low, so these were omitted from the analyses. This assumption was reasonable, as Rodgers et al. (2010) found that over 80% of total reactive phosphorus (TRP) was released during storm events over a 4-year period.

With the exception of TON, which was found to better conform with the pre-requisites of the analysis if not transformed, the data were log transformed (when being used as a response variable) and analysed in R (version 3.0, 32 bit). (Note that where the size of an effect is quoted, the values have been converted back to the raw values, i.e. they are not on the log scale). Date (i.e. pre or post clear-felling), location of the sample site, SS, DRP and NH_4^+-N were included as explanatory variables. A general linear model was used to analyse the data. This method of analysis may be regarded as an extension of ANOVA and ANCOVA; it permits that significance and the extent of the effect of the various dependent variables on the response variable to be assessed independently. Of particular interest to this study, is that it allows the effects of clear-felling itself to be assessed, eliminating possible effects due to measurements being taken pre- or post the clear-felling date.

3. Results and Discussion

3.1. Nutrient and SS concentration

Throughout the entire study period, there was no significant difference between the pre-CF and post-CF nutrient concentrations in the CC, and prior to clear-felling, there was no significant difference between the CC and SC for any of the nutrient concentrations measured, with the exception of NH_4^+-N, which was higher in the study than the control site ($p < 0.001$), and was higher post the clear felling date than before that date (after the effect of clear-felling has been accounted for) ($p = 0.011$), and in TON, which was higher post the clear felling date than before that date (after the effects of clear-felling have been accounted for) ($p = 0.014$). A summary of nutrient concentrations post-CF is shown in Table 2 and a summary of the nutrient and SS loads exported from the CC and SS post-CF (as kg ha^{-1} yr^{-1}) is shown in Table 3.

3.1.1 Dissolved Reactive Phosphorus and Total Phosphorus

The DRP (Figure 3) was significantly higher ($p < 0.005$) in the SC than the CC due to clearfelling. Total phosphorus was not recorded prior to clearfelling, but was higher in the SC than the CC after clearfelling (Figure 4; $p < 0.005$). The limit for MRP, which is similar to DRP (Haygarth et al., 1997), for good status of surface water bodies is ≤ 35 µg L^{-1} (S.I. No. 272 of 2009). The FWMCs and peak concentrations of DRP pre-CF were well below this limit for both sites. Although the FWMC concentration from the post-CF SC only exceeded this limit on one of the ten sampling dates (39 µg L^{-1} P on October 31, 2011), seven of the eight sampling times post-CF had peak DRP concentrations in excess of 35 µg L^{-1} (peak DRP concentrations, usually only attained for an hour, ranged from 42 µg L^{-1} to 471 µg L^{-1}). Flow-weighted and peak concentrations of TP measured in the SC and CC streams in the period prior to the start of clearfelling in the SC were below the EPA critical threshold limit for TP of 62 µg L^{-1} (Coillte, 2008). During and after clearfelling of the SC, the flow-weighted and peak concentrations of TP exceeded this limit on six of the ten sampling dates, but returned below the critical limit for the final two sampling dates. Similar P concentrations were released from a similar sized catchment (20 ha) during the restoration (clearfelling of conifers followed by drain blocking) of a blanket bog in the southwest of Ireland (Coillte, 2008). Increases in DRP and TP of greater magnitude than the present study were measured after clearfelling of a 1-km^2 and a 1-ha peat catchment in the west of Ireland by Cummins and Farrell (2003a). They found that maximum (non-flow-weighted) concentrations of MRP increased from 9 µg L^{-1} (1 km^2 catchment) and 93 µg L^{-1} (1 ha catchment) to 256 µg L^{-1} and 3530 µg L^{-1}, respectively, within a few weeks of clearfelling, and the median values obtained were just over 100 µg L^{-1} (1 km^2 catchment) and 1000 µg L^{-1} (1 ha catchment). However, unlike the present study, which has mineral content in its stream bed, the stream and drain beds of the Cummins and Farrell (2003a) study consisted of purely peat-based matter and the flowing water had no interaction with mineral material, therefore giving little opportunity for adsorption of P to mineral layers. Rodgers et al. (2010) measured average FWMC of 14 ± 10 µg TP L^{-1} in the receiving waters, prior to the clearfelling of a peatland study site, using BMPs. A peak in the FWMC of TP of 201 µg L^{-1} was reached 5 weeks after the end of clearfelling, but this concentration had reduced back to pre-clearfelling concentrations 10 weeks after felling. The concentrations of P in the receiving waters can return to pre-clearfelling levels within 4 years of harvesting (Rodgers et al., 2010).

Exports of DRP, estimated using measured data and Eqn. [1], indicated that the DRP export from the study catchment was 0.4 kg ha^{-1} yr^{-1}versus 0.2 kg ha^{-1} yr^{-1} for the control catchment one year following clearfelling (Table 3). Similarly, TP exports were greater for the study catchment than the control catchment. The main mechanism of P release was not erosion (exports of SS were similar for both catchments were similar; Table 3), but was most likely P release from harvested residues. The poor P adsorption capacity of the peat meant that most of it was transported off site. The only other study to date that has quantified the export of P from forested blanket peat was Rodgers et al. (2010), who measured a total export load of approximately 5 kg total reactive phosphorus (TRP) kg ha^{-1} over a 4-year study period (Table 3) and attributed it to decomposing logging residues and poor P sorption capacity of the peat. No study has previously quantified export loads of DRP or TP from forested blanket peat sites.

The water extractable phosphorus (WEP) concentration, indicating the potential of a soil source to release P into runoff water, may be high under brash material (Finnegan et al., 2012), and is a function of the length of time brash is left on site and the time taken for regeneration of vegetation to occur (Macrae et al., 2005). The export of P post-CF is therefore linked to the amount and management of brash material on site. This P export is due to the poor adsorption capacity of peat (O'Driscoll et al., 2011) and fast (within one year after felling) and extensive (over 30% of P in brash material) mineralisation of P from the logging residues (Stevens et al., 1995). Phosphorus export from logging residues, spread evenly throughout the site, was also noted on a clearfell site in Finland, where the P leaching was as much as 17 times greater after clearfelling than before clearfelling (Piirainen et al., 2004). It is also common practice in Ireland to leave the brash mats across the site post-CF and arrange it into windrows once machinery is on site for reforestation, 1 ½ to 2 years after clearfelling (Collins et al., 2000). It was expected that the degradation of the extra brash placed on the rutted extraction routes for water management control would increase the dissolved P concentration in the stream post-CF in the SC, but this has not occurred to date.

3.1.2 Ammonium-Nitrogen and Total Oxidised Nitrogen

Whilst the NH_4^+-N (Figure 5) was significantly higher (p<0.005) in the SC than the CC after clear-felling, there is no evidence that this was due to the clear-felling itself (p>0.05), and may have been due to variation in levels between study and control sites, and variations pre and post the clear-felling date (see Section 3.1). There is, however, significant evidence of a mean increase in TON levels due to clear-felling (p<0.005) and further significant evidence of an increase in TON levels due to being post the clear-felling date (p=0.0148). The FWMC of NH_4^+-N and TON (Figure 6) in the CC and SC before clearfelling was below 0.1 mg L^{-1}. Peak concentrations of NH_4^+-N in the CC and SC, attained for a maximum of one hour and only on one sampling date pre-CF, were 0.18 and 0.26 mg L^{-1}. Post-CF, the FWMC of NH_4^+-N and TON rose to a maximum of 0.17 and 0.18 mg L^{-1}, respectively, and peak concentrations for NH_4^+-N and TON were 1.36 and 0.19 mg L^{-1}.

There are no critical limits for NH_4^+-N for river water bodies in Ireland. As a proxy value for NH_4^+-N, the critical limit for total ammonia (ionic-NH_4 + un-ionic NH_3) is used, which has a mean value of 0.065 mg L^{-1}, or 0.14 mg L^{-1} 95% of the time, for good status of river water bodies (S.I. No. 272 of 2009). The flow-weighted and peak concentrations of NH_4^+-N in the SC post-CF exceeded this value. The maximum threshold for NH_4^+-N in groundwater is 0.175 mg L^{-1} (S.I. No. 9 of 2010). Although the maximum FWMC of NH_4^+-N in the SC post-CF was below this threshold, peak concentrations exceeded this threshold on five of the eight sampling periods after clearfelling had finished.

Elevated levels of N are generally associated with forestry clearfelling (Nieminen, 1998; Cummins and Farrell, 2003b), but these increases normally do not occur until 1 year after clearfelling and may continue for up to 3 years (Cummins and Farrell, 2003b). Unlike P, initial high concentrations of N do not come from the degradation of brash material (Stevens et al., 1995). The delay in the release of N concentrations is due to the initial high N immobilization of the brash material, which has a high carbon (C):N ratio (Nieminen, 1998). The increase in N after clearfelling is a combination of the subsequent biological mineralisation of organic matter and the reduced uptake from biomass following the removal of the trees (Nieminen, 1998; Cummins and Farrell, 2003b).

Neal et al. (1999) noted that elevated levels of N post-CF on forestry sites across Britain was on a minority of sites, and leaching depended on local conditions. This was also noted by Kreutzweiser et al. (2008) in their review of logging impacts in Boreal regions. Ammonium-N has a high adsorption capacity to exchange sites, which retains it on site, therefore N release post-CF is generally in the form of NO_3^--N (Nieminen, 1998). The production of NO_3^--N is largely due to nitrification, which requires an aerobic zone, and is generally limited in peatland sites due to shallow watertables (Von Arnold et al., 2005). Consequently, N leaching is higher from nutrient-rich, well drained minerotrophic peatlands (Nieminen, 1998) than from the ombrotrophic peats found on the present study site. This could be a possible reason for the lower N export from the Glennamong catchments.

3.1.3 Suspended Sediment

Flow-weighted mean concentrations of SS in the SC increased only once in samples taken during clearfelling and returned to pre-CF levels, or below pre-CF levels, within 6 months of the commencement of clearfelling (Figure 7). This rise during clearfelling was not significant for SS, and there was no significant difference in date or location of sampling for OSS (Figure 8) or MSS (Figure 9). The total amount of SS exported from both sites were similar (approximately 200 kg ha^{-1} yr^{-1}; Table 3) over the year following clearfelling, which indicated that the majority of SS was exported soon after clearfelling began and there was no longer any material for subsequent transportation.

Large increases in SS were only noted during one storm, which occurred during the end of the clearfelling period in early April 2011 (Figure 10). Over a period of 12 hours, 18.4 mm of rain fell, producing an average flow in the stream of 21.3 L s^{-1} with an associated median SS concentration of 35.4 mg L^{-1} (the maximum release at the peak of the storm was 481 mg L^{-1} SS of which 68% was organic in nature). During this storm, the highest concentrations of SS and TP were measured during the increasing limb of the hydrograph, with lower concentrations been measured during the decreasing limb of the hydrograph. The highly mobile DRP appeared to be less dependent on increasing or decreasing limbs of the hydrograph. The recommended level of SS in salmonid waters is 25 mg L^{-1} (European Community, 1988), therefore the release at peak storm levels was over 19 times greater than the recommended level. Following installation of silt traps and extra brash placement on rutted extraction routes at the end of clearfelling, the FWMCs of SS returned to pre-CF levels or below pre-CF levels. However, peak concentrations of SS post-CF in the SS exceeded this threshold on six of the eight sampling occasions, when a maximum SS concentration of 63 mg L^{-1} was measured. Peak SS concentrations in the CC also exceeded the threshold limits on two of the sampling occasions post-CF, when a maximum concentration of 52 mg L^{-1} was measured. Similar patterns in SS concentrations were noted by Nieminen (2003) on a peatland clearfell site in southern Finland, with the only significant increase in SS coming from the most productive, highly fertile mire which was ditch-mounded in such a way that the ditches reached down into the fine textured mineral soil below the peat layer. Rodgers et al. (2011) also found that clearfelling, in line with BMPs, on a peat catchment did not result in a significant SS concentration increase after clearfelling although higher daily peak SS concentrations were observed. Furthermore, no adverse impacts on the receiving waters were noted in their study.

Increased sediment export after clearfelling, following implementation of BMPs, has been reported by other studies (Kirby et al., 1991; Ensign and Mallin, 2001; Aust and

Blinn, 2004; McBroom et al., 2008; Ryder et al., 2011). Variations in results can relate to different site slopes, weather conditions and the rate of vegetation growth post-CF (Rodgers et al., 2011). Higher rates of sediment loss are associated with steeper slopes (McBroom et al., 2008) and the rapid regeneration of vegetation within clearfelled areas can reduce SS export (Aust and Blinn, 2004). However, establishing ground vegetation can be slow on sites where brash material has not been removed (Broadmeadow and Nisbet, 2004).

3.2. Water parameters: DO, EC, pH and temperature

3.2.1 Dissolved Oxygen

Prior to clearfelling, DO levels at both sites were significantly different from each other ($p<0.05$), with the CC having significantly higher values (Figure 11). During clearfelling, the DO dropped to zero in the SC, and continued to fluctuate for up to one month after the end of felling. During this time the DO saturation was below the Irish EPA range for acceptable DO saturation (between 80% and 120% saturation; Bowman, 2009). The higher concentrations of OSS measured during clearfelling may also have affected the DO concentration within the receiving waters due to the organic component being biologically active and thus utilising oxygen during decomposition (Rodgers et al., 2011). Extra light to the stream, provided by the removal of the tree canopy, may also have enhanced algal blooms in the stream of the SC. A similar pattern was noted by Ensign and Mallin (2001) on a wetland clearfell site in the eastern US, which they attributed to an increased BOD load from logging residues and algal blooms.

3.2.2 Electrical Conductivity

There was no significant difference in the EC of both streams pre-CF, and the EC of the SC was generally above, or the same as, that of the CC. During periods of low flow or dry weather, the EC dropped to zero due to the sonde being exposed to the atmosphere (these values have been removed from the graphs for clarity). During clearfelling, the EC dropped in the SC and stayed below that of the CC for the remainder of the study (Figure 12; $p<0.05$). An identical pattern was found in the restoration of a blanket bog in the southwest of Ireland (Coillte, 2008), and Cummins and Farrell (2003a) found, on a clearfelled peatland site, that values of EC reduced after clearfelling.

3.2.3 pH

The pH was consistently higher in the CC than the SC pre-CF. By the end of clearfelling, this pattern swapped, with the SC having a higher pH (Figure 13). The pH measured in a stream during restoration (clearfelling of conifers followed by drain blocking) of a blanket bog in the southwest of Ireland (Coillte, 2008) varied from 7.5 during low flow to approximately 4.3 during peak storm events, which is characteristic of acid sensitive blanket bogs. Similarly, in the present study, the initial high pH (seen at the end of October 2010) and the observed peaks in April 2011 followed dry periods when the pH was elevated due to more interaction with the bedrock in the stream. Rodgers et al. (2008) attributed the higher values of pH during low flow to a greater residence time within their study site, and interaction with an aquifer located above their sampling point.

There are few other long-term data in Ireland on changes in pH levels following harvesting (Johnson et al., 2008). Long-term studies in the UK (Neal et al., 1992) all show a slight decrease, or no change, in the pH after clearfelling. Dissimilar to these studies, Cummins and Farrell (2003b) observed an elevated pH immediately after clearfelling on a peatland site, and attributed this to the road side location of the sampling point which may have allowed dust, caused by road works or increased traffic, to enter water samples. However, elevated pH levels have been reported by other researchers (Ryder et al., 2011) on peatland sites which were not attributable to the roadside location of the sampling point. The increase in pH post-CF could be due to the decomposition of brash material on site (Staaf and Olsson, 1991), which allowed the return of base cations to the soil (Thiffault et al., 2011).

3.2.4 Temperature

The temperature of the stream water on both sites pre-CF was not significantly different from each other, and both sites responded well to the air temperature changes. Post-CF saw a significant rise in the stream water temperature in the SC and was likely due to the removal of the tree canopy, and more light and solar radiation entering the stream (Rodgers et al., 2008) (Figure 14). However, this rise in temperature was not significant ($p=0.0725$). A rise in stream water temperature was also noted by Stott and Marks (2000) in a forest clearfell study of a similar size (20 ha) on a peaty gley catchment in mid-Wales, and by Rodgers at al. (2008) in a clearfell study in Ireland. Changes to stream water temperature impacts most on the aquatic fauna of a waterbody (Mellina et al., 2002), and studies have shown results ranging from little recovery of invertebrates after clearfelling (Gee and Smith, 1997) to an increase in the number of mayflies (Kirby et al., 1991). The influence of the increase in stream temperature on the aquatic fauna of the Glennamong catchment was not investigated in the current study.

3.3. Outlook for implementation of best management practices

Best management practices in clearfelling operations, as recommended by the forest management organisation in Ireland (Coillte) and the Forest Service guidelines (Forest Service, 2000a,b,c,d,e,f), were generally followed in this study. The results of this study indicated that BMPs were not effective in reducing P loads from the clearfelled site. The release of TP and DRP from the clearfelled site was not mitigated by any BMPs employed, and was most likely released from decaying forest residues and poor soil P adsorption. Although the implementation of BMPs in forestry clearfelling has been shown to be effective at decreasing non-point source pollution to receiving watercourses (Ensign and Mallin, 2001; Wallbrink and Croke, 2002; Aust and Blinn, 2004; Johnston et al., 2008), none of these studies have attempted to estimate the export of nutrients or SS off site in terms of kg ha^{-1}. Suspended solids concentrations increased during clearfelling, but quickly reduced. This may have been more to do with the lack of easily erodible material than the efficacy of BMPs employed.

Whole tree harvesting (WTH) may reduce the export of nutrients from harvested sites, but this technique leads to the removal of base cations and may have consequences for future rotations (Nisbet et al., 1997). In addition, WTH may further compound the acidification of peatland forested catchments (Ågren and Löfgren, 2012) and, therefore, is unadvisable in the acid sensitive catchments of the west of Ireland. The leaching of cations from degrading foliage may reverse the effect of acidification in low N-releasing sites (Neal et al., 1999). Nutrient export from nutrient-poor peat, similar to that in the current study, is less likely than from highly productive mires (Nieminen, 2003).

4. Conclusions

The hypothesis of this study was that BMPs are effective in mitigating the transport of nutrients and sediment off site in clearfelled forested peatlands. Following clearfelling, DRP and TP concentrations rose in the clearfelled site and loads released were greater than the control (unharvested) site. This indicated that BMPs may not have been effective in reducing P releases from the clearfelled site. Most of the SS was released soon after clearfelling began and any reductions measured may not necessarily have been due to BMPs than the lack of easily erodible material on site. Site-specific parameters, such as the depth of peat or the slope of a site, and other potential confounding factors, such as the time of felling and weather conditions at the time of felling, may impact on nutrient and sediment release rates, and cognisance should be taken of these factors when drafting a harvest plan.

As recommended in the BMPs, a site should be thoroughly inspected prior to clearfelling. However, this should take place during, or immediately after, a period of prolonged rainfall. In the present study, a stream draining the study site, not identified on an Ordinance Survey 6-inch map and not visible during a site inspection which took place in dry weather, carried large volumes of water from the catchment to the receiving waterbody during adverse weather conditions.

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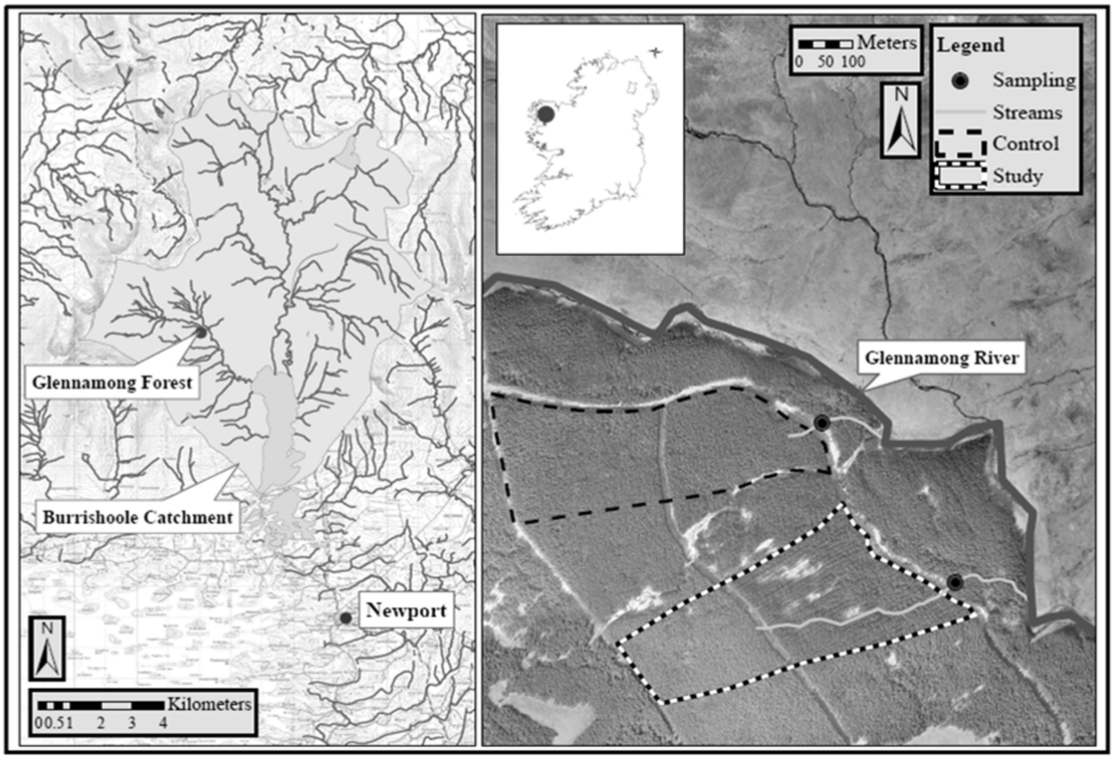

Figure 1: Location of the Glennamong Forest control catchment (CC) and study catchment (SC).

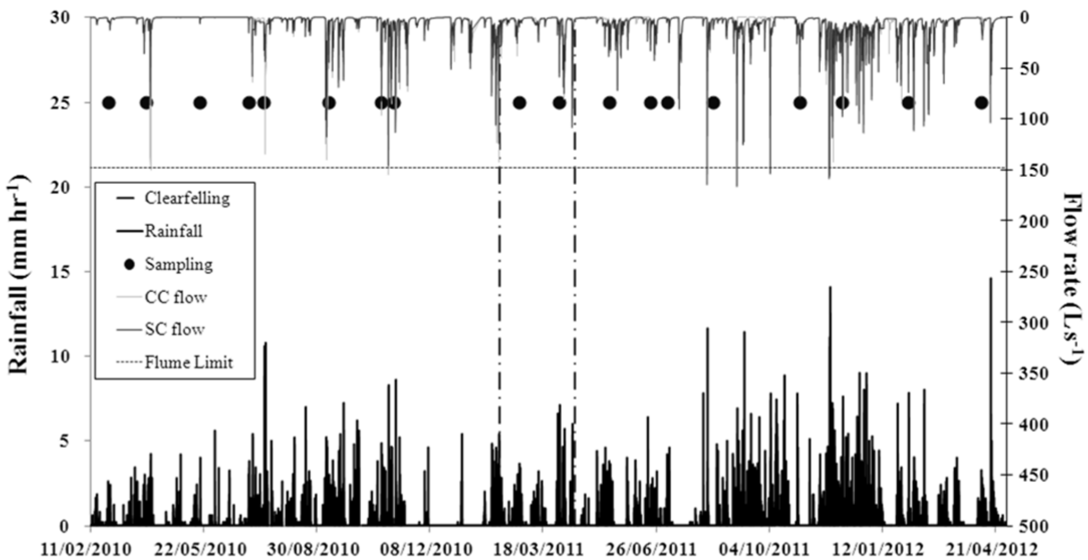

Figure 2. Rainfall (mm hr^{-1}) from the Glennamong weather station and stream water sampling dates from February 2010 to May 2012. Flow rates (L s^{-1}) from the control catchment (CC) and study catchment (SC) are on the inverted secondary axis.

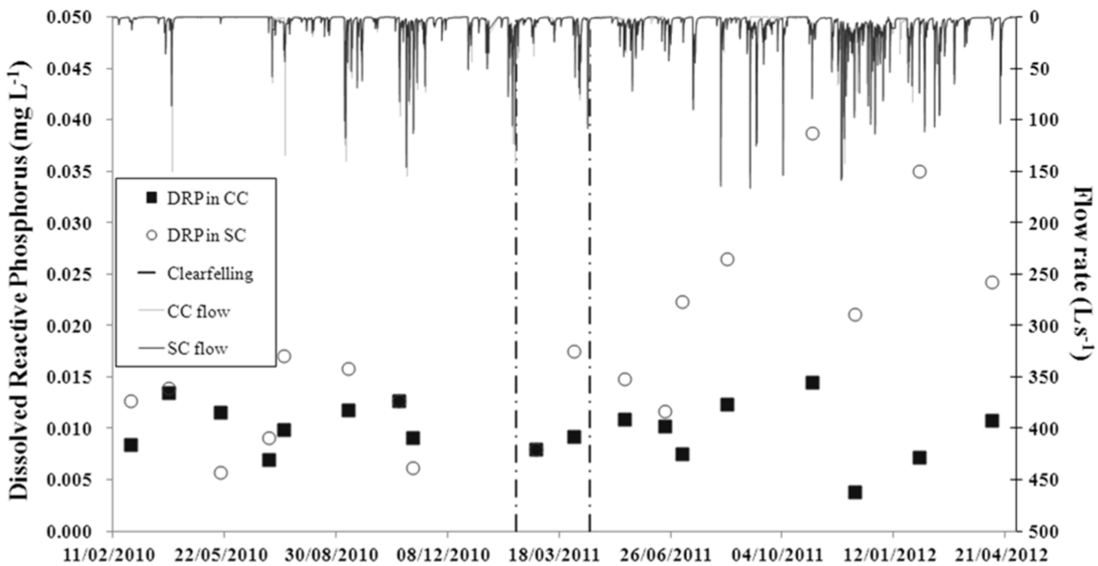

Figure 3. Flow-weighted mean concentrations of dissolved reactive phosphorus (DRP) (mg L^{-1}) measured in the control catchment (CC) and the study catchment (SC) from February 2010 to May 2012. Flow rate (L s^{-1}) is on the inverted secondary axis.

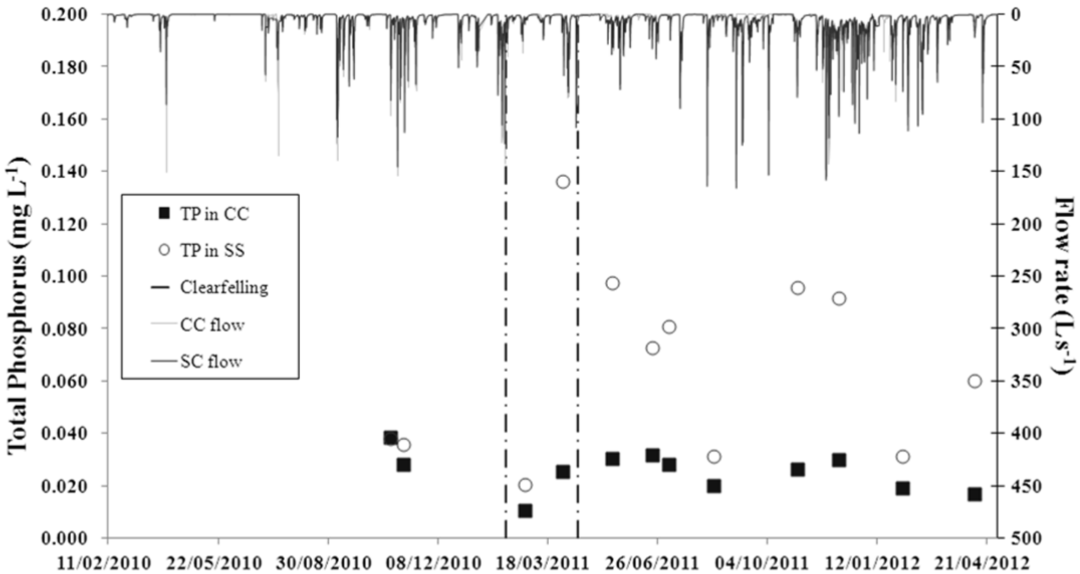

Figure 4. Flow-weighted mean concentrations of total phosphorus (TP) (mg L^{-1}) measured in the control catchment (CC) and the study catchment (SC) from February 2010 to May 2012. Flow rate (L s^{-1}) is on the inverted secondary axis.

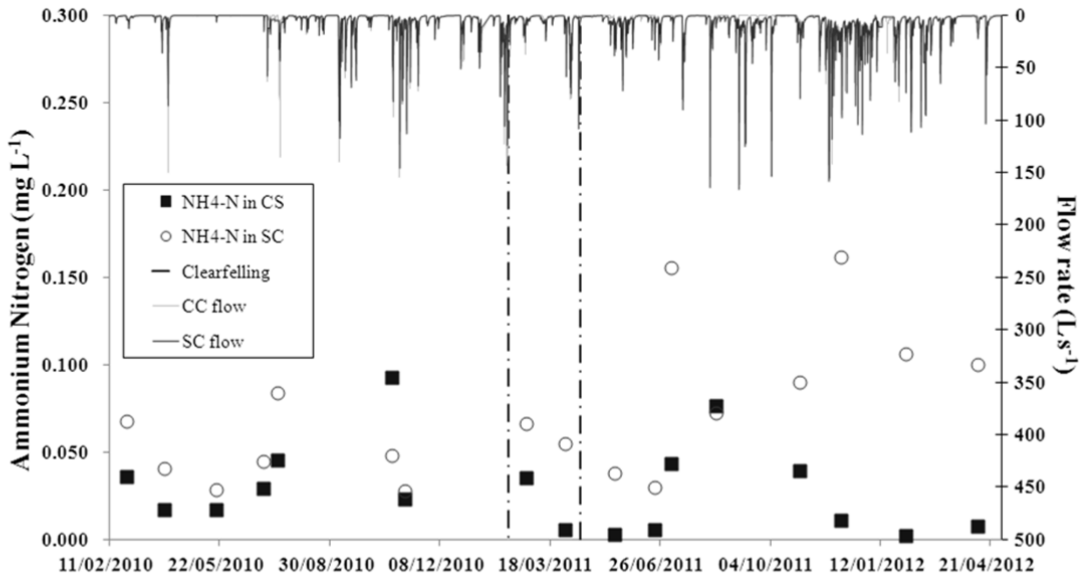

Figure 5. Flow-weighted mean concentrations of ammonium-nitrogen (NH_4^+-N) (mg L^{-1}) measured in the control catchment (CC) and the study catchment (SC) from February 2010 to May 2012. Flow rate (L s^{-1}) is on the inverted secondary axis.

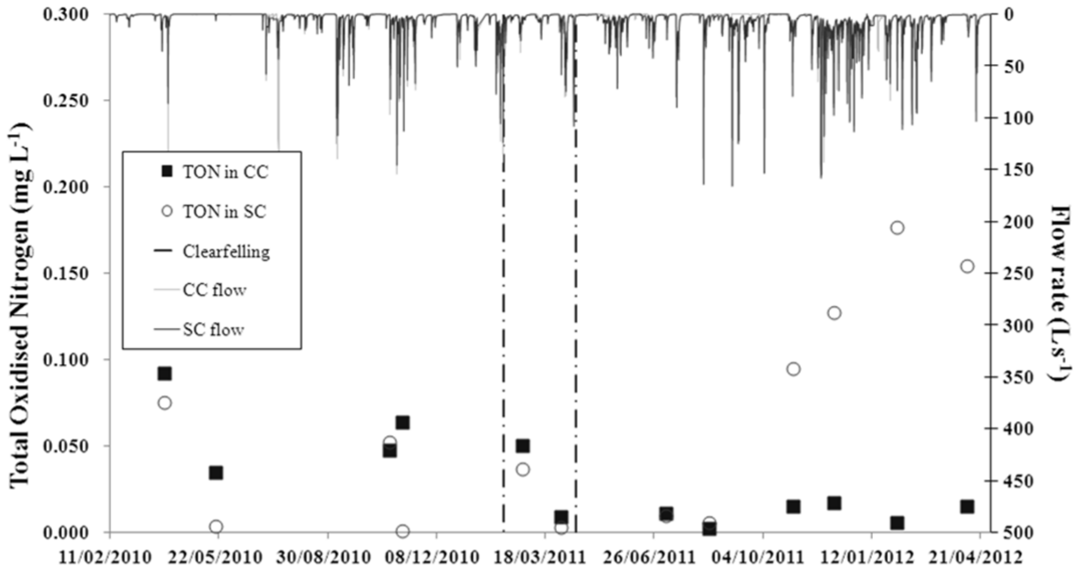

Figure 6. Flow-weighted mean concentrations of total oxidised nitrogen (TON) (mg L^{-1}) measured in the control catchment (CC) and the study catchment (SC) from February 2010 to May 2012. Flow rate (L s^{-1}) is on the inverted secondary axis.

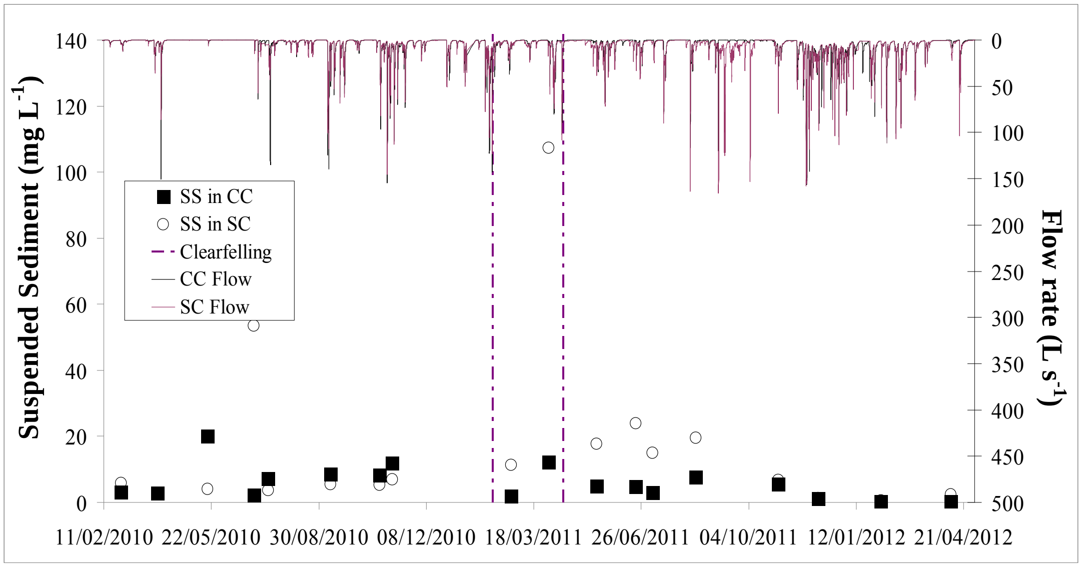

Figure 7. Flow-weighted mean concentrations of suspended sediment (SS) (mg L^{-1}) measured in the control catchment (CC) and the study catchment (SC) from February 2010 to May 2012. Flow rate (L s^{-1}) is on the inverted secondary axis.

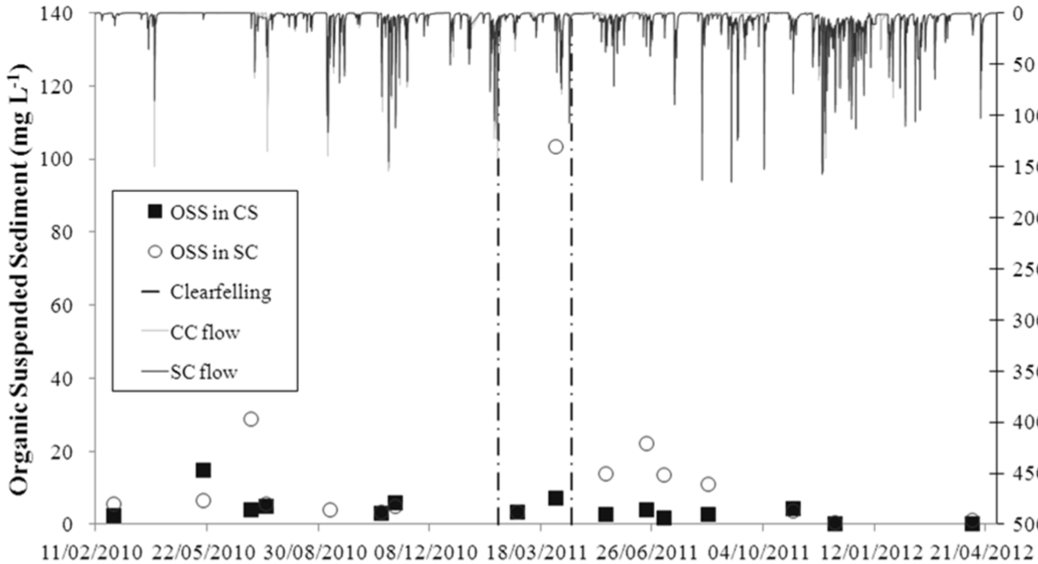

Figure 8. Flow-weighted mean concentrations of organic suspended sediment (OSS) (mg L^{-1}) measured in the control catchment (CC) and the study catchment (SC) from February 2010 to May 2012. Flow rate (L s^{-1}) is on the inverted secondary axis.

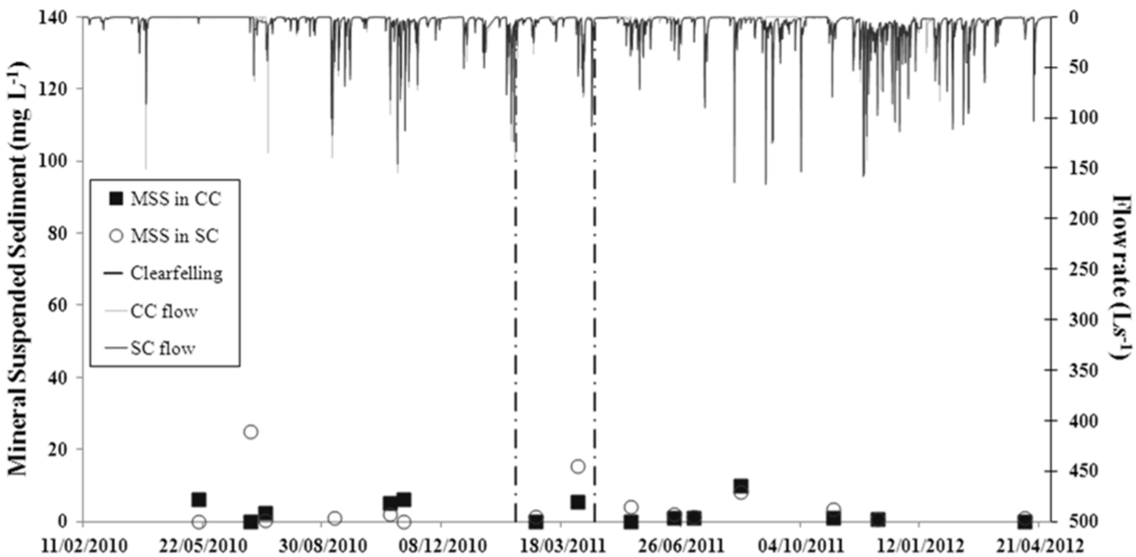

Figure 9. Flow-weighted mean concentrations of mineral suspended sediment (MSS) (mg L^{-1}) measured in the control catchment (CC) and the study catchment (SC) from February 2010 to May 2012. Flow rate (L s^{-1}) is on the inverted secondary axis.

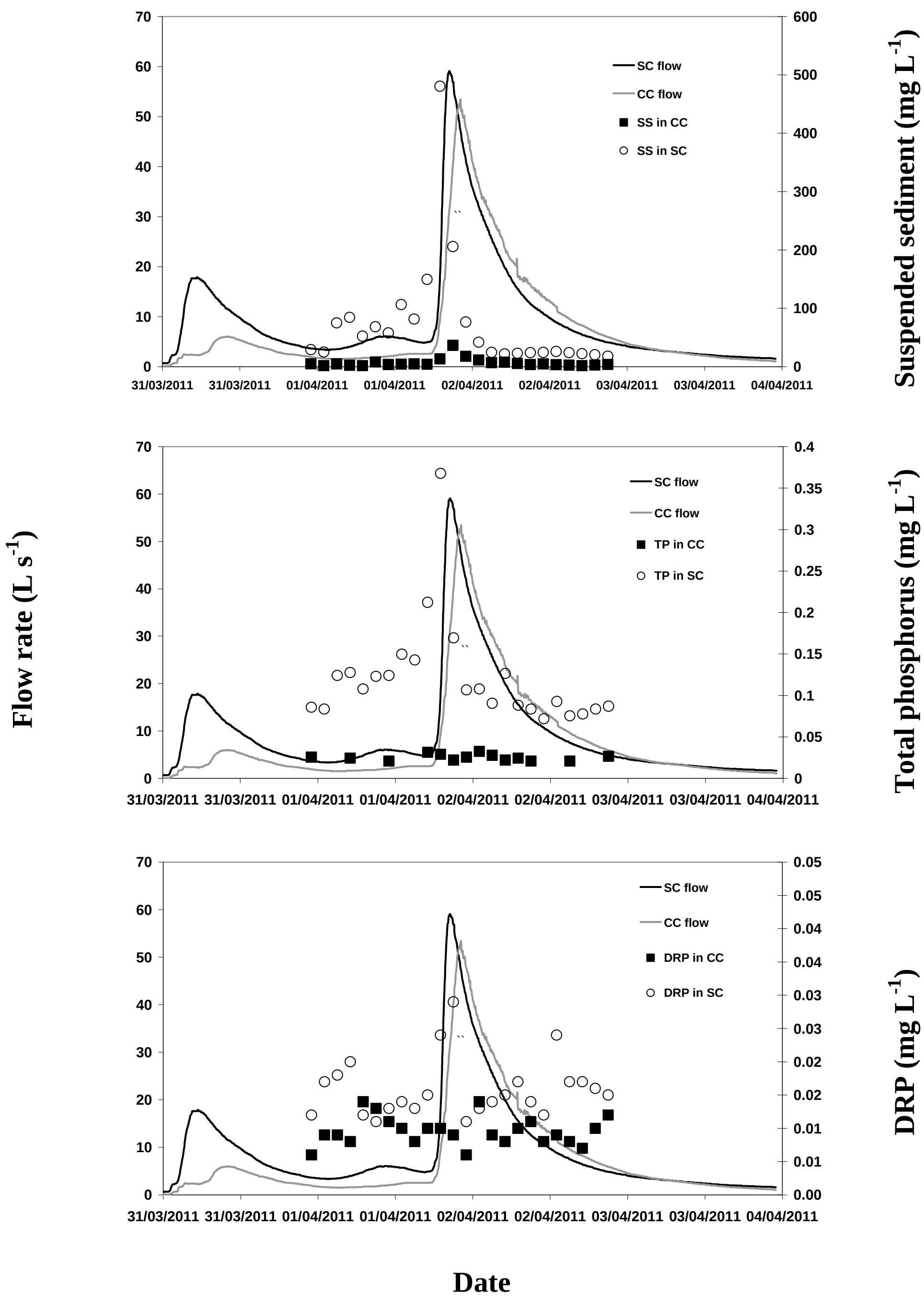

Figure 10. Suspended solids (top), total phosphorus (middle) and dissolved reactive phosphorus (DRP; bottom) release in a storm event towards the end of clearfelling.

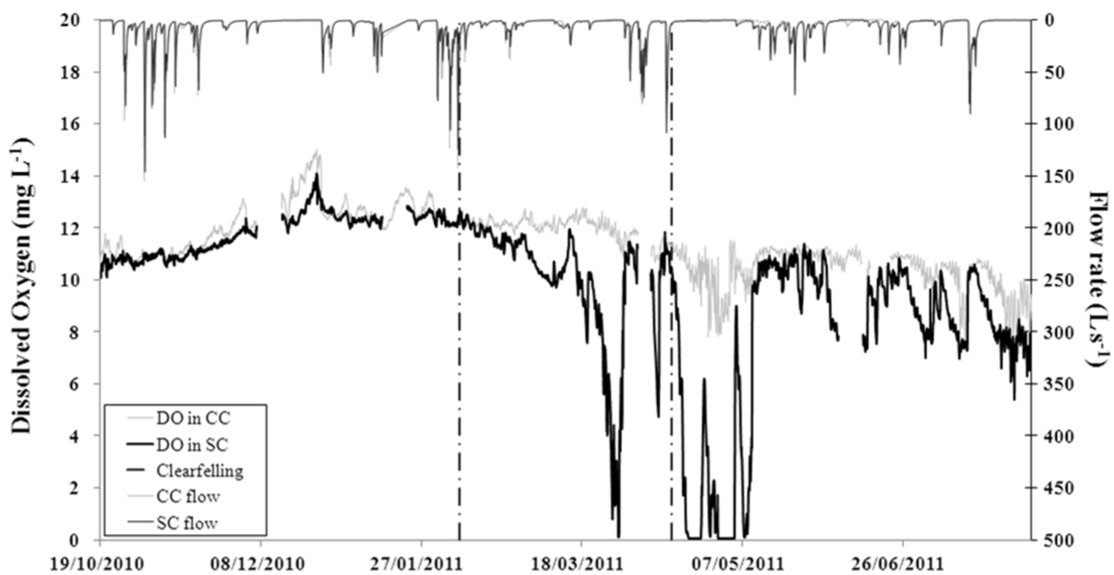

Figure 11. Dissolved oxygen (DO) (mg L^{-1}) at 5-minute intervals measured in the control catchment (CC) and the study catchment (SC) from October 2010 to July 2011. Flow rate (L s^{-1}) is on the inverted secondary axis.

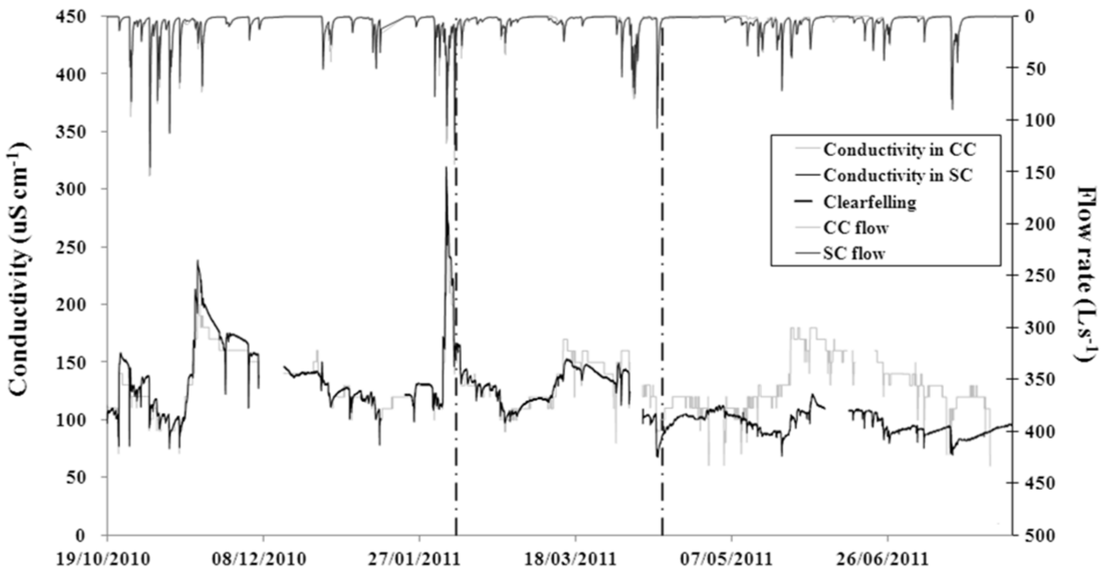

Figure 12. Electrical conductivity (EC) (µS cm^{-1}) at 5-minute intervals measured in the control catchment (CC) and the study catchment (SC) from October 2010 to July 2011. Flow rate (L s^{-1}) is on the inverted secondary axis.

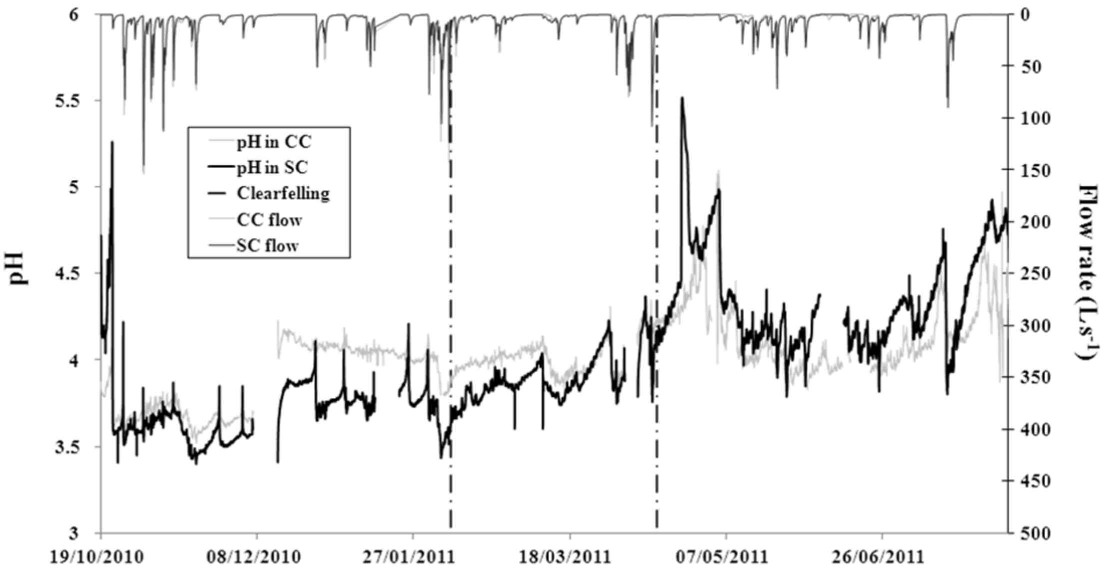

Figure 13. pH at 5-minute intervals measured in the control catchment (CC) and the study catchment (SC) from October 2010 to July 2011. Flow rate (L s^{-1}) is on the inverted secondary axis.

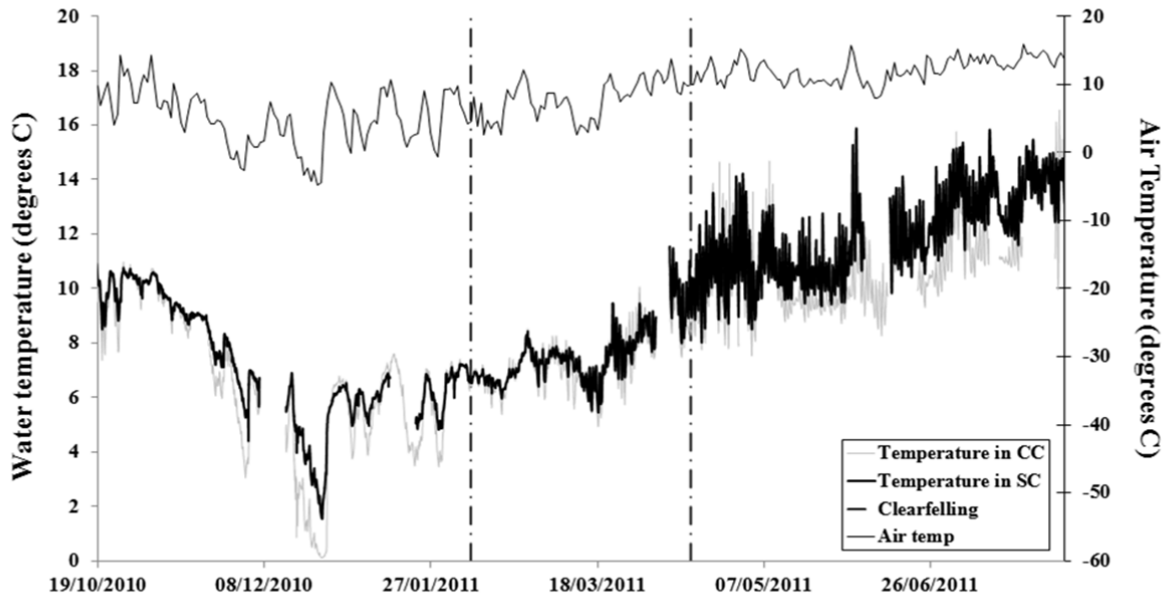

Figure 14. Stream water temperatures (°C) at 5-minute intervals measured in the control catchment (CC) and the study catchment (SC) from October 2010 to July 2011. Air temperatures (°C) from the weather station are on the inverted secondary axis.

Table 1: Best management practices (BMPs) from 'Forest Harvesting and the Environmental Guidelines' (Forest Service, 2000c) and 'Forest and Water Quality Guidelines' (Forest Service, 2000a) with applied BMPs at the Glennamong study site.

Best Management Practice	Compliance (Yes / No)	Comments
Harvest planning		
• Establish relevant environmental issues and liaise with authorities	Yes	
• Terrain inspection and draft harvest plan for size and shape of felling coupe	Yes	• Terrain inspection and harvest plan drafted with appropriate felling size and shape
• Felling sequence and contingency plan	Yes	• Felling sequence followed as per plan
• Equipment to be used and structures required	Yes	
Harvest operation		
• Adequate brash mats to limit damage to soil from heavy machinery	No	• Use of brash mats, but rutting occurred due to heavy rainfall and lack of maintenance
• Installation of ancillary structures and provision of buffer zones to watercourses	No	• Temporary silt traps installed but only at end of clearfelling, so SS was released during clearfelling
• Limit load size	Yes	
• Prevent accumulation of brash in drains and aquatic zones	No	• Brash allowed to gather in stream on site
• Establish new buffer zones at end of clearfelling operations and clean drains	No	• No cleaning of brash from stream in SC post-CF due to a risk of increased sediment
• Consider suspending operations during periods of heavy rain	No	• No suspension of clearfelling during wet weather due to time constraints
Harvest site restoration		
• Repair to road and drains	N/A	• Road repair was not necessary and brash was removed from road drains
• Remove temporary structures and install permanent ones if necessary	Yes	• Permanent silt traps installed
• Remove hazardous compounds	Yes	• All logging equipment was removed from site
• Carry out water management on extraction routes	Yes	• Extra brash placed on rutted areas on extraction routes
Road planning	N/A	• Not necessary
Road construction	N/A	• Not necessary
Machine servicing		
• Storage of materials and maintenance and refuelling away from watercourses (min 50 m)	Yes	• Servicing and maintenance away from watercourses, and any spillages were cleaned with pollution control kits

Table 2: Maximum concentrations ($\mu g\ L^{-1}$) pre- and post- clearfelling for dissolved reactive phosphorus (DRP), total phosphorus (TP), total oxidised nitrogen (TON) and ammonium-nitrogen (NH_4^+-N) from the current study site and comparable study sites worldwide.

Reference	Location	Area of CF (ha)	Type of harvesting	Soil type	Average Annual Rainfall	Max concentrations pre-clearfelling ($\mu g\ L^{-1}$)				Max concentrations post-clearfelling ($\mu g\ L^{-1}$)			
						TP	DRP	TON	NH_4-N	TP	DRP	TON	NH_4-N
Cummins and Farrell (2003 a, b)	Galway, Ireland	1	Bole only clearfelling	Peat	1600	-	13[a]	≈ 400[b]	≈ 300	-	4164[a]	2500[b]	≈ 1800
Ensign and Mallin (2001)	Northern Carolina, USA	52.6	Clearcut with track cutter and shovel logger	Swamp soils	1270	188	47[a]	581[b]	146	427	297[a]	191[b]	440
Neal (2004)	Plynlimon, Mid-Wales	< 1	Bole only clearfelling	Peaty gley	2500	-	30[a]	-	160	-	550[a]	-	1120
Nieminen (2003)	Southern Finland	7	Bole only clearfelling	Peat	600	-	< 10[c]	< 20[b]	< 25	-	100[c]	< 20[b]	< 15
Rodgers et al. (2010)[d]	Mayo, Ireland	14.5	Bole only clearfelling	Peat	2000	28	-	-	-	201	-	-	-
Present study	Mayo, Ireland	9.4	Bole only clearfelling	Peat	3000	80	33	128	182	611	471	194	1336

[a] measured as molydbate reactive phosphorus in these studies.

[b] measured as NO_3^--N by ion chromatography in these studies.

[c] measured as PO_4^{3-}-P by ion chromatography in these studies.

[d] reported as flow-weighted mean concentrations.

Table 3: Comparison of nutrient and sediment loads exiting harvested peat areas.

Reference	Location	Area of CF (ha)	Duration of study after felling (yr)	Type of harvesting	BMP used?	Exports from site (kg ha^{-1} yr^{-1})							
						Control (unharvested)				Study (harvested)			
						Tot-P	DRP	TRP	SS	Tot-P	DRP	TRP	SS
Rodgers et al. (2010[a])	Mayo, Ireland	14.5	4	Bole only clearfelling	Yes			< 0.06				≈ 1.3	
Present study	Mayo, Ireland	10	1	Bole only clearfelling	Yes	0.6	0.2		202	0.9	0.4		200

[a] Total reactive phosphorus (TRP) loads peaked in the second year (2.3 kg ha^{-1}), but decreased in the third and fourth years of their study. Value quoted is study average.