ASSESSMENT AND MITIGATION OF FOREST CLEARFELLING IMPACTS ON SALMONID RECEIVING WATERS

by

Connie O'Driscoll

Supervisor:

Dr. Liwen Xiao, Civil Engineering, National University of Ireland, Galway, Ireland

Co-supervisors:

Dr. Elvira de Eyto, Marine Institute, Newport, Co. Mayo, Ireland

Dr. Mark Healy, Civil Engineering, National University of Ireland, Galway, Ireland

Professor of Civil Engineering: **Padraic E. O'Donoghue**

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Civil Engineering, College of Engineering and Informatics, National University of Ireland, Galway, Ireland

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Abstract

It was estimated that about 500,000 hectare (ha) of peatland was afforested between the 1950s and 1990s in the UK and 300,000 ha in Ireland. Many of these blanket peat forests are now reaching harvestable age and concerns have been raised about the potential release of phosphorus (P) to the receiving aquatic systems as a result of harvesting. These areas contain the headwaters of rivers, many of which contain Red List species (e.g. salmonids and freshwater pearl mussels), which make them important biodiversity refuges. Despite the fact that the sensitivity of clearfelling upland peat catchments has risen to prominence in recent years in terms of economic and conservational viability, sustainable protection methods are poorly researched and proven. The objectives of this study are to investigate the impacts of forestry clearfelling on the ecology and flow regime of receiving waters, and to assess the performance of buffer zones, phased felling, brash removal and a novel grass seeding method on ameliorating any negative clearfelling impacts. The study was based in the Burrishoole Catchment, Newport, Co. Mayo.

Hydrological, physical, chemical and biological parameters, including rainfall, stream flow rate, pH, temperature, dissolved oxygen (DO), electrical conductivities (EC), P, nitrogen (N), suspended solids (SS), macroinvertebrates and diatoms, were monitored for two years before and one year after clearfelling took place, in two sub-catchments. The results indicated that with the implementation of best management practices (BMPs), peatland forest harvesting activities could (1) have no significant impact on SS concentration in the receiving water; (2) increase catchment water yield, but not increase flood risk; (3) increase P and N concentrations in the study streams; and (4) affect the macroinvertebrate and diatom assemblages in the rivers.

Buffer zones (BZs) have been recommended internationally as a mitigation measure for tackling pollution sources and transport. However, large areas of upland blanket peat were afforested in the UK and Ireland before the importance of the riparian buffer areas was realised. In order to reduce the possible negative impact of harvesting activities on receiving waterbodies, the creation of BZs along receiving water courses prior to the clear-felling of the main plantation has been proposed. In this study, a small BZ, with the effective area of about 0.1 ha, was established and seeded with native grass species, onto which runoff from an upstream forest, with an area of

about 10 ha, was spread. One year later, the upstream forest was harvested. The results indicated that the BZ removed 45.3 % of SS, 33.7 % of TON and 17.6 % of total reactive phosphorus (TRP), respectively, in the first year of harvesting.

To reduce nutrient leaching from forest catchments to receiving water, a novel practice – grass seeding clearfelled areas immediately after harvesting – was proposed in this study. It was hypothesised that if the vegetation could quickly recover after forest harvesting, the nutrients would be retained *in situ* through vegetation uptake. A field trial was conducted to identify the successful native grass species that could grow quickly in the recently clearfelled blanket peat forest. The two successful grass species, *Holcus lanatus* and *Agrostis capillaris*, were sown in three harvested blanket peat forest study plots with areas of 100 m^2, 360 m^2 and 660 m^2 immediately after harvesting. Areas without grass seeding were used as controls. One year later, the P contents in the above ground vegetation biomass of the three respective study plots were 2.83 kg P ha^{-1}, 0.65 kg P ha^{-1} and 3.07 kg P ha^{-1}. These values were significantly higher than the value of 0.02 kg P ha^{-1} observed in the control plots. The average concentrations of water extractable phosphorus (WEP) in the three study plots were 8.44 mg (kg dry soil)$^{-1}$, 9.83 mg (kg dry soil)$^{-1}$ and 6.04 mg (kg dry soil)$^{-1}$, respectively, which were lower than the value of 25.72 mg (kg dry soil)$^{-1}$ in the control sites. These results indicate that grass seeding of the peatland immediately after harvesting can quickly immobilise significant amounts of P and warrants additional research as a new BMP following harvesting in the blanket peatland forest to mitigate P release.

To further examine the grass seeding practice, experimental plots with defined boundary conditions were established. In addition, other mitigation approaches, such as whole tree harvesting, were also tested using these plots. Three sets of five treatments were compared as follows: (1) no brash and no seeded grass; (2) brash without seeded grass; (3) brash with seeded grass; (4) seeded grass only and (5) brash mats. The results indicated that (1) the brash mat was a significant source of nutrient release; (2) whole tree harvesting could significantly reduce nutrient release, and (3) grass seeding could be a sustainable practice for nutrient release control after forest harvesting.

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Table of Abbreviations

ACID	Acid Index for Diatoms
AFDM	Ash Free Dry Mass
ANOVA	Analysis Of Variance
BM	Brash Mats
BMP	Best Management Practices
BZ	Buffer Zone
BZS	Buffer Zone Station downstream of buffer zone
CA	Correspondence Analysis
CCA	Canonical Correspondence Analysis
CTL	Cut To Length
DCA	Detrended Correspondence Analysis
DO	Dissolved Oxygen
DOC	Dissolved Organic Carbon
DRP	Dissolved Reactive Phosphorus
DS	Downstream Station
DSC	Downstream Station of the Confluence
DSF	Number of Days Since the last Flood
EC	Electrical conductivities
ECN	Environmental Change Protocols
EPT	Ephemeroptera, Plecoptera, Trichoptera
EQR	Ecological Quality Ratio
GMP	Ground Mineral Phosphate
GSS	Glennamong Study Station upstream of buffer zone
ha	Hectare
IBD	Indice Biologique Diatomique
L	Lower Sites
LSD	Least Significant Difference
M	Middle Sites

MBACI	Multiple Before After Control Impact design
MRP	Molybdate Reactive Phosphorus
MU	Mid-upper Sites
N	Nitrogen
NGSM	Native Grass Seeding Method
NH^4-N	Ammonia
P	Phosphorus
PO^4-P	Orthophosphate
SAC	Special Area of Conservation
SI	Acid Index for Macroinvertebrates
SPA	Special Protection Areas
SS	Suspended Solids
SSSI	Sites of Specific Scientific Interest
TDI	Trophic Diatom Index
TI	Trophienindex
TON	Total Oxidised Nitrogen
TRP	Total Reactive Phosphorus
UP	Upper Sites
US	Upstream Station
USC	Upstream Station of the Confluence
WEP	Water Extractable Phosphorus
WFD	Water Framework Directive

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Chapter One

Introduction

1.1 Background

The Water Framework Directive (WFD) (2000/60/EC) requires EU Member States to achieve 'good ecological status' for all water bodies by 2015 (European Union, 2000). Assessing, maintaining and restoring good ecological status of aquatic ecosystems have become priorities for river basin management and water protection in Europe (Kelly and Wilson, 2004; Leira and Sabater, 2005; Johnson et al., 2007; Kelly et al., 2008; Urrea and Sabater, 2009). In implementing the WFD, member states are required to carry out risk assessments, and devise appropriate cost effective mitigation measures against identified pressures (European Union, 2000). Risk assessments on receiving waters have shown that forest operations can result in increased soil and nutrient release (Nisbet, 2001; Cummins and Farrell, 2003; Nieminen, 2003; Rodgers et al., 2010, 2011), increased acidity (Jenkins et al., 1990; Ormerod et al., 1991; Allott et al., 1997; Hutton et al., 2007); and peat degradation, leading to increased dissolved organic carbon (DOC) export, decreased pH in receiving waters, and altered flow regimes (Fealy et al., 2010; Cantonati and Lange-Bertalot, 2011). This poses a risk to the ecological status of receiving waters.

The forest industry is continuously examining the cost effectiveness of forestry on peatlands, in terms of: timber quality, harvesting and road building costs, along with the requirement of very careful management practices that have to be put in place to safeguard environmentally sensitive areas such as upland peat catchments. These areas contain the headwaters of rivers, many of which contain Red List species (e.g. salmonids and freshwater pearl mussels), which make them important biodiversity refuges. Due to the ecological sensitivity of upland blanket peat catchments, current BMPs need to be monitored and assessed through the measurement of forestry activity impacts. Neal et al. (2003) recommended that a multi-disciplinary approach toward scientific research is needed incorporating ecology, hydrology, hydrochemistry and fluvial geomorphology in the development of guidelines for forestry activities.

1.2 Literature Review

1.2.1 Forest harvesting

For forestry harvesting to be sustainable on sensitive sites, mechanised machine harvesting is essential, and central to this is the practice known as CTL (Cut-To-Length). Cut-To-Length harvesting is carried out initially by a harvester machine that fells the tree, removing the crown and associated tree residues (i.e. needles, twigs and branches) and slicing the bole into pre-determined lengths. Some of the tree residues (i.e. needles, twigs and branches) are collected together to form brash material mats (BMs), thus protecting the soil surface, and reducing erosion. The remainder is left on the soil surface and collected together to form windrows for reforestation purposes. The BMs are approximately 4 m wide. The distance between two BMs is about 12 m. During harvesting, the boles are stacked beside the windrow for collection. A forwarder machine collects the timber and carries it to a log stacking point alongside the road. The majority of disturbance to the peat floor is caused by the forwarder, as it is the main traffic source in the operation, travelling up and down a site many times on a main trail leading up to the stacking point. Rutting, where it occurs, can be a major source of disturbance especially when it channels surface water and enters the water course.

1.2.2 Impacts of forest harvesting

It is widely accepted that forest harvesting can influence the hydrology of a catchment (Bosche and Hewlett, 1982). Studies from the Western Cascades, USA, reported an increase in peak flows (Thomas and Megahan, 1998; Beschta et al., 2000) following forest harvesting. In southern British Columbia, Canada, significant increases in annual and monthly water yields and annual peak flows were observed following harvesting (Cheng, 1989). Bosche and Hewlett (1982) reviewed results from 94 experimental catchments and concluded that stream flow response to harvesting is dependent on climate and, in particular, precipitation. Similarly, Robinson et al. (2003) considered 28 catchments across Europe and concluded that, even within Europe immense variations exist between countries in terms of climate, tree species, and catchment physiography. Bruijnzeel (1988) indicated that stream base flow increase or decrease after harvesting depends on the surface infiltration and evapo-transpiration rates of the catchment. With shallow water tables and low hydraulic conductivity at depth, blanket peatlands tend to generate rapid runoff and have shorter lag times and higher peak flows in response to rainfall events (Evans et al., 1999; Müller, 2000; Rosa and Larocque, 2008; Grayson et al., 2010). Robinson et al. (2003) found commercial conifer plantations on peaty soils in north-west Europe were associated with short-term increases in peak flows and baseflows at the local scale, but were not detectable at larger catchment scale. Clearfelling in the Plynlimon catchments, mid-Wales, demonstrated that adhering to forest BMPs can result in no impact on storm peak flows (Robinson and Dupreyat, 2005). Hydrological analyses in a harvesting study in the Burrishoole Catchment, Co. Mayo (Robinson et al., 2003) demonstrated very little impact on flood risk downstream from a forest clearfelling and harvesting catchment. Holden and Burt (2003) found saturation-excess overland flow was dominant in upland peat catchments with flashy flow regimes. Saturation-excess overland flow is produced when the soil profile is completely saturated and can occur at much lower rainfall intensities. The water at the surface is a mixture of water that has been within the soil mass that is returning to the surface from upslope and fresh rainwater. Saturation excess overland flow can occur for long periods after rainfall has ceased, particularly at the foot of a hillslope where the soil continues to be supplied by water draining from upslope.

The hydrological regime is of great importance to the biota of rivers and so any changes can have implications (Richter et al., 1996). Stream flow is a continuous source of delivery of oxygen, food and removal of waste materials. Increased currents restrict macrophyte establishment and periphyton distribution. Discharge from variable flood events cause major reductions in periphyton standing crops due to scouring (Allan, 1995). Flow extremes impact trout, with low survival rates coinciding with high discharges occurring during the alevin stage (Jensen and Johnsen, 1999). Milner et al. (2003) reported a high egg washout rate at high flows and desiccation at low flows. Variable flows can cause intense and frequent disturbances to gravel substrates, which may destroy fish eggs during incubation and consequently limit salmonid production (Lisle, 1989). Salmonids are also affected indirectly by the temporary destruction caused to their food source and the benthic macroinvertebrate populations (Milner et al., 1981) and, directly, by the foraging inhibition of the juvenile salmonids (Lapointe, 2000).

Eutrophication refers to enrichment of biological systems by nutrients most notably nitrogen and phosphorus, and to the enhanced production of algal and higher plant biomass that the added loads stimulate (Reynolds, 1992). In forested catchments in Ireland, lake and stream health are threatened by the overloading of nutrients in runoff water (EPA, 2004). However, fertilisation is, and has been, essential for successful conifer growth as the majority of land available for forestry in Ireland is lacking in P (www.coillte.ie). The efficiency of P uptake by plants depends on environmental factors such as soil temperature, moisture, aeration and nutrient status. Phosphorus availability to tree stands is reduced by complexation in peat soil by iron (Fe) and aluminium (Al) at low pH. In standing forest catchments, P is conserved, with P quantities inputted via rainfall being greater than the P output via the stream (Taylor et al., 1971). The quantity of P transported out of a catchment is dependent on the amount of P added initially as a fertilizer, the soil P content and surface runoff. With the fluctuation of the water table that can occur after forest harvesting, soluble P in peat soil can also be transferred into deeper ground water layers and, subsequently, to drainage channels (Sapek et al., 2007). Cummins and Farrell (2003) investigated the impacts of clearfelling with regard to P on blanket peatland streams and noted an increase in stream P at three harvested study catchment areas from pre-felling to clearfelling. Similarly, Rodgers et al. (2010) observed an increase in P from 6 μg TRP (total reactive phosphorus) L^{-1} pre-clearfelling to 429 μg TRP L^{-1} post-clearfelling, falling to 100 μg TRP L^{-1} eight months later.

Eutrophication risk is highest in low flows as the residence time of P is increased (Jarvie et al., 2005). House and Denison (1998) identified a build-up of P in surface sediments in the spring and summer, followed by a decrease in the autumn and winter, which confirmed that seasonal changes strongly influenced P transport in their study. Higher flows contained higher concentrations of TRP than lower flows (Dr. Michael Rodgers, NUI Galway, *pers comm*.). Nutrient limitation is less of an issue in upland faster-flowing waters where there is a continual supply of nutrients and turbulent oxygenated waters. Over time, there is a gradual enrichment of lowland aquatic systems such as lakes, which in turn can initiate a change in biotic species and can give rise to algal blooms. Algal blooms can cause mass fish kills, suffocate aquatic organisms through oxygen depletion, and release algal toxins that can cause the failure of organs, gill irritation, stress, and associated secondary infections in fish. Recovery from the eutrophic state is often slow, but is highly variable among water bodies (Carpenter et al., 1998; Nisbet, 2001).

Assessing, maintaining and restoring good ecological status to aquatic ecosystems has become a priority for river basin management and water protection in Europe (Eloranta and Soininen, 2002; Kelly and Wilson, 2004; Leira and Sabater, 2005; Kelly et al., 2008; Urrea and Sabater, 2009). Ecological status incorporates chemical parameters in unison with ecological dynamics such as light availability and flow regimes (Karr et al., 2000; Leira and Sabater, 2005). Macroinvertebrates and the phytobenthos have been used successfully as indicators of the ecological status of aquatic ecosystems worldwide (Kelly et al., 1998; Leira and Sabater, 2005; Clarke and Hering, 2006; Chen et al., 2008).

Qualitative sampling and analysis techniques are well developed for macroinvertebrates, as they have been used as indicators of environmental change for over a century (Growns and Davis, 1991, 1994; Davies and Nelson, 1994; Quinn et al., 2004; Banks et al., 2007; Carter et al., 2007). Studies have shown macroinvertebrate assemblages are influenced by forest harvesting (Davies and Nelson, 1994; Quinn et al., 2004; Ryder et al., 2011). The impacts include reduced diversity and changes in species composition. Tierney et al. (1998) highlighted the fact that taxonomic groups such as Ephemeroptera were absent from forested areas. Taxa found throughout Irish aquatic systems include families of Ephemeroptera, Plecoptera, Coleoptera, Trichoptera, Diptera, Chironomidae and Oligochaeta. Water bodies with a high abundance of Ephemeroptera, Trichoptera and Plecoptera are indicative of relatively unpolluted conditions. In contrast, pollution tolerant species include Chironomidae and Oligochaetae. Rodgers et al. (2008) found no significant changes in the macroinvertebrate assemblages following clearfelling activities in the Burrishoole catchment, but they did find that baseline data for the same stream was "depauperate" of assemblages, possibly owing to forestry acidification effects on the stream, the temporal nature of steam, and closed canopy cover with reduced primary productivity feeding ground.

Periphyton is a complex mixture of an algal majority (primarily diatoms), bacteria, fungi and meiofauna, attached to substrata (e.g., rocks, sand, mud, logs, and plants). This biotic cluster is contained within a mucilaginous polysaccharide matrix, and is also referred to as aufwuchs, biofilm and benthic algae (Steinman et al., 2007). Periphyton responds predictably and quickly to changes in environmental conditions at a large range of spatial scales (Hill et al., 2000; Gaiser, 2009). Studies have reported elevated levels of benthic chlorophyll a (Chl a), ash free dry mass (AFDM) and total P (TP) content of periphyton tissue in nutrient enriched streams (Gaiser et al., 2004; Greenwood and Rosemond, 2005; Veraart et al., 2008). Diatoms have been accepted as a proxy for phytobenthos (Kelly et al., 1998) in determining a 'good' ecological status boundary. According to Stevenson and Yan (1999), monitoring diatoms for environmental assessments is useful for the following reasons: (1) being well established in the food web underlines their importance in ecosystems; (2) diatoms respond rapidly to the majority of physical, chemical and biological changes in water bodies; (3) having a one-stage life cycle and a very short generation makes them especially useful as biological indicators. Diatoms reside in nearly all water bodies, even those where water only occasionally present (Stoermer and Smol, 2000).

1.2.3 Mitigation methods

Buffer zones, which can filter the runoff before it reaches the receiving water, are widely used by water quality managers in the protection of freshwater aquatic systems. They can protect aquatic systems by controlling runoff in the following ways: (1) mechanically, by increasing deposition through the slowing down of flow; (2) chemically, through reactions between incoming nutrients and soil matrices and residual elements; and (3) biologically, through plant and microbial nutrient processes. Buffer zones have been recognised as an efficient method to remove SS and attached P, and could remove 14 % to 91.8 % of TP. However, their effectiveness has been controversial. Vought et al. (1994) found that buffer strips were very efficient in DRP (dissolved reactive phosphorus) removal, with the removal efficiency of 95 %. In contrast, Uusi-Kämppä (2005) found that their naturally vegetated BZ became a P source, responsible for 70 % of DRP release. Stutter et al. (2009) indicated that vegetated BZs increased soil P solubility and the potential amount of P released. Rodgers et al. (2010) found that in the Burrishoole catchment, most of the P release after harvesting occurred in soluble form during storm events, raising concerns about the effectiveness of BZs in blanket peatland catchments. In Ireland and the UK, many of the earlier afforested upland blanket peat catchments were established without any riparian BZs, with trees planted to the stream edge (Ryder et al., 2011). Ryder et al. (2011) concluded that it was technically challenging to create riparian BZs prior to felling.

In order to reduce nutrient export to receiving water following forest harvesting, whole-tree harvesting (WTH) is recommended (Nisbet et al., 1997). In the UK, WTH is usually 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). In Ireland, in experimental trials conducted by Coillte, an adapted WTH procedure was adopted where the forest harvest residues are bundled and removed from the selected sites after the conventional harvesting of stem wood (Dr. Philip O'Dea, Coillte Teoranta, 2010, *pers comm*). Needles and branches have much higher nutrient concentrations than stem wood, and WTH may reduce nutrient sources by 2–3 times more than bole-only harvesting (Nisbet et al. 1997). Rodgers et al. (2010) found higher WEP content in the areas below BM material than the BM free areas in the harvested upland peat forest catchment, indicating that WTH could be used as a means to decrease the P release. Yanai (1998) reported negligible P losses to streams over 3 years from harvesting using the WTH method at the Hubbard Brook Experimental forest in New Hampshire. However, WTH removes not only most of the nutrients, but also base cations (Nisbet et al., 1997), which could have a negative impact on the next crop rotation, especially in blanket peat catchments. Walmsley et al. (2009) found that removal of forest residues can reduce second rotation productivity through nutrient shortage.

Phased felling is recommended in the UK (Forestry Commission, 1988) and Ireland (Forest Service, 2000) to diminish the negative impact of harvesting on the receiving salmonid sensitive aquatic systems. Harvesting appropriately sized coupes in a catchment at any one time can minimize the nutrient concentrations in the main rivers (Rodgers et al., 2010). Cummins and Farrell (2003) found higher P concentrations in the smaller drains, which covered a higher proportion of the harvesting area. Rodgers et al. (2010) carried out a study on the impact of a small stream draining a 25 ha harvested forest coupe on the downstream receiving confluence

(DSC) of a river draining a catchment of approximately 200 ha. They found that although the P concentrations in the study stream rose to about 420 μg TRP L^{-1}, the average P concentrations in the receiving water of the main river were 7 ± 5 μg TRP L^{-1} 10 m upstream of the confluence of the study stream with the main river, and 9 ± 8 μg TRP L^{-1} 30 m downstream of this confluence. During a particular storm event, when the TRP in the study stream increased from about 3 to 292 μg TRP L^{-1}, the TRP concentrations at the DSC in the main river increased from about 5 μg TRP L^{-1} to about 11 μg TRP L^{-1}, which was much lower than the critical value of 30 μg TRP L^{-1}. Phased felling is being used widely in Ireland and the UK (Forestry Commission, 1988; Forest Service, 2000). However, this management strategy does not reduce the TP load leaving the harvested catchment, which could be bound to the bed sediment of the receiving waters. If the P concentration in the river bed or lake sediment increases above the saturation point, it could be released and become available to primary producers (EPA, 2004).

1.3 Research objectives

This PhD research project aimed to assess the impact of forest harvesting on acid sensitive catchments and to investigate mitigation methods. The key objectives of this study were:

1. To investigate the impacts of forest clearfelling on the hydrology and chemistry of streams and rivers draining blanket peatland forests.
2. To investigate the impacts of forest clearfelling on the biology of the receiving streams.
3. To investigate grass seeding as a novel method to mitigate nutrient release from peatlands after forest clear-felling.
4. To investigate the efficiency of a buffer zone in mitigating phosphorus and sediment release from forest clearfelling activities.
5. To investigate and compare plot-scale grass seeding, whole tree harvesting and the absence of any mitigation method and provide scientific data on novel BMPs to water managers.

1.4 Burrishoole Catchment

The study sites were based mainly in the Burrishoole Catchment, Newport, Co. Mayo (Figure 1.1). The Burrishoole catrchment (53º 55’ N 9º 55’ W; c. 100 km^2) comprises approximately 55 km of rivers and streams. The mean annual rainfall in the catchment is more than 2000 mm and the mean air temperature is about 11 °C. The Burrishoole Catchment is historically significant as fish stocks (salmon, char and eels) have been monitored here since the mid-1950s. Blanket peats are extensive in the catchment accounting for 23 % of the land cover (Coillte *pers. comm*). The main land uses are forestry and sheep grazing. Coniferous plantations were planted in forest blocks or coupes since the 1970s, and make up 31 % of the land use of the 10,000 ha catchment area. The peat has a gravimetric water content of approximately 80 %. The depth of the water table fluctuates between 0.2 m and 0.7 m from the soil surface. The catchment system is described as acid oligotrophic and has a low buffering capacity (Byrne et al., 2004).

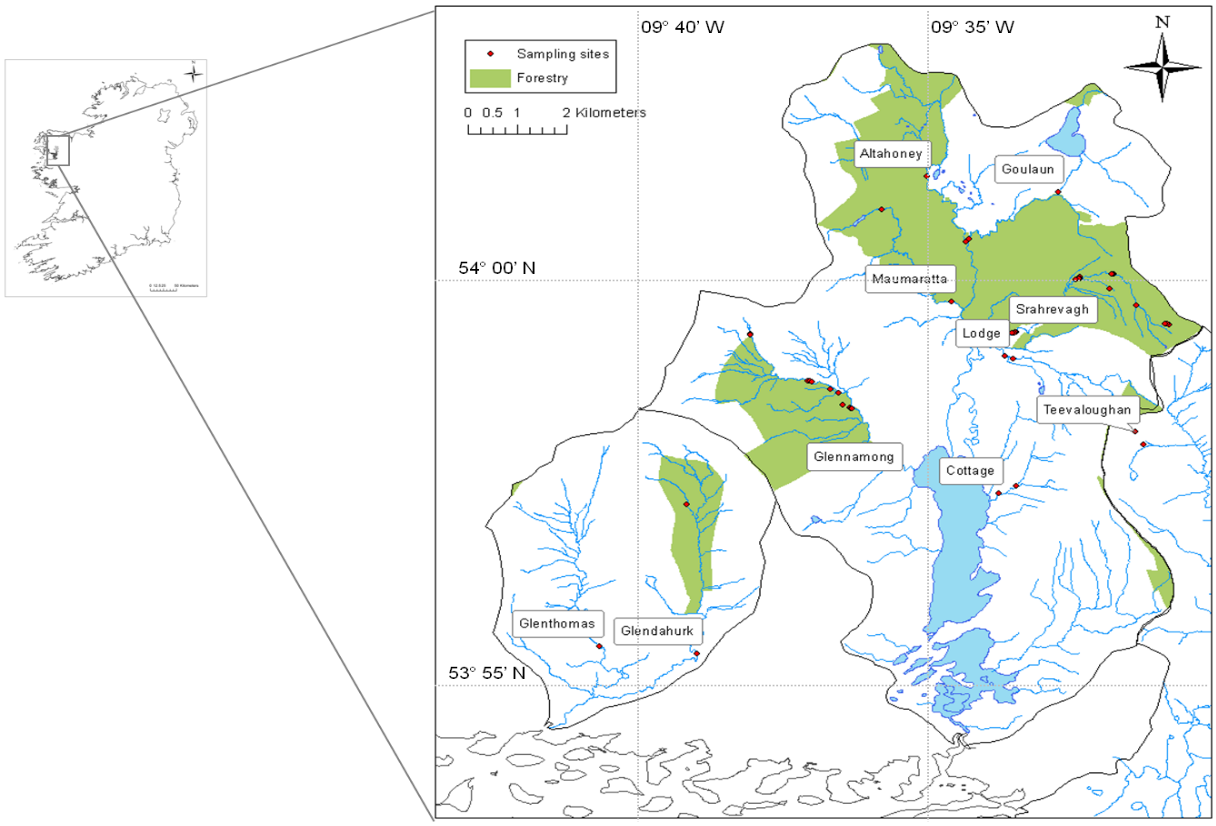

Figure 1.1 Map inclusive of all study sites referred to in this thesis.

Geology in the catchment

Late Precambrian metamorphic rocks and smaller areas of sandstone and limestone characterise the geology of the catchment (Parker, 1977; Long et al., 1992; Irvine et al., 2001). Western parts of the catchment (Glenamong, Altahoney and Maumaratta subcatchments) predominantly comprise quartzite/schist, acidic in nature and with poor buffering capacity, whereas the geology is much more complex in the east (Rough, Lodge, Goulaun and Cottage subcatchments), and also includes veins of volcanic rock, dolomite, wacke and pure schist. The additional minerals available ensure greater buffering capacity and aquatic production.

1.5. Structure of dissertation

Chapter 1 presents a literature review of the harvesting operations on upland peat catchments, the potential impacts of forest clearfelling including the hydrological, chemical and biological impacts along with current BMPs and mitigation methods.

In Chapter 2, the impact of upland blanket peat forest harvesting on the flow regime and phosphorus release are detailed in case studies.

Chapter 3 characterises the diatom assemblages in rivers draining upland forested peat and attributes the main driving factors along with highlighting the natural variation occurring in these assemblages underlining the importance of understanding spatial and temporal factors. The impact of upland blanket peat forest harvesting on the ecological status is described in a case study.

Chapter 4 presents a novel grass seeding method with a potential to reduce P export from harvested sites, investigates the application of a constructed buffer zone in ameliorating the negative impacts of forest harvesting, and details a plot-scale study approach to investigating mitigation methods.

Chapter 5 gives the conclusions with recommendations for further research.

Chapter 6 contains the relevant bibliography for all chapters.

Chapter Two

The impacts of harvesting on the flow regime and nutrient release

2.1 Introduction

This chapter comprises two research papers, which are contributions to the projects, SILTATION and SANIFAC. The first paper "Impact of blanket peat forest harvesting on stream flow regimes – a case study in the Burrishoole Catchment, Co. Mayo" was published in the Irish National Hydrology Conference (Xiao, L., Robinson, M., Rodgers, M., O'Connor, M., O'Driscoll, C., Asam, Z.-Z., 2011. Impact of blanket peat forest harvesting on stream flow regimes – a case study in the Burrishoole Catchment, Co. Mayo. Irish National Hydrology Conference. Athlone, Ireland 15 November 2011). Connie O'Driscoll assisted with the instrumentation, boundary definition, data collection, and data analysis. She also contributed to drafting the submission. The research undertaken in this paper contributed to the development of this thesis, the main findings of which are detailed in the subsequent chapters.

The second paper, "Phosphorus release from forest harvesting on an upland blanket peat catchment", has been published in the peer-reviewed, international journal, Forest Ecology and Management (Rodgers, M., O'Connor, M., Healy, M.G., O'Driscoll, C., Asam, Z.-Z., Nieminen, M., Poole, R., Müller, M. and Xiao, L., 2010. Phosphorus release from forest harvesting on an upland blanket peat catchment. *Forest Ecology and Management*, 260 (12): 2241-2248.). Connie O'Driscoll was involved in the maintenance of equipment, collection of samples, water testing and analysis of the acquired data from 2008, when she began her research. She also contributed to drafting the submission. The research carried out in this paper formed the basis for the development of the research carried out in the subsequent chapters.

2.2 Impact of blanket peat forest harvesting on stream flow regime – a case study in the Burrishoole Catchment, Co. Mayo

Liwen Xiao[a*], Mark Robinson[b], Michael Rodgers[a], Mark O'Connor[a], **Connie O'Driscoll[a]**, Zaki-ul-zaman Asam[a]

[a] Civil Engineering, National University of Ireland, Galway, Ireland

[b] Centre for Ecology and Hydrology, the UK

Abstract

Approximately 11 % of Ireland's land area is covered by forest, planted mostly since the 1950s, with approximately 80 % of this being coniferous (EPA, 2012). The land chosen for forestry was of poor quality and generally unsuited to agriculture; planting on blanket peat land was especially targeted. Today, 43 % of the total forest estate is located on peat soils with blanket bogs accounting for the largest proportion of afforested peatlands (62 % or some 218,850 ha) and raised bogs a further 74,080 ha (Renou-Wilson, 2011). Many of these upland blanket peat forests contain the headwaters of salmonid and freshwater pearl mussel rivers and drinking water sources, and are sensitive to water pollution and hydrological alterations. As the forests planted before the 1980s are reaching harvestable age, great attention has been paid to the possible impacts of harvesting these forests on the receiving water quality and hydrology (Coillte Teo, 2007; Chapter 2, section 2.3; Rodgers et al., 2011).

The aim of this study was to investigate the impact of upland blanket peat forest harvesting on the flow regimes in the Burrishoole Catchment. The Burrishoole catchment, located in County Mayo in the west of Ireland, consists of important salmonid productive rivers and lakes (Figure 1a). About 18 % of the catchment is covered by forests that were planted in the 1970s and which are now being, or about to be, harvested. The study site (9°55'W 35°55'N), which is a sub-catchment of the Burrishoole catchment, is drained by a small first-order stream (Figure 1a) and was planted with lodgepole pine (*Pinus contorta*) between January and April, 1971. The stream is equipped with two flow monitoring stations at stable channel sections, one upstream and the

other downstream of the experimental area (Figure 1a). An H-flume, a water level recorder and a data logger were installed at both US and DS stations, along with a tipping bucket rain gauge at the DS station (Figure 1b). The US station measures flows from the control area of 10.8 ha (area A in Figure 1a) and the DS station receives flow from the control and experimental areas, giving a total combined area of 25.3 ha (areas A and B in Figure 1a). The study site has an average peat depth of more than 2 m. In the catchment, the mean annual rainfall is greater than 2000 mm and the mean air temperature is about 11° C. Hill slope gradients in areas A and B (Figure 1a) average 8^{o} and range between 0^{o} – 16^{o}. Bole-only harvesting was conducted in area B (Figure 1a) from July 25th to September 22nd, 2005. Continuous measurements of stream flow upstream and downstream of the felling coupe commenced over a year earlier in spring 2004. The results indicated that forest harvesting increased water yields and base flows, but had very limited impact on flood risk downstream (Figure 2).

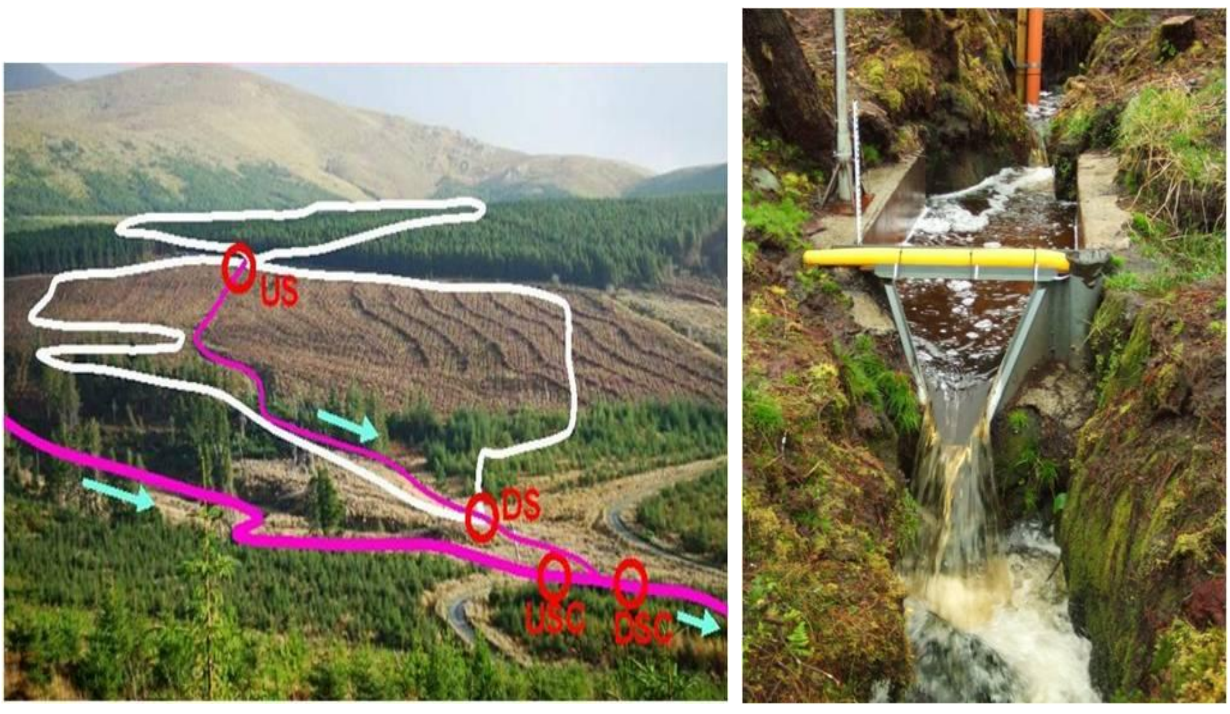

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Figure 1 a and b. The study site and flume

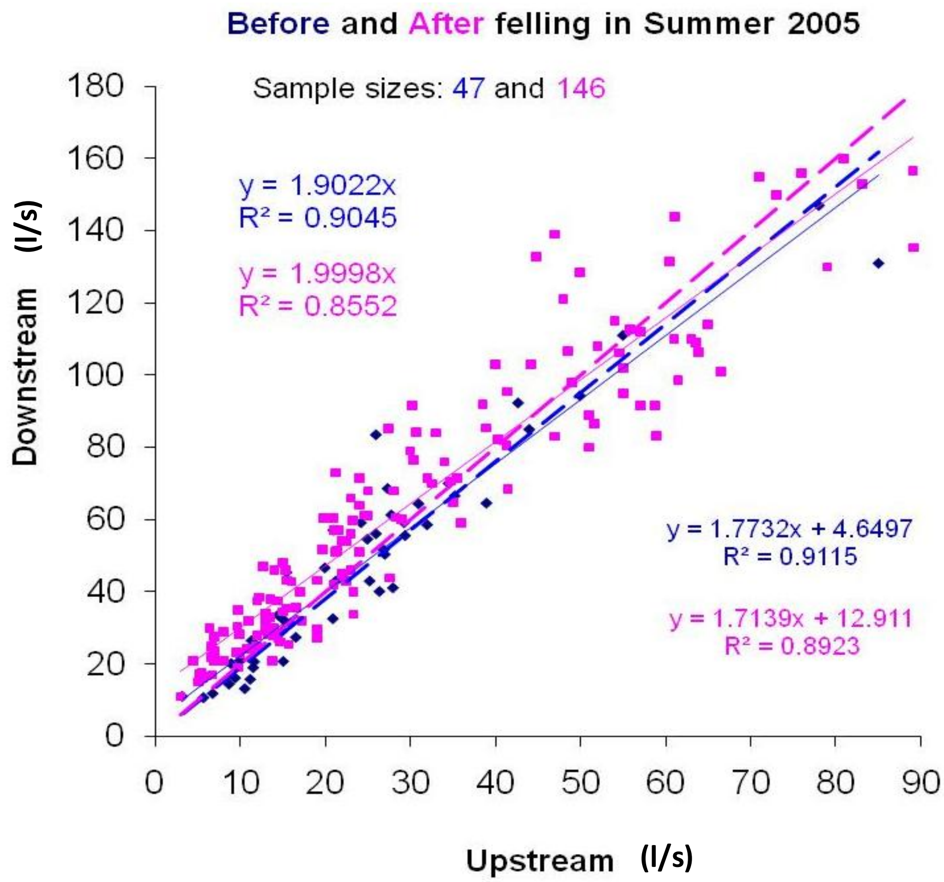

Figure 2 Comparison of peak flows before and after harvesting

Introduction

By the end of 2000, about 300,000 ha of blanket peat was afforested in Ireland (EEA, 2004). Many of these blanket peat forests contain the headwaters of salmonid and freshwater pearl mussel rivers and drinking water sources, which are sensitive to water pollution and hydrological alterations. As the forest planted before 1980s are reaching harvestable age, great attention has been paid on the possible impact of harvesting these forests on water quality and stream flows (Chapter 2, section 2.3; Coillte Teo, 2007).

Previous studies have indicated that forest harvesting would impact stream flow regime of the catchment. Bosche and Hewlett (1982) reviewed 94 experimental catchments and concluded that stream flow response to harvesting depends on the climate, especially the precipitation. Bruijnzeel (1988) indicated that whether the stream base flows increase or decrease after harvesting depends on the surface infiltration and evapo-transpiration of the catchment. In temperate zones, base flow increase after harvesting was almost uniformly observed (Hornbeck et al., 1993, Brown et al., 2005). While increase in the intensity of peak flow and decrease in the time of concentration of flow after deforestation was reported (Hubbart and Matlock, 2009), the forest harvesting impacts on floods may be small when soil conditions were maintained (DeWalle, 2003; Robinson and Dupeyrat, 2005). In Britain, Robinson and Dupeyrat (2005) carried out the first comprehensive study on the impact of harvesting on stream flow regimes in the Plynlimon research catchments in Wales and found that (1) the cutting of the forest increased total annual flows and augmented low flows and (2) there was lack of impact of harvesting on storm peak flows. However, very few studies were conducted in upland blanket peat sites with temperate wet conditions. With shallow water tables and a low hydraulic conductivity at depth, blanket peatlands tend to generate rapid runoff and have shorter lag times and higher peak flows in response to rainfall events (Evans et al., 1999; Rosa and Larocque, 2008; Grayson et al., 2010). Due to differences in tree species, soil types, forest management, catchment characteristics and climate, it is unclear how harvesting upland blanket peat forests affect stream flows in Ireland. Therefore, the objectives of this paper were to assess the impact of harvesting on water yield and flow regimes in the upland blanket peat in the west of Ireland.

Site description

The Burrishoole catchment, located in Co. Mayo, in the west of Ireland, consists of important salmonid productive rivers and lakes (Figure 3). About 18 % of the catchment is covered by forests that were planted in the 1970s and which are now being, or are about to be, harvested. The study was carried out in two areas – the Srahrevagh and the Glennamong, which are sub-catchments of the Burrishoole catchment. The distance between the two sub-catchments is about 5 km. The Srahrevagh study site is drained by a small first order stream (Figure 1a), was planted with Lodgepole Pine (*Pinus contorta*) between January and April 1971. The stream is equipped with two flow monitoring stations at stable channel sections, one upstream (US) and the other downstream (DS) of the experimental area (Figure 1b). The US measures flows from the control area (area A in Figure 3) of 7.2 ha and the DS covers the control coupe and the experimental coupe (coupes B in Figure 3) with a total combined area of 17.7 ha. In August 2005, a windblown tree blocked one of the collector drains, resulting in an increase of the upstream forest control area (coupe D), to about 10.8 ha (coupes A plus D in Figure 3). Meanwhile, the downstream harvested area increased to about 14.5 ha due to the blockage of a drain by brash mat during the harvesting, incorporating another part of the total harvested area (coupe C in Figure 3). Fortunately, in both cases the additional area had the same characteristics of vegetation and soils, and the relative sizes of US and DS remained unchanged – US increasing only marginally from 41 % of the total area to DS before harvesting and 43 % afterwards. All unit area depths in this paper have been calculated using these values. The blanket upland peat soil in all four areas A - D had been double mouldboard ploughed by a Fiat tractor on tracks 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. The trees were planted on the ribbons at 1.5 m intervals, giving an approximate soil area of 3 m^2 per tree. The catchment had an average peat depth of more than 2 m above the bedrock of quartzite, schist and volcanic rock, and the peat typically had a gravimetric water content of more than 80 %. The mean annual rainfall is more than 2000 mm and the mean air temperature is about 11 °C. Hill slope gradients in areas B and C average 8° and range between 0° – 16°. Bole-only harvesting was conducted in area B and C from July 25th to September 22nd 2005. The timber was harvested using a Valmet 941 harvester, and the residues (i.e. needles, twigs and branches) were left on the soil surface and collected together to form windrows. During harvesting, the boles were stacked beside the windrow for collection. A Valmet 840 forwarder delivered the boles to truck collection points beside the forest service road. To minimise soil damage, the clearfelling and harvesting were conducted only in dry weather conditions during the period from July to September 2005. Tree residues (i.e. needles, twigs and branches) were collected together to form brash mats on which the harvesting machines travelled, thus protecting the soil surface, and reducing erosion. In the lowest part of the site where the stream is deeply incised, the trees were cut with a chain saw and left behind. The non-harvested upstream area of A and D was used as a control area in this study, as it had the same type and age of trees, similar soil, hydrologic characteristics and size as the harvested experimental area of B and C. In the experimental area, the furrows and BMs - formed from the harvest residues – are, in general, parallel with the study stream, which is at right angles to the contours. The surface water flows along the furrows, is collected by collector drains (arrows in Figure 3) and joins the study stream.

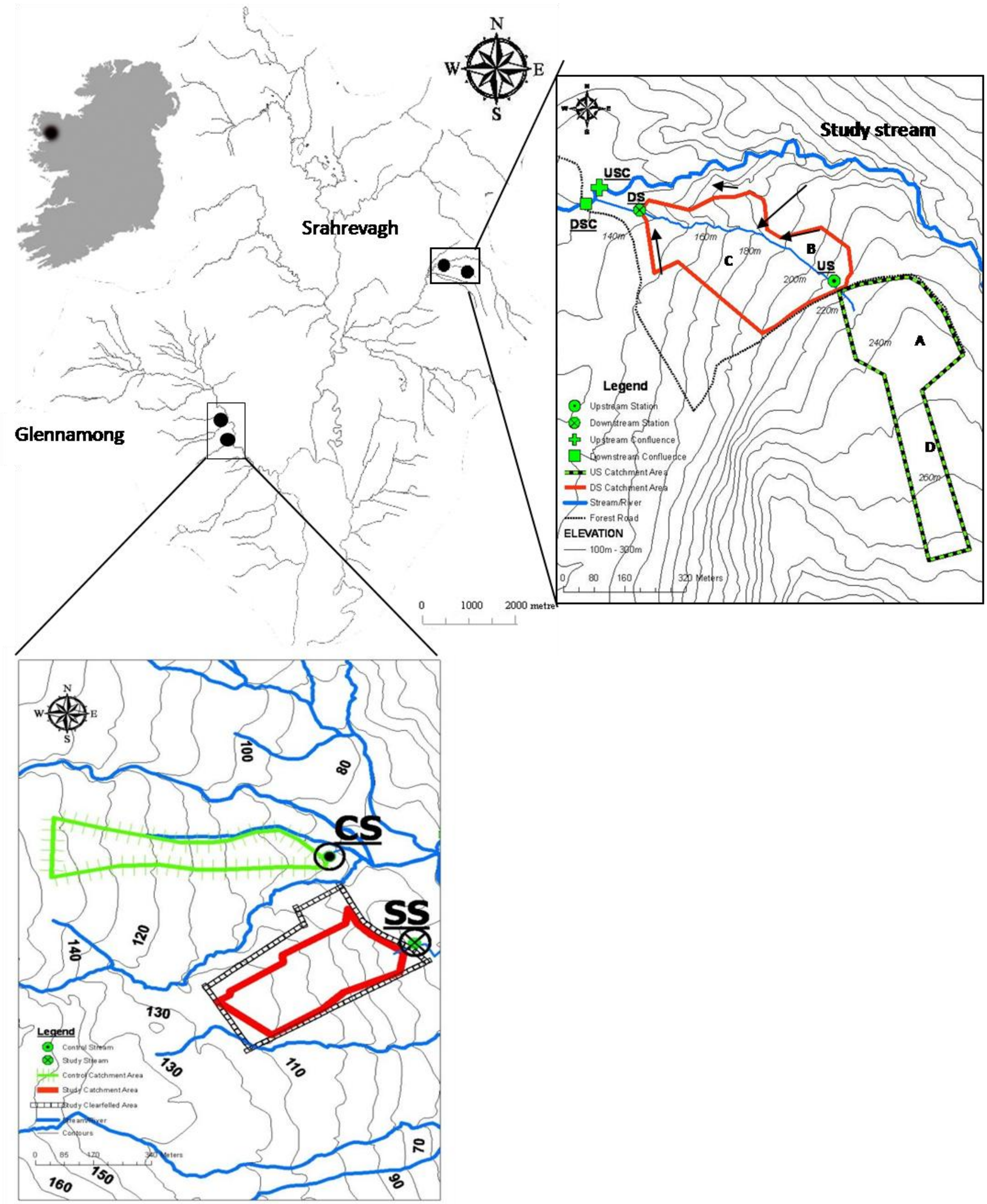

Figure 3 Location of the Burrishoole catchment and the two study sub-catchments – the Srahrevagh and the Glennamong

In Glennamong, two 10 ha sites which were planted with Lodgepole Pines (*Pinus contorta*) in 1975 and are drained by semi-natural drains were chosen for this study. Two monitoring stations were established at the sites out flow (Figure 3). The area is covered by peat with the depth of about 0.5 - 1 m. Bole-only harvesting was conducted in the study site from February 2011 to April 2011.

Sampling, measurement and analysis

From April 2004 - March 2005, continuous water levels in the study stream were recorded at both the US and DS in Srahrevagh, and converted to flows by a rating equation based on dilution gauging and current meter measurements. In April 2005, H-flume flow gauges were installed at the sites for flow measurement. In June 2009, H-flume flow gauges were also installed at the control and study sites in the Glennamong.

Analysis

This is a paired catchment study. One of the advantages of paired catchment studies is that they remove climate variability through the comparison of two catchments subject to the same climatic condition under different land uses (Brown et al., 2005). The impact of the felling on stream flow was assessed for monthly water yields, peak flow and base flow. The data were divided into before and after harvesting periods. The periods March 2004 – July 2005 and July 2009 – February 2011 were used for the pre-felling control period in the Srahrevagh and the Glennamong, respectively. September 2005 – December 2009 and April 2011 – September 2011 were used for post harvesting period in the Srahrevagh and the Glennamong, respectively. Harvested areas (DS-US in the Srahrevagh and the Glennamong) were used as the study sites. Control sites were untouched and used to remove the effects of any climate variability during the study period. The pre-harvesting data was used to establish the calibration equation, where the monthly water yield in the control area (US) was the independent variable and monthly flow in the study area (DS-US) was the dependent variable. The difference between the observed and estimated data in the control sites, therefore, was attributed to the harvesting. The statistical significance of any differences was determined by using Student's t-test.

Results and discussions

After harvesting, a close linear relationship was observed between the monthly water yields of the two areas. Slightly higher monthly flows from the harvested area were observed in the Srahrevagh (Figure 4a). The calibration equation was used to predict the ‘no-felling’ monthly water yield in study area after harvesting, using the observed water yields in US in the same period as the independent variables. The estimated and observed water yields at DS-US after harvesting were then compared using a paired samples t-test at the 95 % significance level (p = 0.05) (SPSS version 18, 2010), which indicated that the observed water yield increase was not significantly higher than the ‘predicted’ water yield ($p < 0.05$). Robinson and Dupeyrat (2005) studied the impact of commercial timber harvesting on stream flow regimes in four nested catchments in mid-Wales and detected an increase of total annual flows. Johnson (1998) observed a 25 % to 30 % increase in water yield when 100 % forest area was harvested in the precipitation range of 800 - 2400 mm per year. In another study, Hornbeck et al. (1970) found that annual flow increased by 40 % in the year following a 100 % forest clearance. The water yield increase in the harvested area was attributed to reduced canopy interception and virtual elimination of transpiration (Johnson, 1998). The impact of harvesting on water yield changes depends on the reduction of forest cover in the catchment basin. A reduction in forest cover of 20 % was necessary before any changes were observed (Hornbeck et al., 1993). In their study, Robinson and Dupeyrat (2005) found that the smallest catchment with the largest proportion of area felled had the greatest flow increase. Significant monthly flow increases were also observed in the Glennamong after harvesting ($p < 0.1$), though the after harvesting period (6 months) was very short (Figure 4b).

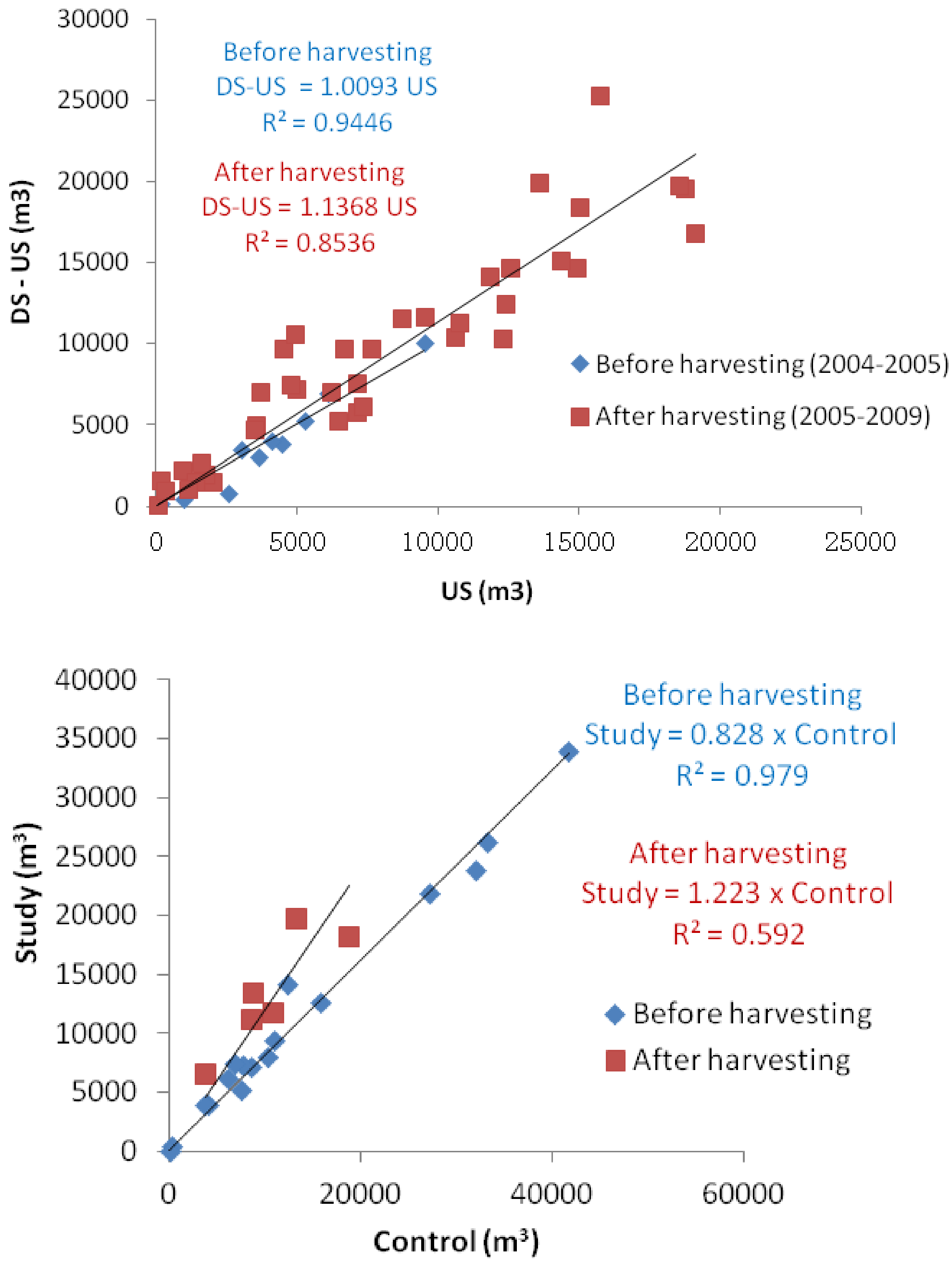

Figure 4 a and b Monthly water yield between the harvested and control areas in the Srahrevagh (a) and the Glennamong (b) respectively, before and after harvesting.

Slight peak flow increases were observed at both sub-catchments after harvesting (Figure 5a and b). However, statistical analysis indicated that the increase was not significant. Statistical analysis on accumulative peak flows further highlighted the lack of impact on peak flows

(Figures 6a and b). In their studies across Europe, Robinson et al. (2003) found that the impact of forest harvesting on extreme flows was relatively small and difficult to detect in North Western European conifers. They completely harvested one of their study sites - Glenturk (which is in close proximity to the Srahrevagh and the Glennamong study sites), and only observed moderate peak flow increases. They attributed the lack of peak flow response to (1) minimum soil disturbance and (2) the presence of harvesting residues on the felled area (Robinson et al., 2003). Peak flow increases are usually due to the reduced infiltration which can be caused by soil compaction and disturbance. DeWalle (2003) also noted that where forest felling significantly increased flood event flows, soil had suffered severe disturbance. In this study, good management practices such as proper use of brash mats and harvesting only in dry weather were implemented, and soil surface disturbance were minimised (Chapter 2, section 2.3).

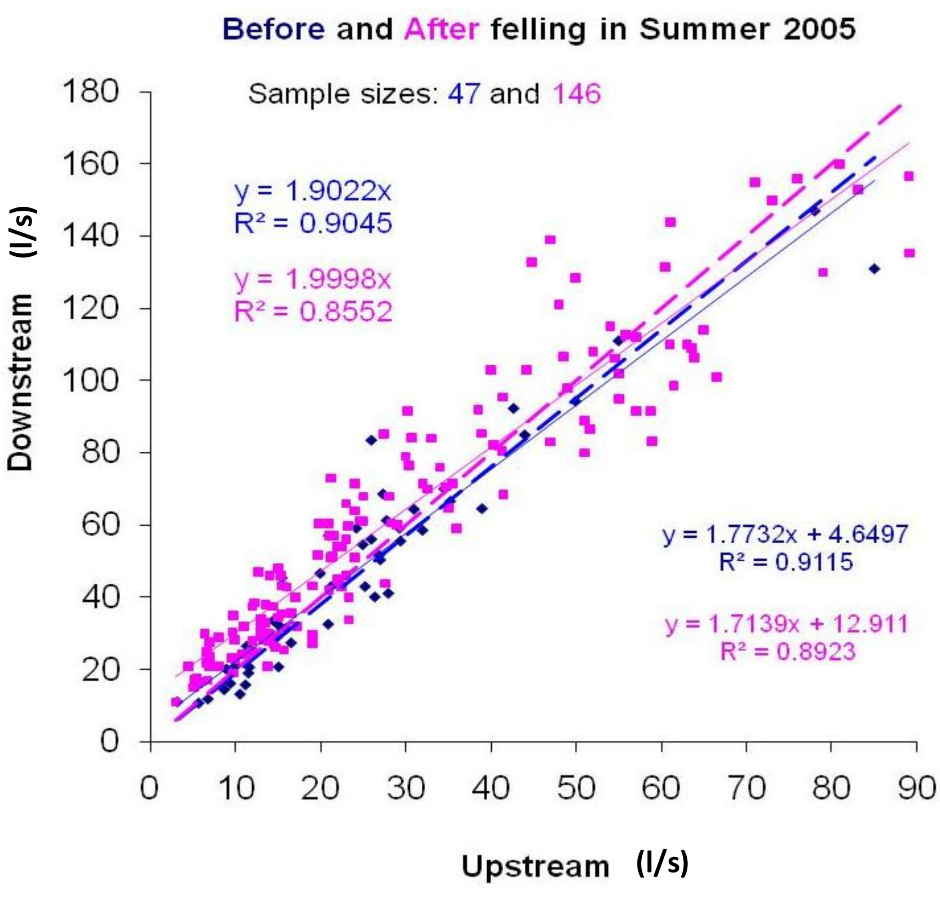

Figure 5a Peak flows before and after harvesting in Srahrevagh (Robinson et al., 2013)

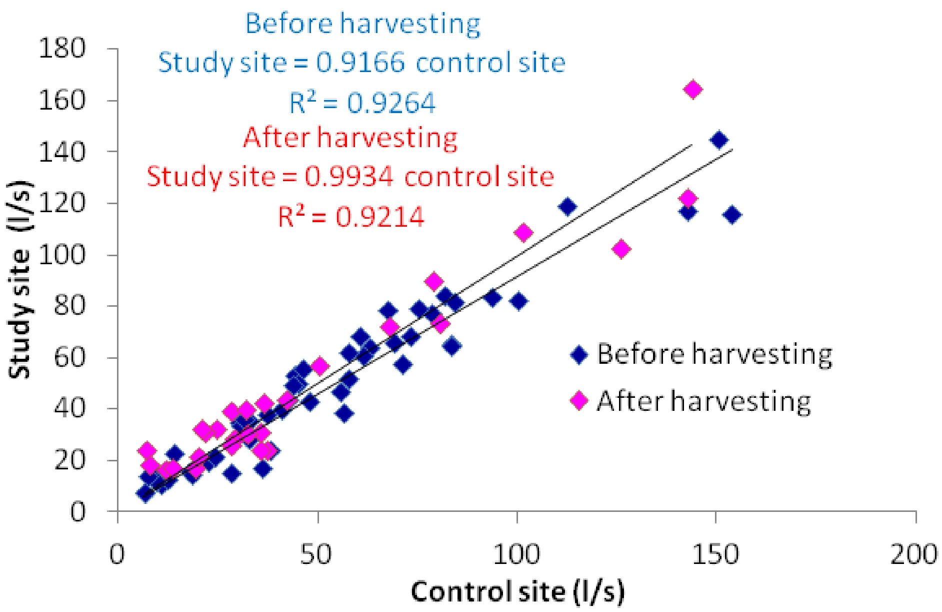

Figure 5b Peak flows before and after harvesting in Glennamong

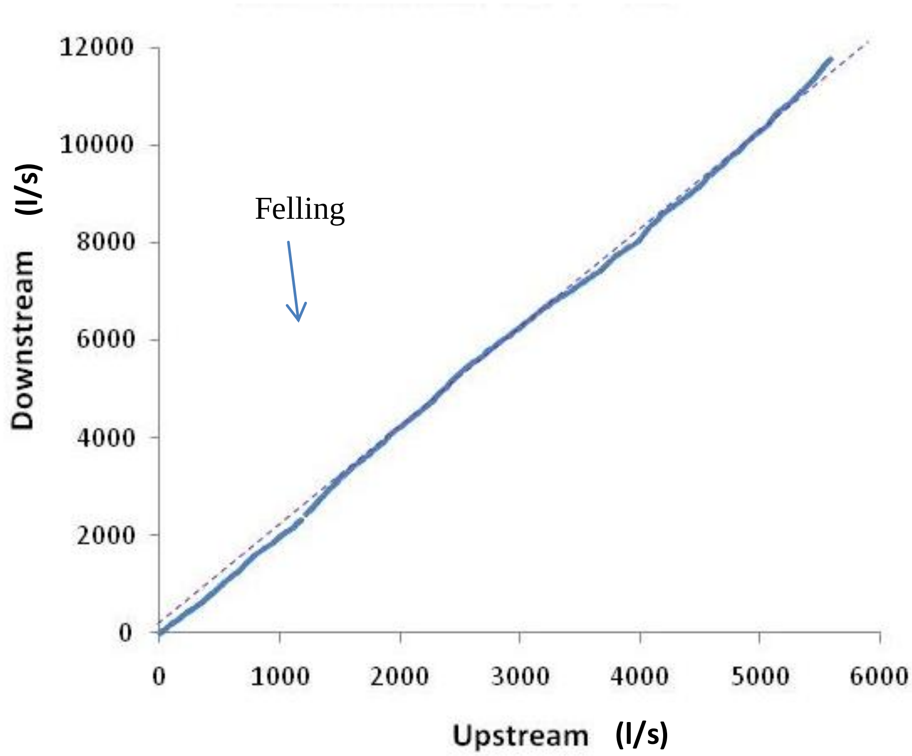

Figure 6 a Chronological accumulated peak flows in Srahrevagh (Robinson et al., 2013)

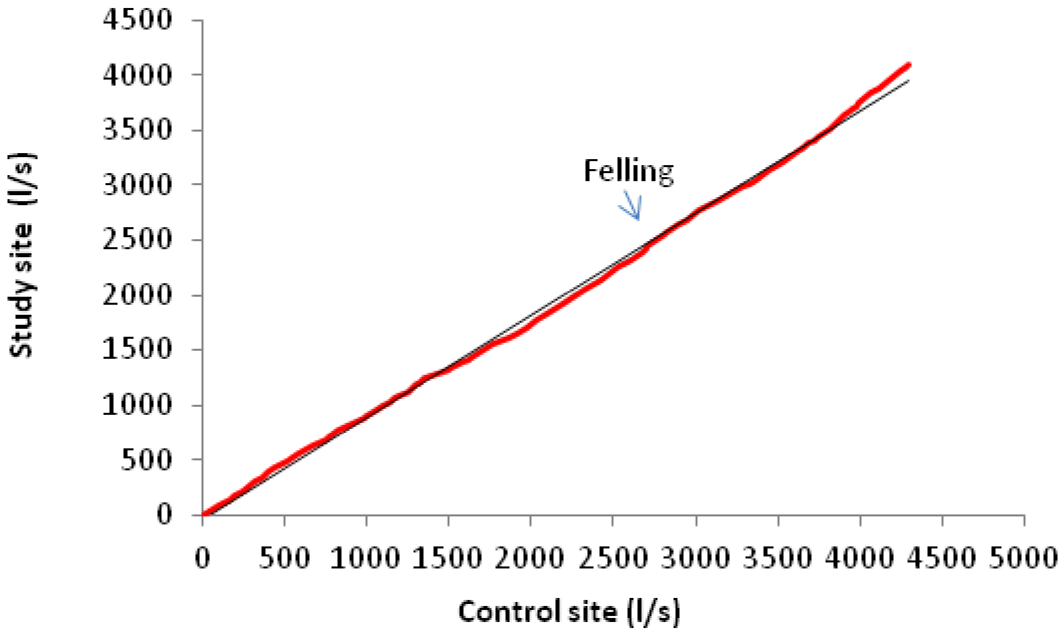

Figure 6 b Chronological accumulated peak flows in the Glennamong

The 95-percentile flow was used to determine the base flow at the control sites in the two sub-catchments. In both sub-catchments, harvesting significantly increased the base flows (Figure 7a and b). Robinson et al. (2003) also observed baseflows increased in three of their experimental catchments attributed to less evaporation from the removed crop.

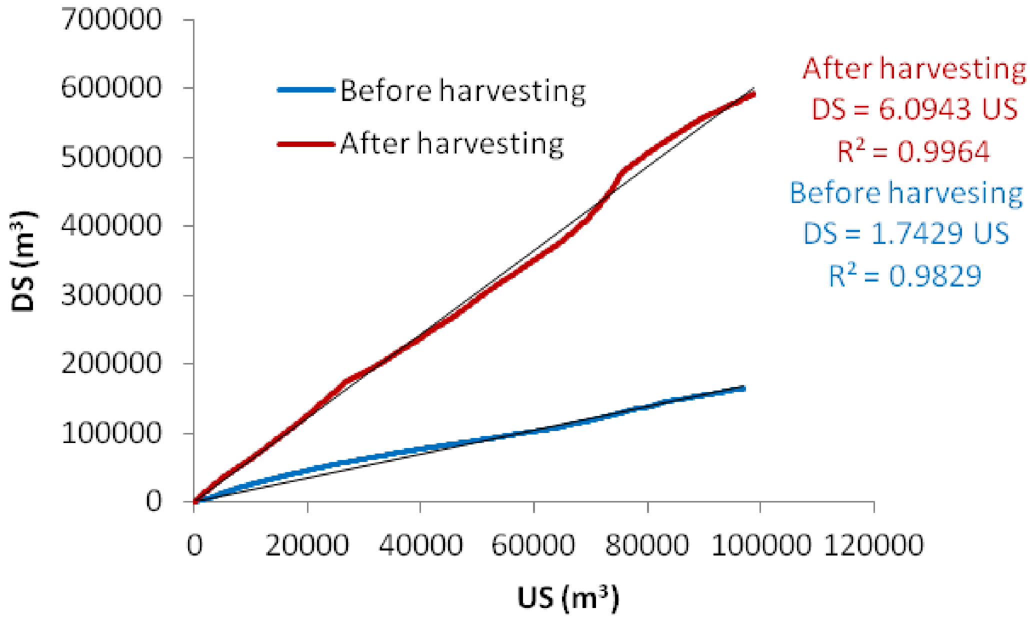

Figure 7a Chronological accumulated base flow before and after harvesting in Srahrevagh

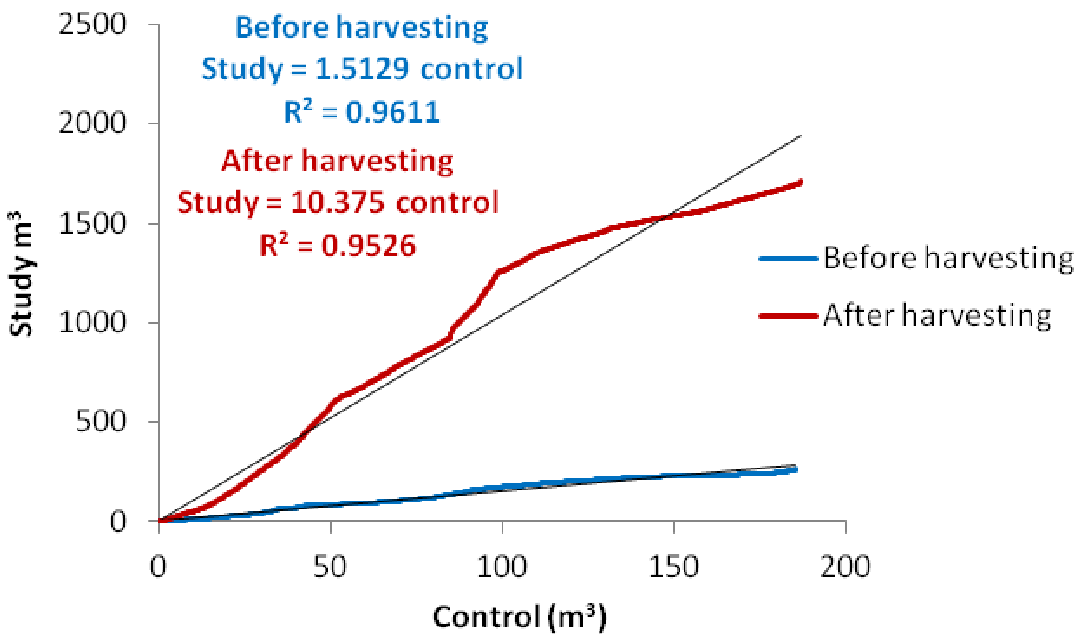

Figure 7b Chronological accumulated base flow before and after harvesting in the Glennamong

Conclusion

Two paired catchment studies were carried out in the Burrishoole Catchment in the west of Ireland to investigate the impact of upland blanket peat forest harvesting on stream flow regimes. Monthly water yield, peak flow and base flow were used as the impact indicators. The results indicated that while forest harvesting increased the monthly water yield and base flow significantly, it had very little impact on the peak flows. This could be due to the implementation of the good management practices such as 1) use of brash mats and 2) harvesting only in dry weather minimising soil surface disturbance.

Acknowledgements

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Bibliography

See Chapter Six pp. 187 - 222.

2.3 Phosphorus release from forest harvesting on an upland blanket peat catchment

Michael Rodgers[a], Mark O'Connor[a], Mark Gerard Healy[a], **Connie O'Driscoll[a]**, Zaki-ul-Zaman Asam[a], Mika Nieminen[b], Russell Poole[c], Markus Müller[a] and Liwen Xiao[a*]

[a] Civil Engineering, National University of Ireland, Galway, Republic of Ireland

[b] Finnish Forest Research Institute, Finland

[c]Marine Institute, Newport, Co. Mayo, Republic of Ireland

Abstract

The aim of this study was to investigate the release of P to receiving waters resulting from harvesting 34-year-old lodgepole pine trees in an upland peat catchment. The study site was located within a 25.3 ha area, and was drained by a stream that received flows from ploughed furrows, mainly, via collector drains, and discharged directly to the salmonid Srahrevagh river, Burrishoole, Co. Mayo, Ireland. The study site was divided in two parts: the upstream part was left intact and the downstream part was harvested in early autumn 2005, following implementation of forest guidelines. Good management practices such as proper use of brash mats and harvesting only in dry weather were implemented. Two instrumented stations were established (US and DS). The measurement of P concentrations at the two stations commenced in May 2005, two months before the harvesting started. The daily mean P concentration at the DS station increased from about 6 μg L^{-1} of TRP during pre-clearfelling to 429 μg L^{-1} in August 2006. By October 2009, four years after clearfelling, the P concentrations at the DS station had returned to pre-clearfelling levels. In the first three years after harvesting, up to 5.15 kg ha^{-1} of TRP were released from the harvested catchment to the receiving water; in the second year alone, 2.3 kg ha^{-1} of TRP were released. Linear regression can be used to describe the relationship between TRP load and water discharge. About 80 % of the TP in the study stream was soluble and more than 70 % of the P release occurred in storm events, indicating that traditional buffer strips with widths of 15-20 m might not be efficient for P immobilization. The P concentrations were affected by antecedent weather conditions, and highest concentrations occurred during storm events following prolonged drought periods. The WEP contents in the soil

were significantly higher below windrow/brash material than in brash-free areas, and whole-tree harvesting should be studied as one of the means to decrease P export from blanket peats.

Keywords: Phosphorus release; blanket peat forest; storm flow; buffer strip; forest harvesting; good management practice

Introduction

Phosphorus (P) at a concentration of about 30 μg L^{-1} is the limiting nutrient for algal growth in freshwaters (Carpenter et al., 1998; Boesch et al., 2001). According to the U.S. Environmental Protection Agency (USEPA, 2004), agriculture is the primary source of non-point source pollution degrading the quality of streams and lakes. In Ireland, almost half the eutrophication of rivers is due to agricultural sources (EPA, 2004). It was reported that in the Shannon River catchment, about 50 % of the export load of P – measured as Molybdate Reactive Phosphorus (MRP) - comes from diffuse sources (Kirk, 1999).

Forests and forest management practices have been identified as potentially important diffuse sources of water pollution in upland areas of the United Kingdom (Nisbet, 2001). Forest operations such as drainage, fertilisation, harvesting and reforestation result in increased P release (Lebo and Hermann, 1994; Ensign and Mallin, 2001; Cummins and Farrell, 2003), and may increase the P concentration of receiving water bodies (Paavilainen and Päivänen, 1995; Ahtiainen and Huttunen, 1999; Nisbet, 2001; Cummins and Farrell, 2003). Clearfelling disrupts P cycling and significantly reduces the uptake of P by plants, resulting in an increased labile P pool in the harvested area (Walbridge and Lockaby, 1994; Herz, 1996). The decomposition of logging residues (i.e. needles, twigs, roots and branches) left on the harvested area further increases the labile P pool in the surface soil layer (Hyvönen et al., 2000; Piirainen et al., 2004).

Blanket peat has extremely low P adsorption capacity, low hydraulic conductivity, and is anaerobic to within a few centimetres of the surface (Tamm et al., 1974; Cummins and Farrell, 2003). Phosphorus in peat soil can be easily transferred to receiving water by runoff (Cummins and Farrell, 2003).

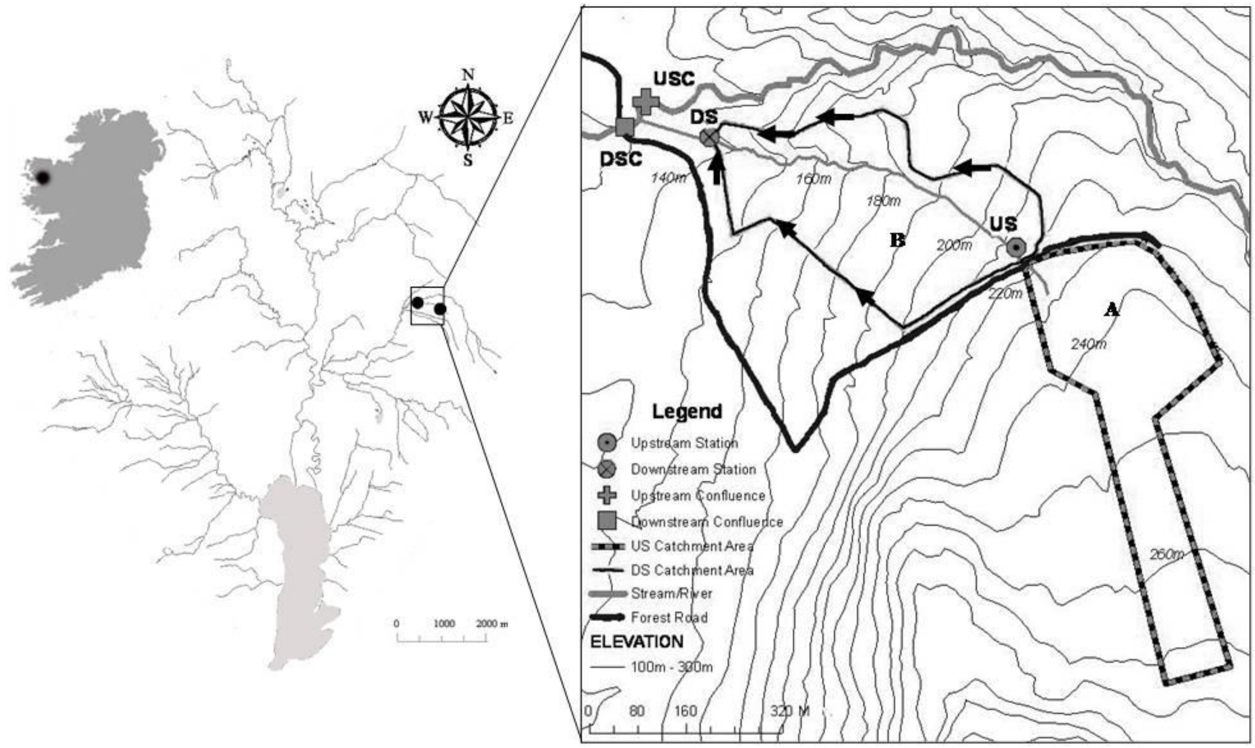


Figure 1 Locations of the Burrishoole catchment and the study stream (DS: downstream station; US: upstream station; USC: upstream of the confluence; DSC: downstream of the confluence; A: intact area - control site; B: harvested area-study site; arrows: indicating the collector drains and flow direction)

With the fluctuation of the water table, soluble P in peat soil can also be transferred into deeper ground water layers and, subsequently, to drainage channels (Sapek et al., 2007). Since the 1950s, large areas of upland peat were afforested in northern European countries. It was estimated that about 500,000 ha of peatland were afforested between the 1950s and 1990s in the UK and 300,000 ha in Ireland (Farrell, 1990; Hargreaves et al., 2003; EEA, 2004). Many of these blanket peat forests are now reaching harvestable age and concerns have been raised about the potential release of P to the receiving aquatic systems as a result of harvesting.

In this paper, P release from an upland blanket peat forested area in the Burrishoole Catchment, Co. Mayo, Ireland was studied for four years after harvesting. We hypothesise that P release is increased significantly due to a combination of poor P adsorption capacity in blanket peat soil, high rainfall (> 2000 mm) and runoff in the study area, and labile P sources being available on site after harvesting. Buffer strips with a width of 15-20 m are recommended as one of the means to reduce P release to recipient water courses. However, their effect may be limited if most of the P release occurs during storm events, coincident with low residence times for the vegetative uptake of soluble P. Thus, a specific aim of the study was to investigate the P release pattern in storm events, and to quantify the P release occurring during storm events and base flow conditions. Whole-tree harvesting has been recommended as another means of decreasing P release. To increase the understanding of the effect of WTH on P release, a small-scale pilot survey was also performed to investigate if the WEP contents in soil below windrow/brash material are significantly higher than for areas without windrow/brash material.

Study site description

The Burrishoole catchment, located in Co. Mayo in the west of Ireland, consists of important salmonid productive rivers and lakes (Figure 1). About 18 % of the catchment is covered by forests that were planted in the 1970s and which are now being, or about to be, harvested. The study site (9°55'W 35°55'N), which is a sub-catchment of the Burrishoole catchment, is drained by a small first-order stream (Figure 1) and was planted with lodgepole pine (*Pinus contorta*) between January and April, 1971. The stream is equipped with two flow monitoring stations at stable channel sections, one upstream (US) and the other downstream (DS) of the experimental area (Figure 1). An H-flume, a water level recorder and a data logger were installed at both US and DS stations, along with a tipping bucket rain gauge at the DS station. The water levels in the H-flumes at both stations were recorded every 5 minutes, facilitating the quantification of water flowing through the two stations. The maximum flow rate for the two H-flumes was 158 L s^{-1}. The US station measures flows from the control area of 10.8 ha (area A in Figure 1) and the DS station receives flow from the control and experimental areas, giving a total combined area of 25.3 ha (areas A and B in Figure 1). The blanket upland peat soil in the study area was double-mouldboard ploughed by a Fiat tractor on tracks 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. The trees were planted on the ribbons at 1.5-m-intervals, giving an approximate soil area of 3 m^2 per tree. The initial stand density was about 2800 trees per ha, but was reduced to about half by thinning in the late 1980s and natural die-off before clearfelling. The area was fertilised manually immediately after planting at a rate of 80 kg ground mineral phosphate (GMP) per ha - equivalent to 12 kg P per ha. This rate is low compared with the normal rate of 250 kg GMP per ha. The catchment has an average peat depth of more than 2 m above bedrock of quartzite, schist and basic volcanic rock, and the peat typically has a gravimetric water content of more than 80 %. Rocks are found in some sections of the study stream bed. The depth of the water table fluctuates between 0.2 and 0.7 m from the soil surface. In the catchment, the mean annual rainfall is more than 2000 mm and the mean air temperature is about 11 ^{o}C. Hill slope gradients in areas A and B (Figure 1) are on average 8^{o} and range between 0^{o} – 16^{o}.

The volume of lodgepole pine upon harvesting in area B (Figure 1) was about 400 m^3 ha^{-1}. Bole-only harvesting was conducted in area B (Figure 1) from July 25th to September 22nd, 2005. The timber was harvested using a Valmet 941 Harvester, and some of the tree residues (i.e. needles, twigs and branches) were collected together to form the brash material mats, thus protecting the soil surface, and reducing erosion. The rest were left on the soil surface and collected together to form windrows. During harvesting, the boles were stacked beside the windrow for collection. A Valmet 840 Forwarder delivered the boles to truck collection points beside the forest road. To minimise soil damage, clearfelling and harvesting were conducted only in dry weather conditions during the period from July to September, 2005. This time period is recommended for harvesting since ground conditions tend to be drier (Forest Service, 2000). Mechanised operations were suspended during and immediately after periods of particularly heavy rainfall. Another important BMP used during the harvesting operation was the proper use of BMs for harvest machine travelling. In the lowest part of the site where the stream is deeply incised, the trees were cut with a chain saw and left behind. In the harvested area, the BMs formed from the harvest residues – lie parallel to the study stream and furrows, which is at right angles to the contours. The BMs are approxiametly 4 m wide. The distance between two adjacent BMs is about 12 m.

The surface water flows along the furrows and is collected by collector drains (Arrows in Figure 1) which join the study stream.

The second rotation of lodgepole pine was planted in December, 2005 at a density of 2,800 per ha with no cultivation and no new drainage. No fertilizer was applied in the replanting operation. A BZ was established by replanting birch, rowan, alder and willow (instead of pine), in a 15–20 m-wide strip on each side of the stream furrows, ribbons, drains and brash/windrows were left *in situ*. Very little revegetation growth was observed in the harvested area until late summer, 2008.

Sampling, measurement and data analysis

Water

From May 2005 to September 2009, water samples at the US and DS stations were taken hourly during flood events and, on selected days, in base flow conditions using an ISCO automated water sampler. Grab water samples were taken at the USC and DSC stations above and below the confluence of the study stream (Figure 1) about once every two weeks. Rainfall water samples were also collected by placing an open and clean plastic container near the DS station during storm events for P analysis. All water samples were frozen at -20 ^{o}C in accordance with standard methods (APHA, 1998) until water quality analyses were conducted. The following analyses were carried out on the water samples: TRP, (DRP – filtered using Whatman Cellulose Nitrate Membrane Filters (pore size 0.45 μm) - and TP - after digestion with acid persulfate – using a Konelab 20 Analyser (Konelab Ltd., Finland).

Soil

Sites of about 1 ha in areas A and B were chosen for soil sampling. Forty and thirty-eight 100-mm-deep soil cores consisting of the humic and upper peat layers were collected using a 30-mm-diameter gouge auger from the ribbons in A and B in May 2005, April 2006, March 2007, April 2008 and March 2009. 15, 26, 25 and 28 more soil cores were taken under the windrow/ brash in the DS harvested area in April 2006, March 2007, April 2008 and March 2009, respectively.

Since the brash mats/ windrows - formed from the harvest residues – are parallel to the study stream and furrows, and along the slope, P from the BMs didn't enrich the brash-free soil. Soil samples were analysed for gravimetric water content and WEP. The core samples were placed in bags, hand mixed until visually homogenised, and subsamples of approximately 0.5 g (dry weight) were removed and extracted in 30 ml of distilled deionised water, and measured for P using a Konelab 20 Analyser. The remaining core samples were dried to determine their gravimetric moisture contents (Macrae et al., 2005).

Analysis methods

Storm flow was defined as the total flow (including the base flow) from the time where stream flow begins to increase on the rising limb to the time when the flow on the falling limb intercepts the serparation line with a constant slope of 0.0055 L s^{-1} ha^{-1} h^{-1} (Yusop et al., 2006). Monthly TRP loading was calculated in base flow and storm flow periods as follows (Yusop et al., 2006):

$$Q_{TRP} = CQ \qquad \text{Equation 1}$$

where Q_{TRP} is monthly TRP load (μg $month^{-1}$); C is the discharge-weighted mean concentration (μg L^{-1}) and Q is the total flow (L $month^{-1}$). For each month, C (μg L^{-1}) values at base flows and storm flows were calculated separately, using the following equation (Fergusson, 1987):

$$C = \sum_{n=1}^{n} c_i q_i / \sum_{i=1}^{n} q_i \qquad \text{Equation 2}$$

where C is the instantaneous concentration (μg L^{-1}), q the corresponding discharge during sampling (L s^{-1}) and n is the number of low flow or storm flow samples in the respective month. Finally, the annual loading is calculated as the summation of monthly loadings during both low and storm flow periods.

The TRP loads were calculated using the following linear equation:

$$Q_{TRP} = \alpha Q + \beta \qquad \text{Equation 3}$$

where Q_{TRP} represents the TRP yield (μg), Q is the water discharge (L), and α (μg L^{-1}) and β (μg) are obtained by the least squares method using observed TRP yield and water discharge data. At the DS station, the values of α and β in the base flow and storm flow were calculated for the following periods: August 2005 - July 2006, August 2006 - July 2007, August 2007 - July 2008, and August 2008 - July 2009. At the US station, because there was no significant change during the study period, the values of α and β in the base flow and storm flow were calculated from August, 2005 to July, 2009.

The differences in WEP in soil in kg ha^{-1} between the areas without windrow and with windrow were calculated by assuming that windrow comprises 25 % of the harvested area and that soil density is similar in areas below windrow and without windrow. The difference between the daily TRP concentrations at the US and DS in the first four years after harvesting was analysed using a paired samples t-test at the 95 % significance level (p = 0.05). The difference between the soil WEP in A and B before harvesting was analysed using an independent samples t-test at the 95 % significance level (p = 0.05). After harvesting, the differences between the soil WEP in (1) area A and the BM area in B and (2) under the BM and in BM area in B were also analysed using an independent samples t-test at the 95 % significance level (p = 0.05). All t-tests were carried out using the SPSS statistical tool (SPSS version 18, 2010).

Results and discussion

General P concentration trends in the stream water after harvesting

The average P concentrations in the rainfall were 13 ± 6 μg L^{-1} of TP and 4 ± 3 μg L^{-1} of TRP. Measured P concentrations at the US station were low during the study period, with average values of 14 ± 10 μg L^{-1} of TP and 6 ± 5 μg L^{-1} of TRP, which were close to the values in the rainfall. Four weeks after harvesting operations began, daily discharge-weighted mean P concentrations at the DS station started to increase and increased gradually to about 123 μg L^{-1} of TP and 73 μg L^{-1} of TRP at the end of harvesting period, and, on 28th October, 2005, they reached peak mean concentrations of about 201 μg L^{-1} of TP and 183 μg L^{-1} of TRP (Figure 2). By the end of December 2005 – 10 weeks after clearfelling - they decreased to about 15 μg L^{-1} of TP and 10 μg L^{-1} of TRP (Figure 2). From the end of July to the middle of August 2006, P concentrations at the DS station increased dramatically to 530 μg L^{-1} of TP and 429 μg L^{-1} of TRP - the highest concentrations recorded in the study (Figure 2). The release pattern of P concentrations - increasing to a clear peak after harvesting, experiencing a distinct declining tail, and then increasing to the maximum peak in the next summer - was also observed by Cummins and Farrell (2003) in a study carried out in a blanket peatland forest in the west of Ireland. The maximum peak in the next summer after harvesting was also observed by Nieminen (2003) in a Scots pine-dominated peatland in southern Finland.

The P concentration peak in summer 2006 was followed by a long declining tail (Figure 2). The daily discharge-weighted mean P concentrations at the DS station reduced to less than 15 μg L^{-1} of TRP and 20 μg L^{-1} of TP in July 2009, four years after harvesting (Figure 2). Statistical analysis indicated that P concentrations at the DS station were significantly higher than that at the US station ($p = 0.05$) in the 4-year period following harvesting.

Linear regressions were established for DRP and TRP versus TP (Figure 3). TRP and DRP were about 87 % and 77 % of TP, respectively, which indicated that: (1) the majority of TP was reactive and (2) particulate P concentrations were low. Renou-Wilson and Farrell (2007) found that in water samples with high organic matter content, TRP may be equal to TP.

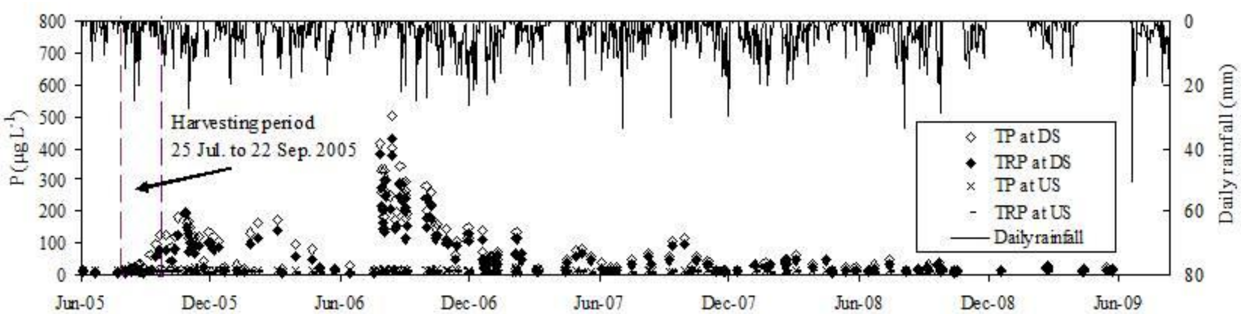


Figure 2 The daily rainfall and discharge-weighted mean total phosphorus (TP) and total reactive phosphorus (TRP) concentrations at the downstream (DS) and upstream stations (US) during the study period

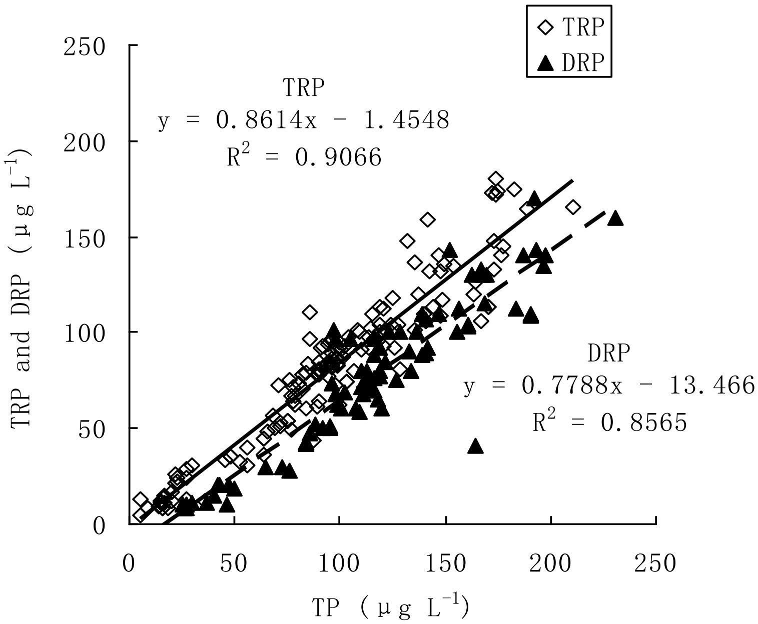
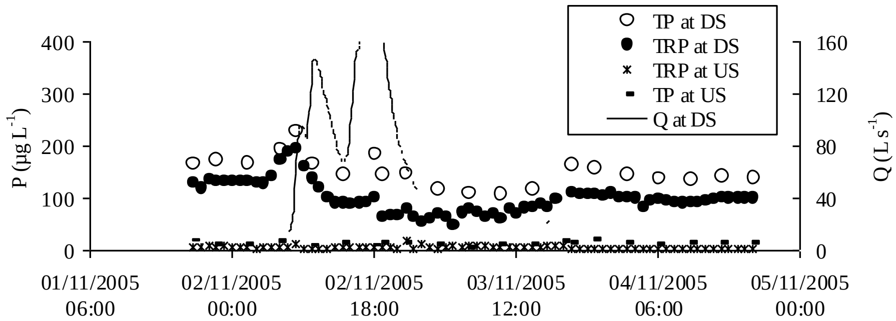


Figure 3 The relationship between the instantaneous concentrations of dissolved reactive phosphorus (DRP), total reactive phosphorus (TRP) and their linked total phosphorus (TP) concentration at downstream station (DS) during the study period

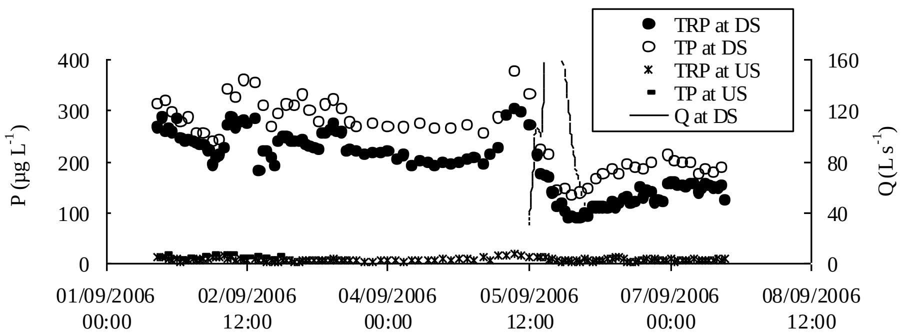
Effect of storm flow events

Over 120 storm events were analysed in this study. In addition to being influenced by the elapsed time after harvesting (Figure 2), P concentrations were also affected by the flow rates (Figure 4a – d). In over 80 % of the monitored storm events, P concentrations increased at the discharge rising stage, reached the maximum prior to the peak flow rate, and then reduced to a relatively stable value (Figure 4a – d).

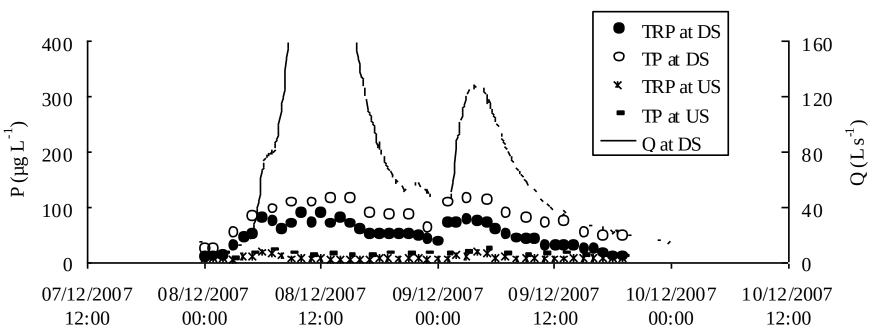
The P concentrations at the DS station increased from 131 μg L^{-1} of TRP, 220 μg L^{-1} of TRP and 20 μg L^{-1} of TRP at the beginning of the storm events in 2005, 2006, and 2007, respectively, to peak values of 193 μg L^{-1} of TRP, 300 μg L^{-1} of TRP and 100 μg L^{-1} of TRP, before reducing to about 80 μg L^{-1} of TRP, 110 μg L^{-1} of TRP and 30 μg L^{-1} of TRP at the end of the events. The major part of the P loading in receiving waters after harvesting activities was derived from the P movement from the topsoil to the stream during overland flow events (McDowell and Wilcock, 2004; Monaghan et al., 2007). Shigaki et al. (2007) and Quinton et al. (2001) found that high rainfall intensity resulted in a greater degree and depth of interaction between runoff and surface soil, including high runoff DRP concentrations, compared to what occurs during low rainfall intensities. The P concentrations were also affected by antecedent weather conditions. In the storm event of November 2nd 2005, peak TRP concentrations were 197 μg L^{-1}, 106 μg L^{-1} and 113 μg L^{-1} in Events 1, 2 and 3 (Figure 4a). The peak TRP concentration in Event 2 was lower than in Event 1, although the flow rate was higher, which could be due to less labile P sources being available for release in Event 2.



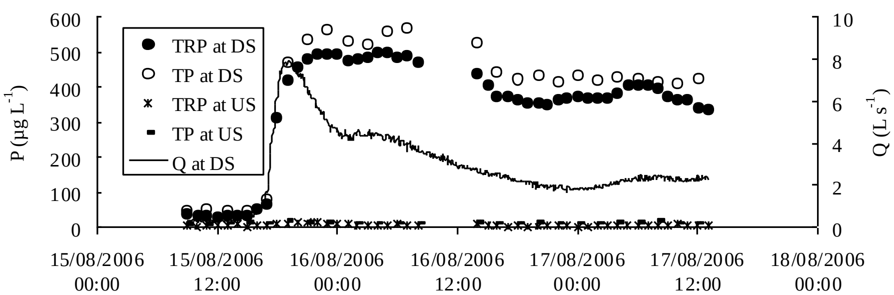
(a)



(b)



(c)



(d)

Figure 4 The instantaneous P concentrations at the upstream station (US) and downstream station (DS) with the instantaneous DS flow rate (Q) in four separate storm events (The maximum flow rate of the flow measurement equipment is 158 L s $^{-1}$).

When a storm event follows immediately after a previous storm event, much of the labile P has already been removed by the previous flood (Bowes et al., 2005). Similar phenomena were also observed in other storm events. The highest P concentrations observed during this study occurred in August 2006 (Figure 4d). There was a drought period before this storm event, resulting in little release of P by hydrological flushing, and large amounts of the labile P pool had accumulated. The TRP concentrations increased from about 10 $\mu g\ L^{-1}$ to about 550 $\mu g\ L^{-1}$ when the flow rate increased from about 0.5 $L\ s^{-1}$ to the peak of 8 $L\ s^{-1}$ (Figure 4d). The P concentrations maintained high values at the end of the storm event, which could be due to the relatively small water discharge that couldn't remove the large amount of mobile P that had accumulated before the storm event.

P concentrations in downstream river

In the present study, the P concentration at the DS station in the study stream did not have a large impact on the P concentration in the main river, which covers an area of 200 ha above its confluence with the study stream and should have a dilution factor of about 8 for the study stream. In their study, Cummins and Farrell (2003) found that the study streams had P concentrations well above critical levels for eutrophication, but they didn't know what implications these pollutions had for downstream river-water quality in larger channels. When the TRP at the DS station increased from about 3 $\mu g\ L^{-1}$ to 292 $\mu g\ L^{-1}$, the TRP concentrations at the DSC increased from about 5 $\mu g\ L^{-1}$ to about 11 $\mu g\ L^{-1}$, giving a measured dilution factor of about 26 (Figure 5). The higher iron concentrations and pH in the main river, which could increase P precipitation (Seida and Nakano, 2002), might contribute to the higher measured dilution factor.

Phosphorus loads

Annual TRP loads from the control area were steady and low during the study period, with values of less than 60 $g\ ha^{-1}$. A total of about 5.15 $kg\ ha^{-1}$ of TRP was released from the harvested area in the four years after harvesting, and mainly occurred in the first three years (Figure 6). The highest TRP load of 2303 $g\ ha^{-1}$ was recorded in the second year after harvesting

(Figure 6). During the study period, more than 80 % of annual water discharge occurred in storm flow (Figure 7). Due to the large water discharges, most of the TRP was released in storm events. In the 1st, 2nd, 3rd and 4th years after harvesting, it was calculated that the respective annual storm-flow TRP releases were about 80.3 %, 85.2 %, 82 % and 80.9 % of the total annual TRP release.

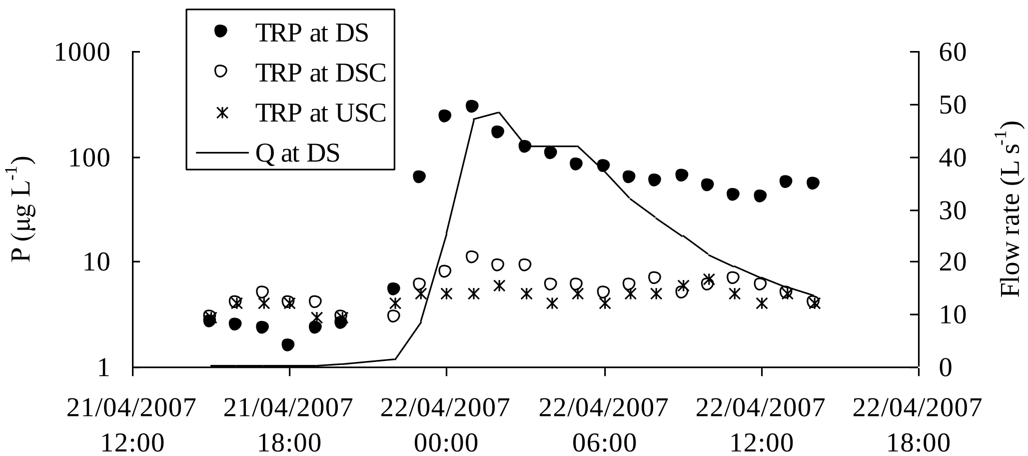


Figure 5 The instantaneous total reactive phosphorus (TRP) concentrations at downstream station (DS), downstream of the confluence (DSC) and upstream of the confluence (USC) with DS flow rate (Q) in a storm event

A linear regression equation (Equation 3) can be used to describe the relationship between the TRP load (*α)* and water discharge (β). Parameter *α* was high in the first three years following harvesting, indicating that more P was released in the first three years (Table 2.1). The TRP loads from the harvested area, estimated using Equation 3, were 1920 g ha^{-1}, 2893 g ha^{-1}, 1146 g ha^{-1} and 370 g ha^{-1}, respectively, in the 1st, 2nd, 3rd and 4th years after harvesting, which were close to the values calculated by Equation 1.

Water extractable P concentrations of the soil after harvesting

The independent samples t-test indicated that (1) before harvesting (in May, 2005), the difference between the WEP concentrations in area A and area B was not significant; (2) after harvesting (in April, 2006 and March, 2007), WEP concentrations were significantly higher in the brash/windrow-free soils in area B than in area A (p = 0.05); (3) in the harvested area B, the WEP concentrations under the windrows/brash were significantly higher than those in the windrow/brash-free area in April 2006, March 2007, April 2008 and March 2009 (p = 0.05) (Figure 8).

Table 1 The values of α and β in base and storm events in Equation 3 at downstream (DS) and upstream (US) stations during the study period

Time	Storm flow				Base flow			
	Catchment	α (μg L^{-1})	β (μg)	R^2	Catchment	α (μg L^{-1})	β (μg)	R^2
August 2005 - July 2006	DS (262)	95.8	20.6	0.85	DS (132)	192	-191	0.73
August 2006 - July 2007	DS (577)	129.8	211.1	0.85	DS (151)	141	153.6	0.68
August 2007 - July 2008	DS (266)	63.9	-284	0.9	DS (171)	77.2	-26.8	0.74
August 2008 - July 2009	DS (234)	12	106	0.41	DS (143)	8.9	91	0.55
August 2005 - July 2009	US (432)	4.5	12.6	0.77	US (379)	6.9	23.7	0.63

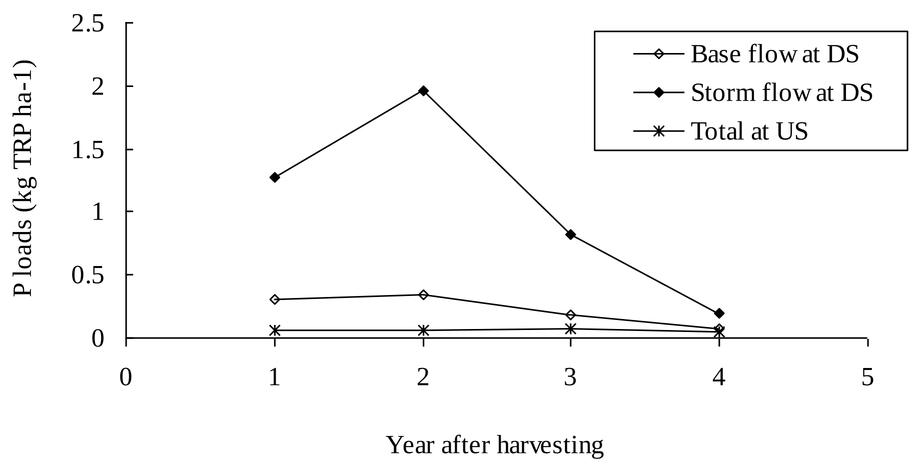


Figure 6 Yearly total reactive phosphorus (TRP) loads from the control site (US) and from the harvested area (DS) in base flow and storm flow after harvesting

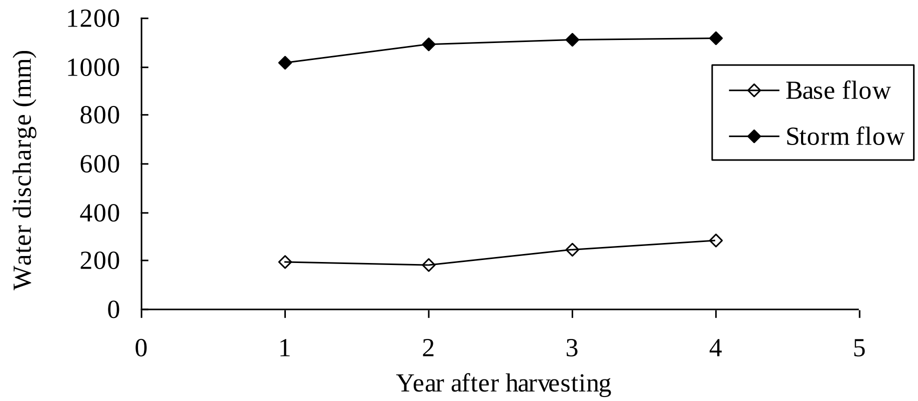


Figure 7 Yearly water discharged from the harvested area in base flow and storm flow after harvesting

Before harvesting, the WEP values in the US and DS areas were similar, at about 17 and 18 mg (kg dry soil)$^{-1}$. Most of this P was cycled in the forest system since very little P was leaving the catchments in runoff (Figure 8). It has been reported that in undisturbed forests, nutrients are effectively retained in the ecosystem, and leaching of P to receiving water is small (Mattson et al., 2003; Finér et al., 2004). After harvesting, both WEP in the soils covered and not covered by brash/windrow material increased, reaching peaks of 67 and 40 mg (kg dry soil)$^{-1}$, respectively, in 2007 (Figure 8). The WEP under the windrows/brash was about 136 %, 152.3 %, 235 % and 188.9 % of the WEP in the windrow/brash-free soil in 2006, 2007, 2008 and 2009, respectively. Higher WEP concentrations, found under the windrows/brash material, were due to P release from decomposing logging residues. The WEP was 1.5 kg ha^{-1}, 2.5 kg ha^{-1}, 1.8 kg ha^{-1}, and 1.3 kg ha^{-1} under the windrow/ brash material in 2006, 2007, 2008, and 2009, respectively, accounting for about 31 %, 36 %, 39 %, and 34 % of the total WEP in the harvested area. This observation is for soil only and ignores P remaining in the decomposing brash mats/windrows.

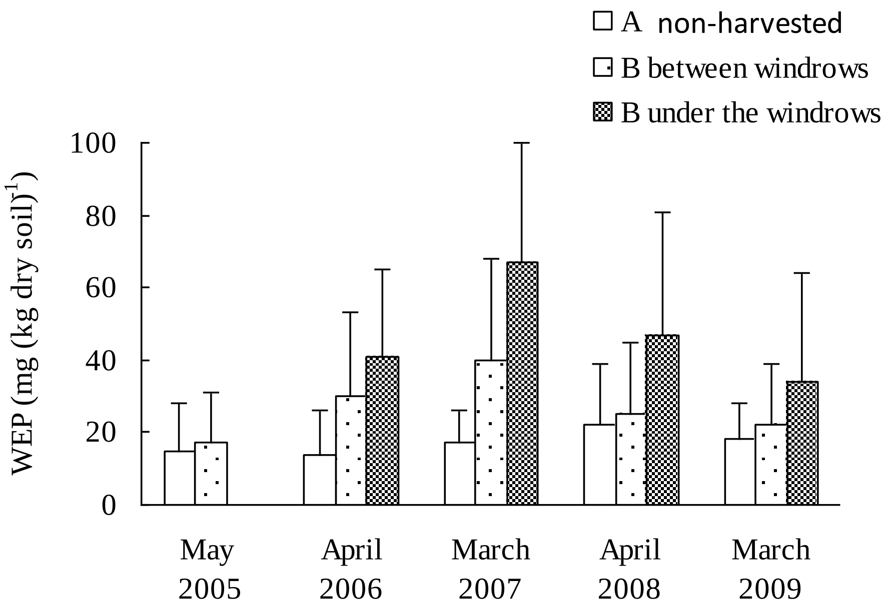


Figure 8 Soil water extractable phosphorus (WEP) in non-harvested (A) and harvested areas (B) between the windrow and under the windrow in May 2005, April 2006, March 2007, April 2008 and March 2009 (The bars indicate the standard deviation)

Hyvönen et al. (2000) found that the logging residues may contribute 8 - 31 kg ha^{-1} of TP to the harvested area. The high WEP value under the windrows/brash material lasted longer than for the windrow-free areas, which could be due to the relatively low decomposition rates of bark and branches (Ganjegunte et al., 2004). This indicates that whole-tree harvesting could, at least to some extent, be used as a means to decrease P release from blanket peats. Since the windrows/brash lie parallel to the furrows and are at right angles to the contours, it is unlikely that the WEP in the BM free area is influenced by P release from the brash area. The increase in WEP in the soil not covered by the windrow/ brash material after harvesting could be due to the decay of the surface organic layer, dead fine roots and lack of plant uptake. After clearfelling, a rise in soil temperature due to increased light penetration to the forest floor can increase decomposition rates (Messina et al., 1997; Perison et al., 1997), which increases the labile P sources (Walbridge and Lockaby, 1994).

Besides the increase in labile P after harvesting, P transport is also strongly linked to the P adsorption capacity of the soil (Tamm et al., 1974). In a clearfelled and harvested catchment covered with Podzols, Yanai (1998) found that, although the leaching of P from the forest floor organic residues to the soil was 0.7 kg ha^{-1} $year^{-1}$, only 0.07 kg ha^{-1} $year^{-1}$ was released from the soil in stream water and sediment, due to high P adsorption capacity of the soil. However, in the present study on upland peat soils, P concentrations in the study stream at the DS station increased significantly after harvesting (Figure 2), with most of the P loading taking place during storm events. In wet climates such as in Ireland, there is a great risk of P release from peat forests, due to a combination of poor P adsorption capacity in the peat soil, high runoff, and P sources being available after harvesting.

Possible mitigation methods

This study showed that the harvesting of the blanket peat forest increased the TRP export in the study stream, and this impact could last for more than four years. Phosphorus concentrations increased from 6 μg L^{-1} of TRP pre-clearfelling to a peak value of 429 μg L^{-1}, one year after harvesting. The results of this study were comparable to those of Cummins and Farrell (2003), who monitored P concentrations in forest drains and streams on blanket peatland in western Ireland weekly from 1996 to 2000, by using continuous depth proportional passive sampling. Their study catchment had similar soil type and weather condition as the present study. They found that catchment harvesting led to substantial increases in P concentrations. The MRP in their three study drains with the areas of 100 ha, 1 ha and 1 ha increased from 9 μg L^{-1}, 13 μg L^{-1} and 93 μg L^{-1} before harvesting to peak values of 265 μg L^{-1}, 3530 μg L^{-1} and 4164 μg L^{-1}, respectively, one year after harvesting (Cummins and Farrell, 2003). In Finland, Nieminen (2003) also found that, due to low Al and Fe content of peat, harvesting of peatland forest increased the leaching of P. As most of the blanket peat forests planted in the UK and Ireland before the 1980s are reaching their harvesting age, efficient and feasible practices are required to minimise the possible P release after harvesting to receiving waters.

In order to reduce nutrient sources after harvesting, WTH is recommended (Nisbet et al., 1997). Needles and branches have much higher nutrient concentrations than stem wood. Whole tree harvesting may reduce nutrient sources by 2 to 3 times more than bole-only harvesting (Nisbet et al., 1997). This study found higher WEP contents in harvested areas below windrow/brash material than for the brash-free sites, indicating that WTH could be used as a means to decrease P release. A BZ is an area adjacent to an aquatic zone and managed for the protection of water quality (Forest Service, 2000). Within BZs, natural vegetation and/ or planted suitable tree species are allowed to develop. Buffer zones have been widely used by forestry practitioners in the protection of freshwater aquatic systems (Newbold et al., 2010). It can protect aquatic systems by controlling runoff using the following methods: (1) mechanically, by increasing deposition through the slowing down of flow; (2) chemically, through reactions between incoming nutrients and soil matrices and residual elements; and (3) biologically, through plant and microbial nutrient processes. However, this study shows that traditional BZs with a width of 15-20 m may not be an efficient method to mitigate the P release from all harvested areas, since, in this study, about 80 % of TP in the study stream was soluble (Figure 3) and more than 70 % of the P release occurred in storm events when there would have been low residence times for the uptake of soluble P in the BZs. If BZs are used to mitigate P release, larger buffer areas than those used presently may be needed (Väänänen et al., 2008).

Phased felling and limiting catchment size to minimise negative effects has been recommended in the UK (Forestry Commission, 1988) and Ireland (Forest Service, 2000). Harvesting reduced catchment sizes at any one time can reduce the nutrient concentrations on aquatic systems. This study found that due to the dilution capacity of the main river, the P concentrations in the river were low after harvesting; indicating that catchment-based selection of the harvesting coupe size could limit the P concentrations in the receiving waters after harvesting. However, the management strategy does not reduce the total P load leaving the harvested catchment. Before the replanted trees mature, vegetation could immobilise the nutrients throughout the harvested catchment. As ground vegetation develops, P uptake and recycling can be expected to diminish leaching over time (Piirainen et al., 2007). In the present study, vegetation such as *Molinia caerulea* cover was observed in 2008 and became well established in 2009. Since the development of the vegetation, P release to the receiving water was reduced, though the WEP in the harvested area was still high. It could take 3 to 4 years for natural re-vegetation of the blanket peat harvested forest area to occur. Stimulation of vegetation cover immediately after harvesting, e.g., through seeding the harvested area with faster growing native grasses, should also be studied as a practice to minimise the P release from the blanket peat forest after clearfelling.

Conclusions

This study showed that the harvesting of the blanket peat forest increased the TRP export in the study stream, and this impact could last for more than four years. In the first three years following harvesting, up to 5.15 kg ha^{-1} of TRP were released from the catchment to the receiving water; in the second year alone after harvesting, 2.3 kg ha^{-1} were released. P concentrations increased from 6 μg L^{-1} of TRP during pre-clearfelling to a peak of 429 μg L^{-1} one year after harvesting. About 80 % of TP in the study stream was soluble and more than 70 % of the P release occurred in storm events. Due to the dilution capacity of the main river, the P concentrations in the river were low during the study period, indicating that rational sizing of the harvesting coupe could be an efficient practice to limit the P concentration in the receiving waters following harvesting. However, the study comprised only one experimental catchment. In future research, a paired catchment approach should be investigated.

Acknowledgements

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Bibliography

See Chapter Six pp. 187 - 222.

Chapter Three

Impacts of forest harvesting on the ecology of receiving waters

3.1 Introduction

This chapter comprises three research papers, which are part of the project SANIFAC (RSF 07 552) funded by COFORD and the Irish Department of Agriculture, Fisheries and Food. Dr. Michael Rodgers and Dr. Liwen Xiao were the Project PI and Project Coordinator, respectively, and contributed to the overall project design.

The first paper has been published in the peer-reviewed, international journal *Ecological Indicators* (O'Driscoll et al., 2012. Diatom assemblages and their associated environmental factors in upland peat forest rivers. *Ecological Indicators*, 18: 443-451). Connie O'Driscoll collected, analysed and synthesised experimental data, and was the primary author of this article. Dr. Elvira de Eyto assisted with data analysis and paper editing; and Mark O'Connor and Zaki ul Zaman Asam assisted with sampling.

The second paper "Influence of natural variation for biomonitoring in upland peat rivers" has been reviewed and provisionally accepted by the peer-reviewed, international journal *Hydrobiologia*. Connie O'Driscoll collected, analysed and synthesised experimental data, and was the primary author of this article. Dr. Elvira de Eyto and Dr. Martyn Kelly assisted with analysis and editing, Mark O'Connor and Zaki ul Zaman Asam assisted with sampling.

The third paper "Responses of macroinvertebrates and diatoms communities in acid sensitive and oligotrophic streams to peatland forest clearfelling" has been submitted to *Ecological Indicators*. Connie O'Driscoll collected, analysed and synthesised data, and was the primary author of this article. Dr. Elvira de Eyto assisted with analysis and editing, Mark O'Connor and Zaki ul Zaman Asam assisted with sampling.

3.2 Diatom assemblages and their associated environmental factors in upland peat forest rivers

Connie O'Driscoll[a], Elvira de Eyto[b], Michael Rodgers[a], Mark O'Connor[a], Zaki-ul-Zaman Asam[a], Liwen Xiao[a*],

[a] Civil Engineering, National University of Ireland, Galway, Ireland

[b] Marine Institute, Newport, Co. Mayo, Ireland

* Corresponding author: Liwen Xiao; Email: liwen.xiao@nuigalway.ie; Tel: +353 91 494188

Abstract

Acid-sensitive upland blanket peat catchments are important habitats for diatom assemblages. In this study, the distribution patterns of epilithic diatom assemblages in the streams of upland forested blanket peat in the north-west of Ireland are presented and the associated environmental factors are discussed. A total of 43 sites in 16 rivers were sampled. Multivariate analysis highlighted alkalinity and EC as the main physicochemical drivers of riverine diatom assemblages. Contrary to expectations, nutrients were not found to have a major influence on the diatoms. A major flood event had a significant impact on the diatom assemblage, and one year after the event, long stalked diatom taxa were still largely absent from the river, indicating that floods could be one of the important factors affecting diatom assemblages. However, the ecological status of the affected sites, as determined by the EQR, did not alter from before to after the flood. The results of this study could be applied to similar acid-sensitive upland peat forest catchments and used as the benchmark to assess the impact of forest operations and peat degradation on ecological status.

Introduction

The WFD requires EU Member States to achieve 'good ecological status' for all water bodies by 2015 (European Union, 2000). Assessing, maintaining and restoring good ecological status of aquatic ecosystems have become priorites for river basin management and water protection in

Europe (Eloranta and Soininen, 2002; Kelly and Wilson, 2004; Leira and Sabater, 2005; Kelly et al., 2008; Urrea and Sabater, 2009). Diatoms have been used successfully as indicators of the ecological quality of aquatic ecosystems worldwide (Kelly et al., 1998; Leira and Sabater, 2005; Hering et al., 2006; Chen et al., 2008). This is due to their (1) being well established in the food web (2) responding rapidly to the majority of physical, chemical and biological changes in water bodies and (3) having a one-stage life cycle and very short generation time (Stevenson and Pan, 1999). Many diatom indices such as the Trophic Diatom Index (TDI) in the UK (Kelly and Whitton, 1995) and Germany (Coring et al., 1999); the Trophienindex (TI) in Austria (Rott et al., 1999) and the Indice Biologique Diatomique (IBD) in France have been developed for the assessment of trophic conditions. However, to meet the requirements of the WFD, trophic indices must be compared to reference conditions. Ecological quality ratios (EQRs) have recently been derived for diatom indices, specifically the TDI, to allow the observed TDI to be compared with an expected reference index value (Kelly et al., 2006). For the EQR to be considered reliable and to meet WFD requirements, representative reference conditions need to be defined for a complete range of ecoregions and habitats. However, acid-sensitive and low alkalinity sites are underrepresented in the EQR (Kelly-Quinn et al., 2004; Kelly et al., 2006). The importance of characterising diatom assemblages in low alkalinity sites has been highlighted (Camburn and Charles, 2000; Tolotti, 2001; Cantonati and Lange-Bertalot, 2011).

Upland peat catchments in north-western Europe are characterised as acid-sensitive areas. These areas contain the headwaters of rivers, many of which contain Red List species (e.g. salmonids and freshwater pearl mussels) which make them important biodiversity refuges. The main pressures to the rivers in these acid-sensitive areas include forestry operations and peat degradation. Since the 1950s, large areas of upland blanket peat have been afforested in north-western European countries. Risk assessments on receiving waters have shown that forest operations result in increased P release (Nisbet, 2001; Cummins and Farrell, 2003; Nieminen, 2003; Rodgers et al., 2010). Acidification of surface waters draining forested catchments is also a concern (Jenkins et al., 1990; Ormerod et al., 1991; Allott et al., 1997). Peat degradation and projected climate change lead to increased dissolved organic carbon (DOC) export, decreased pH in receiving waters and increased flood events (Fealy et al., 2010; Cantonati and Lange-Bertalot, 2011). In view of these multiple and increasing pressures, it is crucial to have a thorough understanding of the diatom assemblages which will be used in assessing ecological change in the rivers draining these acid-sensitive upland peat catchments.

The main purpose of this study was to characterise the diatom assemblages of these rivers and ascertain the environmental drivers of assemblage composition. To the best of our knowledge, there is no research focused on the diatom assemblages and their associated environmental factors in oligotrophic rivers draining upland forested blanket peat. The rivers in upland blanket peat catchments are usually spatey and prone to flash flooding exhibiting a quick response time to precipitation (Müller, 2000). The diatom assemblages are, therefore, likely to contain taxa well adapted to high flows and frequent changes in water level. It is unknown, however, how these assemblages respond to extreme flood events and the implications of flood events on the ecological status of sites. During the sampling period of this study, an extreme flood event occurred in one of our study rivers, where 52 mm of rain fell in two hours (Dalton et al., 2010). Such events are considered to occur once in 250 years (Fealy et al., 2010). This provided a fortuitous opportunity to assess the impact of an extreme flood on diatom assemblages.

Materials and methods

Study sites and characterisation

This study was based in three adjoining catchments, located in Mayo in the north west of Ireland (Figure 1). Most of the catchments are covered in blanket peat and overlie quartzite and schist bedrock. The catchment systems are described as acid oligotrophic and have a low buffering capacity (Byrne et al., 2004). The main land uses are forestry and sheep grazing and the catchments receive an average precipitation of over 2000 mm per year (Dalton et al., 2010). Commercial coniferous plantations were planted in blocks or coupes starting in the 1950s (Chapter 4, section 4.2). A total of 43 sites in 16 rivers were selected for this study, 36 located in forested areas and the other 7 sites in un-forested peatland.

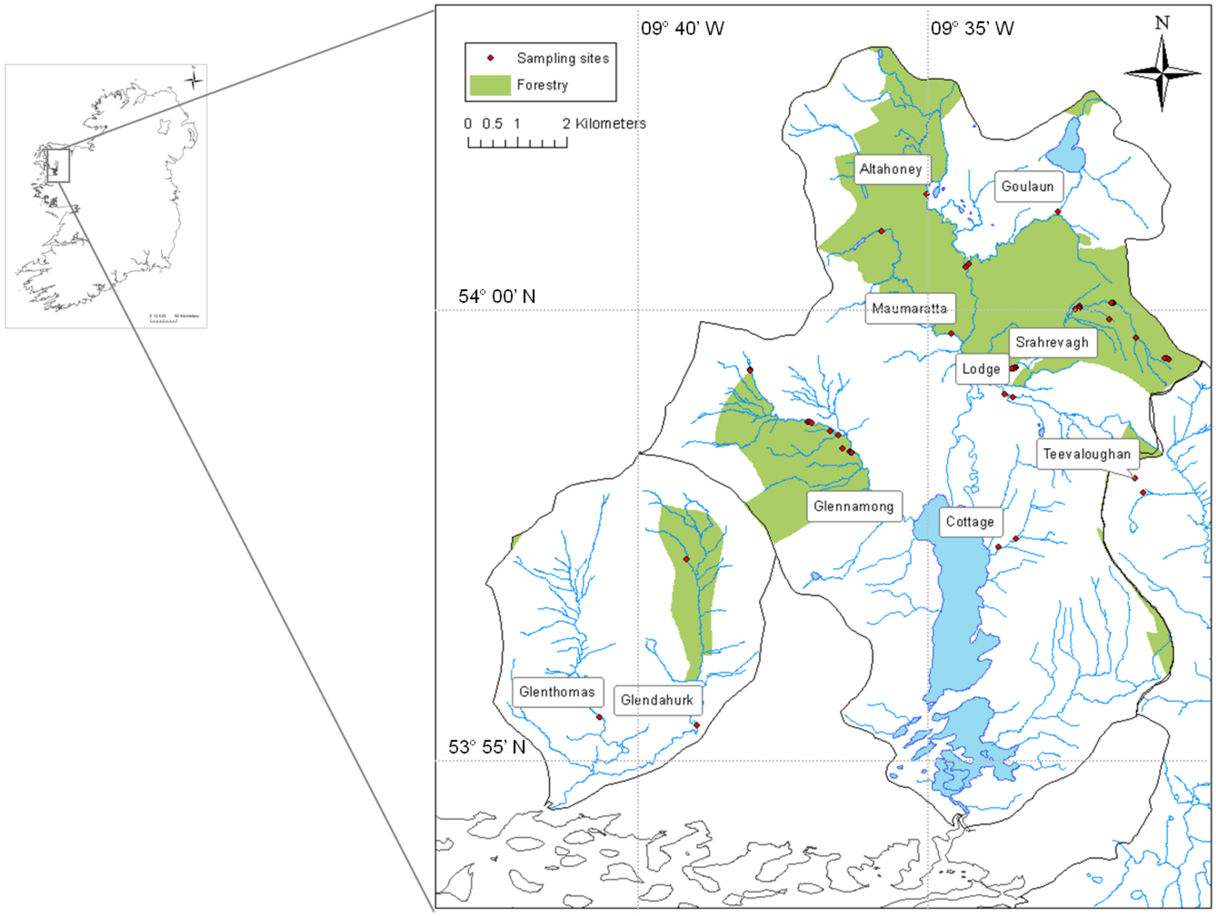

Figure 1 The locations of 43 sites where riverine diatom assemblages were sampled in 2009 in the North West of Ireland

Diatoms were sampled in June 2009. Water samples were taken at each sampling site and analysed on the same day for alkalinity and colour using standard procedures (APHA, 1998). Phosphorus, nitrate-nitrogen and ammonium were measured using a Konelab 20 Analyser (Konelab Ltd., Finland). Water temperature, pH and conductivity were recorded in the field, and altitude, catchment area, stream order (Strahler, 1957), and Shreve stream order (Shreve, 1966) were extracted from GIS maps of the areas. All the sites were sampled in baseflow conditions and so diatoms and environmental variables are representative of conditions at base flow. A subset of samples was taken the following year to examine the impact of an extreme flood event on diatom assemblages in one of the study rivers. On the 2nd of July 2009, 52 mm of rainfall was recorded in two hours, an event considered to occur once in 250 years (Fealy et al., 2010).

Twelve samples were taken in the Srahrevagh and Glennamong rivers in summer 2010, one year after the flood event. The Glennamong is in a neighbouring sub-catchment and was un-impacted by the flood event and so is treated as a control.

Diatom sampling, preparation and identification

Diatoms were collected and prepared in accordance with Kelly et al. (1998). Taxa were identified to species level where possible and counted at x1000 magnification using an Olympus BX-51 microscope equipped with a x100 phase contrast objective (numerical aperture: 1.25). At least 300 valves were identified and counted per slide (Krammer and Lange-Bertalot, 1986, 1997, 2000, 2004). Certain taxa were difficult to identify and the approaches adopted for these species were as follows: *Achnanthidium minutissimum* varieties were split into types based on Potapova and Hamilton (2007). Three types were found in these samples: the 'capitate' morph which corresponds to 'type a' in these samples; 'linear' corresponds to 'type b' and 'narrow-linear' corresponding to 'type c'. These three *Achnanthidium* groups were largely present in girdle view, and so were enumerated separately in girdle view and then divided between the three morphological groups based in proportion to their relative abundance. *Eunotia exigua*, also present in high numbers in girdle view, was difficult to distinguish from *Eunotia tenella* and *Eunotia meisteri* and so the three were combined and considered as *E. exigua* complex. *Gomphonema parvulum* has been described with a number of varieties and attributed environmental preferences. However, populations in these samples had high morphological variability, and so have been termed *G. parvulum* complex.

Data analysis and statistics

Diatom species richness (S) was determined from the number of species counted on each slide. The Shannon–Weiner index (H^1), which measures the proportional abundances of species in a community, was calculated (Shannon and Weaver, 1963). In order to relate the diatom community to water quality, the revised TDI, which quantifies the impact of nutrients on diatom assemblages, was calculated for each site (Kelly et al., 2008). The scores range from 0 (very low nutrients) to 100 (very high nutrients). The TDI was compared with reference assemblage characteristics and the EQR was calculated taking into account the relationship between nutrient gradients and diatom assemblages with season and alkalinity (Kelly et al., 2006). An EQR close to 1 indicates an un-impacted diatom assemblage, whereas a value close to 0 indicates a severe impact (Kelly et al., 2006).

Relationship of diatom assemblages with environmental variables

Diatom data were first analysed with detrended correspondence analysis (DCA) (Hill and Gauch, 1980) to determine the gradient length. The length of the gradient was greater than 2 standard deviation units (4.641), and so unimodal ordination techniques were chosen (ter Braak, 1987). Correspondence analysis (CA) was used to determine the major patterns of variation in species composition data, following which canonical correspondence analysis (CCA) was used to relate diatom assemblages to all predictor environmental variables (ter Braak and Verdonschot, 1995). Data exploration highlighted significant correlation between pH and alkalinity, and so pH was dropped from further analyses. To reduce further the environmental variables to those correlated significantly with the derived axes, step-wise forward selection and Monte Carlo permutation tests were used. Only those taxa that were observed in more than 5% of the samples were included in analyses of taxa abundance to minimise the influence of rare taxa. Taxa abundance was square root transformed in all analyses to reduce the effect of highly variable population densities on ordination scores. Environmental variables were appropriately transformed before analysis to reduce skewed distributions and all ordinations were performed using CANOCO version 4.1 (ter Braak and Šmilauer, 1998).

Impact of extreme flood event on diatom assemblages

To assess the impact of the extreme flood on diatom assemblages, a similar analysis as that for determining the relationship of diatom assemblages with environmental variables was adapted.
The Srahrevagh and Glennamong in 2009 and 2010 were used as environmental variables corresponding to before and after the flood respectively. Taxa abundance was square root transformed to reduce the effect of highly variable population densities on ordination scores.

Results

Environmental characteristics of the sites

Nutrient concentrations in the study sites were very low, with the maximum PO_4-P, NH_4-N and NO_3-N concentrations of 6.27 µg L^{-1}, 122.12 µg L^{-1} and 97.02 µg L^{-1}, respectively (Table 1). Based on the nutrient concentrations, all the study sites can be described as oligotrophic. Electrical conductivities of the sites were between 67 µS cm^{-1} and 251 µS cm^{-1} (Table 1). pH values of the sites ranged from 3.55 to 8.49 (Table 1). Alkalinity in the study sites varied between −2.7 mg L^{-1} $CaCO_3$ and 68.8 mg L^{-1} $CaCO_3$ (Table 1).

Table 1Physical and chemical characteristics of the sites

Sample ID	Electrical Conductivity (µS cm-1)	Alkalinity (mg l-1 $CaCO^3$)	$PO^4_{-3}P$ (µg L-1)	NH+4-N (µg L-1)	NO-3 -N (µg L-1)	pH	Colour (PtCo)	Temperature (°C)	Altitude (m)	Stream Order	Upstream Catchment Area (ha)	Shreve Index
max	251	69.4	6.48	120.05	164.03	8.50	284	21.0	359	4	1549	33
min	64	-4.2	2.94	2.31	0.28	3.54	32	11.2	20	1	10	1
mean	115	15.8	4.21	41.47	23.16	4.71	99	16.5	125	2	529	10
median	94	10.0	3.90	36.06	8.97	7.12	77	16.9	101	2	293	4
std dev	47	19.4	1.06	28.65	37.29	4.21	65	2.6	95	1	499	11

Diatom species composition

Of the 57 taxa found, 52 taxa were observed in more than 5 % of the samples. The five most abundant taxa were *Achnanthes oblongella*, *Fragilaria capucina* var. *gracilis, E. exigua*, *Tabellaria flocculosa*, and *A. minutissimum* Type A, present in 77 %, 60 %, 65 %, 60 %, and 67 % of the sites, respectively. Species richness ranged from a minimum of 5 to a maximum of 25. The Shannon–Weiner diversity index ranged from 0.40 to 2.79. Trophic Diatom Index values ranged from 1.7 to 46.7. There was no correlation between TDI and P concentrations, of which all values were less than 10 µg L^{-1} (Spearmans $rho = -0.049$, $p = 0.75$, d.f. = 41). There was a significant correlation between alkalinity and TDI (Spearman $rho = 0.396$, $p < 0.05$, d.f. = 41). Ecological Quality Ratios values ranged from 0.65 to 1.12, with a median value of 1.

Correspondance analysis ordination results showed that 20.2 % of diatom assemblage variance was explained on axis 1, with a further 12 % explained on axis 2 (Figure 2).

There were two clear groupings from this graph; the first in the top left hand side quadrat and the second in the bottom right hand side quadrat which cluster along a gradient. Taxa situated on the left side of diagram included *A. oblongella*, *F. capucina* var. *gracilis* and *G. parvulum* complex. Taxa situated on the right side included the *E. exigua* complex, *Pinnularia appendiculata*, *Eunotia paludosa* and *T. flocculosa*. Taxa with higher values on axis 2 included *E. paludosa* and *Eunotia microcephala*. Taxa with maximum abundances and lowest values on axis 2 included the *E. exigua* complex and *T. flocculosa*.

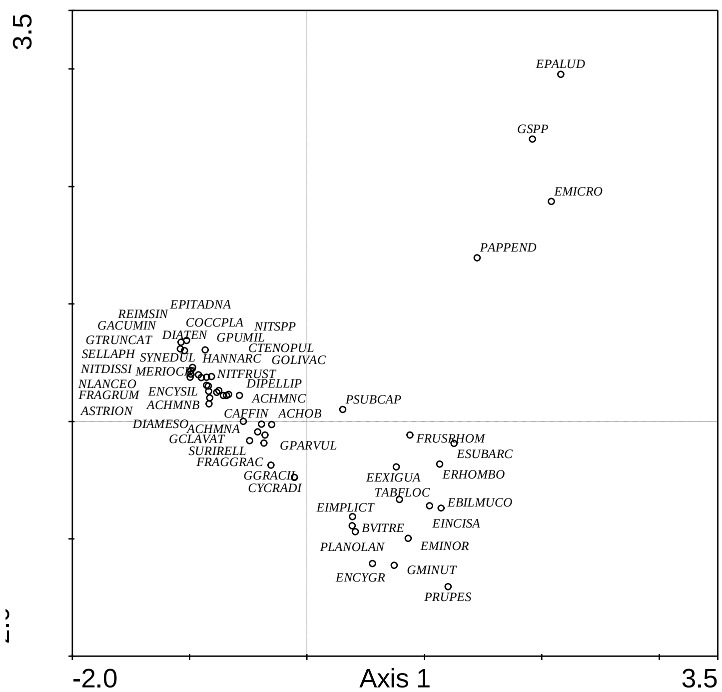

Figure 2 Location in ordination space (correspondence analysis, CA) of the first and second axis of diatom taxa. Taxa shown in the diagram were found in 5 % of all samples (taxa codes correspond to those in Appendix A).

Important environmental variables

The first two axes explained a significant proportion of variance in the diatom taxa data ($p < 0.01$). Canonical correspondence analysis with forward selection identified all variables except colour to be significant ($p < 0.05$) accounting for significant portions (60 %) of the total variance in diatom species composition (Table 2). Alkalinity explained the largest portion (30 %) of the total unconstrained variance. Canonical correspondence analysis ordination plots identified three groupings of sites (Figure 3a, b).

The first group comprised sites with high EC and alkalinity and featured *Reimeria sinuata*, *Gomphonema truncatum*, *Diatoma tenue*, *Cocconeis placentula* and *Ctenophora pulchella*. The second group comprised sites with high conductivity, low alkalinity, and first order streams with small upstream catchment area. These sites were dominated by *E. paludosa*, *P. appendiculata* and *E. microcephala*. The third group comprised sites with low conductivity, low alkalinity and higher stream order and upstream catchment area. *Brachysira neoexilis*, *T. flocculosa*, *E. exigua* complex and *Eunotia implicata* were the common taxa observed at these sites.

Table 2 Weighted correlation matrix showing the relationship between species axes and environmental variables (Figure 3 a, b). The environmental variables listed exerted significant (p < 0.05 influences on algal distributions

Variables	Axis 1	Axis 2	Axis 3	Axis 4
Conductivity (µS cm$^{?1}$)	-0.24	0.83	-0.04	-0.14
Alkalinity (mg L$^{?1}$ CaCO^{3})	-0.89	0.28	0.00	0.00
PO4-P (µg L$^{?1}$)	0.10	0.10	0.22	0.31
NH4-N(µg L$^{?1}$)	0.42	0.07	-0.05	-0.14
NO3-N(µg L$^{?1}$)	0.13	-0.20	-0.41	0.14
Temperature (°C)	-0.49	0.13	-0.34	0.09
Altitude (m)	0.21	0.01	-0.32	0.57
Stream Order	-0.35	-0.30	0.13	-0.39
Upstream Catchment Area (ha)	-0.08	-0.52	0.08	-0.50
Shreve Order	-0.21	-0.45	-0.02	-0.46
Percentage variance of species-environment relationship	34.20	19.80	11.60	8.10
Eigen values	0.60	0.34	0.20	0.14

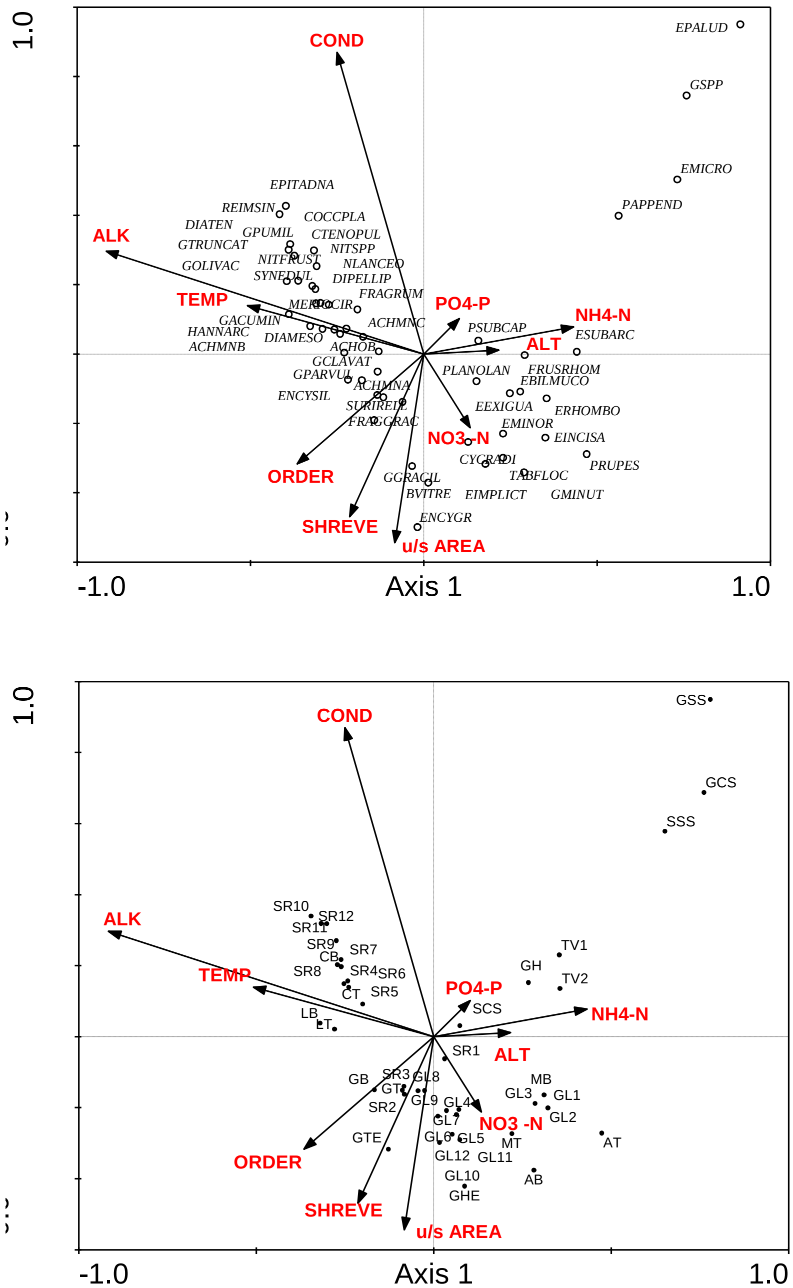

Figure 3a, b Canonical correspondence analysis (CCA) of diatom communities in streams of NW Ireland in the ordination space of first and second axis (ALK, alkalinity; ALT, altitude; COND, conductivity; TEMP, temperature; u/s AREA, upstream area); (a) ordination of diatom species, (b) ordination of sampling locations with significant and independent environmental variables (taxa and sample site codes correspond to those Appendices A and B).

Impact of extreme flood on diatom assemblages

The flood altered the diatom assemblages in the Srahrevagh (flood) (Table 3). The Glennamong (control), which was not affected by the flood, had five out of seven of the most common diatom species occurring in 2009 and 2010 (Table 3). Only one common species, *F. capucina* var. *gracilis*, found in 2009 was replaced by *Eunotia incisa* and *P. appendiculata* in 2010 (Table 3). However, in Srahrevagh, 4 of the 6 common species found in 2009 were replaced in 2010 (Table 3). Canonical correspondence analysis ordination results showed that the environmental variables explain 29 % of the species data (Figure 4).

Table 3 The main diatom taxa in the Glennamong and Srahrevagh rivers a week before and one year after the extreme flood event

	Glennamong		Srahrevagh	
		% of the total abundance		% of the total abundance
	Main diatom species		Main diatom species	
2009	*Achnanthes oblongella*	18.8	*Achnanthes oblongella*	23.7
	Fragilaria capucina var. *gracilis*	18.7	*Fragilaria capucina* var. *gracilis*	11.4
	Tabellaria flocculosa	17.2	*Fragilaria capucina* var. *rumpens*	9.2
	Achnanthidium minutissimum type A	13.2	*Diatoma tenue*	5.7
	Gomphonema parvulum	11	*Epithemia adnata*	5.5
	Eunotia rhomboidea	5.4	*Synedra ulna*	5.5
2010	*Eunotia rhomboidea*	21.9	*Achnanthes oblongella*	19.8
	Tabellaria flocculosa	16.7	*Achnanthidium minutissimum* Type A	16.5
	Achnanthes oblongella	10.7	*Fragilaria capucina* var. *gracilis*	9.6
	Eunotia incisa	7.1	*Gomphonema parvulum*	8.1
	Pinnularia appendiculata	6.7	*Reimeria sinuata*	6.9
	Gomphonema parvulum Complex	6.7	*Cocconeis placentula*	6.6
	Achnanthidium minutissimum Type A	6.3	*Planothidium lanceolatum*	5.4

Axis 1 separated the Glennamong and Srahrevagh rivers and explained 65.4 % of the variation. Axis 2 corresponds to the year 2009 and 2010 and a significantly larger variation can be seen in the impacted Srahrevagh river when compared to the Glennamong. This was owing to several species which were present in 2009 such as *G. truncatum*, *Gomphonema acuminatum*, *Epithemia adnata*, *Hannaea arcus* and *C. pulchella*, and disappear entirely from the samples taken 12 months later in 2010. All sites in the impacted and non impacted rivers had an EQR between 0.8 and 1.

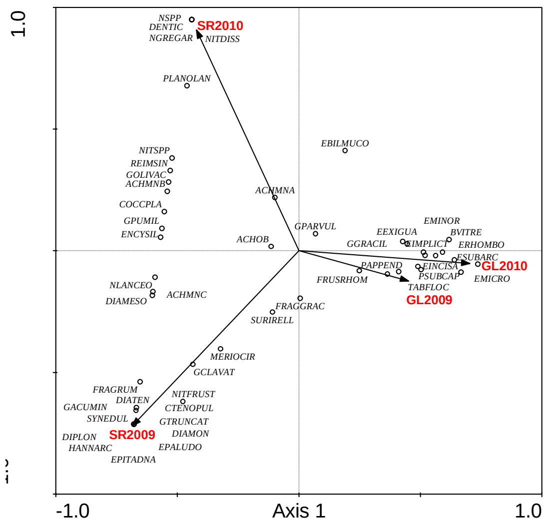

Figure 4 Canonical correspondence analyses (CCA) of diatom assemblages after a major flood in the ordination space of first and second axis

Discussion

Low alkalinity gradient

Multivariate analysis indicated that variables driving the distribution of diatom assemblages in the study catchments were associated with physiography, i.e. alkalinity, ionic concentration, stream order, upstream catchment area and Shreve stream order rather than nutrients. The geology of the western side of the study catchments is mostly comprised of quartzite and schist with the Glennamong, Altahoney and Maumaratta rivers being well represented by taxa with a preference for low pH such as the *Eunotia* genus. Representative diatoms include Red List species such as *E. implicata*, *Eunotia bilunaris* var. *mucophila*, *E. paludosa* and *Eunotia rhomboidea* (Lange-Bertalot, 1996). Cantonati and Lange-Bertalot (2011) highlight the importance of the *Eunotia* group and suggest in the threatened naturally acidic and low-alkalinity waters, *Eunotia* species might be play a fundamental role as indicators of ecological status. Even when the dominant geology is granite/schist, the presence of small veins of buffering minerals (dolomite and wacke) (Fealy et al., 2010) can considerably raise the alkalinity of receiving waters and hence cause a shift to more circumneutral assemblages. For example, *Gomphonema olivaceoides*, abundant in the Srahrevagh, is known to have an ecological preference for upland low nutrient streams with calcareous geology. In-stream longitudinal alkalinity gradients are represented by a shift from upper sites abundant in *A. oblongella* to sites further down the catchment with a more diverse spread of species such as *G. truncatum, R. sinuata* and *C. placentula*. Therefore, in peatland catchments the underlying geology is crucial in determining diatom assemblages. Low alkalinity in rivers is closely correlated with episodic acid pulses and the presence of many pH-tolerant diatom species indicates that acidification may be a main driver of assemblage composition. Anthropogenic acidification has been known to occur due to afforestation (Battarbee et al., 2010) and peat degradation and associated DOC leaching and increased pH (Monteith and Evans, 2005; Jennings et al., 2010). It is also possible that these assemblages represent naturally acidic conditions. However, while many diatom metrics exist to measure acidity (van Dam et al., 1994; Kwandrans, 2007; Andren and Jarlman, 2008), none consider the expected value of sites as required by the WFD. Nevertheless, characterisations of diatom assemblages in these acid sensitive waters will prove crucial in the future in determining acidification impacts.

Low nutrient concentrations

The rivers included in this study can all be defined as oligotrophic, as the annual P concentrations were less than 20 µg PO_4-P L^{-1}. 44 of the 57 diatom taxa found in the study sites are known to be nutrient sensitive (sensitivity values of 1 and 2; Kelly et al., 2005). Although the range of TDI values measured in this study includes some moderately high values (e.g. 40, 42.2, 46.7), the EQR which corrects for alkalinity was found to be close to 1 for all the sites, indicating a 'good' or 'high' ecological status. While the felling of forest sub-catchments in the study areas has been shown to increase the P and sediment loads in receiving waters (Rodgers et al., 2010), it appears that the ecological quality of the diatom assemblages has not been impacted. We can therefore support the characterisation of riverine diatoms in this area as oligotrophic and lacking any sign of anthropogenic nutrient enrichment from forestry activities. These sites therefore represent reference conditions with respect to nutrient status. However, this result should be treated with a certain amount of caution as work is still ongoing on the use of EQRs in Ireland, particularly in low alkalinity sites (i.e. <6.8 mg L^{-1} $CaCO^3$ (B. Kennedy, EPA Ireland, *pers. comm*.).

Spatey nature of rivers

It is highly likely that the lack of impact due to nutrient enrichment on the diatom assemblages is due to the quick flushing of nutrients through these spatey rivers. The diatom assemblages of first-order streams that are liable to flash flooding and receive high annual precipitation (>2000 mm) have high abundances of *A. oblongella*. This species attaches to the substrate by its valve face and mucilage which is likely to give it protection and resilience to floodwaters. *A. oblongella* showed resilience even to the major flood experienced in the Srahrevagh, having a maximum abundance both before and after the flood in 2009 at the upper first order sites. However, the long stalked species such as *G. truncatum*, loosely attached taxa such as *E. adnata* and *F. capucina var. gracilis,* which were abundant at the lower sites before the flood, disappeared from samples taken in 2010. They were replaced by r-strategists with small cell size, low biomass and fast growth such as the *Achnanthidium* types (Biggs et al., 1998), the *G. parvulum* complex and *R. sinuata*, indicating that episodic flood events can be an important structuring factor for diatom assemblages. In their study, Cambra and Goma (1997) noticed a shift from well structured diatom communities with a high diversity index to less well structured, low diversity communities dominated by r-strategist species after a perturbation. Despite the substantial shift in assemblages, no change in ecological status in terms of nutrients was evident the year after the flood. This highlights the sturdiness of the EQR in determining the ecological status of upland rivers draining forested blanket peat. However, these results are based on a subset of samples taken the year after flood. Due to the highly dynamic nature of these rivers, single sampling provides a snapshot overview of ecological status. Kelly et al. (2006) recommends six replicates of samples should be taken over two to three years to eliminate annual, seasonal and spatial variation. Further study should examine the influence of spatial and temporal variation on the diatom assemblages of acid-sensitive upland blanket peat catchments.

Conclusions

This study has provided a characterisation of diatom assemblages in upland peatland rivers characteristic of the west of Ireland, and related alkalinity and conductivity as the main physicochemical drivers of the diatom assemblages. This highlights the importance of the underlying geology in determining diatom assemblage composition. Multivariate analysis indicated that nutrient enrichment from forestry activities did not stand out as having a major influence on the diatom assemblages. Therefore these upland peatland rivers represent reference conditions with respect to nutrient status. Further work needs to be carried out to determine if the acidic nature of the sites is a response to anthropogenic impacts or natural acidity. Spatial gradients highlighted an upstream-downstream trend and future work should concentrate on how the spatial and temporal variation impacts the diatom assemblages in these spate rivers. The results of this study could be applied to similar upland peat forest catchments and used as the benchmark to assess the impact of ongoing forest harvesting on ecological status. The impact of the flood on diatom assemblage structure is evident at the lower sites; however, this had no bearing on the EQR status. Future work needs to be carried out to determine how long (if at all) it takes for these species to return.

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Bibliography

See Chapter Six pp. 187 - 222.

Appendix A

Taxa codes used in the multivariate ordinations

Taxa	Label
Achnanthes oblongella Østrup	ACHOB
Achnanthidium minutissimum type 1 * sensu Potapova et Hamilton	ACHMNA
Achnanthidium minutissimum type 2 * sensu Potapova et Hamilton	ACHMNB
Achnanthidium minutissimum type 3 * sensu Potapova et Hamilton	ACHMNC
Asterionella formosa Hassall	ASTRION
Brachysira neoexilis Lange-Bertalot	BNEOEXILIS
Cocconeis placentula Ehrenberg	COCCPLA
Ctenophora pulchella (Ralfs ex Kützing) Williams & Round	CTENOPUL
Cyclotella radiosa Kützing ex. de Brébisson	CYCRADI
Cymbella affinis Kützing	CAFFIN
Diatoma mesodon (Ehrenberg) Kützing	DIAMESO
Diatoma tenue Agardh	DIATEN
Diploneis elliptica (Kützing) Cleve	DIPELLIP
Encyonema gracile Ehrenberg	ENCYGR
Encyonema silesiacum (Bleisch in Rabenhorst) Mann in Round, Crawford & Mann	ENCYSIL
Epithemia adnata (Kützing) Rabenhorst	EPITADNA
Eunotia bilunaris var. *mucophila* Lange-Bertalot and Norpel	EBILMUCO
Eunotia exigua complex	EEXIGUA
Eunotia exigua (de Brébisson ex Kützing) Rabenhorst	
Eunotia tenella (Grunow in Van Heurck) Cleve	
Eunotia meisteri (Hustedt)	
Eunotia implicata Norpel, Lange-Bertalot & Alles	EIMPLICT
Eunotia incisa Smith ex Gregory	EINCISA
Eunotia microcephala Krasske ex Hustedt	EMICRO
Eunotia minor (Kützing) Grunow in Van Heurck	EMINOR
Eunotia paludosa Grunow	EPALUD
Eunotia rhomboidea Hustedt	ERHOMBO
Eunotia subarcuatoides Alles, Norpel, Lange-Bertalot	ESUBARC
Fragilaria capucina var. *gracilis* (Østrup) Hustedt	FRAGGRAC
Fragilaria capucina var. *rumpens* (Kützing) Lange-Bertalot	FRAGRUM
Frustulia rhomboides (Ehrenberg) De Toni	FRUSRHOM
Gomphonema acuminatum Ehrenberg	GACUMIN
Gomphonema clavatum Ehrenberg	GCLAVAT
Gomphonema gracile Ehrenberg	GGRACIL
Gomphonema minutum (Agardh) Agardh	GMINUT
Gomphonema olivaceoides Hustedt	GOLIVAC
Gomphonema parvulum complex (Kützing) Kützing	GPARVUL
Gomphonema pumilum (Grunow) Reichardt & Lange-Bertalot	GPUMIL
Gomphonema sp.	GSPP
Gomphonema truncatum Ehrenberg	GTRUNCAT
Hannaea arcus (Ehrenberg) Patrick. in Patrick and Reimer	HANNARC
Meridion circulare (Greville) Agardh	MERIOCIR
Navicula lanceolata (Agardh) Kützing	NLANCEO
Nitzschia dissipata (Kützing) Grunow	NITDISSI
Nitzschia frustulum (Kützing) Grunow in Cleve & Grunow	NITFRUST
Nitzschia sp.	NITSPP
Pinnularia appendiculata (Agardh) Cleve	PAPPEND
Pinnularia rupestris Hantzsch in Rabenhorst	PRUPES
Pinnularia subcapitata Gregory	PSUBCAP
Planothidium lanceolatum (Brébisson ex Kützing) Lange-Bertalot	PLANOLAN
Reimeria sinuata (Gregory) Kociolek & Stoermer	REIMSIN
Sellaphora pupula (Kützing) Mereschkowsky	SELLAPH
Surirella sp.	SURIRELL
Synedra ulna (Nitzsch) Ehrenberg	SYNEDUL

Appendix B

Sample codes used in the multivariate ordinations

Site	Label
Srahrevagh 1 (upper)	SR1
Srahrevagh 2 (upper)	SR2
Srahrevagh 3 (upper)	SR3
Srahrevagh 4 (mid-upper)	SR4
Srahrevagh 5 (mid-upper)	SR5
Srahrevagh 6 (mid-upper)	SR6
Srahrevagh 7 (middle)	SR7
Srahrevagh 8 (middle)	SR8
Srahrevagh 9 (middle)	SR9
Srahrevagh 10 (lower)	SR10
Srahrevagh 11 (lower)	SR11
Srahrevagh 12 (lower)	SR12
Glennamong 1 (upper)	GL1
Glennamong 2 (upper)	GL2
Glennamong 3 (upper)	GL3
Glennamong 4 (mid-upper)	GL4
Glennamong 5 (mid-upper)	GL5
Glennamong 6 (mid-upper)	GL6
Glennamong 7 (middle)	GL7
Glennamong 8 (middle)	GL8
Glennamong 9 (middle)	GL9
Glennamong 10 (lower)	GL10
Glennamong 11 (lower)	GL11
Glennamong 12 (lower)	GL12
Glendahurk	GH
Teevaloughan 1	TV1
Teevaloughan 2	TV2
Glennamong control stream	GCS
Glennamong study stream	GSS
Srahrevagh study stream	SSS
Srahrevagh control stream	SCS
Maumaratta top	MT
Maumaratta bottom	MB
Goulaun top	GT
Goulaun bottom	GB
Altahoney top	AT
Altahoney bottom	AB
Cottage top	CT
Cottage bottom	CB
Lodge top	LT
Lodge bottom	LB
Glendahurk	GHE
Glenthomas	GTE

Appendix C

Physical characteristics of the individual sites

	Sample Number	Temp (°C)	Altitude (m)	Stream Order	Upstream Catchment Area (ha)	Shreve Index
Srahrevagh Upper	3	18.0	341	1	13	1
		0.15	18.01	0.00	0.00	0.00
Srahrevagh mid-Upper	3	19.1	181	2	168	3
		0.06	1.15	0.00	0.00	0.00
Srahrevagh Middle	3	20.4	130	2	266	4
		0.55	0.00	0.00	0.00	0.00
Srahrevagh Lower	3	20.0	40	3	690	8
		0.15	0.00	0.00	0.00	0.00
Glennamong Upper	3	14.2	230	2	191	2
		0.12	0.00	0.00	0.00	0.00
Glennamong mid-Upper	3	16.3	107	2	605	14
		0.10	2.31	0.00	0.00	0.00
Glennamong Middle	3	16.9	50	4	1145	29
		0.06	0.00	0.00	0.00	0.00
Glennamong Lower	3	17.5	20	4	1549	33
		0.06	0.00	0.00	0.00	0.00
Glendahurk	3	15.0	101	1	10	1
		0.03	0.00	0.00	0.00	0.00
Teevaloughan1	3	14.0	280	1	20	1
		0.03	0.00	0.00	0.00	0.00
Teevaloughan2	3	14.1	268	1	32	1
		0.10	0.00	0.00	0.00	0.00
Glennamong Study	3	14.5	87	1	10	1
		0.03	0.00	0.00	0.00	0.00
Glennamong Control	3	14.4	56	1	10	1
		0.07	0.00	0.00	0.00	0.00
Srahrevagh Study	3	16.1	220	1	17	1
		0.03	0.00	0.00	0.00	0.00
Srahrevagh Control	3	16.7	240	1	23	1
		0.03	0.00	0.00	0.00	0.00
Maumaratta	3	14.0	95	2	473	3
		1.77	63.64	0.71	254.56	2.12
Goulaun	3	18.1	85	3	754	6
		0.93	35.36	0.71	304.06	2.12
Altahoney	3	12.8	80	3	1144	20
		2.31	28.28	0.00	181.02	2.83
Cottage	3	11.8	60	3	210	6
		0.13	28.28	0.00	0.00	0.71
Lodge	3	12.6	34	3	468	7
		0.07	4.95	0.00	0.00	0.71
Glenthomas	3	18.4	50	3	1429	17
		0.0	0.00	0.00	0.00	0.00
Glendahurk (main)	3	19.1	50	3	1211	25
		0.00	0.00	0.00	0.00	0.00

Appendix D

Chemical characteristics of the individual sites

	Sample Number	Conductivity (µS cm^{-1})	Alkalinity (mg L^{-1} CaCO3)	PO$^{4}_{-3}$P (µg L^{-1})	NH+4-N (µg L^{-1})	NO-3 -N (µg L^{-1})	pH	Colour (PtCo)
Srahrevagh Upper	3	93	13.5	3.21	27.47	12.15	7.4	38
		3.51	*0.83*	*0.21*	*7.84*	*0.73*	*0.04*	*3.48*
Srahrevagh mid-Upper	3	141	37.0	4.66	26.06	7.10	7.7	49
		0.00	*0.55*	*0.22*	*5.10*	*0.45*	*0.00*	*1.67*
Srahrevagh Middle	3	152	37.9	3.72	19.17	8.79	7.3	41
		12.00	*5.22*	*0.36*	*5.93*	*2.39*	*0.15*	*3.38*
Srahrevagh Lower	3	213	68.8	5.38	6.51	0.56	8.5	56
		0.67	*0.30*	*0.62*	*2.13*	*0.14*	*0.01*	*0.58*
Glennamong Upper	3	67	0.1	4.82	36.61	97.02	5.6	104
		0.33	*0.03*	*0.56*	*1.89*	*33.24*	*0.07*	*5.81*
Glennamong mid-Upper	3	79	6.7	5.44	26.16	88.87	7.0	82
		0.58	*0.42*	*0.54*	*3.44*	*37.58*	*0.05*	*3.71*
Glennamong Middle	3	93	10.7	4.45	29.11	28.43	7.3	81
		1.00	*0.24*	*0.82*	*3.76*	*9.72*	*0.02*	*2.73*
Glennamong Lower	3	94	9.4	3.72	21.39	10.92	7.2	68
		0.00	*0.17*	*0.26*	*7.53*	*1.01*	*0.02*	*1.20*
Glendahurk	3	89	2.8	3.60	54.37	49.49	5.6	74
		2.19	*1.08*	*0.17*	*11.32*	*11.21*	*0.12*	*0.58*
Teevaloughan1	3	144	1.5	3.28	117.02	6.72	3.8	262
		0.58	*0.06*	*0.06*	*3.52*	*1.72*	*0.01*	*0.33*
Teevaloughan2	3	122	1.7	3.54	77.73	34.12	4.8	141
		0.33	*0.08*	*0.23*	*2.42*	*2.31*	*0.01*	*1.00*
Glennamong Study	3	173	-1.7	4.04	73.91	39.03	3.9	256
		0.33	*1.29*	*0.26*	*4.47*	*2.05*	*0.01*	*0.33*
Glennamong Control	3	251	-2.7	3.75	39.28	3.43	3.5	264
		0.33	*0.33*	*0.27*	*3.83*	*0.81*	*0.01*	*0.33*
Srahrevagh Study	3	201	-2.4	6.27	110.65	2.00	3.6	283
		0.88	*0.30*	*0.18*	*4.00*	*1.08*	*0.01*	*0.33*
Srahrevagh Control	3	115	9.9	4.57	122.12	49.86	5.8	168
		0.88	*0.04*	*0.17*	*1.24*	*1.07*	*0.01*	*0.58*
Maumaratta	3	68	-0.5	4.92	43.41	0.00	5.8	138
		3.50	0.50	1.57	0.12	4.38	0.00	5.00
Goulaun	3	102	17.0	3.67	59.37	0.00	7.1	66
		16.00	7.00	0.18	6.56	0.73	0.22	7.50
Altahoney	3	75	-2.0	4.53	42.27	0.00	5.6	166
		1.50	1.00	0.63	5.04	0.53	0.00	3.00
Cottage	3	98	18.0	3.29	43.41	9.81	7.3	72
		4.00	0.00	0.15	18.24	0.78	0.19	8.50
Lodge	3	97	21.0	3.08	58.92	2.51	7.8	90
		1.00	0.00	0.10	2.28	3.69	0.02	2.00
Glenthomas	3	82	6.0	3.22	21.46	5.10	6.2	110
		0.67	*0.30*	*0.21*	*1.05*	*0.54*	*0.01*	*0.58*
Glendahurk (main)	3	87	11.4	3.46	65.12	2.05	6.3	64
		0.33	*0.30*	*0.21*	*1.05*	*0.54*	*0.01*	*0.58*

3.3 Influence of natural variation in benthic invertebrate and diatom communities for biomonitoring in upland peat rivers

Connie O'Driscoll[1], Elvira de Eyto[2], Michael Rodgers[1], Mark O'Connor[1], Zaki-ul-Zaman Asam[1], Martyn Kelly[3] and Liwen Xiao[1]*

[1] Civil Engineering, National University of Ireland, Galway, Republic of Ireland

[2] Marine Institute, Newport, Co. Mayo, Republic of Ireland

[3] Bowburn Consultancy, Bowburn, Durham, U.K.

* Corresponding author: Liwen Xiao; Email: liwen.xiao@nuigalway.ie; Tel: +353 91 494188

Abstract

Upland blanket peat catchments are important biodiversity refuges and are increasingly recognised as important carbon stores. Key pressures on peatland catchments include forestry, artificial drainage, wildfire and vegetation burning, windfarm development, live-stock grazing, and climate change. Assessment of these pressures and mitigation of impacts requires sensitive monitoring programmes, particularly in the aquatic habitats draining such catchments. In this study, two rivers draining upland peat catchments were monitored to examine the natural variation existing in benthic invertebrates and diatom assemblages. Samples were collected seasonally over 2 annual-cycles (2009-2010) at twelve sampling sites along each of the two rivers. Multivariate analysis confirmed these peatland fed streams show characteristic spatial patterns in the longitudinal distribution of invertebrate and diatom species assemblages downstream and away from the constraining influence of the peat. A temporal trend was observed in the invertebrate assemblages reflecting seasonal water temperature fluxes. Despite observed relationships in these biotic indices highlighting spatio-temporal variation, the ecological quality ratios for the associated waterbodies were consistent in status throughout season and year as well as along rivers. This study provides valuable information for creating and/ or validating bioassessment protocols for upland blanket peat catchments.

Keywords: Upland blanket peat; Diatom; Invertebrate; WFD; Seasonal and spatial variation.

Introduction

Peatlands are predominantly found in boreal, subarctic and tropical zones, and cover approximately 3 % of the Earth's surface (Ramchunder et al., 2009; Dise, 2009). Peatlands are important economically, in terms of forestry, windfarm development, turf-cutting, tourism, flood mitigation and maintenance of clean water supplies (Renou-Wilson et al., 2011). Peatlands are also important as biodiversity hotspots and carbon pools. It is estimated that the carbon held in peatlands represents 25 % of the world soil carbon pool (Parish et al., 2007). In Ireland, peat soils cover 21% of the national land area and contain more than 75 % of the national soil organic carbon (Renou-Wilson et al., 2011). Upland blanket peatlands in north-western Europe contain internationally rare plant species such as *Erica tetralix* L., *Calluna vulgaris* (L.) Hull, *Eriophorum vaginatum* L. and *Molinia caerulea* (L.) Moench. The headwaters draining peatlands feed into rivers, many of which contain Red List species such as Atlantic salmon *Salmo salar* L. and freshwater pearl mussels *Margaritifera margaritifera L.* As a result of threats to these species, many peatland catchments have been designated EU Natura 2000 sites, Special Areas of Conservation (SACs), Special Protection Areas (SPAs), and Sites of Special Scientific Interest (SSSIs) making them important biodiversity refuges (O'Driscoll et al., 2012). Peatlands are subject to multiple and increasing pressures such as artificial drainage, wildfire, vegetation burning, windfarm development, livestock grazing, forestry activities and projected climate change (Jenkins et al., 1990; Ormerod et al., 1991; Allott et al., 1997; Nisbet, 2001; Cummins and Farrell, 2003; Nieminen, 2003; Wallage et al., 2006; Ramchunder et al., 2009; Fealy et al., 2010; Rodgers et al., 2010; Cantonati and Lange-Bertalot, 2011). The importance of protecting and managing peatland ecosystems is therefore fundamental to supporting the biodiversity and services they provide (Holden et al., 2006; Ramchunder et al., 2009; Renou-Wilson et al., 2011), and some restoration management techniques are being investigated (Ramchunder et al., 2009; O'Driscoll et al., 2011).

Ecological assessment has become an integral component of new environmental legislation in Europe (European Union, 2000), the United States (Gibson et al., 1996), and Australia and New Zealand (ANZECC, 2000), with observed conditions being compared against expected conditions, in order to determine the level of anthropogenic change. The expected conditions are calculated from either ‘reference sites’ i.e. pristine sites lacking in anthropogenic perturbation or ‘reference conditions’ i.e. historical data collected before significant anthropogenic impacts (Stoddard et al., 2006; Pardo et al., 2012). The WFD requires all member states to achieve at least good ecological status in all water bodies by 2015 (Directive, 2000, Article 4, clause 1). “The water body is the basic compliance reporting and management unit for the WFD into which all rivers, lakes, ground, transitional and coastal waters are divided” (EPA, 2006). River systems are further divided in water bodies based on hydro-morphological characteristics such as altitude, depth, size and flow and catchment rock type. Biotic indicators such as benthic invertebrates and diatoms have been adopted worldwide as superior assessors of ecological quality as they integrate the physical and chemical characteristics of rivers and lakes (Rott, 1991; Stevenson and Pan, 1999; Leira and Sabater, 2005; Carter et al., 2007; Kelly et al., 2008; Gaiser, 2009). Invertebrates have been used for over a century as indicators of environmental change and as a result, qualitative sampling and analyses techniques are well developed (Carter et al., 2007). Reported impacts range from reduced diversity and density to changes in species composition (Growns and Davis, 1991, 1994; Davies and Nelson, 1994; Miller and Golladay, 1996; Tierney et al., 1997; Quinn et al., 2003; Banks et al., 2007; Baldigo et al., 2009; Couceiro et al., 2010). Diatoms, being present in all types of aquatic systems and having short generation times, are reportedly one of the most successful contemporary groups of photosynthetic eukaryotic microorganisms being used as bioindicators (Rott, 1991). Studies have shown impacts from forest harvesting (Lowe et al 1986; Naymik et al., 2005) municipal and industrial impacts (Beyene et al., 2009) and acidification (Battarbee et al., 2010). However, the impacts of peatland stressors on stream ecosystems have received much less attention (Durance and Ormerod, 2007: Ramchunder et al., 2009).

Many studies have recorded spatial and temporal variation in invertebrate and diatom assemblages (Vannote et al 1980; Mullholland et al., 1995; Potapova and Charles, 2002; Šporka et al., 2006; Passy, 2007; Yashi et al, 2011). This variation needs to be quantified and accounted for, so that the risk of misclassification of the true status of the water body can be minimised (Kelly et al., 2009; Clarke, 2011). Recently, studies have begun to compare invertebrates and diatoms in stream bio-assessments, the results of which provide unexpected and contrasting results (Passy et al., 2004; Sandin, 2004; Feio et al., 2007; Lewis et al., 2007; Beyene et al., 2009). The general conclusion that can be derived from these studies is that various taxonomic groups respond differently to ecological gradients and given that interactions occur between taxonomic groups, research should focus on multiple groups in assessing environmental pressures (Heino, 2010; Kelly, 2011).

A few studies have addressed environmental gradients influencing the diatom (O'Driscoll et al., 2012) and invertebrate (Ramchunder et al., 2009) distributions in upland peat fed streams. However, to the best of our knowledge, no study attempts to examine the extent of temporal and spatial variation of these two groups concurrently in relation to water quality indices of upland blanket peat streams. Upland peat catchments have been underrepresented in the derivation of EQRs (Kelly-Quinn et al., 2004; Kelly et al., 2006; O'Driscoll et al., 2012). The main purpose of this study therefore is to (1) assess the spatial and temporal variation in the diatom and invertebrate assemblages in light of interactions between these two groups and their associated environmental factors in upland peat catchments; (2) investigate relationships between spatial and temporal variation and commonly used biotic indices; (3) explore the consequence of these relationships on the associated waterbody ecological status class. Such insight will be crucial in to the development of management strategies for the restoration and protection of water bodies. The river is the primary unit of study here to exam assemblage variation and secondary the WFD defined waterbody as there should be an expectation that all sites within this defined unit should behave similarly.

Materials and methods

Study sites

This study was based on two rivers (the Glennamong and Srahrevagh) located in the Burrishoole Catchment, Mayo (NW Ireland) (Figure 1). The Burrishoole catchment, which is an important habitat for salmon (*Salmo salar* L.), trout (*Salmo trutta* L.) and eel (*Anguilla anguilla* L.) (Whelan et al., 1998) has been a significant site for fisheries research since the mid-1950s. Most of the catchment is covered in blanket peat and overlies mainly quartzite and schist bedrock. The Burrishoole receives an average precipitation of about 2,000 mm per year (Dalton et al., 2010).

The catchment system is naturally acidic and oligotrophic and has a low acid buffering capacity (Byrne et al., 2004). Both of the study rivers have significant amounts of commercial coniferous forestry in their catchments (the Srahrevagh > 90 % and the Glennamong 22 %).

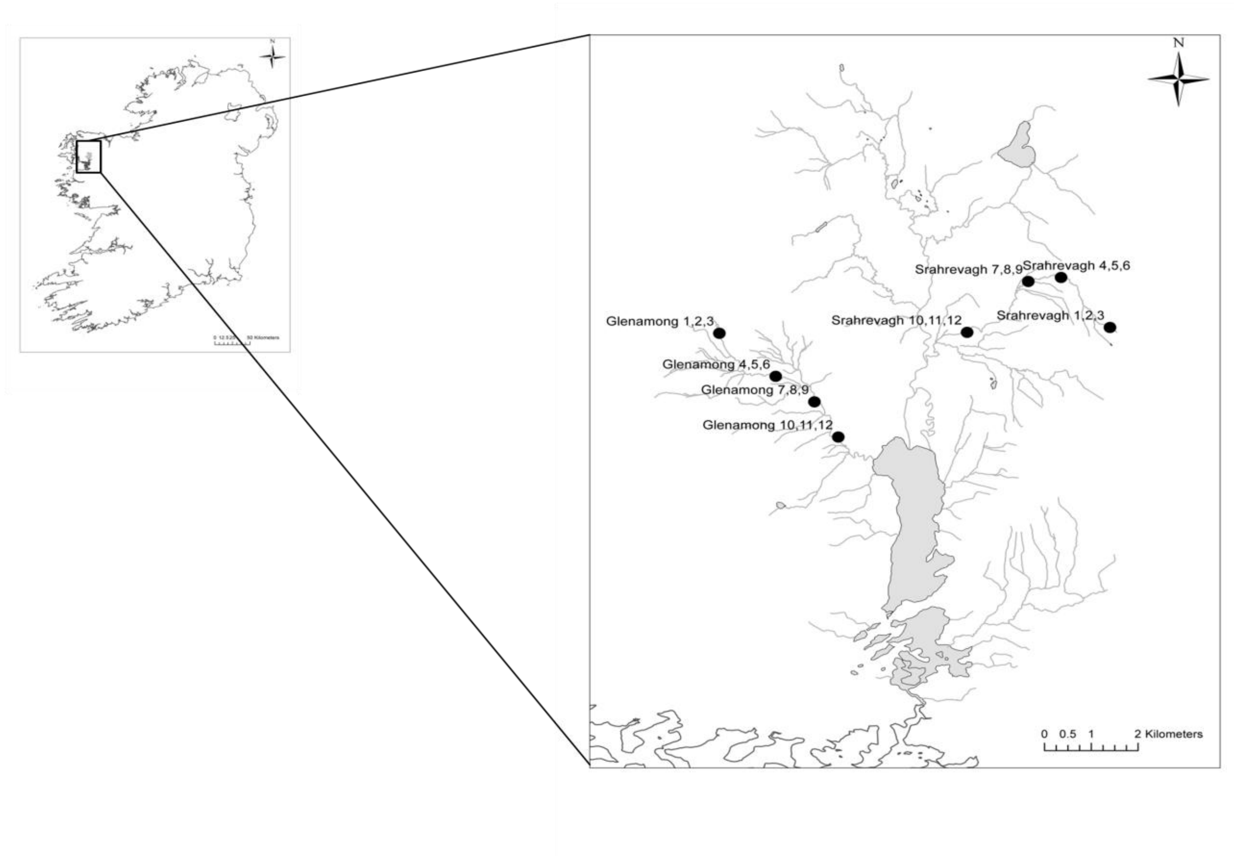
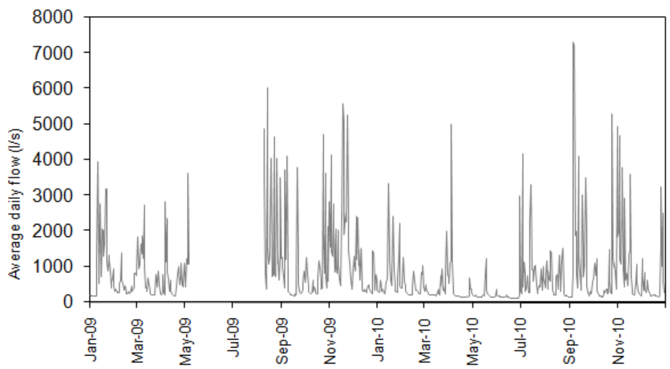


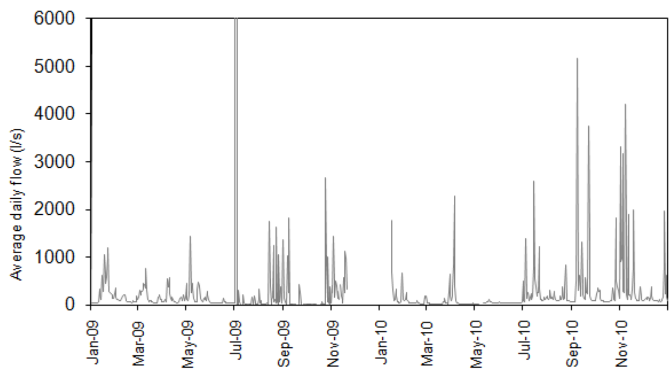
Figure 1 Geographical location of the study site in Ireland (left) and of the 12 sampling sites along each of the Glennamong and Srahrevagh rivers.

Sample collection and preparation

Twelve sites were sampled along each river from the headwaters to the lake four times a year (seasonally - April, June, October and January) for two years (2009, 2010) (Figure 1). Sites 1, 2 and 3 were referred to as the upper sites (UP) and were located above the forestry on tributaries of the main river; 4, 5 and 6 were the mid-upper sites (MU), located in the forestry; 7, 8 and 9 the middle sites (M), located in the forestry; and 10, 11 and 12 the lower sites (L) were located below the forestry. For the Srahrevagh river, all 12 sites were within one water body as defined for the WFD (Irish water body code: WE 32 781). The Glennamong river is split into three waterbodies for the purpose of the WFD and the 12 sampling sites selected for this river were situated within 2 of these waterbodies (WE 32 2767 and WE 32 2441). WE 32 2767 is the upper tributary of WE 32 2441. Water samples were taken at each sampling site and analysed on the same day for alkalinity and colour using standard procedures (APHA, 1998). Soluble reactive phosphorus and ammonia were measured using a Konelab 20 Analyser (Konelab Ltd., Finland). Water temperature, pH and conductivity were recorded in the field, and upstream catchment area information was extracted from GIS maps of the area. Catchment area was used as a proxy for discharge as it was not feasible to measure discharge at all sites. Day's since the last flood (DSF) was calculated by counting the number of days since the last flood event (Biggs, 2000). Flood events were taken to be at water volumes >1000 l/s in the Srahrevagh and > 3000 l/s in the Glennamong following visual inspection of annual hydrographs (Figure 2).



(a)



(b)

Figure 2. Hydrographs of the Glennamong (a) and Srahrevagh (b) rivers over the duration of the study.

Benthic invertebrates were sampled using a 60-second kick sample (standard 1 mm mesh sized pond net) in river riffles (Armitage et al., 1975). The number of grazers was calculated by summing the total number of individuals that could be classified as grazers. Periphyton was removed from five cobble surfaces with 100 ml of stream water in accordance with Kelly et al. (1998). Orthogonal measurements of each stone were taken in the field, and converted to stone surface area using the equation of Dall (1979). The samples were stored in the dark and analysed in the laboratory later the same day for periphyton ash free dry mass (AFDM g m^2) and total phosphorus (TP) (APHA, 1998). Sub-samples were taken from the periphyton sample and cleaned using the cold acid permanganate method (Kelly et al., 1998). Permanent slides were prepared using Naphrax (refractive index. = 1.74).

Identification

Invertebrates were sorted and identified to species level where possible following the keys of Hynes (1977); Elliot et al. (1988); Friday (1988); Edington and Hildrew (1995) and Wallace et al. (2003). Diatoms were identified to species level where possible and counted at x1000 magnification using an Olympus BX-51 microscope equipped with an x100 phase contrast objective. At least 300 valves were identified and counted per slide using Krammer and Lange-Bertalot (1986; 1997; 2000; 2004). Certain taxa were difficult to identify and the approach adopted for these species was as follows: Three types of *Achnanthidium minutissimum* varieties were recognized and split into types based on Potapova and Hamilton (2007): the 'capitate' morph/ 'type a'; 'linear'/ 'type b' and 'wide linear-lanceolate'/ 'type c'. *A. minutissimum* present in girdle view was enumerated separately and then divided between the three morphological groups based in proportion to their relative abundance in valve view. *E.exigua* (de Brébisson ex Kützing) Rabenhorst, also present in high numbers in girdle view, was difficult to distinguish from *Eunotia tenella* (Grunow in Van Heurck) Cleve, and *Eunotia meisteri* (Hustedt) and so the three were combined and considered as *Eunotia exigua* complex as recommended by DeNicola (2000). A number of varieties of *Gomphonema parvulum* have been described; however, populations in these samples had high morphological variability and so have been termed "*Gomphonema parvulum* complex".

Data analysis

Data exploration highlighted a clear separation of the two rivers due to different ranges in alkalinity and so when analysed together, all other variation was concealed. As a result, the two rivers are analysed separately. Commonly used indices such as EPT (Ephemeroptera, Plecoptera and Trichoptera) and TDI (Trophic Diatom Index, Kelly et al., 2008) were calculated for each site. Less commonly used indices such as SI (Medin's acid index for invertebrates; Henrikson and Medin, 1986) and ACID (Acidity index for diatoms, Andrén and Jarlman, 2008) were chosen as measures of acidity in peatland receiving waters. The Irish Q Index (McGarrigle et al., 2002) and an EQR calculated from the TDI (Kelly et al., 2008) were used as measures of ecological status based on invertebrates and diatoms, respectively, and can be reported in terms of WFD status classes: bad, poor, moderate, good and high. All biological variables were expressed as proportional abundances and fourth-root transformed to reduce the influence of dominant species. Bray Curtis similarities between samples were calculated to form a resemblance matrix. Species with an occurrence in less than 5% of samples were excluded. PERMANOVA (permutational analysis of variance) (Anderson, 2005, version 1.6) was carried out on each set of biological data to determine the main sources of spatial and temporal variation in the data, with three explanatory variables (year, season and site). 4999 permutations, using raw data were carried out in all cases. All interactions were included, and pairwise posthoc comparisons were used to further elucidate the significant patterns. MDS plots were created using Primer version 6(Clarke and Gorley, 2006), and the BVSTEP routine was used to highlight important explanatory environmental variables. Environmental variables were normalised. SIMPER analysis was used to highlight important species differences between groups. The biotic indices (EPT, SI, ACID and TDI) and EQRs (Q Index and TDI-EQR) were examined in terms of the variation that exists within a WFD waterbody (WE 32 781- Srahrevagh; WE 32 2767 – upper tributary of the Glennamong; and WE 32 2441 main Glennamong river). Univariate analysis was carried out using ANOVA (analysis of variance) and pairwise comparisons were made using LSD *post hoc* tests with site and season as explanatory variables (Datadesk version 6.1).

Results

Environmental characteristics of the study sites

The rivers in both these catchments are spatey in nature and are prone to flash flooding (Figure 2). Nutrient values recorded in both the Glennamong and Srahrevagh were low and stable throughout the duration of the study (SRP < 10 µg L^{-1} PO_4-P and Ammonium < 60 µg L^{-1} NH4-N) (Table 1). Conductivity and alkalinity values increased downstream. Colour, conductivity and alkalinity values varied with season. The highest alkalinity values were obtained during periods of higher temperatures and lower flows in summer. Spearman rank correlation analysis indicated that temperature has a strong positive correlation with conductivity, pH and alkalinity ($p < 0.05$). The main difference between the two rivers is the alkalinity which ranged in the Srahrevagh from 8.2 mg L^{-1} to 68.8 mg L^{-1} and from 0.1 mg L^{-1} to 10.7 mg L^{-1} in the Glennamong (Table 1).

Species composition

44 diatom taxa were found in the Glennamong river, 28 of which were observed in more than 5 % of the samples. The five most abundant taxa were *Achnanthes oblongella* Østrup, *E. rhomboidea* Hustedt, *E. exigua*, *Tabellaria flocculosa* (Roth) Kützing, and *G. parvulum*, present in 96 %, 89 %, 83 %, 79 % and 76 % of the samples, respectively (see Appendix A and B for species origins). 56 diatom taxa were found in the Srahrevagh river, 43 of which were observed in more than 5 % of the samples. The five most abundant taxa were *Achnanthes oblongella*, *Gomphonema parvulum* Group, *A.minutissimum* type a, *A. minutissimum* type b and *Fragilaria capucina var. gracilis* (Østrup) Hustedt present in 99 %, 98 %, 92 %, 83 % and 74 % of the samples, respectively.

31 invertebrate taxa were found in the Glennamong river, 22 of which were observed in more than 5% of the samples. The five most abundant invertebrate taxa were *Baetis rhodani* Pictet, *Leuctra hippopus* Kempny, Simulidae *spp.*, *Protonemura meyeri* Pictet, and Chironomidae *spp.*, present in 98 %, 89 %, 88 %, 85 % and 82 % of the samples, respectively. 35 invertebrate taxa were found in the Srahrevagh river, 26 of which were observed in more than 5% of the samples.

The five most abundant invertebrate taxa were *Baetis rhodani*, Simulidae *spp, Rhithrogena semicolorata* Curtis,, *Leuctra hippopus*, and *Gammarus dubenii* Lilljeborg present in 99 %, 93 %, 85 %, 73 % and 65 % of the samples respectively.

Table 1 Physical and chemical characteristics of the sites studied in spring, summer, autumn and winter

Site	Upper				Mid-Upper				Middle				Low er			
Season	Spring	Summer	Autumn	Winter	Spring	Summer	Autumn	Winter	Spring	Summer	Autumn	Winter	Spring	Summer	Autumn	Winter
Sample No.	9	9	6	6	9	9	6	6	9	9	6	6	9	9	6	6
Srahrevagh																
Altitude (m)	342	/	/	/	186	/	/	/	129	/	/	/	33	/	/	/
Upstream catchment area (ha)	13	/	/	/	168	/	/	/	266	/	/	/	690	/	/	/
Temperature	8.8	15.1	9.7	4.6	9.9	15.4	10.5	5.0	11.5	17.6	10.6	5.5	12.0	17.9	12.4	5.5
	0.41	*0.86*	*0.21*	*0.27*	*0.42*	*0.51*	*0.34*	*0.26*	*0.48*	*0.85*	*1.38*	*0.13*	*0.65*	*0.61*	*0.70*	*0.12*
Conductivity (µS cm-1)	83	81	68	62	111	107	79	85	115	114	82	85	164	149	122	112
	2.95	*3.60*	*8.83*	*5.75*	*4.60*	*9.72*	*9.34*	*12.24*	*6.61*	*11.82*	*10.03*	*11.07*	*6.09*	*18.48*	*17.00*	*12.45*
Alkalinity (mg L-1 CaCO3)	13.3	13.5	16.8	8.2	23.9	37.0	19.0	14.0	22.5	37.9	17.9	14.9	50.6	68.8	45.8	32.8
	0.18	*0.41*	*0.21*	*0.07*	*0.64*	*0.28*	*0.15*	*0.50*	*2.46*	*2.61*	*2.67*	*1.76*	*0.45*	*0.15*	*0.16*	*0.48*
PO4-P (µg L-1)	5.27	6.75	4.07	5.54	7.05	7.47	6.84	6.60	7.68	7.12	6.94	7.88	8.44	9.52	6.91	7.58
	0.67	*1.68*	*0.22*	*0.56*	*0.90*	*1.24*	*1.39*	*1.58*	*0.83*	*1.28*	*0.64*	*0.46*	*0.78*	*2.11*	*0.73*	*0.41*
NH4-N (µg L-1)	45.08	40.18	17.59	29.21	45.56	32.38	18.53	51.02	48.87	28.95	25.88	32.19	43.73	26.18	12.89	31.57
	11.11	*8.98*	*6.51*	*6.12*	*4.47*	*8.42*	*7.54*	*17.61*	*5.83*	*8.69*	*10.93*	*1.37*	*3.56*	*9.89*	*6.10*	*5.55*
pH	6.24	6.70	7.09	5.47	7.09	6.99	7.02	6.38	6.86	6.86	6.85	5.58	7.33	7.32	6.93	6.68
	6.54	*7.08*	*7.70*	*5.71*	*7.58*	*7.47*	*7.55*	*7.08*	*7.47*	*7.52*	*7.47*	*5.70*	*7.73*	*7.73*	*7.28*	*7.10*
Colour (PtCo)	16	38	29	38	29	49	70	68	41	41	74	75	42	56	87	98
	1.32	*1.74*	*3.12*	*6.54*	*2.52*	*0.83*	*0.42*	*1.84*	*0.88*	*1.69*	*1.28*	*5.53*	*0.93*	*0.29*	*0.42*	*0.97*
Glennamong																
Altitude (m)	237	/	/	/	90	/	/	/	52	/	/	/	17	/	/	/
Upstream catchment area (ha)	191	/	/	/	605	/	/	/	1145	/	/	/	1549	/	/	/
Temperature	9.6	13.4	10.7	5.4	11.0	14.9	11.4	5.3	12.6	15.4	13.2	4.9	13.1	15.6	15.1	4.5
	0.32	*0.22*	*0.51*	*0.35*	*0.36*	*0.40*	*0.74*	*0.77*	*0.70*	*0.46*	*1.12*	*0.30*	*0.80*	*0.58*	*1.28*	*0.02*
Conductivity (µS cm-1)	64	54	62	66	72	61	59	60	80	68	76	64	80	70	67	64
	4.48	*3.71*	*2.91*	*1.12*	*3.94*	*5.25*	*1.57*	*2.17*	*4.11*	*7.13*	*13.51*	*3.24*	*4.19*	*6.98*	*1.72*	*0.40*
Alkalinity (mg L-1 CaCO3)	0.1	0.1	0.1	0.1	6.7	6.7	6.7	6.7	10.7	10.7	10.7	10.7	9.4	9.4	9.4	9.4
	0.01	*0.01*	*0.02*	*0.02*	*0.21*	*0.21*	*0.27*	*0.27*	*0.12*	*0.12*	*0.15*	*0.15*	*0.08*	*0.08*	*0.11*	*0.11*
PO4-P (µg L-1)	6.55	10.04	4.51	6.29	9.18	8.13	5.72	6.65	6.29	8.05	5.26	6.53	6.83	6.99	4.54	5.66
	0.97	*2.11*	*0.80*	*0.49*	*1.38*	*1.43*	*0.60*	*0.90*	*0.89*	*1.54*	*0.52*	*0.55*	*0.96*	*1.39*	*0.63*	*0.26*
NH4-N (µg L-1)	47.46	43.31	17.38	30.58	38.35	36.01	18.64	30.58	46.80	33.73	23.75	27.58	59.44	32.56	21.60	29.99
	6.92	*9.59*	*4.79*	*6.00*	*6.29*	*7.41*	*7.15*	*6.69*	*7.51*	*7.29*	*12.27*	*3.20*	*12.12*	*7.79*	*12.81*	*5.77*
pH	5.29	4.66	4.84	4.95	6.66	5.65	5.61	5.86	7.14	5.99	5.60	5.90	6.26	5.88	5.23	5.97
	5.84	*5.12*	*5.73*	*5.45*	*7.30*	*6.03*	*5.94*	*6.47*	*7.89*	*6.29*	*5.75*	*6.33*	*6.62*	*6.26*	*5.55*	*6.83*
Colour (PtCo)	73	104	136	107	52	82	104	104	41	81	82	108	43	68	86	102
	2.19	*2.91*	*2.92*	*3.94*	*1.96*	*1.86*	*9.41*	*5.15*	*0.50*	*1.36*	*0.63*	*1.52*	*2.08*	*0.60*	*0.21*	*1.32*

Multivariate analysis

Glennamong Diatoms

PERMANOVA indicated that in the Glennamong river all the main explanatory variables were significant sources of variation (year, site and season ($p_{mc} < 0.05$)). The majority of the interaction terms were also significant. However, in detailed pairwise tests, year was not significant ($p_{mc} = 0.767$), nor was season ($p_{mc} > 0.05$ in all comparisons). The main source of variation for the diatom assemblages in the Glennamong was site, with the upper site being significantly different from the three bottom ones ($p_{mc} < 0.001$ in all cases). This is apparent in an MDS plot, with the upper sites being clearly separated from the rest (Figure 3).

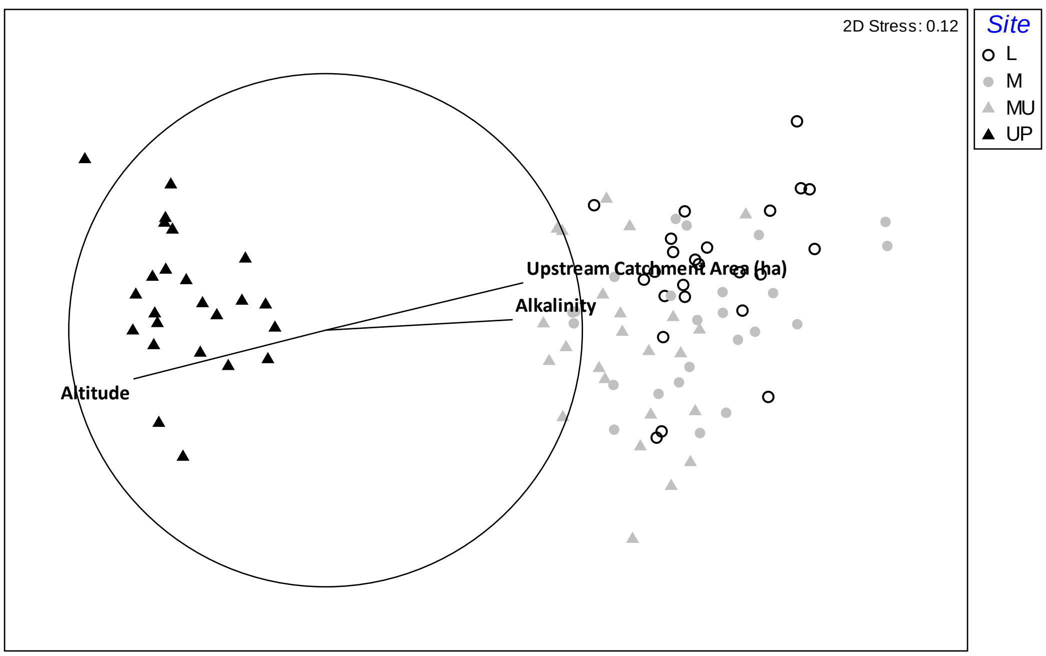


Figure 3. MDS plot of diatom samples taken from the Glennamong river at four sites (L-lower, M-Middle, MU – Mid upper and Up- Upper). Vectors indicate the most significant explanatory environmental variables as indicated by BVSTEP.

The BVSTEP routine indicated that the environmental variables best correlated with the patterns in the Glennamong diatom samples were altitude, upstream catchment area and alkalinity (Spearman's *rho* = 0.786). SIMPER analysis indicated the split in the MDS plot was a result of the dominance of *Eunotia rhomboidea* at the upper site and a corresponding absence or low abundance of *Achnanthes oblongella*, *Gomphonema parvulum* and *Eunotia exigua*, which were the more common species further down the river (Figure 4).

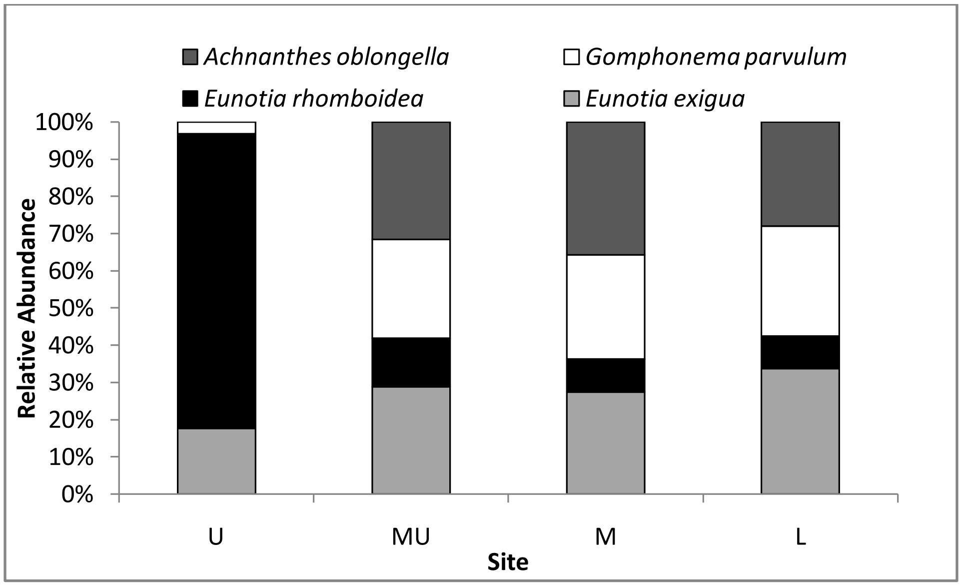


Figure 4. SIMPER analysis results highlighting the average relative abundance of species most responsible for the site variation in the Glennamong diatoms.

Srahrevagh Diatoms

Site differences were obvious in the diatom data collected in the Srahrevagh river, with a notable gradient in species assemblages following an upstream to downstream trend (PERMANOVA, p_{mc} <0.001 in all cases). This is apparent in the MDS plot, with, the upper sites being clearly separated from the rest (Figure 5).

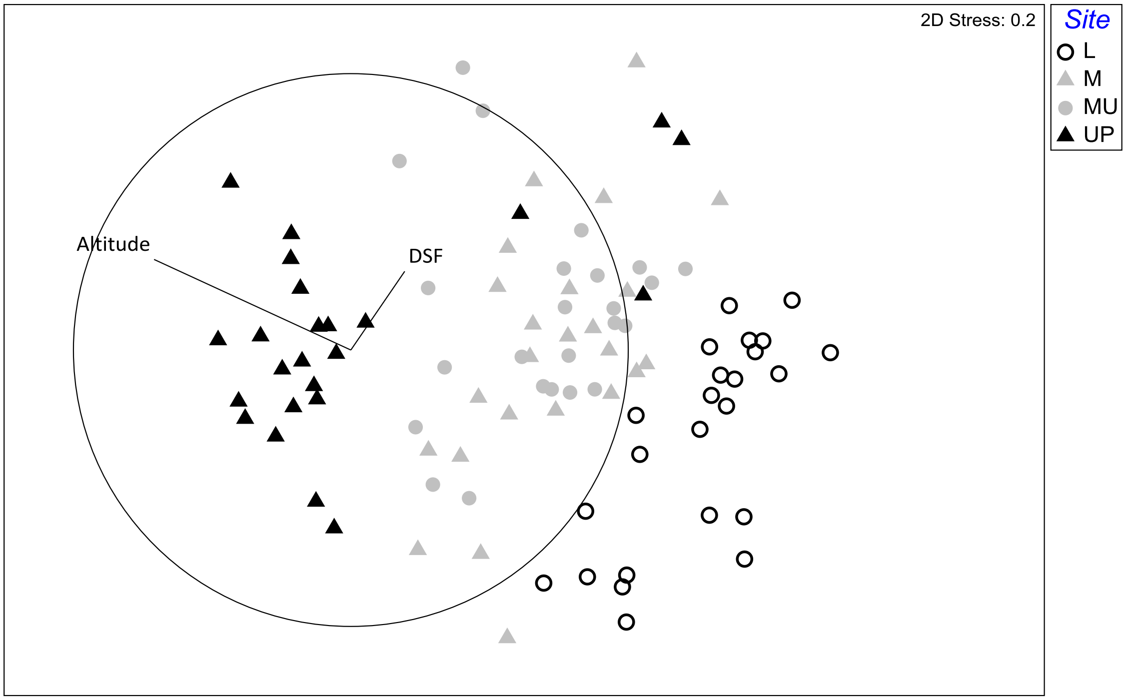


Figure 5. MDS plot of diatom samples taken from the Srahrevagh river at four sites (L-lower, M-Middle, MU – Mid upper and Up- Upper). Vectors indicate the most significant explanatory environmental variables as indicated by BVSTEP.

The BVSTEP routine indicated that the environmental variables best correlated with the patterns in the Srahrevagh diatom samples were altitude and DSF (days since flood) (Spearman's *rho* = 0.605). SIMPER analysis indicated the split in the MDS plot was a result of the dominance of *Achnanthes oblongella* at the upstream site progressing to increased abundances of *Gomphonema olivaceoides* Hustedt and *Planothidium lanceolatum* (Brébisson ex. Kützing) Lange-Bertalot and the introduction of *Reimeria sinuata* (Gregory) Kociolek and Stoermer at the lower site (Figure 6).

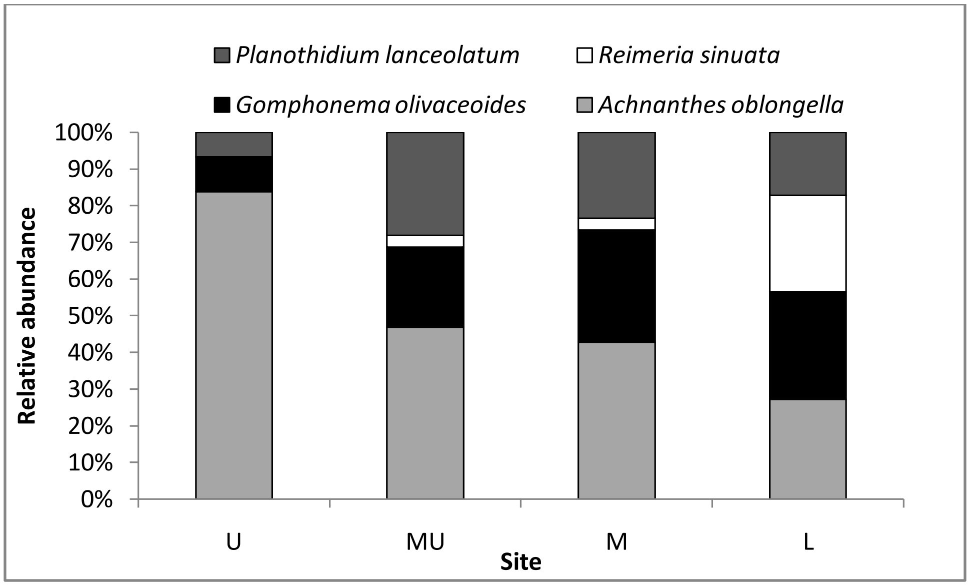


Figure 6. SIMPER analysis results highlighting the average relative abundance of species most responsible for the site variation in the Srahrevagh diatoms.

Glennamong Invertebrates

Site differences were apparent in the invertebrate data collected in the Glennamong river, with the upper site again having quite distinct assemblages compared to those lower down the river (PERMANOVA, $p_{mc}<0.05$). In contrast to the diatoms, however, seasonal variation was also a significant source of variation, with all seasons having quite distinct assemblages (PERMANOVA, $p_{mc}<0.05$) apart from spring and winter. The BVSTEP routine indicated that the environmental variables best correlated with the patterns in the Glennamong invertebrate samples were water temperature and altitude (Spearman's *rho* =0.464), with temperature splitting the biological data along the y-axis (with season) and altitude splitting the samples according to site (x-axis) (Figure 7).

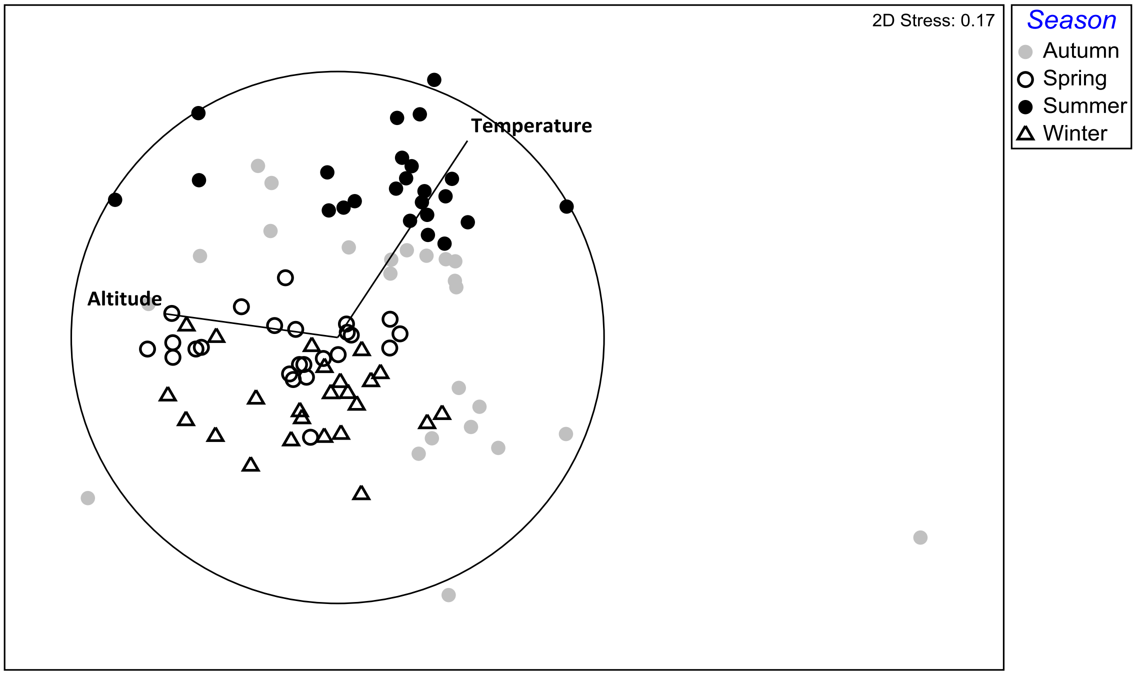
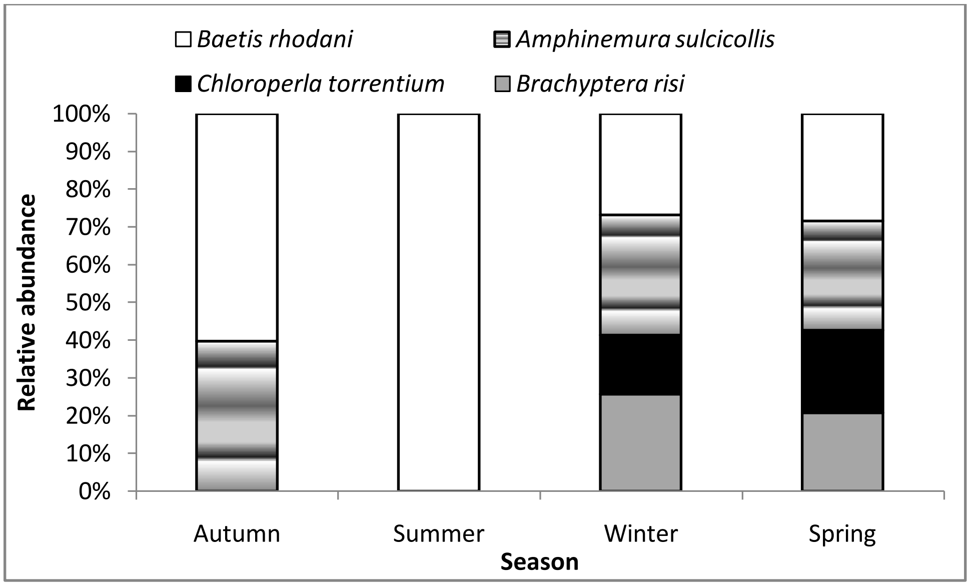
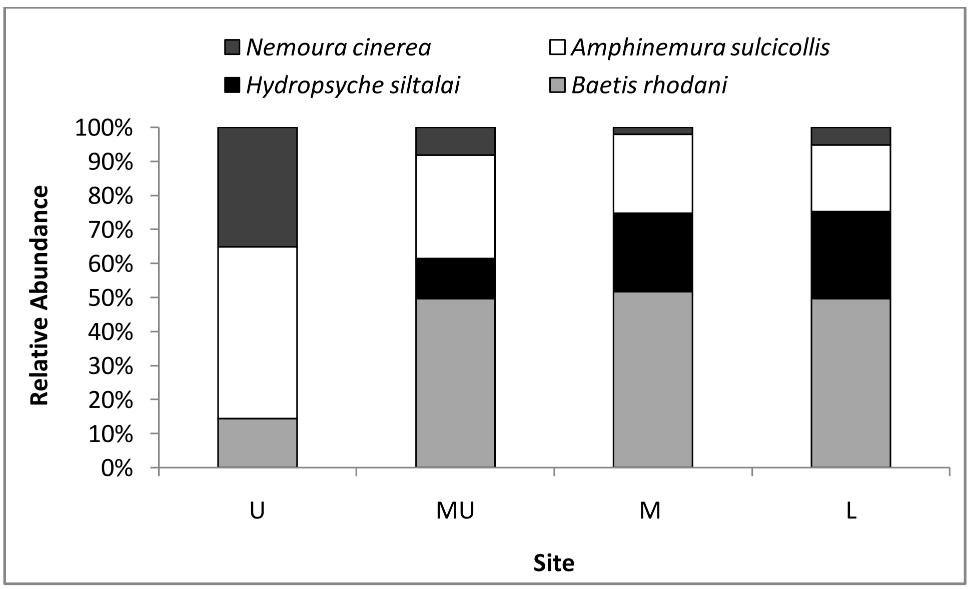


Figure 7. MDS plot of invertebrate samples taken from the Glennamong river at four seasons (Autumn, Spring, Summer and Winter). Vectors indicate the most significant explanatory environmental variables as indicated by BVSTEP.

SIMPER analysis indicated that the main seasonal difference in species were due to increased abundance of *Baetis rhodani* in summer, and increased plecopteran species such as *Amphinemura sulcicollis* Stephens, *Chloroperla torrentium* Pictet and *Brachyptera risi* Morton out of the summer season (Figure 8a). SIMPER analysis highlighted that the main site differences in species were due to increased *Nemoura cinerea* Retzius and *Amphinemura sulcicollis* at the upper site and increasing abundances of *Baetis rhodani* progressing downstream (Figure 8b).



a)



b)

Figure 8 a and b. SIMPER analysis results highlighting the average relative abundance of species responsible for the seasonal variation (a) and spatial variation (b) in the Glennamong invertebrates.

Srahrevagh Invertebrates

The Srahrevagh invertebrate assemblages also notably divided into the upper sites and those further downstream (PERMANOVA, p_{mc} <0.03 in all cases). Similarly to the Glennamong invertebrates, season was also a significant source of variation with summer being different to the winter, spring and autumn (PERMANOVA, p_{mc}<0.05 in all cases). The BVSTEP routine indicated that the environmental variables best correlated with the patterns in the Srahrevagh invertebrate samples were water temperature, altitude and P (Spearmans *rho* =0.358), with temperature splitting the biological data along the y-axis (with season) and altitude splitting the samples according to site (x-axis) (Figure 9).

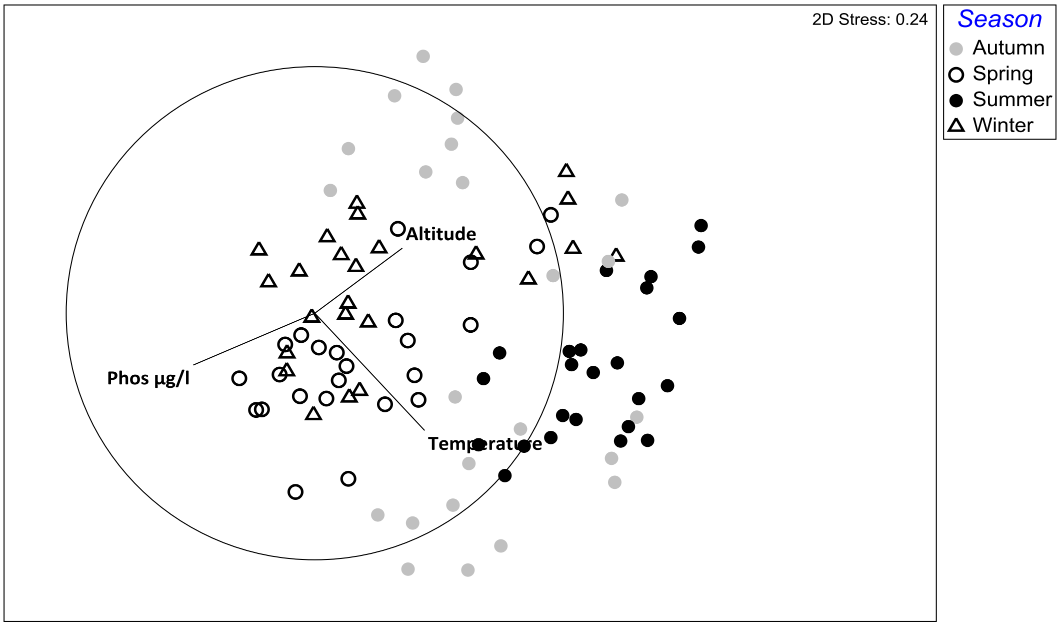
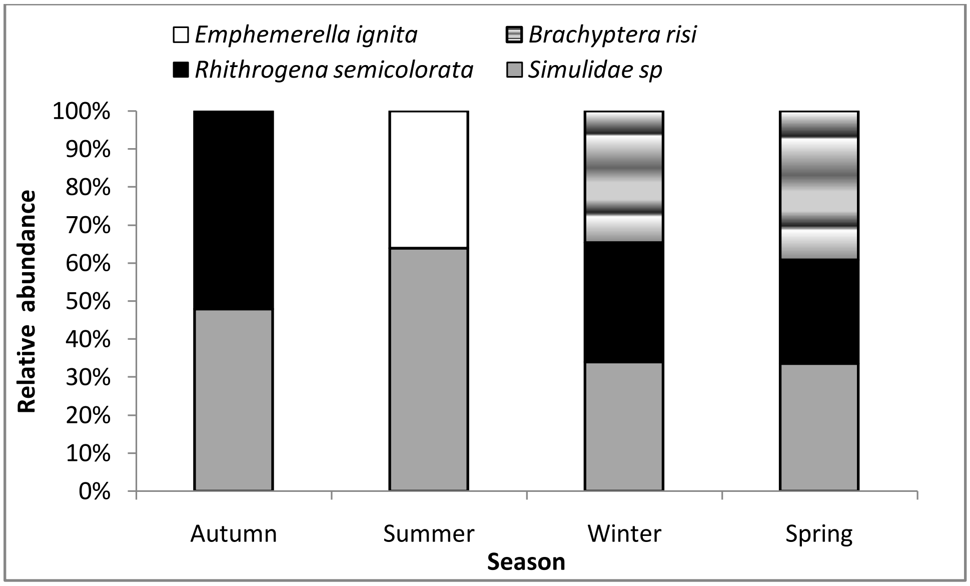


Figure 9. MDS plot of invertebrate samples taken from the Srahrevagh river at four seasons (Autumn, Spring, Summer and Winter). Vectors indicate the most significant explanatory environmental variables as indicated by BVSTEP.

SIMPER analysis indicated that the main seasonal difference in species were due to increased abundances of Simulidae *spp* and *Seratella ignita* Poda in summer being replaced by *Brachyptera risi* and *Rhithrogena semicolorata* out of the summer season (Figure 10a). SIMPER analysis also highlighted that the main site differences in species were due to relatively higher abundance of *Gammarus dubenii* at the upper sites and increased abundance of Elmidae *spp* and *Rhithrogena semicolorata* further downstream (Figure 10b).

a)



b)

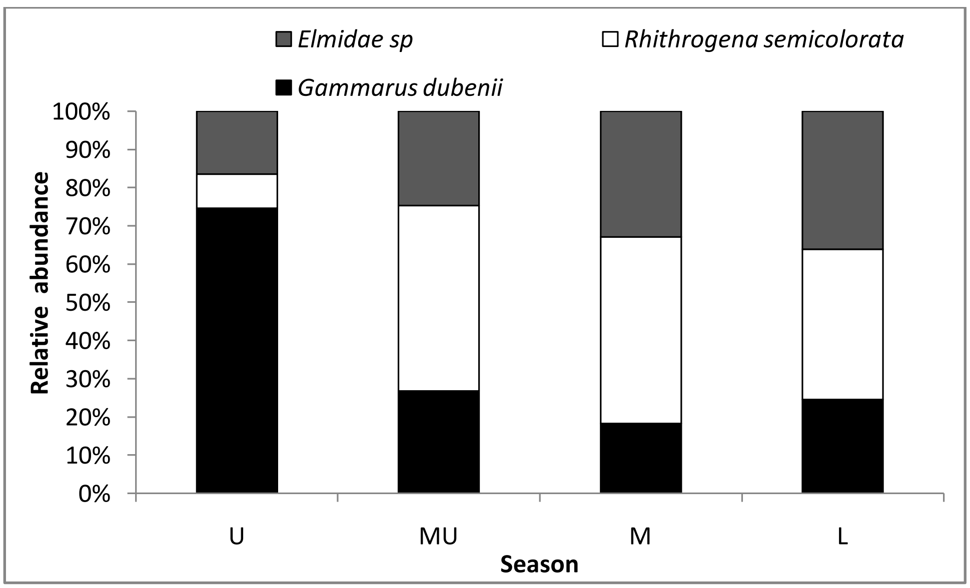


Figure 10 a and b SIMPER analysis results highlighting the average relative abundance of species responsible for the seasonal (a) and site (b) variation in the Srahrevagh invertebrates.

Univariate analysis

Phytobenthic AFDM and TP were highly variable throughout the sampling period and so were dropped from further analysis. TDI-EQR values were consistent across site and season, and demonstrated high status for all samples. The Q-index values varied between season only 0.8 in winter, spring and autumn and 0.6 in summer.

Analysis of variance (ANOVA) indicated that in the WE32 2441 waterbody (Glennamong lower sites), site was a significant source of variation in ACID scores (Table 2), with significantly higher values of ACID at the lowest site (L) and lower values at the mid upper site (MU) (LSD *post hoc* tests, $p < 0.05$). ACID values also varied according to season (Table 2) with significantly higher values of ACID values in summer (LSD *post hoc* tests, $p < 0.0001$). Season was a significant source of variation in TDI scores within the WE32 2441 waterbody (Table 2), with significantly higher values of TDI in summer and lowest values in autumn (LSD *post hoc* tests, $p < 0.0007$). Trophic Diatom Index values within this waterbody did not vary significantly with site.

Analysis of variance (ANOVA) indicated that in the WE32 2781 waterbody (Srahrevagh), site was a significant source of variation in ACID scores (Table 2), with significantly higher values of ACID at the lower (L) site and lower values at the upper (UP) site (LSD *post hoc* tests, $p < 0.0001$). ACID values within this waterbody did not vary significantly with season. Site was also a significant source of variation in TDI scores within the WE32 2781 waterbody (Table 2), with significantly higher values of TDI at the lower (L) site and lower values at the upper (UP) site (LSD *post hoc* tests, $p < 0.0001$). Season had only a marginally significant influence on the TDI scores (Table 2), with higher values in winter (LSD post hoc tests, $p < 0.001$).

Analysis of variance (ANOVA) indicated that in the WE32 2441 waterbody (Glennamong lower sites), season was a significant source of variation in EPT scores (Table 2), with significantly higher values of EPT in winter, and lowest values in summer (LSD *post hoc* tests, $p < 0.0001$). EPT values within this waterbody did not vary significantly with site. SI values within this waterbody also did not vary significantly with season or site.

Analysis of variance (ANOVA) indicated that in the WE32 2781 waterbody (Srahrevagh) season was a significant source of variation in EPT numbers (Table2), with significantly higher values of EPT in winter, and lowest values in autumn (LSD *post hoc* tests, $p <0.05$). EPT values within this waterbody did not vary significantly with site. Season was also a significant source of variation in SI scores within the WE32 2781 waterbody (Table 2), with significantly higher values of SI in summer and lower values of SI in winter (LSD *post hoc* tests, $p < 0.0003$). SI values within this waterbody did not vary significantly with site.

Table 2 ANOVA results for within waterbody variation in biotic indices.

		WE 322441 (Glennamong)			WE 322781 (Srahrevagh)		
	Source of variation	d.f.	F-ratio	*p*	d.f.	F-ratio	*p*
Diatoms							
Acid	Season	3	7.92	<0.01	-	-	n.s.
	Site	2	7.21	<0.01	3	10.53	<0.01
TDI	Season	3	8.62	<0.01	3	2.96	=0.04
	Site	-	-	n.s.	3	60.41	<0.01
Invertebrates							
EPT	Season	3	18.41	<0.01	3	10.584	<0.01
	Site	-	-	n.s.	-	-	n.s.
SI	Season	-	-	n.s.	3	8.6892	<0.01
	Site	-	-	n.s.	-	-	n.s.

Discussion

General spatial and temporal patterns and driving factors of species assemblages

Diatoms

On the whole, multivariate analysis confirmed the longitudinal patterns reported earlier by O'Driscoll et al. (2012) and supported the consensus that alkalinity is the primary physicochemical driver of riverine diatom assemblages in upland blanket peat catchments, followed by disturbance due to hydrological properties such as altitude and catchment area. A distinct shift from a dominance of *Eunotia rhomboidea* (ACID score 2) at the upper reaches to higher abundances of the more acid tolerant *Eunotia exigua* (ACID score 1) further downstream was apparent in the Glennamong. However, this shift was accompanied by increases in circumneutral species such as *Achnanthes oblongella* (ACID score 3) and *Gomphonema parvulum* (ACID score 3) downstream with increasing alkalinity and dilution with larger catchment area. Overall, season did not appear to have a significant influence on the diatom assemblages in the Glennamong.

Similarly, diatoms were not found to vary seasonally in the Srahrevagh river, although the variable DSF (days since last flood) was significant at the Srahrevagh lower site. Although frequency of floods often increases in certain seasons (e.g. winter), this is not the case in the west of Ireland, where floods occur at any stage of the year (Müller, 2000), and so the DSF variable cannot be classed as a 'seasonal' factor, but rather should be considered a hydro-morphological feature of these types of catchments. During periods of low flows, the lower sites were characterised by larger diatom species capable of forming tall structures that overtop the base layer of colonist taxa, e.g. *Ctenophora pulchella* (Ralfs ex. Kützing) William and Round, *Synedra ulna* (Nitzsch) Ehrenberg and *Gomphonema truncatum* (Ehrenberg) and *Epithemia adnata* (Kützing) Rabenhorst (Biggs et al., 1998). The upper reaches of the same river are dominated with r-strategist species such *Achnanthes oblongella* and *Achnanthidium minutissimum* types, which are quick to re-colonise after a flood. The rivers in upland blanket peat catchments are usually spate, prone to flash flooding and exhibit a quick response time to precipitation (Müller, 2000). Long periods of time elapsing without a flood are necessary for a stable periphyton matrix where k-strategist species characterising the larger diatom taxa can grow (Biggs et al., 2000). The number of grazers did not stand out as a significant variable; however, a shift from *Achnanthes oblongella* to *Achnanthidium* types did occur between the upper and mid-upper sites in the Srahrevagh. *Achnanthes oblongella* lies flat against the stone surface and has highest abundances at the upper site, where grazers are less dominant and collector/ gatherers are most predominant. *Achnanthidium* types are associated with increased grazing pressure (Biggs et al., 1998). As *Achnanthidium* types increase downstream, *Achnanthes oblongella* decreases suggesting a possible competitive association between these two taxa.

Invertebrates

On the whole, multivariate analysis highlighted spatial and seasonal variation in the invertebrates. Sites in the upper reaches of the Glennamong had greater abundances of acid-tolerant Plecoptera species, reflecting the lower alkalinity. The lower reaches demonstrated an increase in alkalinity and were represented by a shift to more circumneutral species such as *Baetis rhodani*. A feeding response can also be inferred from the upstream-downstream gradient and was reflected in the higher abundance of Plecoptera shredders at the upper sites where the dead leaf detritus is readily made available from the surrounding perennial *Molinia caerulea* L. and *Eriophorum vaginatum* L. grasslands. An increase in Ephemeroptera grazers at the lower sites coincided with a slightly more stable periphyton matrix. Seasonal guilds in the invertebrates largely reflect life-histories (Butler, 1984; Giller and Twomey, 1993). Higher temperatures in the summer coincided with the appearance of *Seratella ignita* which disappears in the autumn, winter and spring samples. Lower seasonal temperatures coincide with the appearance of *Brachyptera risi* and *Rhithrogena semicolorata*. Bivoltine *Baetis rhodani* appeared in samples throughout the year and at all sites in the Srahrevagh. In the Srahrevagh river, a disturbance guild was highlighted by representation from *Philopotamus montana* and *Gammarus dubenii* at the upper site, consistent with first order streams, waterfalls and high dissolved oxygen (Elliott, 1981) and probable scouring. Sites further downstream showed an increase in the grazer *Rhithrogena semicolorata*.

Implications for WFD waterbodies

The data presented in this study indicated that in upland blanket peat catchments biotic indices fluctuated in response to catchment characteristics along a river. Higher diatom ACID and TDI scores were associated with increased alkalinity and greater pH dilution downstream away from the constraining influence of the peat. The relationship observed between the TDI and alkalinity in both the Srahrevagh and Glennamong river supported the earlier work of Kelly et al. (2008), which showed that the TDI of reference samples was correlated positively with alkalinity. However, the TDI-EQR corrects for alkalinity with relatively few low alkalinity sites (Kelly et al., 2008; Chapter 3, section 3.2). There may be interactions between metrics designed to evaluate nutrients and underlying pH gradients (Schneider et al., submitted). The implications of this are that if one were to take a sample downstream in a waterbody draining an upland blanket peat catchment, a higher value would be obtained than if one sampled further upstream. The definition of waterbodies for use in ecological monitoring programs appears, from this data, to be a crucial part of accounting for site variation, particularly in regions with low alkalinity. The data presented here show significant downstream variation in diatom derived acid scores, even within a waterbody, which is undoubtedly linked to the rapid rising and falling of pH values in lower reaches of these types of rivers, and the dilution capacity of the water body. Derivation of EQRs from a waterbody from diatom acid indices therefore needs to be carried out with care, and with the knowledge that these scores may be naturally lower at the upper reaches of a waterbody.

The seasonal variability in diatom index scores of waterbodies was very small indicating low temporal heterogeneity. These results suggest that the reliability of diatom based biotic indices is largely independent of seasonal variation, and a fairly consistent index value will be obtained, irrespective of sampling time. This is in contrast to Kelly et al. (2006), who suggest that at least six replicates over two/ three years are required to provide a reliable status class using the TDI-EQR. However, this study focuses on upland blanket peat catchments exclusive to low alkalinity areas and Kelly et al. (2006) considers a more complete alkalinity gradient. The lack of seasonal variation in the diatom assemblages and resulting biotic indices may be a reflection of hydro-morphology of these types of rivers, where frequent floods do not allow enough time for a stable periphyton matrix to develop (Biggs et al., 2000). This highlights the importance of catchment level studies and shifts the focus from studies with a large number of sites to problem-solving within individual catchments.

The TDI-EQR, which measures eutrophication, demonstrated consistent scores across season and site indicating that the inclusion of the alkalinity and season correction factor accounts for the natural variation occurring in these streams. This result however has to be considered with caution as the TDI-EQR have not yet been fully calibrated for low alkalinity sites (i.e. < 6.8 mg L^{-1} $CaCO^3$ (O'Driscoll et al., 2012).

Significant seasonal variation was presented in the EPT index scores with higher values obtained in winter and spring and lower values obtained in the summer/ autumn period. These values equate to a Q-Index EQR of 0.8 (unpolluted) to 0.60 (moderately polluted) (EPA, 2006). In contrast with previous work (Sprules, 1947; Vlek, 2006; Šporka et al., 2006), the EPT index in upland blanket peat catchments does show seasonal variation. Sprules (1947) reported that while the number of Plecoptera species decreases with increasing temperature, Ephemeroptera species increase, thus avoiding seasonal differences in EPT index scores. However, there is a relatively low number of sensitive Ephemeroptera observed in the acid sensitive Glennamong river and so when the number of Plecoptera species decrease, they are not replaced with increased numbers of Ephemeroptera species. It is clear that the invertebrates of upland blanket peat catchments are more dependent on phenological cues such as daylight hours and water temperature, than diatoms. Many patterns appeared in the analysis of this data and controlled experiments may be necessary to disentangle the environmental variables driving these trends. Sampling twice a year would incorporate this natural variation, however, sampling twice a year can be resource consuming. In Ireland, invertebrates are generally sampled for water quality monitoring programs in the summer/autumn period when the pressures (high temperature, low DO and high nutrient/ organic pressure) are likely to be at their greatest and the invertebrate community is most likely to be impacted (B. Kennedy, EPA Ireland, *pers. comm*.). The data presented here indicate that this sampling period will give the lowest biotic scores for upland peat rivers, and so are most likely to capture any potential impacts.

Conclusions

- Diatom indices seem relatively unaffected by seasonal variation in waterbodies draining peat catchments
- Diatom indices are sensitive to downstream variation associated with increasing alkalinity even within a waterbody, and biologists using one sampling location within a waterbody need to be aware of this when drawing conclusions about ecological quality.
- However, whilst spatial variation in the diatoms is apparent in the raw biotic indices, assessment in terms of the EQR scale indicate that these rivers are typically oligotrophic and acidic irrespective of natural variation.

- Seasonality is the major factor affecting invertebrate assemblages, and the raw biotic indices and EQRs. The data presented here indicate that lowest values would be expected in the summer.

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Bibliography

See Chapter Six pp. 187 - 222.

3.4 Responses of macroinvertebrate and diatom communities in acid sensitive and oligotrophic streams to peatland forest clearfelling

Connie O'Driscoll[1], Elvira de Eyto[2], Michael Rodgers[1], Mark O'Connor[1], Zaki-ul-Zaman Asam[1] and Liwen Xiao[1*]

[1] Civil Engineering, National University of Ireland, Galway, Republic of Ireland

[2] Marine Institute, Newport, Co. Mayo, Republic of Ireland

* Corresponding author: Liwen Xiao; Email: liwen.xiao@nuigalway.ie; Tel: +353 91 494188

Abstract

Harvesting of peatland forests increases stream phosphorus and suspended sediment concentrations, water temperature and sunlight availability. However, few studies have addressed the responses of phytobenthic and benthic invertebrate assemblages to forest clearfelling in upland peat catchments, characterised as oligotrophic and naturally acidic. In this study, a multiple before – after – control - impact (MBACI) experiment was carried out in three neighbouring peatland catchments to evaluate the impact of forest harvesting on the phytobenthos and benthic invertebrates. Water quality, benthic invertebrate and diatom assemblages and their associated biotic indices were monitored in five first order peatland forest streams before and after harvesting for three years. The results indicated that forest clearfelling shifted the dominant invertebrate species from *Nemoura cinerea* (47.7 %), Simulidae species (19.5 %) and *Leuctra hippopus* (12.7 %) before harvesting to Chironomidae species (98 %) after harvesting and reduced the invertebrate biotic indices EPT (Ephemeroptera Plecoptera Trichoptera), diversity and species richness significantly from 4, 1.3 and 7 pre-harvesting to 1, 0.2 and 2 after harvesting, respectively. In contrast, forest clearfelling did not alter diatom assemblages significantly. The dominant diatom species were *Eunotia exigua* and *Pinnularia appendiculata* both before and after harvesting. This raises concerns about the accuracies of using diatoms to test nutrient impacts in naturally acidic streams.

Introduction

Land-use changes negatively affect the ecological integrity of river systems (Allan, 2004). Peatland afforestation was practiced in the UK and Ireland, Fennoscandia, and North America, during the late 20th century (Paavilainen and Päivänen, 1995). Many of these blanket peat forests are now reaching harvestable age and concerns have been raised about the potential impact of harvesting to the receiving aquatic systems. Studies in Ireland, Finland and the UK have found that peatland forests harvesting changed stream flow regimes and deteriorated receiving water quality (Cummins and Farrell, 2003: Niemen, 2003: Rodgers et al. 2008, 2010, and 2011). After 5 years study in the Burrishoole catchment in the west of Ireland, Rodgers et al. (2011) found that whilst best management practices were strictly implemented the daily mean TRP concentration in a study stream increased from about 6 $\mu g\ l^{-1}$ pre-harvesting to 429 $\mu g\ L^{-1}$ one year after harvesting (Chapter 2, section 3.3). An impact of harvesting on the SS concentration and stream flow regime was also observed (Rodgers et al., 2008, 2010). In addition, in Ireland and the UK, many of the earlier afforested upland blanket peat catchments were established without any riparian buffer areas, with trees planted to the stream edge (Ryder et al., 2010). Harvesting of these catchments to the stream edge reduces canopy cover over the streams and increases sun light availability, resulting in an increase in the amount of autochthonous energy production (Chizinski et al., 2010) and daily maximum temperatures (0.05 – 1.1 ^{o}C) (Rodgers et al., 2008). The physical and chemical changes due to harvesting activities can modify stream habitats and change biota communities. Protection of the aquatic systems in these upland peat forest catchments is vital as the WFD stipulates EU Member States must maintain "high and good ecological status" where it exists and to restore at least "good status" for all water bodies by 2015 (European Union, 2000).

Though the assessment of peatland forest harvesting on hydrology, soil erosion and nutrient release has been studied widely, few studies have focused on the impact of peatland forest harvesting on water ecological status, which requires incorporation of chemical parameters in unity with ecological dynamics such as light availability and flow regimes (Karr et al., 2000; Leira and Sabater, 2005). Benthic invertebrates and diatoms have been used successfully for assessment of ecological quality and aquatic ecosystems worldwide (Kelly et al., 1998; Leira and

Sabater, 2005; Clarke and Hering, 2006; Chen et al., 2008). They are considered to be useful indicators for assessing potential effects of timber harvest due to their well-described responses to environmental conditions (i.e. light, water temperature, nutrients, sediment) that may change during land management for timber harvest (Naymik and Pan, 2005; Reid et al., 2010). Reid et al. (2010) carried out a study on the response of stream invertebrate communities to forest harvesting in catchments predominated by clay and yellow-brown earths in New Zealand, and found that the impact of harvesting increased as the proportion of upstream catchment harvested increased and after riparian vegetation was harvested. They contributed the changes in macroinvertebrate communities to the increase of temperature, fine sediments and algal biomass (Reid et al., 2010). In the U.S., Naymik and Pan (2005) used diatom assemblages as indicators to assess the impact of timber harvesting on coastal Oregon streams, and found that changes in diatom assemblages was positively correlated with the percentage of upstream area harvested and water quality variables such as N and P. However, the responses of invertebrate and diatom assemblages to forest harvesting can be affected by forest management strategies, climate and landscape such as soil types and bedrocks. In their study, Richards et al. (1996) found that geology was one of the most important factors determining stream habitat in a study of 45 watersheds in Michigan. Similarly, Leland (1995) related the periphyton community with the rock type rather than land use. To the best of our knowledge, the response of benthic invertebrates and diatom assemblages in naturally acidic upland blanket peat streams to forest harvesting upland blanket peat catchments has not been well explored.

In this study, the impact of forest harvesting on the phytobenthos and benthic invertebrates was investigated in the Nephin Beg Range in the west of Ireland by using a multiple before- after-control-impact (MBACI) experimental design. The Nephin Beg Range is a Natura 2000 site (IE0004098) and one of the most important Atlantic salmon and freshwater mussel areas in Europe. Headwater streams in these catchments play a crucial role in detecting water quality deterioration because they provide the link with the catchment experiencing the land use change (Richardson and Danehy, 2006; Louhi et al., 2010) and which contain the threatened Red List species Salmo salar L. and Margaritifera margaritifera L.

As a part of BMPs, phased felling is recommended in the UK (Forestry Commission, 1988) and Ireland (Forest Service, 2000) to reduce the negative impact of harvesting on water quality. Harvesting smaller sized coupes in a catchment at any one time reduces the impact to the receiving water (Neal et al., 2003; Rodgers et al., 2011). Assessments of phased felling on stream water quality have been carried out by monitoring water chemical parameters (Neal et al., 2003; Rodgers et al., 2010). There is much less information available on the short term effects of phased felling on the ecological status of headwater streams and consequently the larger rivers into which they feed. Therefore another objective of this study is to investigate the effectiveness of the phased felling approach on the ecological status of the receiving salmonid rivers.

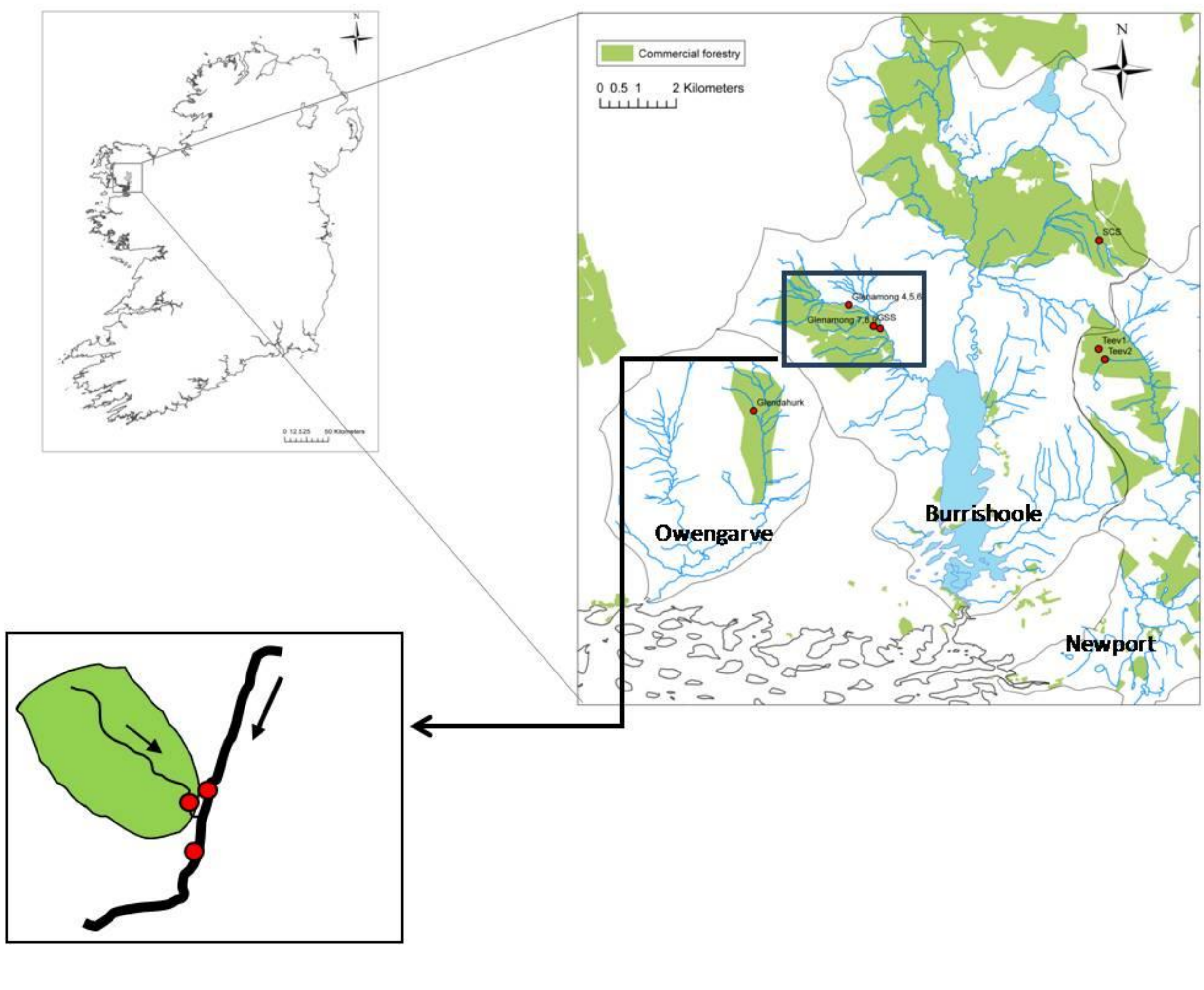


Figure 1 Geographical location of the study streams and a schemtic of the sampling sites above and below the confluence of the small first order stram and larger river.

Materials and methods

Study sites, characterisation and experimental design

This study was based in five completely forested streams – Glendahurk, Teev1, Teev2, GSS and SCS in three adjoining catchments in Nephin Beg Range, located in Mayo in the north west of Ireland (Figure 1). The catchments are covered in blanket peat and overlie quartzite and schist bedrock. The catchment systems are described as oligotrophic and have a low acid buffering capacity (Byrne et al., 2004). The main land uses are forestry and sheep grazing, and the catchments receive an average precipitation of over 2,000 mm per year (Dalton et al., 2010). Commercial coniferous plantations were planted in blocks or coupes starting in the 1950s (O'Driscoll et al., 2011). The streams selected for this study are approximately 50 – 100 cm wide with 50 cm-height banks on either side. They have upstream catchment sizes ranging from 10 ha to 32 ha and typically flow over bedrock, but in some sections have a peat floor. *Pinus contorta* (lodgepole pine) was the main tree species in Glendahurk, GSS and SCS. Teev1 and Teev2 consisted of *Picea sitchensis* (Sitka spruce) and *Pinus contorta* (Lodgepole pine). The five streams were analogous in terms of slope, type of substrata, and land management use. They were first order streams and originated in the forestry. To determine the response of macroinvertebrate and diatom to forest harvesting, the MBACI design was employed in this study. GSS, Teev1 and Teev2 served as study streams. The Teev1 and Teev2 impact streams were clearfelled in autumn 2009 and the GSS was clearfelled in 2011. Glendahurk and SCS served as control streams for Teev1 and Teev2 and GSS, respectively, and their upstream areas were not harvested during the study. Beginning in March 2009, the five reaches were monitored for bio-physico-chemical parameters seasonally for three years.

In addition, the salmonid river GL – the receiving river of the impact stream GSS - was selected to study the effectiveness of the phased felling approach on the ecological status of the receiving river. The bio-physico-chemical status of GLa and GLb, which are about 30 m above and below the confluence of the river GL with stream GSS, were monitored during the experiment.

Sample collection and preparation

Benthic invertebrate and phytobenthic samples were collected at the 5 forested study streams (control n = 2; harvested n = 3) and one river between March 2009 and June 2011. Water samples were taken at each sampling point and analysed on the same day for alkalinity and SS using standard procedures (APHA, 1998). Total reactive phosphorus was measured using a Konelab 20 Analyser (Konelab Ltd., Finland). Water temperature was recorded in the field. Benthic invertebrates were sampled using a standard 1 mm pond net and a 1-minute kick sample in river riffles. Invertebrates were elutriated in the field from associated sand, stones and woody debris, and preserved in 70% industrial methylated spirits. Periphyton was removed from five cobble surfaces with 100 ml of stream water in accordance with Kelly et al. (1998). Orthogonal measurements of each stone were taken in the field, and converted to stone surface area using the equation of Dall (1979). The samples were stored in the dark and analysed in the laboratory later the same day for periphyton AFDM and TP (APHA, 1998). Sub-samples were taken from the periphyton sample and cleaned using the cold acid permanganate method for diatom analysis (Kelly et al., 2005). Permanent slides were prepared using Naphrax (r.i. = 1.74).

Identification

Invertebrates were sorted and identified to species level where possible following the keys of Hynes (1977); Elliot et al. (1988); Friday (1988); Edington and Hildrew (1995) and Wallace et al. (2003). Diatoms were identified to species level, where possible, and counted at x1,000 magnification using an Olympus BX-51 microscope equipped with an x100 phase contrast objective. At least 300 valves were counted and identified per slide using Krammer and Lange-Bertalot (1986, 1988, 1991a, b). Certain taxa were difficult to identify and the approach adopted for these species is as follows: three types of *Achnanthidium minutissimum* varieties were recognized and split into types based on Potapova and Hamilton (2007): the 'capitate' morph/ 'type a'; 'linear'/ 'type b' and 'wide linear-lanceolate'/ 'type c'. These three *A.*groups were largely present in girdle view and so were enumerated separately in girdle view and then divided between the three morphological groups based in proportion to their relative abundance. *Eunotia exigua*, also present in high numbers in girdle view was difficult to distinguish with *E. tenella* and *E. meisteri* and so the three were combined and considered as *E. exigua* complex as recommended by DeNicola (2000). *Gomphonema parvulum* has been described with a number of varieties and attributed environmental preferences, however, populations in these samples had high morphological variability and so have been termed *G. parvulum* complex.

Data analysis

Multivariate statistical techniques were used to analyse the variation in species abundance and composition at each treatment before and after clearfelling using the PRIMER software package (Plymouth Marine Laboratories, UK). Relative abundances were calculated for the macroinvertebrate and diatom species data, and only species present in more than 5 % of samples were included. The Bray-Curtis similarity matrix was used to generate 2-dimensional plots with the non-metric multi-dimensional scaling (nMDS) technique (Clarke 1993). Whether the treatments differed from one another in species composition before and after clearfelling was tested, and a time treatment interaction was investigated. Treatment comparisons were made with PERMANOVA software (Anderson, 2005). The calculated statistic (pseudo-F) is calculated, like a traditional F-statistic, as the sum of the squared distances among groups divided by the sum of the squared distances within groups (Anderson, 2001; McArdle and Anderson, 2001). Data were untransformed and un-standardised. Analysis was conducted using the Bray–Curtis measure on data expressed in percentages. P-values were calculated by permuting the observations 9999 times, so no assumptions of the distributional form of the data were required. The same procedure was followed to perform the permutational multivariate analysis of variance on the species abundance data for above (GLa) and below (GLb) the confluence in the main GL river. The similarity percentages (SIMPER) procedure was used to identify the major species contributing to the similarity measure obtained (Clarke and Warwick 1994). The macroinvertebrate indices such as EPT and SI (Henrikson and Medin, 1986) and diatom indices such as TDI (Kelly et al., 2008), EQR (Kelly et al., 2006) and ACID (Andrén and Jarlman, 2008) were calculated at each site before and after clearfelling. Diatom and macroinvertebrate species richness (*S*) and Shannon-Weiner index (*H*') (Shannon and Weaver, 1963), Chlorophyll a (Chl a mg m^2), Phaeopigments (mg m^2) and AFDM (mg m^2) were also calculated. Mann Whitney U tests were used to test for differences in indices before and after clearfelling.

Results

Impact of harvesting on environmental variables

In the control streams TRP, water temperature and SS were similar before and after harvesting ($p > 0.1$) (Figure 2). In the impact streams TRP, temperature and SS were significantly higher ($p < 0.05$) after clearfelling, indicating that harvesting had a significant impact on the study sites (Figure 2). In the main river, water quality at GLa and GLb were similar before and after harvesting, indicating that the best management practice of harvesting a smaller proportion of the catchment in one time can mitigate the negative impact of harvesting (Chapter 2, section 2.3).

Table 1 The main macroinvertebrate taxa in the control and study sites before and after clearfelling

	Control			Study	
Before clearfelling	Main macroinvertebrate species	% of the total abundance	Before clearfelling	Main macroinvertebrate species	% of the total abundance
	Baetis rhodani	43.2		*Nemoura cinerea*	47.7
	Leuctra hippopus	27.5		*Simulidae spp*	19.5
	Amphinemura sulcicollis	5.3		*Leuctra hippopus*	12.7
	Simulidae spp	3.6		*Chironomid spp*	5.8
	Brachyptera risi	3.5		*Polycentropus kingi*	3.9
After clearfelling	*Baetis rhodani*	41.5	After clearfelling	*Chironomid spp*	98.0
	Simulidae spp	26.4		*Nemoura cinerea*	1.4
	Brachysira risi	11.6		*Polycentropus kingi*	0.2
	Leuctra hippopus	9.3		*Chloroperla torrentium*	0.2
	Chloroperla torrentium	2.5		*Tipulidae spp*	0.2

Impact of harvesting on macroinvertebrate assemblages

Peatland forest harvesting changed macroinvertebrate assemblages in the study streams. In the control streams, the dominant macroinvertebrate species were *Baetis rhodani*, *Leuctra hippopus* and *Amphinemura sulcicollis* before harvesting and *Baetis rhodani,* Simulidae *spp, Brachyptera risi* and *Leuctra hippopus* after harvesting (Table 1). In the impact streams, the dominant macroinvertebrate species were *Nemoura cinerea*, Simulidae species, *Leuctra hippopus* and Chironomid species before harvesting (Table 1). After harvesting, the Chironomid species dominated the macroinvertebrate community completely, with the abundance of 98 % (Table 1). Harvesting of the peatland forests also reduced EPT, species no. and diversity in the study streams significantly ($p < 0.05$) (Figure 3).

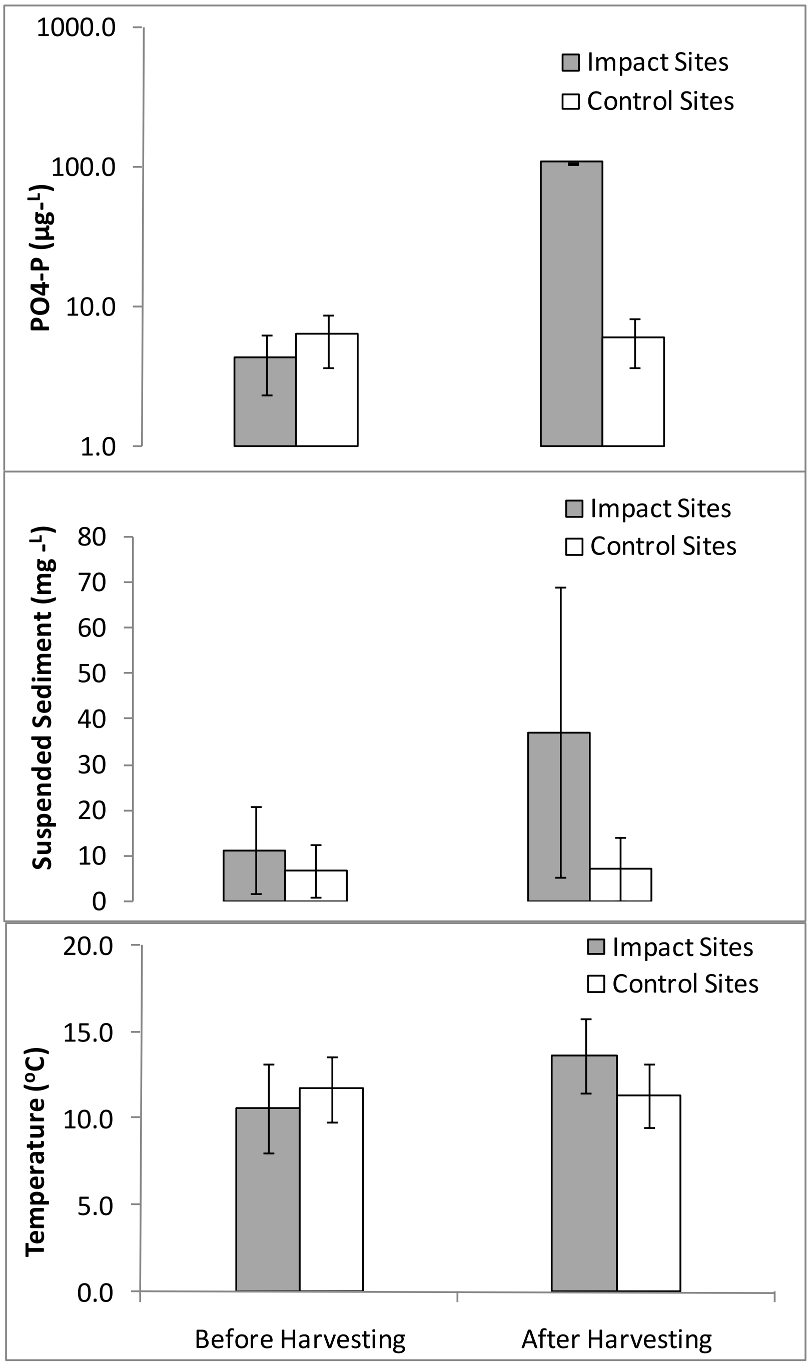


Figure 2 Significant changes in physical and chemical variables at the control and study sites before and after clearfelling

Table 2 Permutation multivariate analysis of variance on macroinvertebrate species composition among levels of the factors treatment (control and study) and time (before and after clearfelling)

a. Source	*df*	SS	MS	F	P(perms)
Time	1	5978.20 (9.32)	5978.20	2.5717	0.025
Treatment	1	12219.00 (19.06)	12219.00	5.2566	0.001
Time x Treatment	1	6278.50 (9.78)	6278.50	2.7009	0.024
Residual	16	37193.00 (58.01)	2324.60		
Total	19	64119.00			

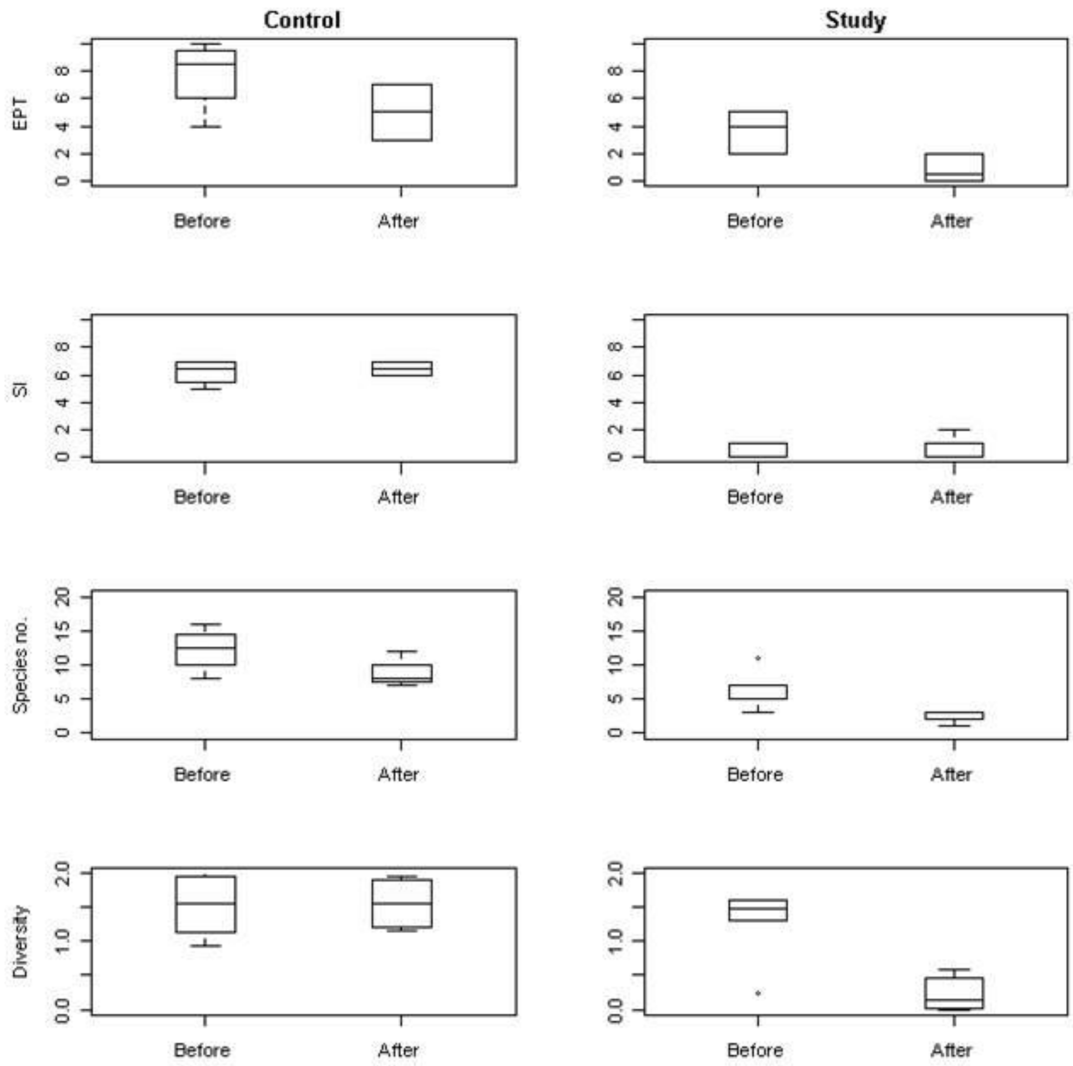


Figure 3 Comparison between treatments control and study, before and after harvesting of macroinvertebrate indices (EPT, SI, Species no. and diversity)

In the study streams, the two-dimensional MDS plots of macroinvertebrate species data showed distinct separation between groupings of the study sites before and after clearfelling. The before clearfelling samples generally clump together with the control samples and away from the after clearfelling samples (Figure 4). This finding agrees with the PERMANOVA, as it revealed a significant ($p < 0.05$) effect on macroinvertebrate time x treatment interaction (Table 2).

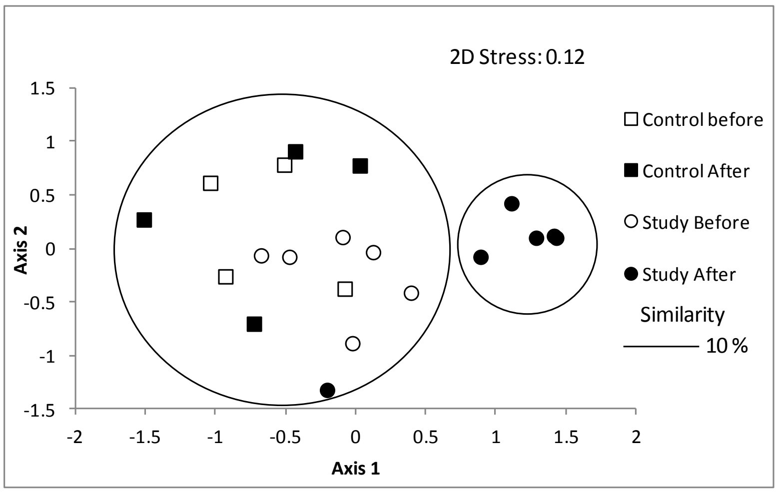


Figure 4 MDS ordination of Bray – Curtis similarities from macroinvertebrates species abundances data for the five study sites shown before and after felling; with superimposed cluster analysis at similarity levels of 10 %.

Results from pairwise comparisons between the control x before and the control x after show no significant variation in the macroinvertebrates, but are highly significant between the study x before and the study x after ($p < 0.01$). The macroinvertebrate SIMPER result of mean similarity within before harvesting samples at the study site was produced by the mean abundances of *Nemoura cinerea*, *Leuctra hippopus*, Simulidae species, Chironomid species, Dicranota species and *Polycentropus kingi*, and after clearfelling mean sample similarity was largely due to Chironomid *spp*, *Nemoura cinerea* and *P. kingi* (Figure 5a). The dissimilarity between times was due to the high abundance of Chironomid species (83 %) after clearfelling (Figure 5b).

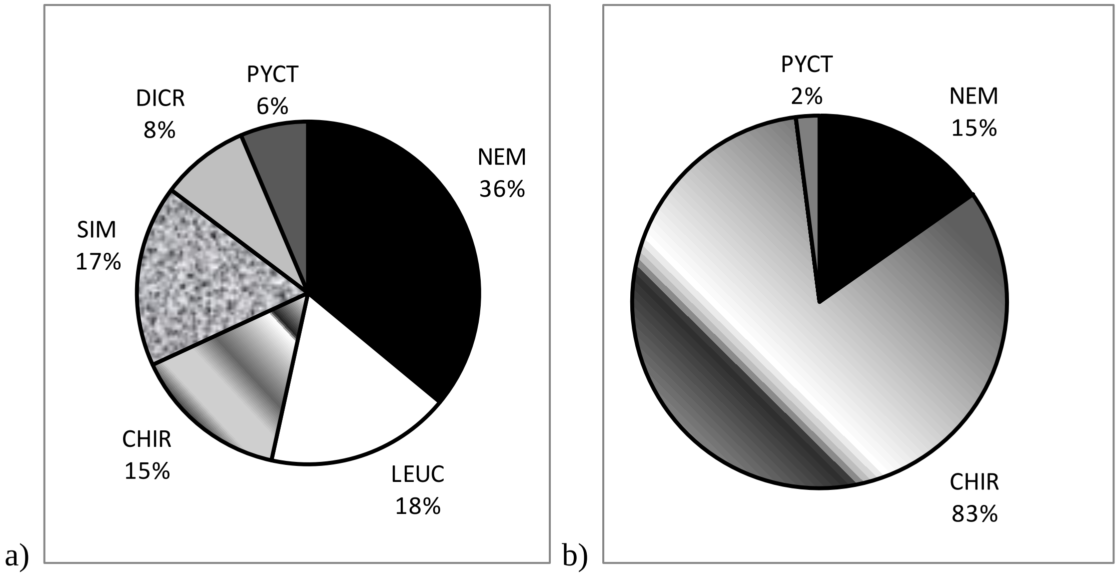


Figure 5a and b Percentage abundance of the species of macroinvertebrates contributing to the similarity measure obtained for the study site before (a) and after (b) felling.

Impact of harvesting on diatom assemblages

Table 3 Permutation multivariate analysis of variance on diatom species composition among levels of the factors treatment (control and study) and time (before and after clearfelling)

b. Source	*df*	SS	MS	F	P(perms)
Time	1	2656.90 (5.72)	2656.90	1.2323	0.261
Treatment	1	6083.20 (13.10)	6083.20	2.8216	0.087
Time x Treatment	1	1948.90 (4.20)	1948.90	0.9040	0.412
Residual	16	34495.00 (74.27)	2156.00		
Total	19	46445.00			

The impact of peatland forest harvesting on river diatom assemblages is not significant (Table 3). In the control streams, the diatom assemblages were dominated by *Achnanthes oblongella*, *Gomphonema parvulum*, and *Eunotia exigua* before and after harvesting (Table 4). In the impact streams, the dominant species were *Eunotia exigua* and *Pinnularia appendiculata* both before and after harvesting.

Table 4 The main diatom taxa in the control and study sites before and after clearfelling

Control			Study		
Before clearfelling	Main diatom species	% of the total abundance	Before clearfelling	Main diatom species	% of the total abundance
	Achnanthes oblongella	59.7		*Eunotia exigua*	70.2
	Gomphonema parvulum	11.7		*Pinnularia appendiculata*	13.9
	Eunotia rhomboidea	8.6		*Eunotia subarcuatoides*	2.3
	Eunotia exigua	6.0		*Eunotia bilunaris* var. *mu*	5.1
	Achnanthidiumum minutissimum type 1	3.3		*Eunotia paludosa*	2.9
After clearfelling			After clearfelling		
	Achnanthes oblongella	58.5		*Eunotia exigua*	63.6
	Gomphonema parvulum	17.2		*Pinnularia appendiculata*	13.3
	Eunotia exigua	6.1		*Eunotia subarcuatoides*	7.9
	Pinnularia appendiculata	3.0		*Eunotia paludosa*	2.7
	Achnanthidiumum minutissimum type 1	3.0		*Pinnularia subcapitata*	2.7

No significant changes to the phytobenthic indices (Chl a, AFDM, ACID, TDI, diversity and species no.) were observed after harvesting ($p > 0.05$). There were two distinct groupings of study streams samples from before and after clearfelling in the two-dimensional diatom MDS (Figure 7). However, samples from the control streams before and after clearfelling intermingled with the after clearfelling study streams samples (white circles Figure 7).

The results from the diatom pairwise comparisons show no significant variation between the control x before and the control x after; and the study x before and study x after wee marginally significant ($p = 0.76$). The diatom SIMPER result of mean similarity within before samples at the impact sites was produced by the mean abundances of *Eunotia exigua*, *Pinnularia appendiculata*, *Eunotia paludosa*, *Eunotia subarcuatoides* and. *Frustules rhomboids;* and after mean sample similarity was largely due to *Achnanthes oblongella*, *Eunotia exigua*, *Pinnularia appendiculata*, *Gomphonema parvulum* and *Meridion circulare* (Figure 8). The dissimilarity between times was due to the after high abundance of *Achnanthes oblongella* (35 %) and the appearance of *Gomphonema parvulum* and *Meridion circulare* (Figure 9).

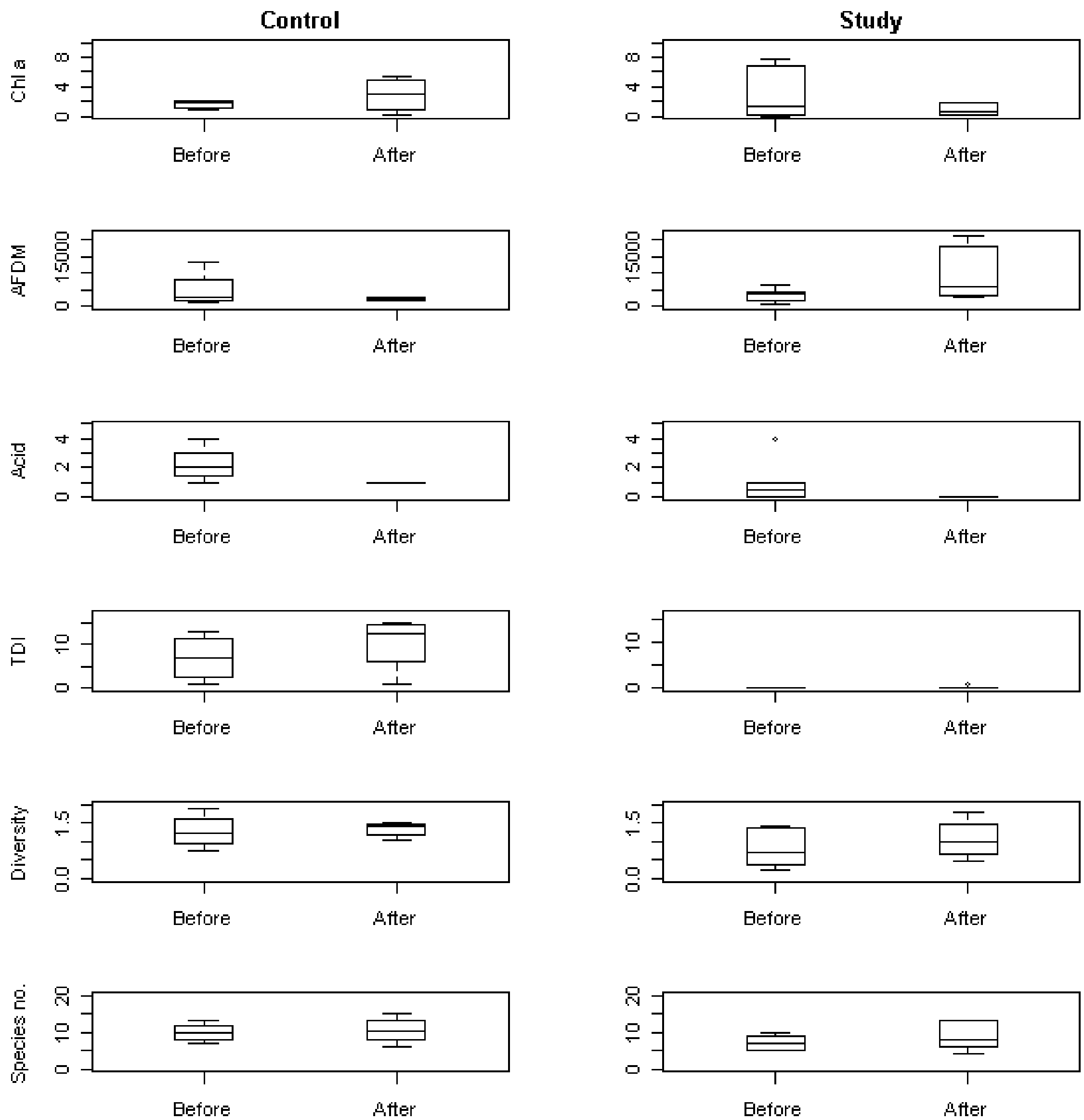


Figure 6 Comparison between treatments control and study, before and after harvesting of phytobenthic indices (Chl a, AFDM, Acid, TDI, diversity and species no.)

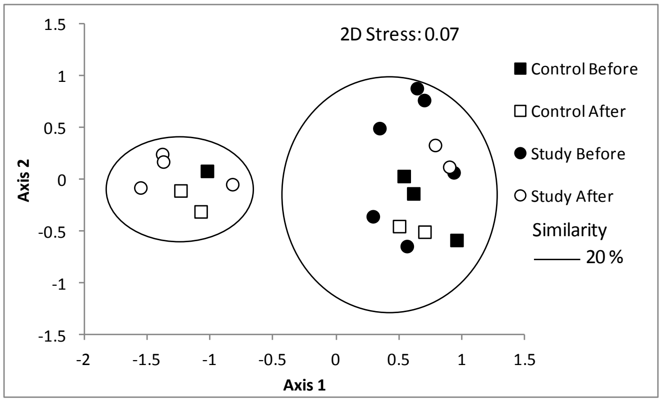


Figure 7 MDS ordination of Bray – Curtis similarities from diatom species abundances data for the five study sites

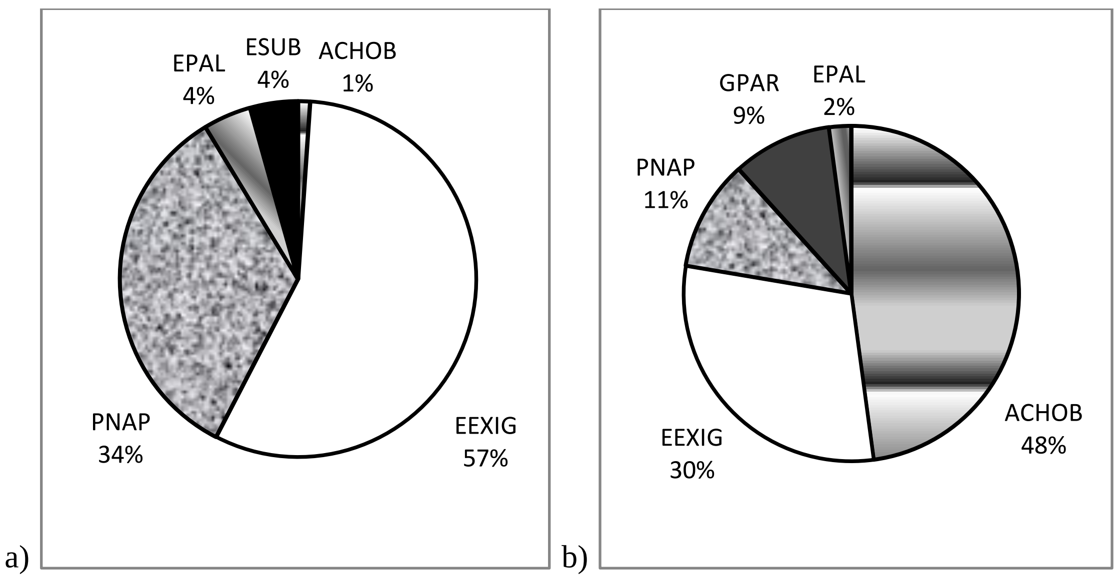
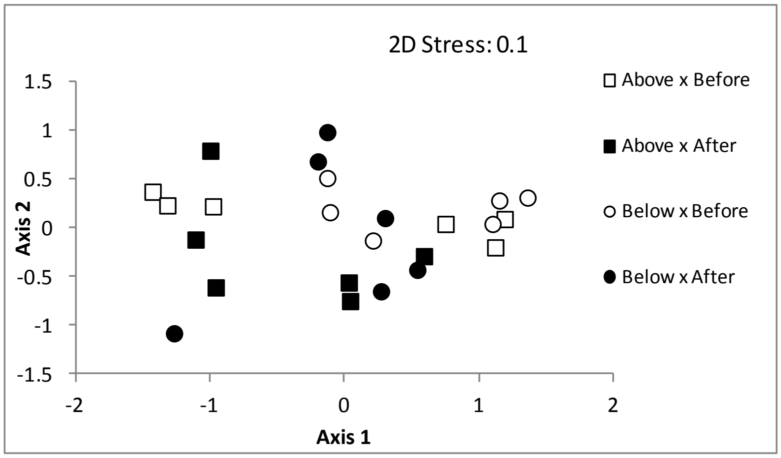


Figure 8 Percentage abundance of the species of diatoms contributing to the similarity measure obtained for and (a) and after (b) felling.

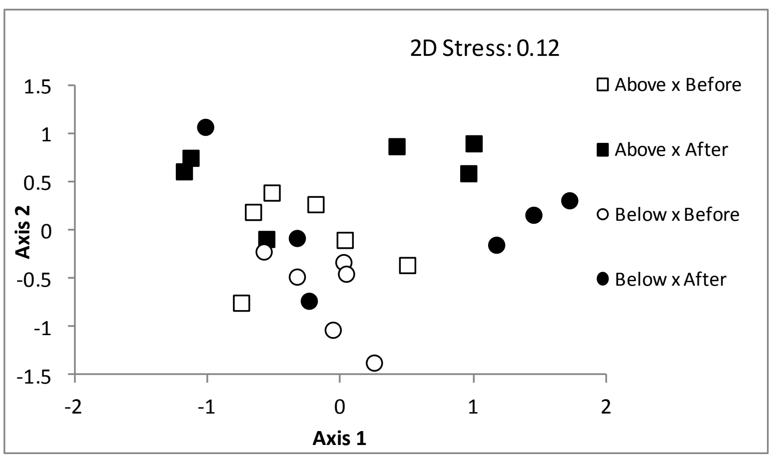
Main river

No significant macroinvertebrate and diatom assemblage change was observed in the main river after harvesting. *Baetis rhodani*, *Leuctra hippopus* and Simulid species were the main macroinvertebrate species in GLa and GLb before and after harvesting. The diatom community was dominated by *Achnanthes oblongella*, *Eunotia exigua*, *Gomphonema parvulum* and

Tabellaria flocculosa in GLa and GLb before and after harvesting. The two-dimensional MDS plots of macroinvertebrate and diatom species data showed no separation between groupings of the above and below sites before and after clearfelling with the before samples (Figure 9a and b).



(a)



(b)

Figure 9 a and b MDS ordination of Bray – Curtis similarities from macroinvertebrate (a) and diatom (b) species abundances data for the sites above and below the confluence before and after felling.

This agrees with the PERMANOVA, as it revealed insignificant ($p > 0.05$) effect on macroinvertebrate (Table 4a) and diatom (Table 4b) time x treatment interactions. Results from pairwise comparisons between the above x before, above x after, below x before, and below x after showed no significant variation ($p > 0.05$).

Table 5 Permutation multivariate analysis of variance on macroinvertebrate (a) and diatom (b) species composition among levels of the factors treatment (above and below) and time (before and after clearfelling).

a. Source	*df*	SS	MS	F	P(perms)
Time	1	1812.90 (8.35)	1812.90	2.1448	0.113
Treatment	1	2477.10 (11.41)	2477.10	2.9306	0.042
Time x Treatment	1	511.69 (2.36)	511.69	0.6054	0.574
Residual	20	16905.00 (77.88)	845.25		
Total	23	21707			

b. Source	*df*	SS	MS	F	P(perms)
Time	1	3438.60 (12.47)	3438.60	3.4845	0.016
Treatment	1	4317.70 (15.65)	4317.70	4.3754	0.010
Time x Treatment	1	91.98 (0.33)	91.98	0.0932	0.980
Residual	20	19736.00 (71.55)	986.81		
Total	23	27584			

Discussion

This study shows that peatland forest harvesting activities significantly increased the TRP and SS in the impact streams. These findings are consistent with the studies carried out in peatland forest catchments by Nieminen (2003), Cummins and Farrell (2003) and Rodgers et al. (2008, 2011, and Chapter 2, section 2.3).

In this study the macroinvertebrate assemblages were affected by peatland forest harvesting activities. Chironomid species dominated the macroinvertebrate community completely after harvesting. Domination of Chironomid species at the forest impacted sites was also observed in other studies and was considered as a pattern characteristic of a severely disturbed aquatic ecosystem (Adamus and Bandt, 1990; Beyene et al., 2009; Ryder et al., 2011). Organisms physiologically adapted to low oxygen tension exploit the excess nutrients available and thus dramatically increase in abundance. Families belonging to the Plecoptera group are clear-water fauna (Bouchard, 2004) and abundances reduce as pollution load increases. A corresponding reduction in EPT, macroinvertebrate diversity and species richness was also observed.

One factor commonly associated with changes in aquatic invertebrate communities is the available sunlight and water temperature (Johnson and Jones, 2000; Chizinski et al., 2010). Miserendino et al. (2011) observed strong relationships between macroinvertebrate metrics with DO, TON and SS. The shift from detritivore Plecopterans to the intense numbers of Chironomid grazers has been reported in the literature. Kobayashi et al. (2010) observed that following clearfelling, benthic invertebrates typically shift from detritivore to grazer-dominated communities due to changes in the trophic base from allochthonous to autochthonous (Nislow and Lowe, 2006), and increase in abundance and production (Stone and Wallace, 1998).

Diatoms have been demonstrated to be sensitive indicators of many kinds of disturbances in streams as they respond quickly to changes in water quality (Stoermer and Smol, 2000). Phosphorus above a concentration of about 30 $\mu g\ L^{-1}$ can trigger eutrophication in freshwaters (Carpenter et al., 1998; Boesch et al., 2001). Many previous studies have shown marked changes of diatom assemblages after harvesting (Naymik and Pan, 2005; Yang et al., 2008; Wang et al., 2009). Lowe et al. (1986) observed higher abundances of 14 diatom species at their harvested sites compared to the control sites six years after harvesting and attributed these differences to increased light availability. In their study, Yang et al. (2008) found that TP was the most important variable in explaining the diatom distributions and accounted for 9.5 % variance of diatoms. Total phosphorus concentrations of 80 – 110 $\mu g\ L^{-1}$ could switch macrophyte dominated to alga-dominated states in the lake (Yang et al., 2008). Similarly, Wang et al. (2009) found that TP and velocity were the most important factors explaining diatom distributions in their study. In this study, as the harvesting activities increased P concentrations and water temperature, it was therefore expected that changes would be observed in diatom assemblages. However, no significant impact was observed in the diatom assemblages or related indices. A slight shift was evident in some of the clearfelled sites with an increase in *Achnanthes oblongella*, *Gomphonema parvulum* and *Meridion circulare*. *Achnanthes oblongella* is reported to be abundant in headwater streams with circumneutral pH and low nutrient concentrations (Chapter 3, section 3.2). *Gomphonema parvulum* is thought to favour high nutrient concentrations optimally occurring between 0.35 and 1 mg l^{-1} P. *Meridion circulare* is reported to be confined to cool running waters where it not tolerant of low pH or high nutrient concentrations. The expected shift from oligotrophic species to nutrient tolerant species was lacking. The possible strong temporal variation observed in the diatom assemblages may have masked or confounded the impacts. However, it is also probable that because of the naturally acidic nature of the catchment and the associatred impairment of the diatom assemblages due to acidity the succession of more nutrient tolerant diatoms is prohibited. O'Driscoll et al. (Chapter 3, section 3.2) investigated diatom assemblages and the environmental variables that are the driving factors in the same peatland catchment and found that alkalinity and conductivity were the main physicochemical drivers of the diatom assemblages and nutrient enrichment from forestry activities did not stand out as having a major influence on the diatom assemblages.

Conclusion

- Forest clearfelling in these acidic upland blanket peat catchmetns appear to have a great effect on the water quality and macroinvertebrates but little effect on the phytobenthos.
- The lack of impact on the diatom assemblage could be due to the low alkalinity and natural acid in the streams.
- The strong temporal variation observed in the diatom assemblage may have masked or confounded the impacts.
- Phased felling in acid sensitive catchments appears to be an efficient BMP in protecting larger salmonid rivers against the additional input of nutrient and sediments after forestry harvesting.

Acknowledgements

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Bibliography

See Chapter Six pp. 187 - 222.

Chapter Four

Mitigation methods

4.1 Introduction

This chapter comprises three research papers, which are part of the project SANIFAC (RSF 07 552) funded by COFORD and the Irish Department of Agriculture, Fisheries and Food. Dr. Michael Rodgers and Dr. Liwen Xiao are the Project PI and Project Coordinator, respectively, and contributed to the overall project design.

The first paper has been published by the peer-reviewed, international journal Water, Air and Soil Pollution (O'Driscoll et al., 2011. A potential solution to mitigate phosphorus release following clearfelling in peatland catchments. *Water, Air and Soil Pollution*, 221:1-11). Connie O'Driscoll collected, analysed and synthesised data, and was the primary author of this article. Dr. Elvira de Eyto assisted with analysis and editing; and Mark O'Connor and Zaki ul Zaman Asam assisted with the sampling.

The second paper titled "Impact of vegetation and whole-tree–harvesting as potential mitigation methods for nutrient export at plot scale in a harvested upland peat catchment" has been submitted to the peer-reviewed, international journal *Forest Ecology and Management*. Connie O'Driscoll collected, analysed and synthesised data, and was the primary author of this article. Dr. Elvira de Eyto assisted with analysis and editing; and Mark O'Connor and Zaki ul Zaman Asam assisted with the plot construction, instrumentation and sampling.

The third paper titled "Buffer Zone Creation in an Upland Peat Forest" has been submitted to the peer-reviewed, international journal *Ecological Engineering*. Connie O'Driscoll collected, analysed and synthesised data, and was the primary author of this article. Dr. Elvira de Eyto assisted with analysis and editing; and Mark O'Connor and Zaki ul Zaman Asam assisted with the buffer zone construction, instrumentation and sampling. Kilian Kelly carried out the biodiversity study in the buffer zone area.

4.2 A potential solution to mitigate phosphorus release following clearfelling in peatland forest catchments

Connie O'Driscoll[1], Michael Rodgers[1], Mark O'Connor[1], Zaki-ul-Zaman Asam[1], Elvira de Eyto[2], Russell Poole[2] and Liwen Xiao[1*]

[1] Civil Engineering, National University of Ireland, Galway, Republic of Ireland

[2] Marine Institute, Newport, Co. Mayo, Republic of Ireland

* Corresponding author: Liwen Xiao; Email: liwen.xiao@nuigalway.ie; Tel: +353 91 494188

Abstract

Since the 1950s, large areas of upland peat have been afforested in northern European countries. Due to the poor P adsorption capacity, low hydraulic permeability in blanket peat soil and increased labile P sources, harvesting these blanket peat forests can significantly increase P concentrations in the receiving aquatic systems. This paper briefly reviews the current management practices on the control of P releases from forestry in Ireland and the UK, and proposes a possible novel practice – grass seeding clearfelled areas immediately after harvesting, which should reduce P release from upland blanket peat forest harvesting. The study was conducted in the Burrishoole Catchment in the west of Ireland. A field trial was carried out to identify the successful native grass species that could grow quickly on recently clearfelled blanket peat forest. The two successful grass species - *Holcus lanatus* and *Agrostis capillaris* – were sown in three blanket peat forest study plots with areas of 100 m^2, 360 m^2 and 660 m^2 immediately after harvesting. Areas without grass seeding were used as controls. One year later, the P content in the above ground vegetation biomass of the three study plots were 2.83 kg P ha^{-1}, 0.65 kg P ha^{-1} and 3.07 kg P ha^{-1}, respectively, which were significantly higher than the value of 0.02 kg P ha^{-1} observed in the control sites. The WEP content in the three study plots were 8.44 mg (kg dry soil)$^{-1}$, 9.83 mg (kg dry soil)$^{-1}$ and 6.04 mg (kg dry soil)$^{-1}$, respectively, which were lower than the average value of 25.72 mg (kg dry soil)$^{-1}$ in the control sites. The results indicate that grass seeding of the peatland immediately after harvesting can quickly immobilise

significant amounts of P and warrants additional research as a new BMP following harvesting in the blanket peatland forest to mitigate P release.

Introduction

Forest harvesting disrupts the P cycle of forest ecosystems and increases labile phosphorus (P) sources in the soil, which could result in an increase of P release. Phosphorus at concentrations of 30 μg L^{-1} could trigger eutrophication in freshwaters (Boesch et al., 2001). Eutrophication has been identified as the most important water quality problem in the UK and Ireland (EPA, 2004), particularly for the generally oligotrophic salmonid rivers and lakes, which are very sensitive to pollution. Therefore, P release after harvesting is of significant concern in upland blanket peat forest catchments, such as the Burrishoole catchment in the west of Ireland, which contains salmonids and has a great risk of P release due to the poor P adsorption capacity and low hydraulic permeability of the peat soil. Since the 1950s, large areas of upland peat have been afforested in northern European countries. Previous studies have documented the effects of peatland forest harvesting on P release. In Southern Finland, Nieminen (2003) found an increase in P release at three out of four peatland forest study sites after harvesting. In the west of Ireland, Cummins and Farrell (2003) investigated the biogeochemical impacts of clearfelling with regard to P on blanket peatland streams and noted that in three drains the MRP increased from 9, 13 and 93 before harvesting to 265, 3530, and 4164 μg L^{-1}, respectively, one year after harvesting. Recently, Rodgers et al. (Chapter 2, section 2.3) carried out a study in the Burrishoole catchment in the west of Ireland and found that the daily mean TRP concentration in a study stream increased from about 6 μg L^{-1} pre-harvesting to 429 μg L^{-1} one year after harvesting, even though best management practices were strictly implemented. Four years after clearfelling, the P concentrations returned to the pre-harvesting concentrations. In the first three years after harvesting, up to 5.15 kg ha^{-1} of TRP was released from the harvested catchment to the receiving water; in the second year alone, 2.3 kg ha^{-1} of TRP was released. These results indicated that the water quality of lakes, rivers and streams in the blanket peat forest catchments could be threatened by possible increases of P in runoff water arising from forest harvesting.

Current mitigation methods

Buffer zones, which can filter the runoff before it reaches the receiving water, are widely used by forestry practitioners in the management of freshwater aquatic systems. They can protect aquatic systems by controlling runoff: (1) mechanically, by increasing deposition through the slowing down of flow; (2) chemically, through reactions between incoming nutrients and soil matrices and residual elements; and (3) biologically, through plant and microbial nutrient processes. Buffer zones have been recognised as an efficient method to remove SS and attached P and could remove 14 % to 91.8 % of TP (Table 1). However, its effectiveness on DRP removal has been controversial. In their study, Vought et al. (1994) found that buffer strips were very efficient in DRP removal, with the removal efficiency of 95 %. In contrast, Uusi-Kämppä (2005) found that their naturally vegetated BZ became a P release source, responsible for 70 % of DRP release. Stutter et al. (2009) indicated that vegetated BZs increased soil P solubility and the potential amount of P release. In Ireland and the UK, many of the earlier afforested upland blanket peat catchments were established without any riparian buffer areas, with trees planted to the stream edge (Ryder et al., 2010). Ryder et al. (2010) carried out a study on the creation of riparian BZs in three blanket peat forests in the west of Ireland and concluded that it was a technically challenging felling operation. In their study, Rodgers et al. (Chapter 2, section 2.3) found that in the Burrishoole catchment, most of the P release after harvesting occurred in soluble form during storm events, raising concerns about the effectiveness of BZs in blanket peatland catchments.

In order to reduce nutrient sources, WTH is recommended (Nisbet et al., 1997). In the UK, WTH is usually achieved by removing the whole tree from the site in a single operation (Nisbet et al., 1997). In Ireland, during WTH the logging residues are bundled and removed from the site after the conventional harvesting of stem wood (*pers. comm*. Dr. Philip O'Dea, Coillte Teoranta). Needles and branches have much higher nutrient concentrations than stem wood and whole-tree harvesting may reduce nutrient sources by 2 to 3 times more than bole-only harvesting (Nisbet et al., 1997). Rodgers et al. (Chapter 2, section 2.3) found higher WEP content in the areas below BMs than the brash-free areas in the harvested upland peat forest catchment and indicated that WTH could be used as a means to decrease P release. Yanai (1998) reported negligible P loss to streams over three years from harvesting using the WTH method at the Hubbard Brook

Experimental forest in New Hampshire. However, WTH can remove most of the nutrients as well as base cations (Nisbet et al., 1997), which could have a negative impact on the next crop rotation, especially in blanket peat catchments. Walmsley et al. (2009) found that removal of forest residues can reduce second rotation productivity through nutrient shortage.

Table 1 Performance of buffer zones on P removal

Soil	Total P removal	Dissolved reactive P removal	Vegetation	Width (m)	References
Clay	40%	0	Grass	10	Uusi-Kämppä, 2005
Clay	40%	-70%	Natural vegetation	10	Uusi-Kämppä, 2005
Silty loam	14%		Grass	5	Syversen and Borch, 2005
Silty loam	26%		Grass	10	Syversen and Borch, 2005
Silty loam	40%		Grass	15	Syversen and Borch, 2005
	70%	75%	Grass	113	Bhattarai et al., 2009
Silt, loam and sand	31%		Grass	2	Abu-Zreig et al., 2003
Silt, loam and sand	89%		Grass	15	Abu-Zreig et al., 2003
	61%		Grass	4.6	Dillaha et al., 1989
	79%		Grass	9.1	Dillaha et al., 1989
	18%		Grass	4.6	Magette et al., 1989
	46%		Grass	9.1	Magette et al., 1989
		66%		8	Vought et al., 1994
		95%		16	Vought et al., 1994
Peatland	67%		Forest		Marttila and Kløve, 2010
Clay and sand		41%	Grass	4.1	Yates and Prasher, 2009
Silt loam	42.90%		Grass	8	Mankin et al., 2007
	67%			20	Mander et al., 1997
	81%			28	Mander et al., 1997
Silt loam	91.80%		Shrub	8	Mankin et al., 2007
Loamy sand		63%	Forest/grass	75	Lowrance et al., 1984

Phased felling is recommended in the UK (Forest Commission, 1988) and Ireland (Forest Service, 2000) to diminish the negative impact of harvesting on water quality. Harvesting appropriately sized coupes in a catchment at any one time can minimise the nutrient concentrations in the main rivers (Chapter 2, section 2.3). In their study, Cummins and Farrell (2003) found higher P concentrations in the smaller drains, which covered higher proportion of harvesting area. Rodgers et al. (Chapter 2, section 2.3) carried out a study on the impact of harvesting on the downstream receiving river. The study stream and the main river have the areas of about 25 and 200 ha, respectively. They found that although the P concentrations in the study stream were up to about 420 µg TRP L^{-1}, the average P concentrations in the receiving water of the main river were 7 ± 5 µg TRP L^{-1} at the USC, and 9 ± 8 µg TRP L^{-1} - about 30 m downstream at the DSC. In a storm event, when the TRP in the study stream increased from about 3 µg TRP L^{-1} to 292 µg TRP L^{-1}, the TRP concentrations at the DSC in the main river increased from about 5 µg TRPL^{1} to about 11 µg TRP L^{-1}, which was much lower than the critical value of 30 µg TRP L^{-1}. Phased felling is being used widely in Ireland. However, this management strategy does not reduce the total P load leaving the harvested catchment, which could be bound to the bed sediment of the receiving waters. If the P concentration in the river bed or lake sediment increases above the saturation point, it could be released and become available to phytoplankton (EPA, 2004).

A possible novel practice – grass seeding

The increase of P release is due to the disruption of the P cycle after harvesting, which reduces the catchment's P conservation capacity. The conservation of nutrients is dependent on a functional balance within the intra-system cycle of the ecosystem and critical to this balance is the uptake of water and nutrients by plants. Previous studies have indicated that vegetation can retain the available P *in situ* and reduce P release from forest activities. In Finland, Silvan et al. (2004) demonstrated that plants are effective in retaining P in peatlands. In China and Australia, vetiver grass in BZs and wetlands has shown a huge potential for removing P from wastewater and polluted water (Wagner et al., 2003). Loach (1968) found that *Molinia caerulea* could uptake 3.4 kg TP ha^{-1} in the wet-heath soils. Sheaffer et al. (2008) reported a P uptake of 30 kg ha^{-1} by *Phalaris arundinacea* in their wastewater treatment sites. However, recovery of blanket peat vegetation following forest harvesting usually takes several years. Connaghan (2007) found that *Juncus effusus* could develop in riparian areas within three years of clearfelling, whereas further away from the river where peat depth increased and soil fertility decreased vegetation took six to ten years to recover.

It appears that natural re-vegetation arising from the seed bank is likely to be too slow to significantly mitigate against the P from felling, which mainly occurs in the first three years after harvesting (Cummins and Farrell, 2003; Rodgers et al., Chapter 2, section 2.3). In order to minimise the release of nutrients to receiving waters after harvesting, a rational approach is to maximise the ground vegetative growth over the first year after harvesting. This can be achieved by seeding the clearfelled area with fast-growing suitable native vegetation. Sowing herbaceous species to reduce soil erosion has been used widely during the first year after forest fire (Ruby, 1989). However, to the best of our knowledge, no research has been carried out on the potential of sowing grass immediately after harvesting upland blanket peat catchments to mitigate nutrient release. In this study, seeding native grasses immediately after harvesting was examined as a potential as a new forestry BMP. It was hypothesised that by sowing the appropriate grass species in the blanket peat forest area immediately after harvesting, significant amounts of P will be quickly taken up and conserved *in situ*, which will result in reduced P release. To test this hypothesis, a trial experiment was first carried out to identify the successful germination grass species in the blanket peatland. The grass species were then sown in three harvested blanket peat forest plots. The biomass and P content of the above ground vegetation were tested one year after grass seeding. In order to compare P uptake by vegetation in seeded versus natural re-vegetated areas, vegetation surveys were also carried out in nine blanket peat forest sites which were harvested 1-5 years prior to the present study in the west of Ireland.

Material and methods

Site description

The study was carried out in County Mayo in the west of Ireland (Figure 1; Table 2). A total of nine sites were surveyed for natural re-vegetation in the blanket peat area after harvesting. All the sites have similar soil type and hydrological conditions. They are covered with blanket peat and overlie mainly quartzite and schist bedrock, and receive an average precipitation of over 2,000 mm per year. During the harvesting operation, boles were removed, and tree residues (i.e. needles, twigs and branches) were collected together to form the brash material mats and windrows. A second rotation of *Pinus contorta* was planted in all sites within 6 months after harvesting, except in the Glennamong and Teevaloughan. No fertiliser was applied in the replanting operation.

Trial and plot-scale experiment

Ten widespread native Irish grass species, which were considered to be suitable for the purpose of this study, were chosen for the trial experiment. They included: (1) *Agrostis capillaris*, (2) *Epilobium angustifolium*, (3) *Eriophorum vaginatum*, (4) *Festuca rubra*, (5) *Holcus lanatus*, (6) *Juncus effusus*, (7) *Lolium perenne*, (8) *Molinia caerulea*, (9) *Phalaris arudinacea* and (10) *Phragmites australis*. Grass seeds were purchased from Emorsgate Seeds, Norfolk, UK.

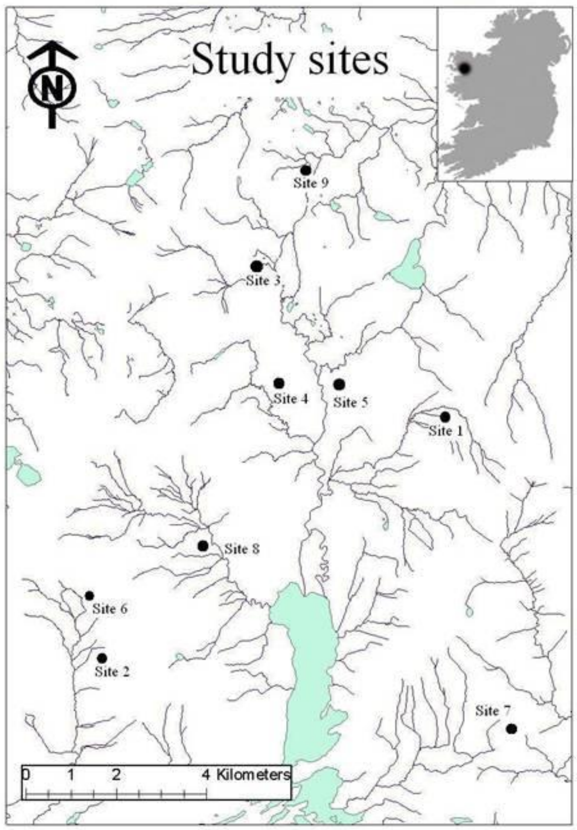


Figure 1 Locations of the study sites (Site 1: Srahrevagh; site 2: Glendahurk-2; site 3: Altahoney; site 4: Maumaratta; site 5: Goulaun; site 6: Glendahurk-1; site 7: Teevaloughan; site 8: Glennamong; site 9: Tawnynahulty).

Prior to the field trial test, a sample of seeds was tested for viability using a controlled laboratory germination test (Rao et al., 2006). For each species, 25 seeds were placed in a petri- dish on 42 mm-diameter Whatman filter paper, with 8 replicates. 3 ml of distilled water was added and the dishes were arranged in cultivation chambers with fluorescent tubes of white light and a light/

darkness timer, at 15–25°C. Dishes were sampled daily during three weeks. A seed was considered germinated when the radicle emerged. Distilled water was added whenever moisture loss was detected.

Table 2 Background information on the study sites

Site No.	Site name	Tree species before harvesting	Year of planting	Year of harvesting
1	Srahrevagh	Lodgepole pine	1971	2005
2	Glendahurk-1	Lodgepole pine	1971	2006
3	Altahoney	Lodgepole pine	1971	2006
4	Maumaratta	Lodgepole pine	1971	2007
5	Goulaun	Lodgepole pine	1971	2008
6	Glendahurk-2	Lodgepole pine	1971	2008
7	Teevaloughan	Lodgepole pine and Sitka spruce	1971	2009
8	Glennamong	Lodgepole pine	1971	2009
9	Tawnynahulty	Lodgepole pine	1971	2009

In the field trial test, a total of thirty three plots with an area of 900 cm^2 each were defined in the brash free area in Teevaloughan site (Site 7 in Figure 1 and Table 2). 300 seeds of each of the ten candidate species were scattered on three replicate plots. Three replicate control plots were also included. The plots were surveyed weekly for four months. Percent seedling emergence was calculated as the number of visible seedlings divided by the total number of seeds scattered on each plot.

In the Glennamong site (Site 8 in Figure 1 and Table 2), an area of about 1 ha was clearfelled in August 2009 and three plots of 100 m^2, 360 m^2 and 660 m^2 were identified for the grass seeding plot-scale study. Each plot received the same sowing treatment, which comprised of a 50:50 ratio of *Holcus lanatus* and *Agrostis capillaris*. The ground was undisturbed and the seed was distributed evenly by hand at an initial rate of 36 kg ha^{-1} on top of the old forest residue layer in October 2009. December 2009 and January 2010 were exceptionally cold months and a layer of snow, measuring 30 cm in depth, was recorded above the seeded area. To eliminate the risk of seed establishment failure, the plots were seeded again in February 2010 at the same rate of 36 kg ha^{-1}. The area which was not seeded was used as control.

Above ground vegetation biomass and P content measurement

To estimate the above-ground vegetation biomass in nine study sites, thirty 0.25 m x 0.25 m quadrats were randomly sampled (Moore and Chapman, 1986) in each site in August 2010. All vegetation lying within the quadrat was harvested to within 1 cm and dried at 80 °C in the laboratory on the day of collection for 48 hours. Samples were then weighed and the biomass was calculated by using Equation 1. Total phosphorus content of the vegetation was measured in accordance with Ryan et al. (2001). About 1 g of dry matter from each sample was weighed, ground and put into a furnace at a temperature of 550 ^{o}C overnight, then 5 ml of 2 N HCl was added to extract the P and subsequently diluted to 50 ml with deionised water. Phosphorus in the solution was analysed using a Konelab 20 Analyser (Konelab Ltd.).

$$B_p = \frac{Wt}{St} \times 10000 \qquad \text{Equation 1}$$

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

Soil WEP measurement

One hundred-millimeter-deep soil cores, consisting of the humic and upper peat layers, were collected using a 30-mm-diameter gouge auger in the Glennamong site. Four, 8, and 14 soil samples were taken from plot 1, 2, and 3, respectively. Soil samples were analysed for gravimetric water content and WEP. The core samples were placed in bags, hand mixed until visually homogenised, and subsamples of approximately 0.5 g (dry weight) were removed and extracted in 30 ml of deionised water, and measured for P using a Konelab 20 Analyser. The remaining core samples were dried to determine their gravimetric moisture contents (Macrae et al., 2005).

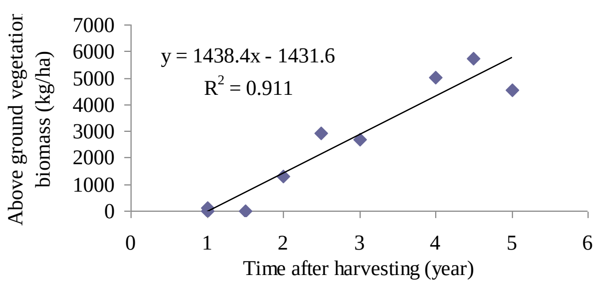
Data Analysis

In order to investigate the effects of grass seeding on total aboveground biomass production, grass phosphorus uptake, and soil water extractable phosphorus, data collected in the sown and control plots were compared by using t tests. All statistical analyses were conducted using the SPSS statistical package for windows (SPSS version 18, 2010).

Results

Biomass and P Content of Natural Re-Vegetation in Blanket Peat Forests after Harvesting

The biomass of the aboveground vegetation has a strong linear relationship with years after harvesting (Figure 2a). Vegetation appears to begin re-colonising about one and half years after harvesting. Five years later, the above-ground vegetation linearly increased to about 6,000 kg biomass per hectare. P content in the aboveground vegetation also linearly increased and reached 3.5 kg TP ha^{-1} 5 years after harvesting (Figure 2b).



a)

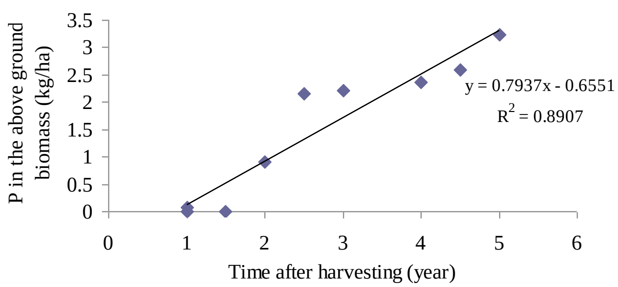


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Figure 2 Relationship between a) biomass and b) P content of the above ground vegetation and years after harvesting

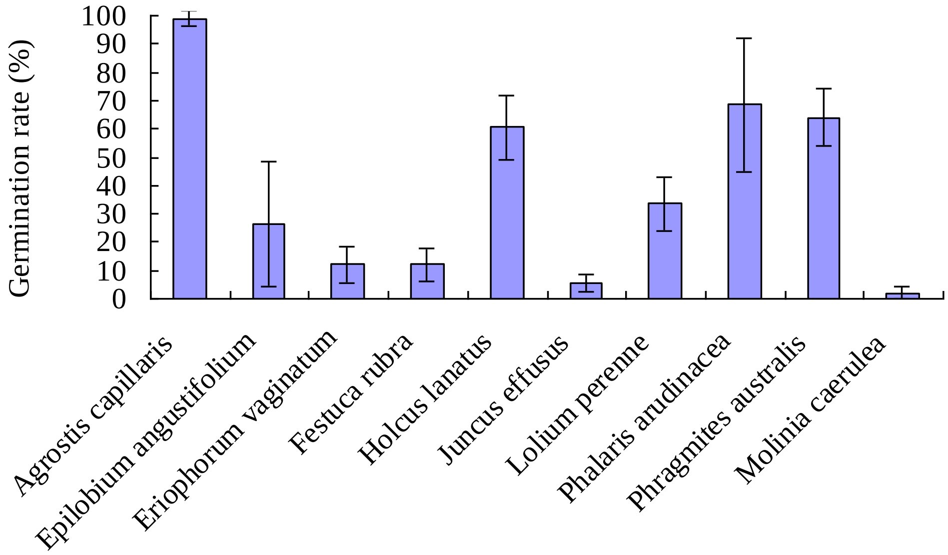
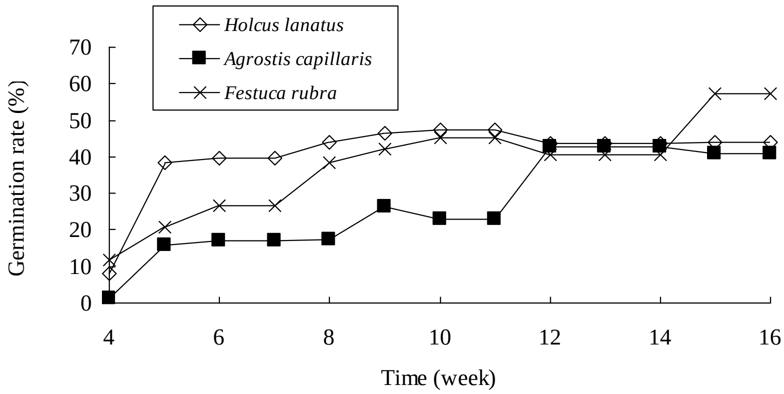


Figure 3 Successful germination rates of ten grass species examined in laboratory conditions (Error bars indicate ± 1standard deviation)

Successful Germination Grass Species

Most species germinated successfully within 3 weeks. *A. capillaris*, *P. arudinacea*, *P. australis*, and *H. lanatus* have the highest viable rates of 99 %, 68.5 %, 64 %, and 60.5 %, respectively (Figure 3). M. caerulea has the lowest rate of only 2 %. Low *M. caerulea* germination rates of 3 % and 9 % were also reported by other researchers (Grime et al., 1981; Grime et al., 1988; Brys et al., 2005). In their study, Grime et al. (1981) believed that the low germination percentage could be due to the low temperature. During the 16-week field trial study in Teevaloughan, no grass growth was observed in the control plots. In the study plots, 7 out of 10 grass species successfully germinated. At the end of the study, *H. lanatus*, *A. capillaris*, *F. rubra*, *P. australis*, *P. arudinacea*, *L. perenne*, and *E. angustifolium* had the germination rates of 44 %, 41 %, 57 %, 8 %, 11 %, 18%, and 3%, respectively (Figure 4). *H. lanatus*, *A. capillaris*, and *F. rubra* had the highest germination rates. However, *F. rubra* was observed to be discoloured toward the end of the study period, as was noted by O'Toole et al. (1964), which could be due to poor nutrients concentrations in the soil. Similar phenomena were also found in *P. australis* and *L. perenne*, which died back after week 7 and week 9, respectively. Only two species— *H. lanatus* and *A.*

capillaris—germinated successfully in the forested peatland habitat, and continued to grow and thrive up to 13 weeks after seeding, and were considered to be suitable for the purpose of this study



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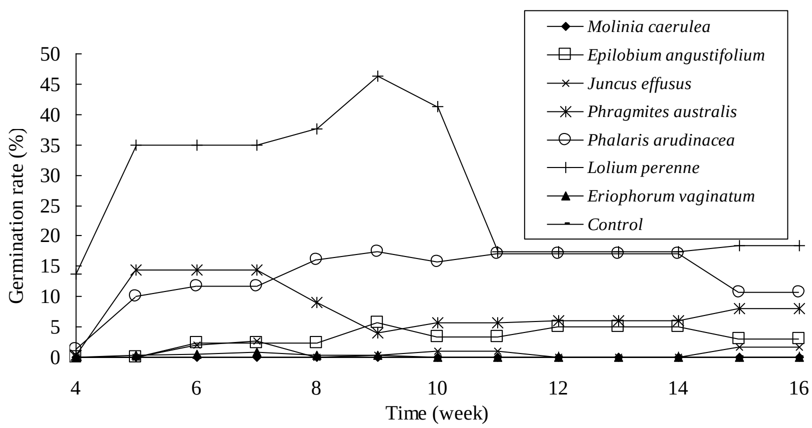
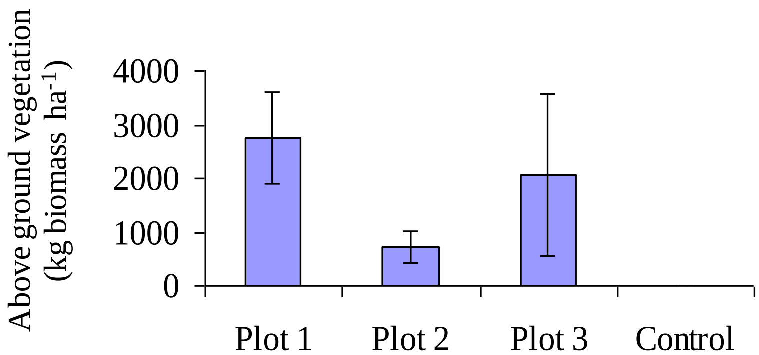


b)

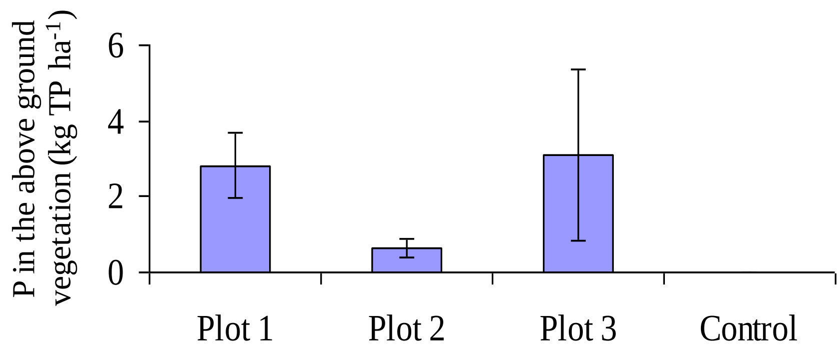
Figure 4 Germination rates of ten grass species planted in the trial experimental

Impact of Grass Seeding on the Biomass and P Content of Above-ground Vegetation

Seeding of *H. lanatus* L. and *A. capillaris* L. increased the above-ground vegetation biomass and P content 1 year after grass seeding (Figure 5). While there was very little vegetation growth in the control plots (22 kg biomass ha^{-1} with P content of 0.02 kg TP ha^{-1}), vegetation biomass of 2,753, 723, and 2,050 kg ha^{-1} were observed in the three study plots, giving the TP content of 2.83, 0.65, and 3.07 kg ha^{-1}, respectively (Figure 5). The above-ground biomass and P content in the sown plots were significantly higher than in the control plots ($p < 0.01$). The vegetation collected for testing was cut to 1 cm above-ground level, so these estimates could be higher when taking below ground biomass production into account (which has been estimated at 30 % of the total plant biomass, Scholes and Hall, 1996).



a)



b)

Figure 5 a, b Biomass and P content of above ground vegetation in the study plots and control in the Glennamong (Plot 1: 100 m^2, Plot 2: 360 m^2, Plot 3: 660 m^2; The bars indicate ± 1 standard deviation)

In the UK, Goodwin et al. (1998) found that *H. lanatus* produced biomass of 3,405 kg ha^{-1} with P concentrations of 1.64 mg TP (g biomass) $^{-1}$, giving the total P content of 5.58 kg P ha^{-1}. Figure 6 shows the WEP concentrations in the sown plots and the control plots. The WEP in the three study plots were 9, 12, and 6 mg P (kg dry soil) $^{-1}$, respectively, which was significantly lower than the value of 27 mg P (kg dry soil) $^{-1}$ in the control areas (Figure 6; t-test, $p < 0.01$).

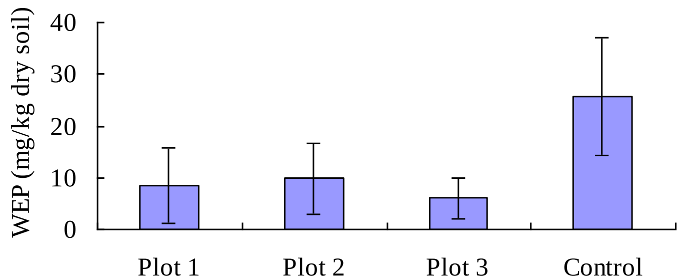


Figure 6 Water extractable phosphorus (WEP) in the study plots and control area in Glennamong (Plot 1: 100 m^2, Plot 2: 360 m^2, Plot 3: 660 m^2; The error bars indicate the standard deviation)

Discussion

In this study, *Calluna vulgaris*, *M. caerulea*, and *J. effusus* are the main species presenting at the natural re-vegetation sites. Similar findings were reported by Connaghan (2007). Recovery of blanket peat vegetation following forest harvesting usually takes several years (Connaghan, 2007). In this study, it took 5 years for the natural re-vegetation to have the above-ground biomass of 6,000 kg ha^{-1}. In a study by Allison and Ausden (2006), where plots were established on pine plantation heathland, which was recently clearfelled, it took 4 years for an increase in percentage frequency of *C. vulgaris* (a native heathland species) to appear. In the west of Ireland, Connaghan (2007) carried out grass surveys in eight blanket peat sites and found that bare soil could still account for 35 % one year after harvesting. The slow vegetation recovery of the harvested blanket peat forest sites could be due to (1) a significant reduction of the seed bank, (2) the burial of the seed bank by a thick layer of needle litter, and (3) the slow germination characteristics of the seeds typically found at these sites (Pywell et al., 2003). In a study to improve the peatland for the purpose of agriculture, O'Toole et al. (1964) highlighted the difficulties involved in attempting to identify successful species to seed peatland in Ireland. Grennan and Mulqueen (1964) sowed seed mixtures of Italian ryegrass (*Lolium multiflorum* L.), perennial ryegrass (*L. perenne* L.), cocksfoot (*Dactylis glomerata*), timothy (*Phleum pratense*), late flowering red clover (*Trifolium pratense*), and white clover (*Trifolium repens* L.) in the blanket peatland and found that when there were no P additions, all sown species died off after germination. In this study, only two grass species—*H. lanatus* and *A. capillaris*— were found to germinate successfully and continue to grow in the harvested blanket peat forest areas. After 10 years of study, O'Toole et al. (1964) found that *H. lanatus* was one of the most suitable species for seeding blanket peatland. In a study on the effects of sowing native herbaceous species on the post-fire recovery in a heathland, Fernández-Abascal et al. (2004) found that *F. rubra* appears before *A. capillaris* and also dies back earlier. They deemed *A. capillaris* a more suitable species than *F. rubra*. In a study investigating spatial and temporal patterns of growth and nutrient uptake of five coexisting grasses, Veresoglou and Fitter (1984) found that *H. lanatus* displayed a maximum nutrient uptake when soil moisture content and extractable P were high. In contrast, they found *A. capillaris* had a tendency to uptake peak P when the soil was drier. The use of these two herbaceous species in this study may complement one another through increasing uptake duration. Piirainen et al. (2007) found that as ground vegetation develops, P uptake and recycling can be expected to diminish leaching over time. In this study, the relatively low WEP in the study plots is likely to be a result of P uptake by the seeded grasses. *H. lanatus* and *A. capillaris* have been reported to have high P uptake capacity. Veresoglou and Fitter (1984) carried out a study on nutrient uptake in five coexisting grasses and found that *H. lanatus* and *A. capillaris* could uptake 16.9 and 2.7 mg TP (m^2 day) $^{-1}$, respectively. As WEP has a strong linear relationship with TP concentrations in the runoff (Schindler et al., 2009) and has been proved to be a useful indicator of soluble P concentrations in peat soil runoff water (Daly and Styles, 2005), it is expected that the reduction of WEP in the grass seeded plots could result in reduction of P runoff release.

Future Research

Future research on the potential of grass seeding as a new forestry BMP should measure stream chemistry to assess the success of the practice at protecting water quality. It is expected that the P measured in the grass would render a corresponding reduction in the P exported by the stream after harvesting. However, this has not been addressed by this study. Sowing grass immediately after harvesting may affect forest regeneration. The inter-specific interactions between seeded grasses and the replanted seedlings can be positive and negative, and require further studies (Goldberge, 1990; Maestre et al., 2004; Liu and Wang, 2008; Maestre et al., 2009). The seeded grasses store significant amount of P released from the peat and the logging residues. When the canopy of the next forest crop gradually closes over, the vegetation decays and releases the nutrients for uptake by the growing trees, which will facilitate forest regeneration. In fact, these nutrients slowly released from grass could be critical for the reforestation in peatlands, because of the poor nutrients of the soil and the low fertilisation rate limited by forest guidelines (Forest service, 2000). In contrast, the sowing grasses may compete for nutrients and lights with replanted seedlings in the first few years after seeding (Li et al., 2010). However, this negative impact can be diminished by choosing the right seeding rates and seeding distance from the seedlings. Future research could be carried out on an appropriate seeding rate, to ensure the nutrient release to the receiving water and the competition with the replanted seedlings, and so that the costs can be minimised.

Conclusion

The results of this study indicate that (1) *H. lanatus* and *A. capillaris* can be quickly established in blanket peat forest areas after harvesting and (2) sowing *H. lanatus* and *A. capillaris* immediately after harvesting has the potential to immobilise the P that would otherwise be available for leaching. One year after sowing, the P contents in the aboveground vegetation biomass could be up to 3.07 kg Pha^{-1}. Further research into the feasibility of grass seeding as a potential new BMP is clearly warranted. Sowing the right grass species at appropriate rates should diminish the deleterious effects of forest harvesting on surface water quality and facilitate the forest regeneration.

Acknowledgements

The authors gratefully acknowledge the funding from the Department of Agricultural, Fisheries and Food in Ireland, COFORD, Ireland EPA, Coillte and the Marine Institute. They also acknowledge the assistance of Dr. Philip O’Dea, Mary O’Brien, Mary Dillane and Liz Ryder.

Bibliography

See Chapter Six pp. 187 - 222.

4.3 Vegetation and whole-tree harvesting as potential mitigation methods for nutrient export control in upland peat catchment - plot scale study

Connie O'Driscoll[1], Michael Rodgers[1], Mark O'Connor[1], Zaki-ul-Zaman Asam[1], Elvira de Eyto[2], Russell Poole[2] and Liwen Xiao[1*]

[1] Civil Engineering, National University of Ireland, Galway, Republic of Ireland

[2] Marine Institute, Newport, Co. Mayo, Republic of Ireland

* Corresponding author: Liwen Xiao; Email: liwen.xiao@nuigalway.ie; Tel: +353 91 494188

Abstract

In north-western Europe nutrient release from forest harvesting is potentially a major environmental problem with respect to degradation of receiving water ecological status. With the implementation of the WFD, water managers are required to employ mitigation methods to prevent deterioration of water quality. In this study, experimental 140 m^2 field plots with three replicates each were monitored for two years (2010–2011) to evaluate the impact of five different management treatments (brash and grass, brash only, brash mat, whole-tree-harvesting, grass only) on nutrient export in a harvested catchment in the west of Ireland. These experiments were designed to comparatively assess the benefits of whole tree harvesting and the native grass seeding method on mitigating nutrient release following forest harvesting. The results indicated that the lowest runoff assoicated TRP (47.7 ± 3 µg L^{-1}) and NH_4-N (93 ± 4 µg L^{-1}) were observed for the grass only treatment plot. The grass only plot also had the highest runoff concentration of NO_3-N (197.7 ± 105 µg L^{-1}). This study presents significant advancement of our understanding of mitigation against nutrient release in harvested peatland forest catchments and offers potential resolution on the controversial issues to forest managers.

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). At maturity, forest catchments are reported to release very low baseline quantities of nutrients (Mattsson et al., 2003; Cummins and Farrell, 2003; Finér et al., 2004; Machava et al., 2007; Chapter 2, section 2.3). However, most of these forests are currently of harvestable age and concerns have been raised about the potential for pollution of surface water resources with fertiliser derivatives such as phosphates and nitrates. These areas contain the headwaters of oligotrophic rivers, many of which contain Red List species (e.g. salmonids and freshwater pearl mussel) which make them important biodiversity refuges. The WFD requires implementation of measures to maintain high status where it exists and achieve good ecological status by 2015 (European Union, 2000). Phosphorus and N are key sources of non-point surface water pollution causing enrichment of rivers and stimulating eutrophication (Reynolds, 1992). Catchment scale studies have shown that harvesting peatland forests increases nutrient export to receiving waters (Lebo and Herrmann, 1994; Paavilainen and Päivänen, 1995; Ahtiainen and Huttunen, 1999; Ensign and Mallin, 2001; Nisbet, 2001; Cummins and Farrell, 2003; Nieminen, 2004; Chapter 2, section 2.3). In comparable studies (Nieminen, 2003; Cummins and Farrell, 2003; and Chapter 2, section 2.3) similar P release patterns after a harvesting event were observed. An immediate peak after harvesting was observed with a declining tail followed by a maximum peak the following summer and a long declining tail. The cause of this trend is not investigated in these catchment scale studies due to the complicity of P release and possible multiple limiters of biological productivity (Cummins and Farrell, 2003). It is suggested that the lack of uptake of nutrients from the labile P by the removed standing forest is a cause for the immediate peak and decomposing logging residues is the cause for the second peak (Hyvönen et al., 2000; Ganjeguntea et al., 2004; Chapter 2, section 2.3).

Current economically feasible opportunities for reducing P losses in blanket peat harvested catchments include: WTH; BZs and constructed wetlands; phased felling; reduction in fertiliser application. Whole-tree-harvesting reduces nutrient export to receiving water following forest harvesting and 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 nutrient concentrations than stem wood, and WTH may reduce nutrient sources by 2–3 times more than bole-only harvesting (Nisbet et al. 1997). Higher nutrients are associated with areas below windrows/ brash material (Niemen, 2004; Chapter 2, section 2.3) in harvested upland peat forest catchment indicating that WTH could be used as a means to decrease the nutrient export. Yanai (1998) reported negligible P loss to streams over 3 years from harvesting using the WTH method at the Hubbard Brook Experimental forest in New Hampshire.

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. (Chapter 2, section 2.3) demonstrated that traditional buffers of 15–20 m may not be adequate to reduce P release as the majority occurred during storm events when the buffer zone would have had low residence time. Constructed wetlands are costly to establish and many of the earlier afforested upland blanket peat catchments in Ireland and the UK were established without any riparian buffer areas and trees planted to the stream edge (Ryder et al. 2010; Chapter 4, section 4.2). O'Driscoll et al. (Chapter 4, section 4.2) investigated a novel method whereby native grass species were seeded on site immediately after harvesting and before restocking with conifers and confirmed that the grass could immobilise the nutrient movement from the harvested catchment.

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). Catchments scale studies are very popular in observing human induced deterioration in water quality; however, they leave many unexplained answers and are difficult to replicate (van Es et al., 1998; Silva and Williams, 2001; Townsend et al., 2004). Plot-scale studies are typically used to elucidate the mechanisms behind nutrient loss from soil to runoff; to evaluate different management practices which are applied as treatments; and provide an experimental design that is statistically valid (van Es et al., 1998; Wainwright et al., 2000).

Despite the fact that the sensitivity of clearfelling upland peat catchments has risen to prominence in recent years in terms of economic and conservational viability, sustainable protection methods are inadequately researched and insufficiently proven. Therefore, the objectives of this study were to (1) investigate the nutrient release from BMs after peatland forest harvesting, (2) assess the performance of WTH on nutrient release control and (3) assess the novel practice – grass seeding immediately after harvesting on nutrients release control using statistically valid experimental field plots in a just-harvested peatland forest catchment. To the best of our knowledge, plot scale studies have not been carried out in peatland standing forests for the purpose of investigating forest harvesting impacts and mitigation methods.

Materials and methods

Site description and instrumentation

The study plots were nested within a 7.65 ha catchment drained by a small first order stream (Figure 1). The study catchment is a sub-catchment of the Burrishoole catchment located in Co. Mayo in the west of Ireland (Figure 1). About 23 % of the Burrishoole catchment is covered by coniferous forests, planted in the 1970s and currently or soon to be harvested. The average peat depth is approximately 2 m and lies on a bedrock of quartzite, schist and basic volcanic rock. The peat typically has a gravimetric water content of greater than 80 %. The Burrishoole has a mean annual rainfall and air temperature of about 2000 mm and 11 °C respectively. The first order stream is equipped with a monitoring station at a stable channel section, downstream of the study area (Figure 1).

An H-flume, a water level recorder, a Datasonde (measuring water temperature, pH, conductivity and DO) and a data logger were installed at the monitoring station, along with a tipping bucket rain gauge. Readings were recorded every 5 minutes.

Harvesting procedures

The study area was felled between September 2010 and January 2011. A Valmet 941 Harvester was used to clearfell the site and following best management practices the crown of the tree and associated tree residues (i.e. needles, twigs and branches) were collected together to form BMs which were used for machine travel to improve the soil bearing capacity against the heavy harvesting machinery, and reduce erosion. During harvesting, the boles were stacked beside the windrow for collection. A Valmet 840 Forwarder delivered the boles to truck collection points beside the forest road. At the end of the harvesting operations the BMs were collected together to form windrows which aim to make the planting of new trees easier. Brash mats and windrows lie parallel to the study stream and furrows on the harvested site, which were at right angles to the contours. The BMs and windrows were about 4 m wide. The distance between two adjacent BMs was about 12 m. The surface water flows along the furrows and was collected by collector drains before joining the study stream.

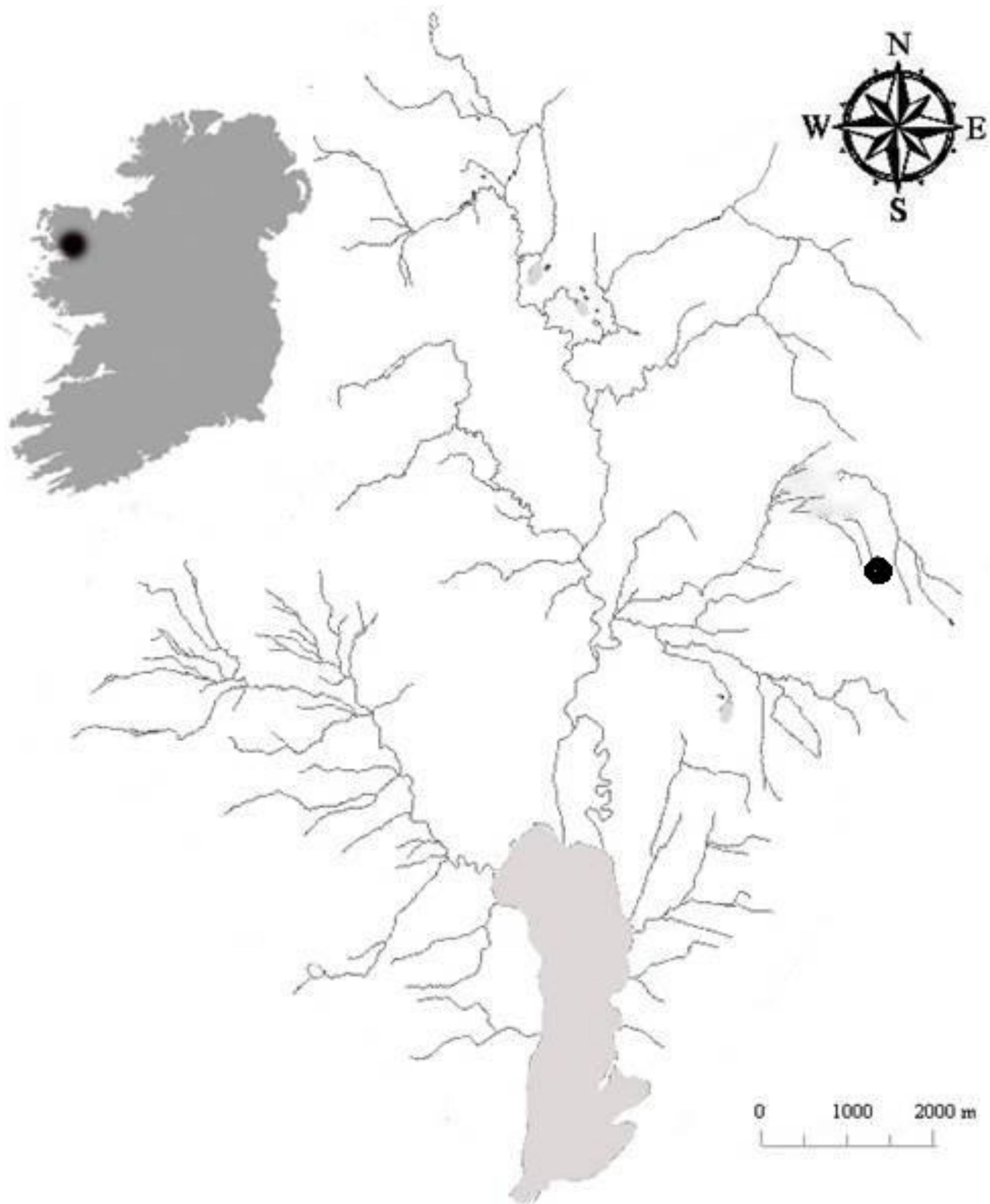


Figure 1 Geographical location of the study plots.

Plot design and instrumentation

Fifteen experimental plots (7 m wide by 20 m long) were constructed in the study area standing forest in January 2010 (Figure 2). To prevent lateral inflow of water the peat plots were isolated from the surrounding peat by inserting corrugated plastic into the furrows (30 cm deep). To prevent inflow of water from upslope, trenches were dug across the top of all fifteen plots (30 cm deep). The end of each plot strip was isolated with corrugated plastic and connected into the bespoke designed outflow surface monitoring system using white PVC piping. Outgoing surface flow rates from the plot strips were measured using manufactured tipping buckets. Water samples were collected in purposely modified ISCO automated samplers. Baseline pre-felling data was collected for one year from the fifteen plots and in January 2011 the 7.62 ha catchment and the 15 experimental plots were clearfelled. The five treatments were established in February 2011 (Table 1).

Table 1 the practices examined in the 5 plots

Plot	Treatment	Objectives
1	Brash and grasses seeding	To investigate the practice of leaving brash on site and seeding the site with native grass species seeding on nutrient release control
2	Brash only	To investigate the nutrient export from the practice of leaving the brash on the site
3	Brash mat	To investigate the impact of the machine travelling on the brash on the nutrient export
4	Whole treat harvesting	To explore the whole tree harvesting method (where all brash material is removed from the site) on nutrient release control
5	Grass only	To study the practice of whole tree harvesting plus grass seeding on nutrients release control

The brash used in plots was applied in accordance with forestry BMPs. The application was as follows: 9 m length of brash mat (equivalent to 53 trees) was collected by the forwarder and placed into the top half of the plot. The brash was composed of the crown of the tree and the branches apart from the main stem. The grass was seeded at a rate of 36 Kg ha^{-1} as described in O'Driscoll et al. (Chapter 4, section 4.2).

Water sampling and analysis

From March 2010 to February 2012, water samples were taken during flood events and base flow conditions over 24 hour periods at each of the outflow surface monitoring systems. Rainfall water samples were also collected by placing an open and clean plastic container near the plots during storm events for P analysis. All water samples were frozen at -20° C in accordance with the standard methods (APHA, 1998) until water quality analyses were conducted. The following analyses were carried out on the water samples: PO_4-P, NO_3-N and NH_4-N using a Konelab 20 Analyser (Konelab Ltd., Finland).

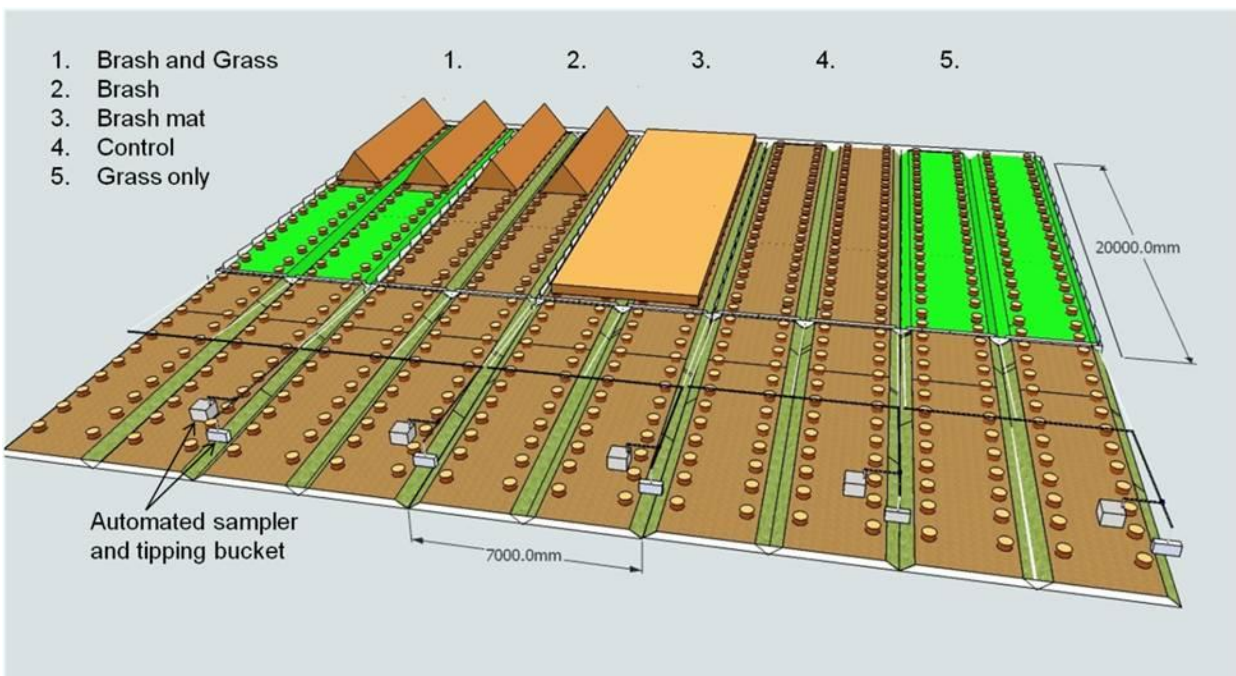


Figure 2 Design of the experimental plots 1 – 5 Brash and grass; brash; brash mat; control and grass only.

Soil water extractable (WEP) and total (TP) phosphorus content measurement

30 cm deep soil cores consisting of the humic (0-10 cm) and upper peat layers (10-20 cm and 20-30 cm) were collected using a 30-mm-diameter gouge auger. Four soil samples were taken from each plot (60 in total), two from between the furrows and two from within the furrows in May 2010 and again in June 2011. Soil samples were analysed for gravimetric water content and WEP and TP. The core samples were divided into three depths: 0-10 cm, 10-20 cm and 20-30 cm, placed in bags and mixed by hand until visually homogenised. 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 minutes. The supernatant was then filtered (0.45 µm) and measured for P using a Konelab 20 Analyser. Second subsamples of approximately 5 g (wet weight) were removed and dried to determine the gravimetric water content (Macrae et al., 2005). The dried subsamples were then put into a furnace at a temperature of 550 °C for 24 hours, then 5 ml of 2 N HCl was added to extract the P and subsequently diluted to 50 ml with deionised water. Phosphorus in the solution was analysed using a Konelab 20 Analyser (Konelab Ltd.).

Aboveground vegetation biomass and P content measurement

To estimate the aboveground vegetation biomass in the plots, four 0.25-m×0.25-m quadrats were randomly sampled in each seeded plot in August 2011 (Moore and Chapman, 1986). All vegetation lying within the quadrat was harvested to within 1 cm and dried at 80°C in the laboratory on the day of collection for 48 hours. Samples were then weighed, and the biomass was calculated by using Equation 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). One g of dry matter from each sample was weighed, ground, and put into a furnace at a temperature of 550°C overnight, then 5 ml of 2 N HCl was added to extract the P and subsequently diluted to 50 ml with deionised water. P in the solution was analysed using a Konelab 20 Analyser (Konelab Ltd.).

$$B_p = \frac{Wt}{St} \times 10000 \qquad \text{Equation 1}$$

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

Results

General nutrient concentration trends in the study plots before and after the establishment of the five treatments

Measured TRP concentrations in the five treatments were low before the harvesting with average values of 14.1 ± 3 µg L^{-1}, 21.1 ± 16 µg L^{-1}, 19.2 ± 10 µg L^{-1}, 25.7 ± 5 µg L^{-1} and 15.9 ± 6 µg L^{-1} in Plots 1, 2, 3, 4 and 5, respectively (Figure 3). Four months after the harvesting operations began, daily discharge weighted mean P concentrations in the plots started to 6 µg L^{-1}, 149.4 ± 116 µg L^{-1}, 378.2 ± 178 µg L^{-1}, 55.5 ± 3 µg L^{-1} and 38.8 ± 22 µg L^{-1}, respectively.

Measured NO_3-N concentrations in the five plots were low before the harvesting with average values of 21.1 ± 15 µg L^{-1}, 37.7 ± 27 µg L^{-1}, 9.9 ± 14 µg L^{-1}, 10.3 ± 14 µg L^{-1} and 33.0 ± 20 µg L^{-1} in Plots 1, 2, 3, 4 and 5, respectively, which were close to the values observed in the rainfall. After harvesting the values increased to 118.9 ± 72 µg L^{-1}, 144.4 ± 88 µg L^{-1}, 117.3 ± 121 µg L^{-1}, 143.5 ± 93 µg L^{-1} and 167 ± 75 µg L^{-1}, respectively.

Measured NH_4-N concentrations in the five plots were low before the harvesting with average values of 87.3 ± 25 µg L^{-1}, 94.6 ± 38 µg L^{-1}, 93.4 ± 43 µg L^{-1}, 82.3 ± 44 µg L^{-1} and 90.4 ± 45 µg L^{-1} in Plots 1, 2, 3, 4 and 5, respectively, which were close to the values observed in the rainfall. After harvesting, the values were 110.7 ± 76 µg L^{-1}, 188.3 ± 102 µg L^{-1}, 136.6 ± 69 µg L^{-1}, 113.0 ± 80 µg L^{-1} and 147.0 ± 103 µg L^{-1}, respectively.

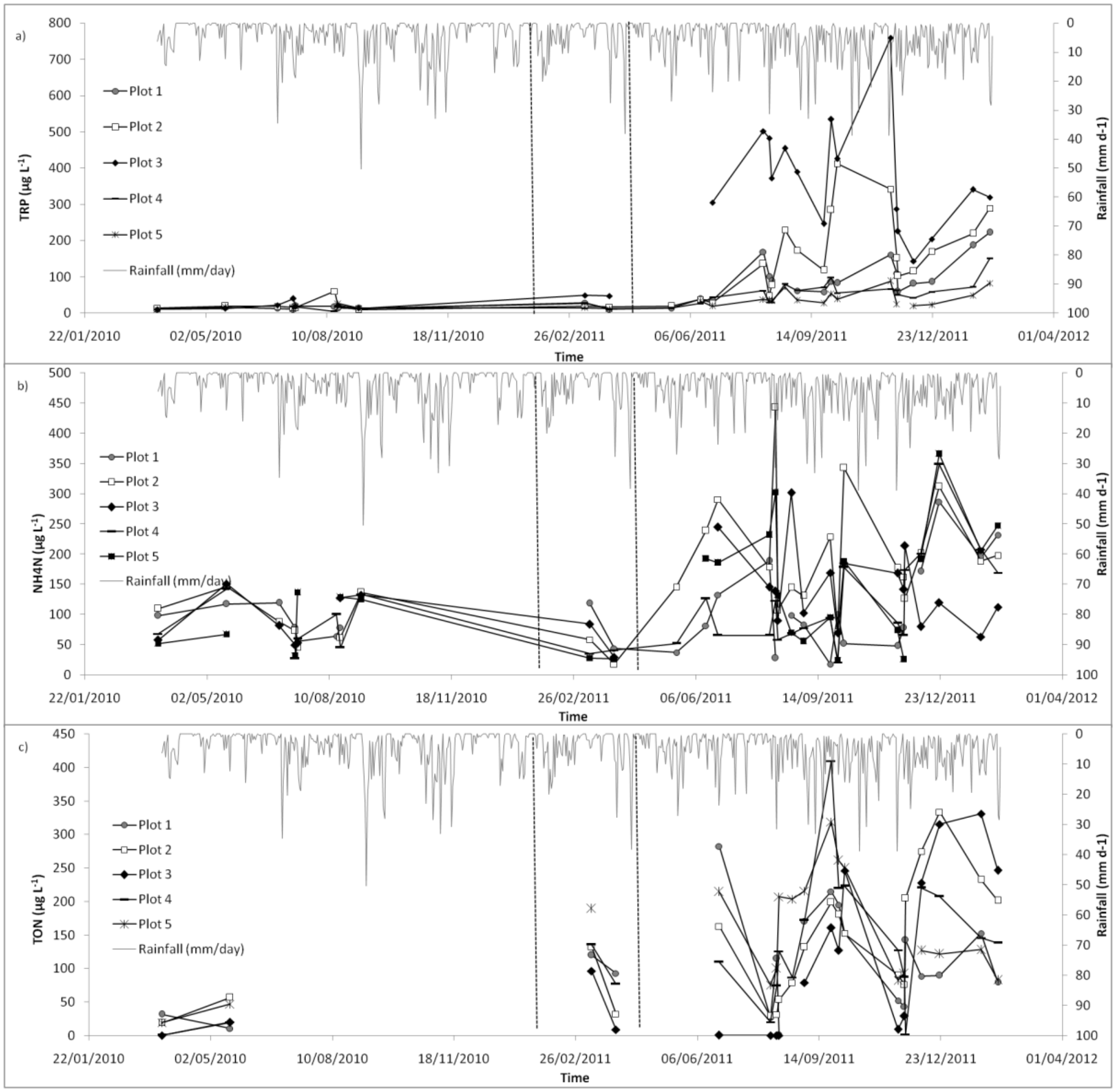


Figure 3. The daily discharge-weighted mean TRP, NH_4-N and TON concentrations of five treatments (3 replicates each) before and after the harvesting and treatment establishment. Dash lines indicate the harvesting period.

Comparison of treatments during storm events

The BM plot exported the highest concentration of P consistently in six measured storm events with an average value of 342.3 ± 22 µg L^{-1} (Figure 4a). The windrow/ brash plot was the next highest with an average value of 214.6 ± 87 µg L^{-1} (Figure 4a). The third highest for P release was the brash and grass plot, followed by the control and lastly the grass plot with average values across six storm events of141.3 ± 58 µg L^{-1},75.3± 38 µg L^{-1} and47.7 ± 31 µg L^{-1} (Figure 4a). The BM plot exported the highest concentrations of NH_4-N followed by the windrow/ brash plot and the control 238.5 ± 76 µg L^{-1}, 230.1 ± 65 µg L^{-1} and 143.9 ± 63 µg L^{-1}, respectively (Figure 4b). The lowest releasing plots were those which were seeded, the grass only and the brash and grass plots with average values across the six storm events of 93.4 ± 40 µg L^{-1} and 89.3 ± 34 µg L^{-1}, respectively (Figure 4b). The plot releasing the highest concentrations of NO_3-N was the grass only plot across six consecutive storms with an average value of 197.7 ± 105 µg L^{-1} (Figure 4c). The control plot was next followed by the brash and grass plot, the brash/ windrow plot and lastly the BM plot with average values of 190.1 ± 90 µg L^{-1}, 163.4 ± 98 µg L^{-1}, 177.9 ± 86 µg L^{-1} and 68.9 ± 44 µg L^{-1}, respectively (Figure 4c).

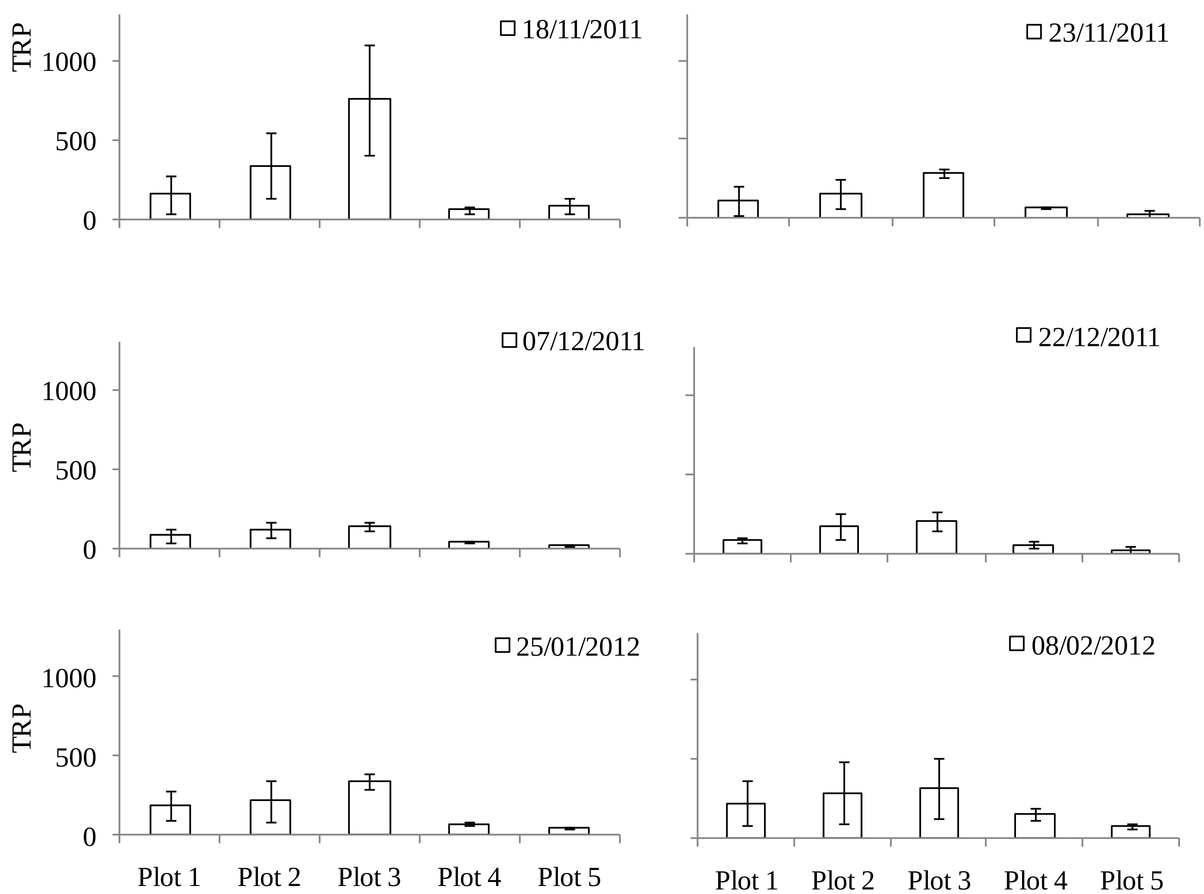


Figure 4a the instantaneous TRP concentrations across the five treatments during 6 storm events. The standard deviations of the replicates of three plots per treatment are shown.

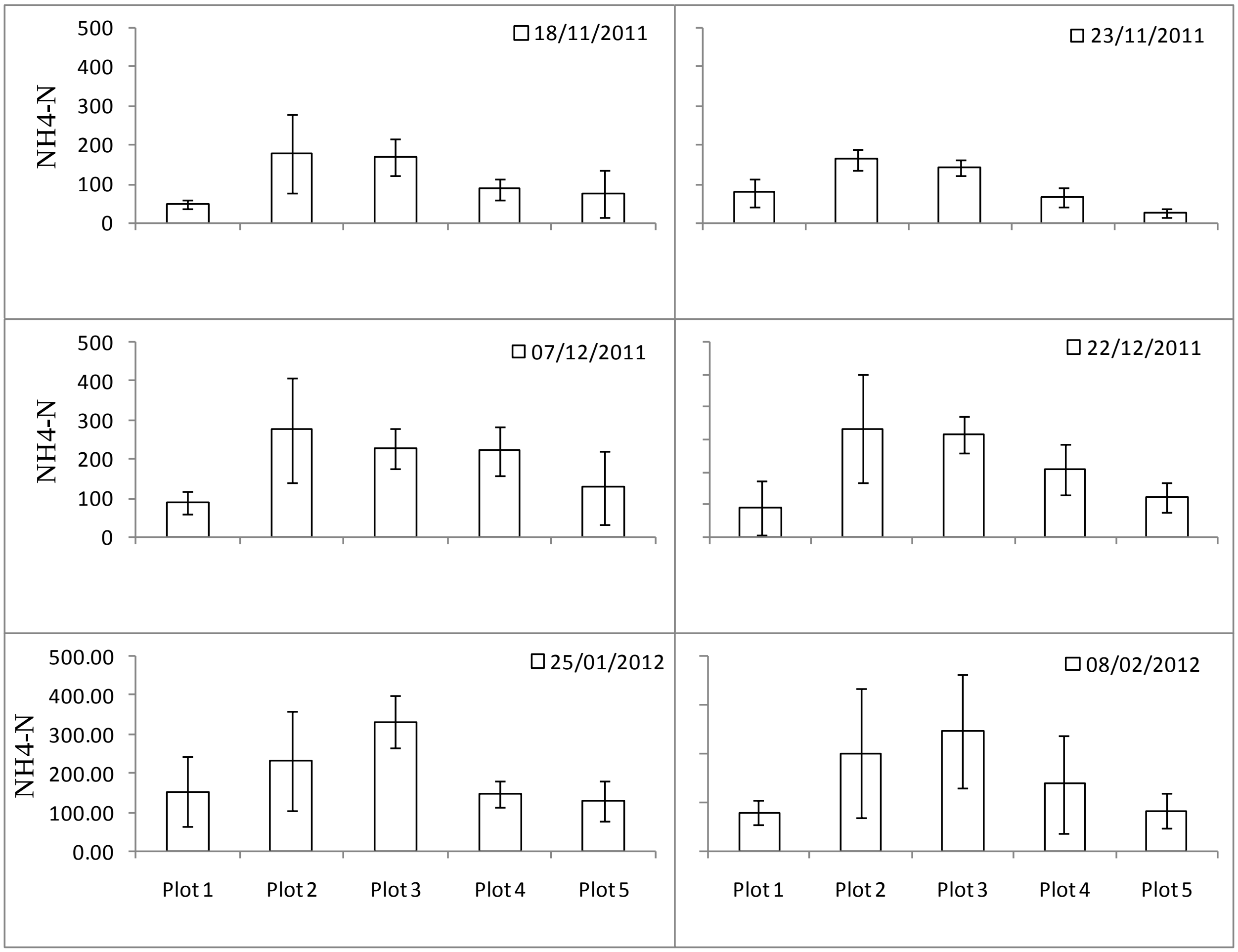


Figure 4b the instantaneous NH4-N concentrations across the five treatments during 6 storm events. The standard deviations of the replicates of three plots per treatment are shown.

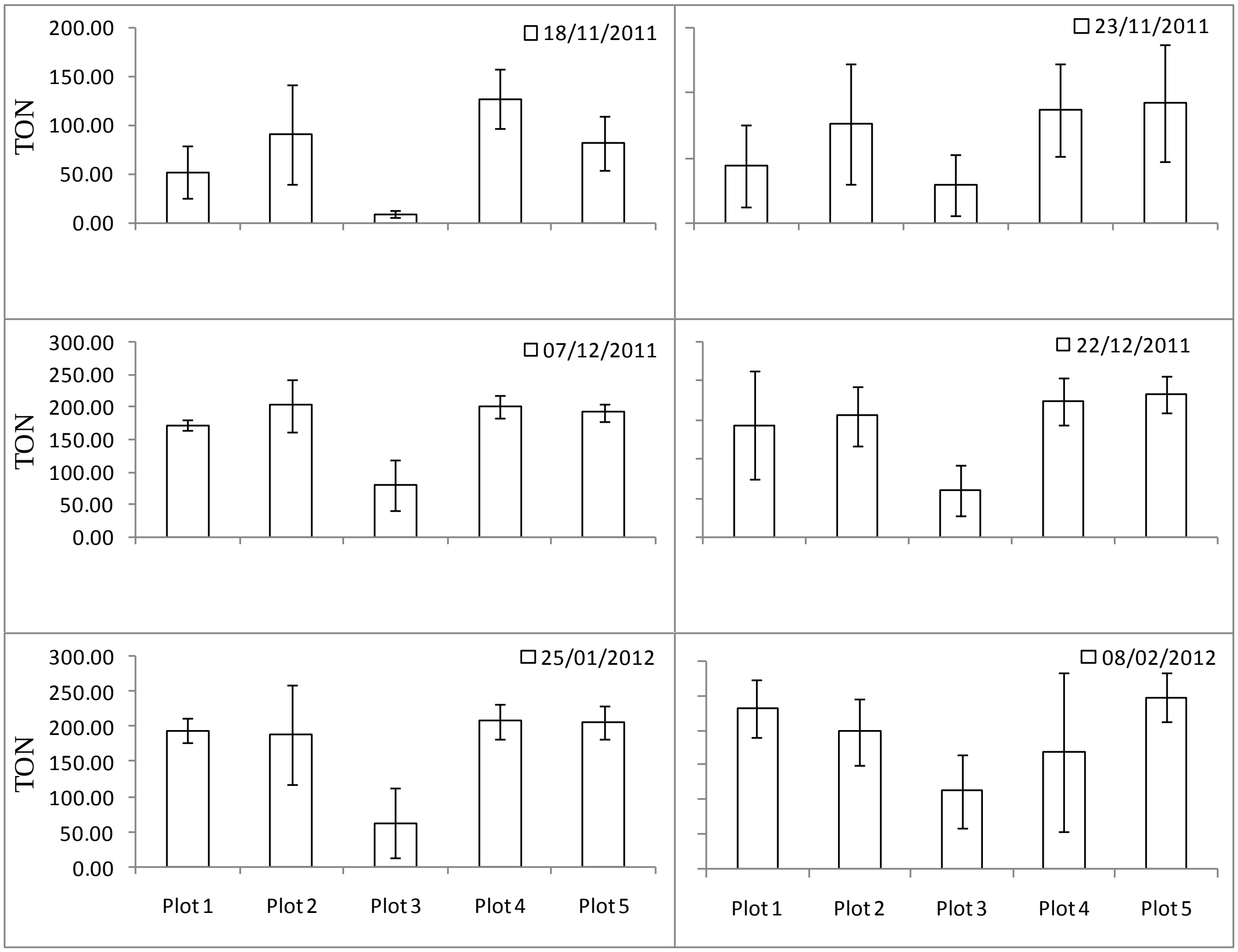


Figure 4c the instantaneous TON concentrations across the five treatments during 6 storm events. The standard deviations of the replicates of three plots per treatment are shown.

Water extractable and total P concentrations of the soil in the study plots before and after harvesting

The BM, brash/ windrow and grass only plots indicate an increase in WEP after clearfelling with differences of 1.67, 0.32 and 1.10 µg (kg dry soil) $^{-1}$, respectively (Figure 5a). The brash and grass and control plots have less WEP concentrations in the soil after harvesting with decreases of 0.56 and 0.52 µg (kg dry soil)$^{-1}$, respectively (Figure 5a). The TP concentrations in the soil increase across all plots from before to after clearfelling however the greatest increase can be seen in the plots with grass, 0.12 and 0.11 mg P g^{-1} dry soil in the brash and grass and grass only plots, respectively (Figure 5b). In general there is an increase of WEP in-the-furrows compared to between-the-furrows and a decrease of TP in-the-furrows compared to between-the-furrows (Figure 6a and 6b). A general increase in WEP can be seen down the 0 cm to 30 cm profile across all treatments except for the BM where WEP concentrations are the same at all depths (Figure 7a). In general, a decrease can be seen in TP down the 0 cm to 25 cm profile across all treatments accept for the grass only plot where a significant increase can be seen in TP concentrations at 25 cm depth (Figure 7b).

Biomass production and P concentration of the vegetation in the study plots after harvesting

Seeding of *Holcus lanatus* and *Agrostis capillaris* increased the above-ground vegetation biomass and P content 6 months after grass seeding (Figure 8). The brash and grass plot contained 100 kg ha^{-1} more aboveground biomass than the grass only plot highlighting the uptake of P from the plot (Figure 8). The grass in the brash and grass and grass only plotup took 9.99 ± 4 and 5.60 ± 2 kg TP ha^{-1}, respectively (Figure 8). No vegetation was present in the plots that weren’t seeded.

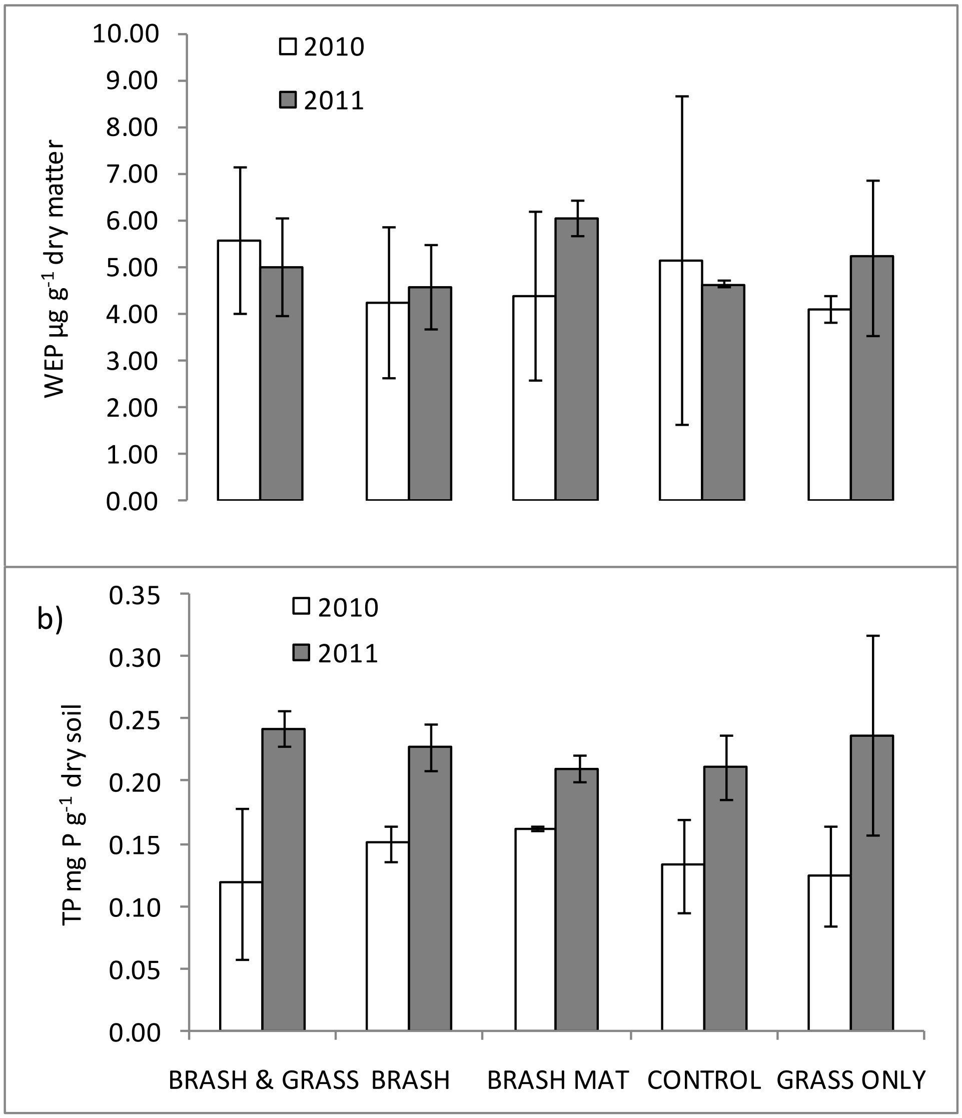


Figure 5 (a) WEP and (b) TP concentrations in the soil before and after harvesting

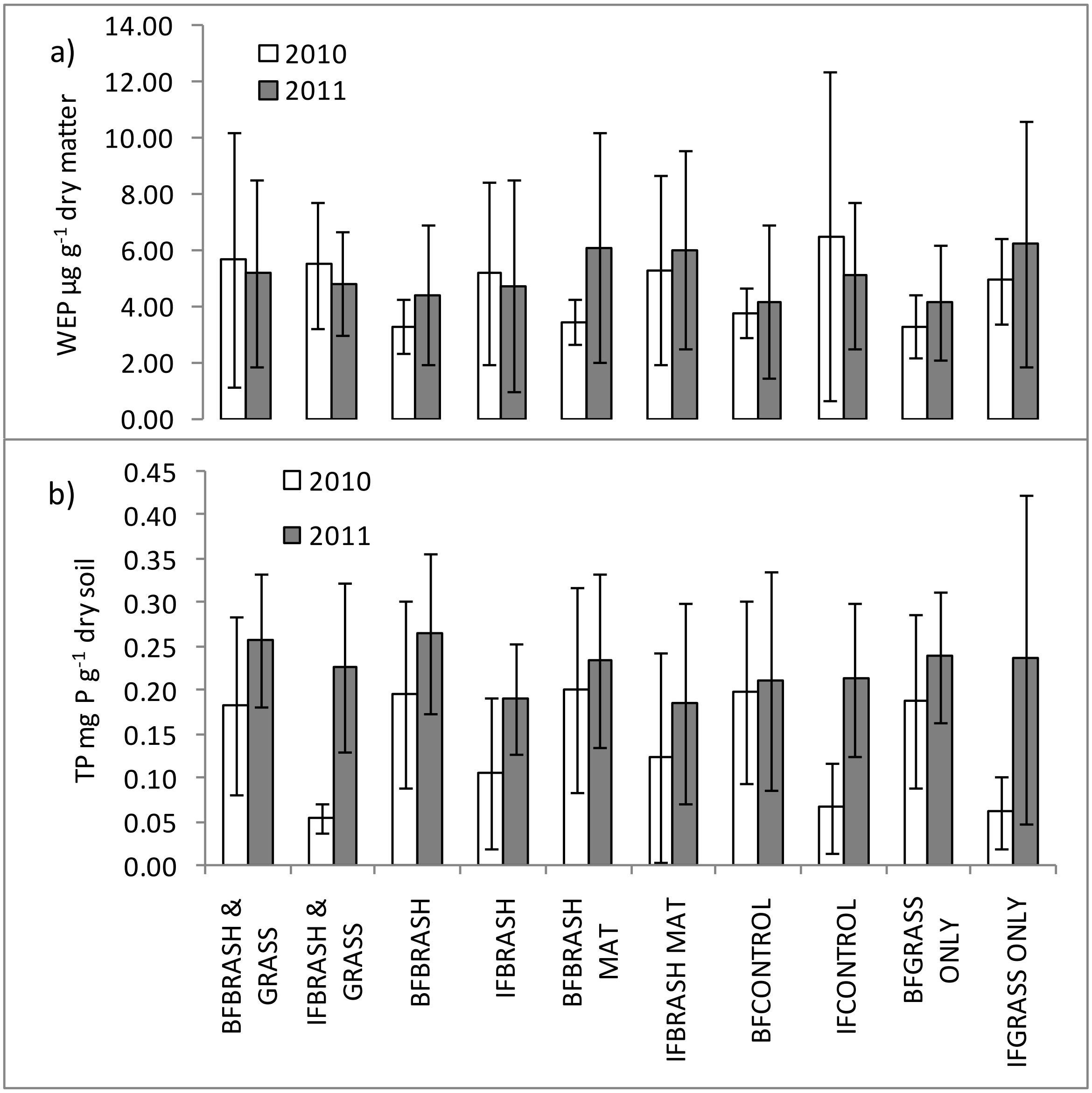


Figure 6 (a) WEP and (b) TP concentrations between-the-furrows and in-the-furrows across all treatments before and after harvesting and treatment establishment.

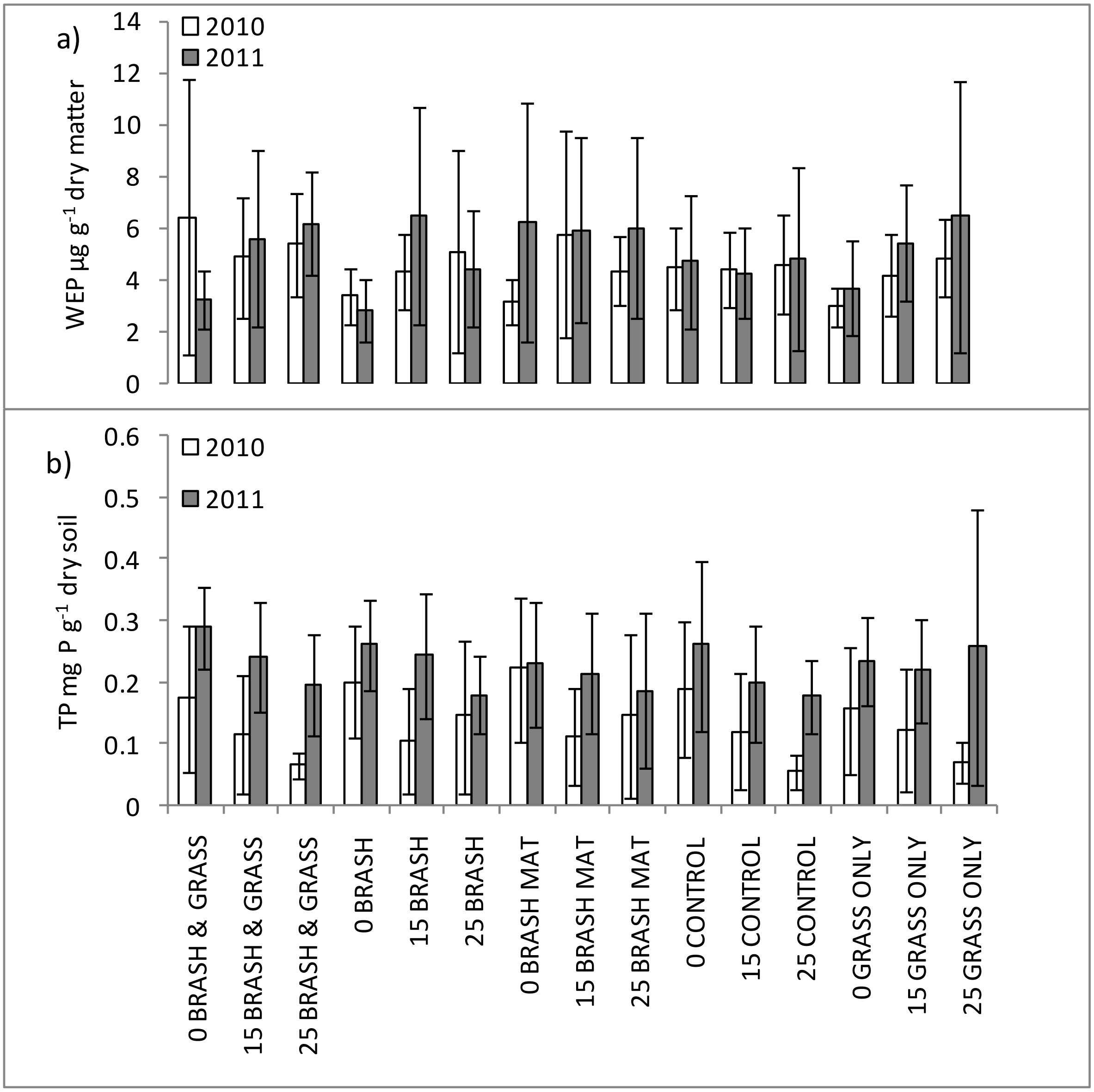


Figure 7 (a) WEP and (b) TP concentrations before and after harvesting at depths of 0-10, 10-20 and 20-30 cm across all treatments

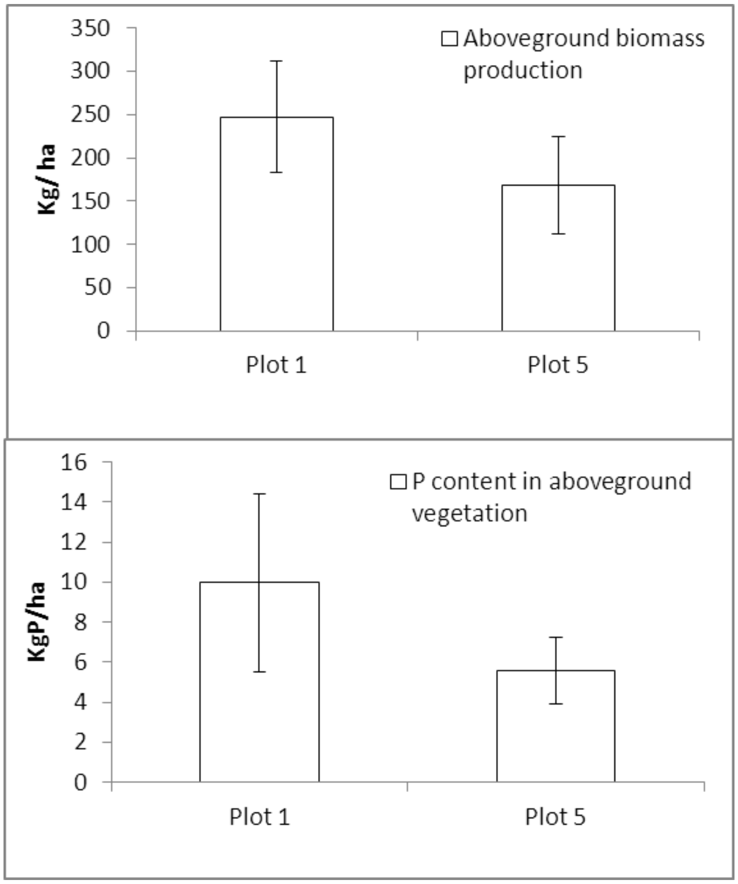


Figure 8 Biomass and P content of above ground vegetation in the study plots brash and grass and grass only. (The error bars indicate ± 1 standard deviation).

Interpretation and discussion

The highest concentrations of P release were observed from brash mat plot. Organic soils exhibit very little strength when they are not frozen and are virtually unable to support heavy equipment (Yong and Townsend, 1981). Equipment operation on organic soils is limited to periods when the soils are frozen in more northern countries where this is an option; however, in the morewestern European countries, weather conditions are fairer and felling can take place all year round. Soil compaction occurs during times of harvesting and extraction machinery traffic (Nugent et al., 2003). The consequences of excessive soil compaction are a decrease in the 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 a current approach to minimising soil disturbance McDonald and Seixas, 1997; Tiernan et al., 2002).

Harvesting residues have been reported to contain approximately half the amount of P and double the normal N application rate of fertiliser prescriptions (Taylor 1991; Titus and Malcolm, 1991; Stevens et al., 1995; Olsson et al., 1996; Hyvönen et al., 2000). With the soil compaction following harvesting and extraction, water may collect beneath the BMs, increasing favourable conditions for more rapid nutrient decomposition. The second highest release of P was observed coming from brash/ windrow plot. Rodgers et al. (Chapter 2, section 2.3) showed WEP concentrations from a harvested area were significantly higher (136 %, 152.3 %, 235 % and 188.9 %, in 2006, 2007, 2008 and 2009 respectively) in the soil under the brash/ windrow than in the brash/ windrow free soils. Similarly, significantly higher NO_3-N and NH_4-N concentrations were found in soil water under piles of cutting residues than respective residue-free areas in the first year after harvesting (Rosén and Lundmark-Thelin, 1987; Emmett et al., 1991; Staaf and Olsson, 1994).

The increase in nutrient concentrations in soil water arises from the decomposition of the logging residues, organic horizon and organic matter in the surface soil and the diminished nutrient uptake by the once sanding forest (Palviainen et al., 2004). Using litter bags to assess the timing of nutrient release from different harvesting logging residues (i.e. needles, twigs and branches) and a comparison of harvesting intensities (i.e. removing all logging residues, leaving all logging residues on site and leaving needles and removing twigs and branches), Hyvönen et al., (2000) established that fresh needles are the most important source of nutrient release immediately after clearfelling and that branch material releases nutrients much slower over the subsequent decade. The third highest P export was observed from the grass and brash plot. Contrary to these findings Staaf and Olsson (1995) reported no significant effect on ground vegetation biomass or amount of biomass between whole tree harvesting and brash left on site. Stevens and Hornung (1990) found a negative correlation between brash cover and re-establishment of ground vegetation. Emmett et al. (1991) observed *Agrostis capillaris* L. to establish quickly in the absence of brash and exhibit greater nitrate loss reduction. The lowest P release was observed coming from the grass only plot. 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 peat catchments this re-growth process can take up to four years (Connaghan, 2007; Chapter 4, section 4.2).

A faster reduction time in nutrient export from the forested catchment may occur by seeding immediately after clearfelling. Grass seeding is applied internationally for a scale of purposes: (1) to mitigate the ecological risks associated with exposed bare soil following intense wildfire and to encourage ecosystem recovery; (2) re-vegetating road corridors to mitigate negative effects of road development; and (3) revegetation and reclamation of abandoned mines. Its use on upland harvested blanket peat for the uptake of nutrients from the decomposition of the organic horizon and organic matter in the surface soil has been illustrated (Chapter 4, section 4.2). The grass seeding method has shown that native species such as *Holcus lanatus* L. and *Agrostis capillaris* L. can be established quickly in recently harvested blanket peat forest areas and can immediately begin to immobilise the P that would otherwise be available for leaching. This highlights that in the year following harvesting the majority of P exported from a catchment comes from the BM. The next highest P exporting treatment was the brash plot where the brash has been left on site. The third highest P exporting treatment was the brash and grass. This underlines that if brash is to be left on site the P export can be reduced by implementing the native grass seeding method (NGSM). The WTH method was illustrated to be effective with regard to the P export. The treatment with the lowest P export was the grass only plot. This combination of whole WTH and the NGSM can be implemented by forestry practitioners on highly sensitive sites such as in freshwater pearl mussel catchments.

With regard to ammonium, the highest export was observed in the brash and BM plots. There was no obvious difference between the grass only, the control and the brash and control plot. This illustrates that the presence of brash increases the NH_4-N export and that the grass is very efficient in up taking NH_4-N. There was a reduced leaching of nitrogen by the BM plot. This has been referred to previously in the literature (Palviainen et al., 2004; Hyvönen et al., 2000; Lundmark–Thelin and Johansson, 1997). Palviainen et al. (2004) found no net release of N within the first three years following harvesting and that soil solution concentrations increased during the same period. The WEP is highest in the brash mat plot following clearfelling. TP increases across all treatments in 2011 highlighting the capacity of the soil itself to retain P after clearfelling. In general there is an increase of WEP in-the-furrows compared to between-the-furrows and a decrease of TP in-the-furrows compared to between-the-furrows.

Conclusion

This study found that in the first year after harvesting, the average P concentrations in the five plots were 378.2 ± 178 $\mu g\ L^{-1}$, 149.4 ± 116 $\mu g\ L^{-1}$, 88.6 ± 59 $\mu g\ L^{-1}$, 55.5 ± 32 $\mu g\ L^{-1}$ and 38.75 ± 22 $\mu g\ L^{-1}$ at the brash mat, brash, brash and grass, control and grass only plot, respectively, indicating that (1) soil disturbance triggered P release, (2) brash was significant P release sources after harvesting and (3) grass seeding practice has great potential in mitigation of P release. The grass in the brash and grass only plot contained 100 kg ha^{-1} more aboveground biomass than the grass only plot. The grass in the brash and grass and grass only plot up took 9.99 ± 4 and 5.60 ± 2 kg TP ha^{-1}, respectively. This study also indicated that whole-tree harvesting and grass seeding have little impact on NO_3-N and NH_4-N release, however, the released concentrations of NO_3-N and NH_4-N are relatively low. A full scale study on grass seeding and whole-tree harvesting practices is needed.

Acknowledgments

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Bibliography

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4.4 Creation and functioning of a buffer zone in an upland peat forest

Connie O'Driscoll[1], Mark O'Connor[1], Zaki-ul-Zaman, Elvira de Eyto[2], Kilian Kelly[3], Michael Rodgers[1] and Liwen Xiao[1]*

[1] Civil Engineering, National University of Ireland, Galway, Republic of Ireland

[2] Marine Institute, Newport, Co. Mayo, Republic of Ireland

[3] Institute of Technology Tralee, Co. Kerry, Republic of Ireland

* Corresponding author: Liwen Xiao; Email: liwen.xiao@nuigalway.ie; Tel: +353 91 494188

Abstract

Large areas of upland blanket peat were afforested in the UK and Ireland before the importance of riparian BZs was realised. These forests are now reaching harvestable age, so in order to reduce the possible negative impact of harvesting activities on receiving water bodies, the creation of buffer zones along receiving water courses prior to the clear-felling of the main plantation has been proposed. In this study, a small buffer zone with the effective area of about 0.1 ha was established and seeded with native grass species and the runoff from the upstream forest with area of about 10 hectare (ha) was spread to the buffer zone. One year later, the upstream forest was harvested. The result indicated that the buffer zone removed 45.3 %, 33.7 % and 17.6 % of the SS, TON and PO_4-P, respectively, in the first year of harvesting. In addition to protecting aquatic systems, the buffer zone also provided a habitat for flora and fauna that would not normally be associated with conifer plantations.

Key words: peatland forest, buffer zone, phosphorus, nitrogen, SS

Introduction

Excess phosphorus (P) and nitrogen (N) input into water bodies could trigger eutrophication, which further deteriorates water quality and changes the aquatic flora and fauna (Vikman et al., 2010). Similarly, high suspended sediment (SS), especially organic solid, could consume oxygen resources and cause damage to the aquatic system (Paavilainen and Päivänen, 1995). It has been reported that organic material deposited in gravels had a great deleterious effect on salmon spawning grounds (Greig et al., 2007). Peatland forest harvesting activities increase labile nutrients sources and create some mechanical disturbance on the ground surface, which could lead to release of P, N and SS to river systems (Cummins and Farrell, 2003; Nieminen, 2003; Uusivuori et al., 2008; Rodgers et al., 2010; Rodgers et al., 2011). Peatland forests can contain the headwaters of many important water resources which can contain salmonids and freshwater mussels (O'Driscoll et al., 2011). The European Union (EU) Water Framework Directive (WFD) requires the EU Member States to achieve 'good ecological status' for all water bodies by 2015 (European Union, 2000), protecting the ecological status of aquatic ecosystem from peatland forest harvesting has become one of the priorities for forestry and river basin managements in European.

Directing the runoff from the upland areas over and through a buffer zone area, before it reaches the receiving water bodies, can reduce nutrient and solid concentrations; a method which is widely used by water quality managers in the protection of freshwater aquatic systems (Correll, 2005; Väänänen et al., 2008; Hoffmann et al., 2009; Vikman et al., 2010). Buffer zones can slow down flow which will increase particle deposition and enhance reactions between incoming nutrients; soil matrices and plant and microbial nutrient processes. Whilst BZs have been recognised as an efficient method to remove SS and attached nutrients, their effectiveness on dissolved nutrients has been controversial (Hoffmann et al., 2009). For example, Vought et al. (1994) reported that buffer strips removed 95 % of the incoming dissolved reactive phosphorus (DRP), but Uusi-Kämppä (2005) found that naturally vegetated BZs increased DRP release by 70 %. These controversial findings could be due to the fact that the performance of the buffer zones on nutrients and SS removal is affected by several factors such as hydraulic retention time, flow paths, soil types, vegetation composition, management history and inlet nutrients concentrations and loads (Väänänen et al., 2008; Hoffmann et al., 2009).

In Ireland and the UK, many of the earlier afforested upland blanket peat catchments were established without any riparian buffer areas, with trees planted to the stream edge (Ryder et al. 2011). Establishing BZs before harvesting by removing the peatland forest beside the water bodies has been recommended as a BMP to protect receiving water quality in Ireland and the UK

(Forest Commission, 1988; Forest Service, 2000). Giving the controversial findings in the literature and that (1) most of the P and N released after harvesting occurred in soluble form during storm events (Rodgers et al., 2010); (2) peat has low hydraulic conductivity and P adsorption capacity; and (3) the spatey nature of upland forest peat catchments in Ireland and the UK, questions have been raised about the potential efficiency of BZs for these upland blanket peat sites (Rodgers et al., 2010; O'Driscoll et al., 2011). To the best of our knowledge no study has been reported on the creation and functioning of a full scale BZ in upland blanket peat catchments.

The objectives of this study are to (1) establish a peatland buffer zone by harvesting a forest area beside a river and seeding it with native grass species and (2) assess its efficiencies on mitigation of nutrients and SS release due to forest harvesting. The buffer zone was harvested and seeded with two native grass species 18 months prior to the main upstream forest being harvested. Following vegetation establishment, runoff was spread through the BZ area by blocking the stream. Flows and water quality in the inflow and outflow of the buffer zone were monitored intensively before and after the main forest was harvested. As most of the nutrients and SS released after harvesting was occurs during storm events (Rodgers et al., 2010; Rodgers et al., 2011), this study mainly focused on storm events. This study will contribute important scientific qualitative and quantitative data on the creation and functioning of BZs in upland blanket peat forested catchments.

In addition to protecting aquatic systems, BZs may provide a habitat for flora and fauna that would not normally be associated with conifer plantations. This study provided a fortuitous opportunity to assessing the biodiversity of established BZs. The diversity of four taxonomic groups - plants, birds, small mammals and ground-dwelling invertebrates – in the grass seeded BZ, a natural vegetated BZ, a mature plantation forest and an area of natural peatland were examined and compared.

Materials and Methods

Sites description and buffer zone construction

The study was carried out in the Glennamong sub-catchment (Figure 1). A forest catchment (10 ha) which was to be felled during the study period was used as the study site. A 1-ha area downstream of this study site was used for the establishment of the BZ. The average depth of the peat in the BZ was about 0.5 m. The peat profile also contained sand deposits of varying thickness. The probable source of these layers was that they were formed of the material that was displaced from the ditches during the initial drainage in the early 1980s. The BZ was constructed by initially clearfelling the standing forest in August 2009. Brash was removed from the site and drainage channels were defined around the 1 ha area. Two months after clearfelling, three plots were identified within the BZ that could potentially have stream water from the study site diverted over and through them. The areas of these plots were as follows: 0.01 ha (plot 1), 0.036 ha (plot 2), and 0.066 ha (plot 3), accounting for 1.12 % of the area of study site. Each plot received the same sowing treatment, which comprised a fifty: fifty ratio of *Holcus lanatus* L. and *Agrostis capillaris* L. The ground was undisturbed, and the seed was distributed evenly by hand at an initial rate of 36 kg ha^{-1} on top of the old forest residue layer in October 2009. December 2009 and January 2010 were exceptionally cold months, and a layer of snow measuring 30 cm in depth was recorded on the ground above the seeded area. To eliminate the risk of seed establishment failure, the plots were seeded again in February 2010 at the same rate of 36 kg ha^{-1}. The area that was not seeded was used as control. The stream was diverted through the buffer zone in September 2010. Areas of preferential flow were identified and blocked using either sandbags or corrugated recycled plastic. The study site was clearfelled in February 2011.

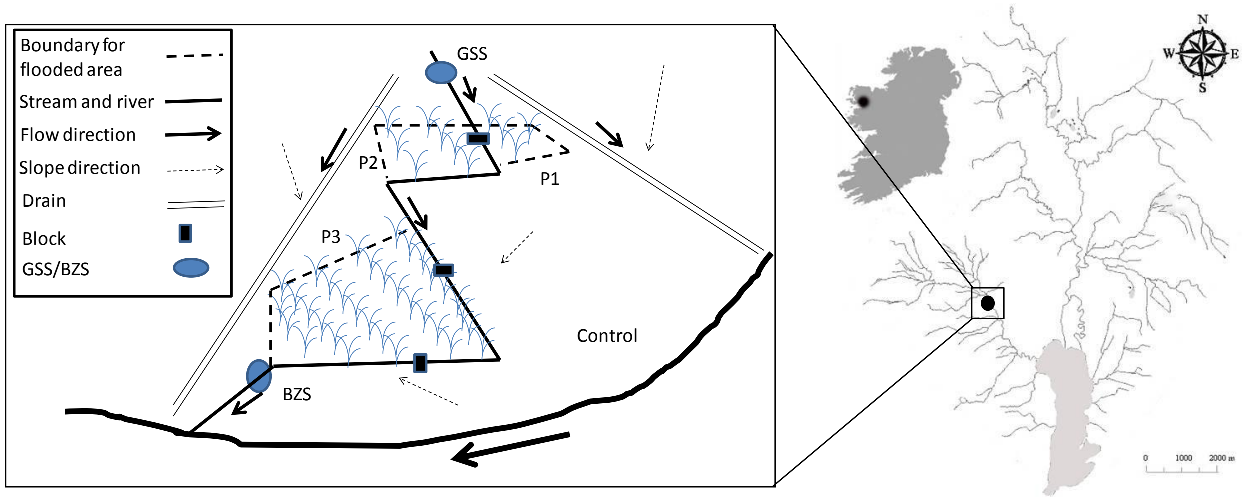

Figure 1 Location of the Burrishoole catchment and the buffer zone catchment boundaries are presented in detail.

Instrumentation setup and water sampling analysis

In January 2010, the stream was equipped with two flow and water quality monitoring stations at stable channel sections, one downstream of the study site and at the inflow to the buffer zone (GSS) and the second further downstream at the outflow of the buffer zone (BZS). An H-flume, a water level recorder, a Datasonde (measuring temperature, pH, conductivity and DO) and a data logger were installed at each station, along with a tipping bucket rain gauge. Readings were recorded every 5 minutes. Water samples were taken for nutrient and sediment analysis at the stations during storm events, on weekly basis over the study period using an ISCO automated water sampler. All water samples were frozen at -20° C in accordance with the standard methods (APHA, 1998) until water quality analyses were conducted. The following analyses were carried out on the water samples: total reactive phosphorus (TRP), total oxidised nitrogen (TON) and ammonium (NH_4-N) using a Konelab 20 Analyser (Konelab Ltd., Finland).

Soil water extractable (WEP) and total (TP) content measurement

30 cm deep soil cores consisting of the humic (0 – 10 cm), upper peat/ sand (10 – 20 cm) and lower peat (20 – 30 cm) layers were collected using a 3-cm-diameter gouge auger in the buffer zone area. 4, 8, and 14 soil samples were taken from plot 1, 2, and 3, respectively, between the

furrows and in the furrows, in April 2009, 2010 and 2011. Soil samples were analysed for gravimetric water content, WEP and TP. The core samples were placed in bags, mixed by hand until visually homogenised, and 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 measured for P using a Konelab 20 Analyser. Second subsamples of approximately 5 g (wet weight) were removed and dried to determine their gravimetric moisture contents (Macrae et al. 2005). The dried subsamples were then put into a furnace at a temperature of 550 °C for 24 hours. 5 ml of 2 N HCl was added to extract the TP and they were subsequently diluted to 50 ml with deionised water. Phosphorus in the solution was analysed using a Konelab 20 Analyser (Konelab Ltd.) to determine the TP content.

Aboveground vegetation biomass and P content measurement

To estimate the aboveground vegetation biomass in the buffer zone, thirty-two 25 cm× 25cm quadrats were randomly sampled (8 in plot 1; 11 in plot 2; 9 in plot 3 and 4 in the control area) in August 2010, and again in August 2011. All vegetation lying within the quadrat was harvested to within 1 cm and dried at 80 °C in the laboratory on the day of collection for 48 hours. Samples were then weighed, for above ground biomass calculation. Dried samples were milled to pass a 0.2 cm sieve and TP content of the vegetation was measured in accordance with Ryan et al. (2001). About 1 g (dry matter) of milled sample was weighed, and put into a furnace at a temperature of 550 °C overnight, then 5 ml of 2 N HCl was added to extract the TP and subsequently diluted to 50 ml with deionised water. Phosphorus in the solution was analysed using a Konelab 20 Analyser (Konelab Ltd.).

Data analysis

Higher concentrations of nutrients and SS in stream water have been observed during episodic storm events carried in high flows (Rodgers et al., 2010; Rodgers et al., 2011). In addition, the performance of BZ on nutrient and SS retention is affected by the hydrological conditions. Therefore, this study mainly focused on storm events and water samples were taken mainly during storm events. A storm event was defined as a block of rainfall that was preceded and followed by at least 12 hours of no rainfall (Hotta et al., 2007). As the flow is variable during a storm event, it is difficult to determine the hydraulic retention time at a certain flow rate by using tracers. As this BZ had a slope of 2-5 degrees and had a relatively small volume of stored water which could be exchanged with the runoff, it was assumed that the faster the runoff passing through the BZ the lower the hydraulic retention time. The time that the runoff needed to pass through the BZ is estimated was the time difference of the peak flow passing through GSS and BZS in storm events (Figure 2).

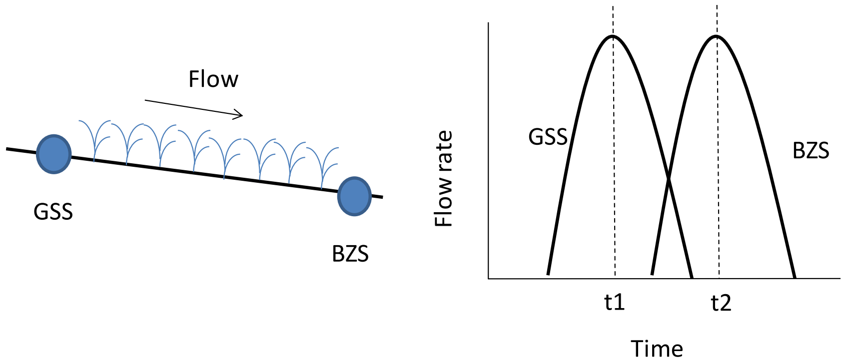

Figure 2. In a storm event the peak flow passed GSS at t1 and BZS at t2. The time difference (t2-t1) is considered as the residence time

In this study, the SS and nutrient loads in the first year after harvesting were estimated by using the methods in Rodgers et al. (2010, 2011). The SS and nutrients loads during each sampled storm event were also estimated by using the following equation:

$$L = \frac{\sum C_i * Q_i}{\sum Q_i} * Q_T$$

where L is the load, C_i and Q_i are the instantaneous concentration and flow at time i and Q_T was the total flow during the storm event.

To further investigate the relationship between the retention efficiency and the flow rate and input concentrations, the peak flow instantaneous retention efficiency was calculated for each sampled storm event by using the following equation:

$$\% \, retention = \left(1 - \frac{Q_{t2}C_{t2}}{Q_{t1}C_{t1}}\right) * 100$$

where Q_{t1} and Q_{t2} were the peak flows rate at the GSS and the BZS during a storm event, respectively; C_{t1} and C_{t2} were the concentrations in the peak flows at the GSS and the BZS during a storm event

Buffer zone biodiversity monitoring

The BZ and two control sites were selected in the Glennamong sub-catchment. Control site (1) was a 1 ha area of mature lodgepole pine (*Pinus contorta*). Control site (2) was an area comprised of a mosaic of wet heath, blanket bog and dry heath. 16 plots were randomly selected within each site. A small mammal survey was carried out using standard Capture-Mark-Recapture methods with Longworth small mammal traps (Barnett & Dutton, 1995). Terrestrial invertebrates were sampled using pitfall traps, adopted from Environmental Change Network (ECN) protocols (Skyes and Lane, 1996). Bird data was collected from the sites over the course of two visits using point counts in April and May, in accordance with methods described by Iremonger et al. (2006). Relative abundance of plants was recorded with randomly placed 1 m^2 quadrats using the DAFOR scale. Simpson's index of diversity was calculated for the invertebrate data. Variables were inspected for conformity to the assumptions of parametric tests. One-way ANOVA and post hoc tests (Bonferroni and LSD) were carried out using SPSS Statistics 18 (SPSS version 18, 2010).

Results and discussion

Runoff passing through the buffer zone before and after water spreading

One year after grass seeding, the vegetation in the BZ was well established. The stream that passed through the BZ was blocked in three locations and the runoff from the study site was directed through the BZ. There was an exponential relationship between the times that the storm runoff passed through the BZ and its peak flow rate in both before and after water spreading (Figure 3). Based on the equation, the storms which had a peak flow rate of higher than 5 L s^{-1} could pass through the BZ within 30 minutes (Figure 3). Diverting the stream water through the BZ significantly increased the lag time ($p < 0.01$) (Figure 3), which was due to the creation of sheet flows. Creation of sheet flows has been considered as a primary condition for buffer zone to achieve efficiency SS and nutrients removal (Väänänen et al., 2008). The buffer zone didn't reduce the storm peak flow rates (Figure 4). Monthly runoff passing through BZS was slightly higher than that of GSS (Figure 5), due to the BZS being inclusive of the study site and the 1 ha BZ. The main flow path in the BZ was overland flow, which is due to the low vertical hydraulic conductivity of the peat soil. It has been reported that the hydraulic conductivity of the peat ranges from 10^{-4} m s^{-1} to 10^{-8} m s^{-1} (Lewis et al., 2011). Dahl et al. (2007) proposed four major flow path types – diffuse flow, overland flow, direct flow and drainage flow – in riparian areas. Compared with the other three path types, the overland flow path type had a lower residence time and smaller contact area with the bio-geochemically active soil and as a result could have a weakened nutrient removal capacity (Hoffmann et al., 2009).

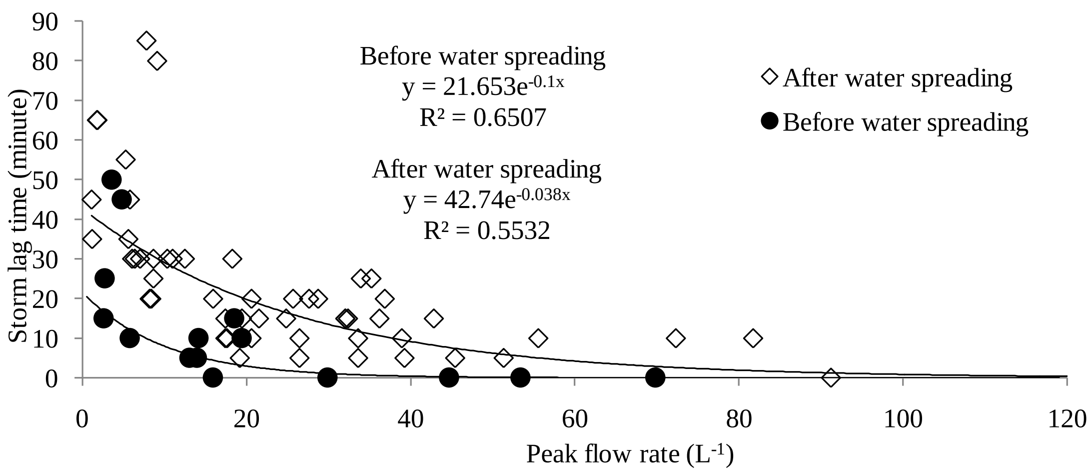

Figure 3. Runoff passing through the buffer zone in different storm events before and after water spreading; 15 and 65 storm events were included before and after water spreading, respectively.

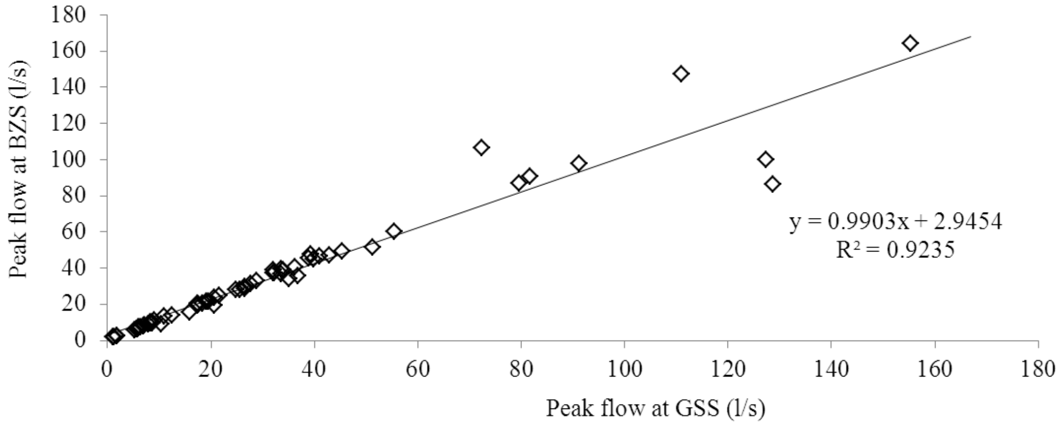

Figure 4. Storm peak flows at the GSS and the BZS after water spreading

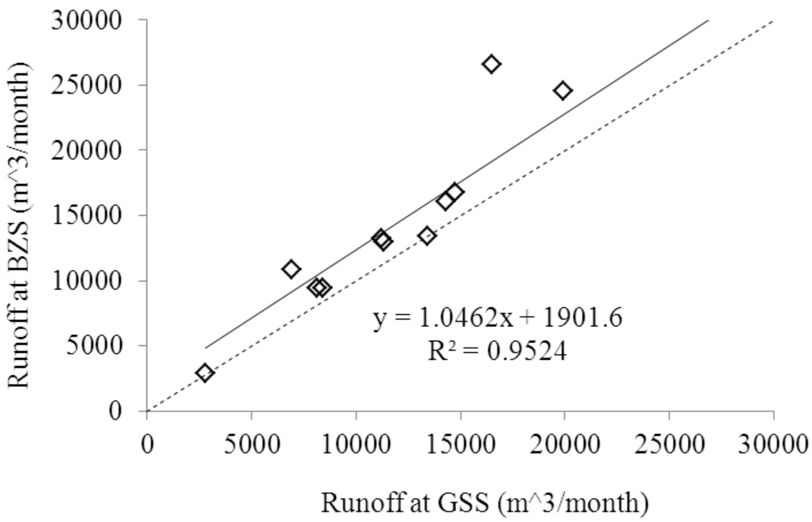

Figure 5. Monthly runoff passing through the GSS and the BZS after water spreading (Dash line is 1:1)

Nutrient and suspended sediment concentrations at GSS and BZS before and after harvesting

The average P concentrations in the rainfall were 4 ± 3 µg L^{-1} of TRP. Measured P concentrations at the two stations were low before the harvesting with average values of 13 ± 4 µg L^{-1} and 12 ± 3 µg L^{-1} at the GSS and BZS, respectively (Figure 6.). Four weeks after the harvesting operations began, daily discharge weighted mean P concentration at the GSS station beganincreasing gradually to a maximum of 101.7 µg L^{-1} by the end of the harvesting period.

During the same period the maximum P concentration in the BZ did not exceed 40 µg L^{-1} (Figure 6).

The average TON concentrations in the rainfall were 69 ± 28 µg L^{-1}. Before harvesting the measured TON concentrations were 35 ± 25 µg L^{-1} and 48 ± 32 µg L^{-1} at the GSS and BZS, respectively. After harvesting the values increased at GSS to 82 ± 40 µg L^{-1}, which was similar to the values of 76 ± 87 µg L^{-1} at BZS (Figure 6), indicating that the buffer zone did not retain TON significantly.

The average NH_4-N concentrations in the rainfall were 74 ± 38 µg $L^{-1.}$ Measured NH_4-N concentrations before harvesting were 72 ± 47 µg L^{-1} and 63 ± 43 µg L^{-1} at the GSS and BZS, respectively. After harvesting the values increased at the GSS and at the BZS (112 ± 170 µg L^{-1} and 125 ± 190 µg L^{-1} respectively), suggesting that NH_4-N was not retained by the buffer zone (Figure 6).

Measured SS concentrations before harvesting were 48 ± 76 mg L^{-1} and 339 ± 806 µg L^{-1} at the GSS and BZS, respectively (Figure 6). The values observed were not typical of reported baseline values for upland peat sites (Rodgers et al., 2011). However, during the pre-felling baseline data collection period a forest road was constructed and tonnes of sediment was imported into the catchment. While best management practices (BMPs) were implemented during the road construction period this data highlights the ongoing and unresolved issue of the impacts of forest road construction. The after harvesting values were insignificant in comparison to the pre-felling values.

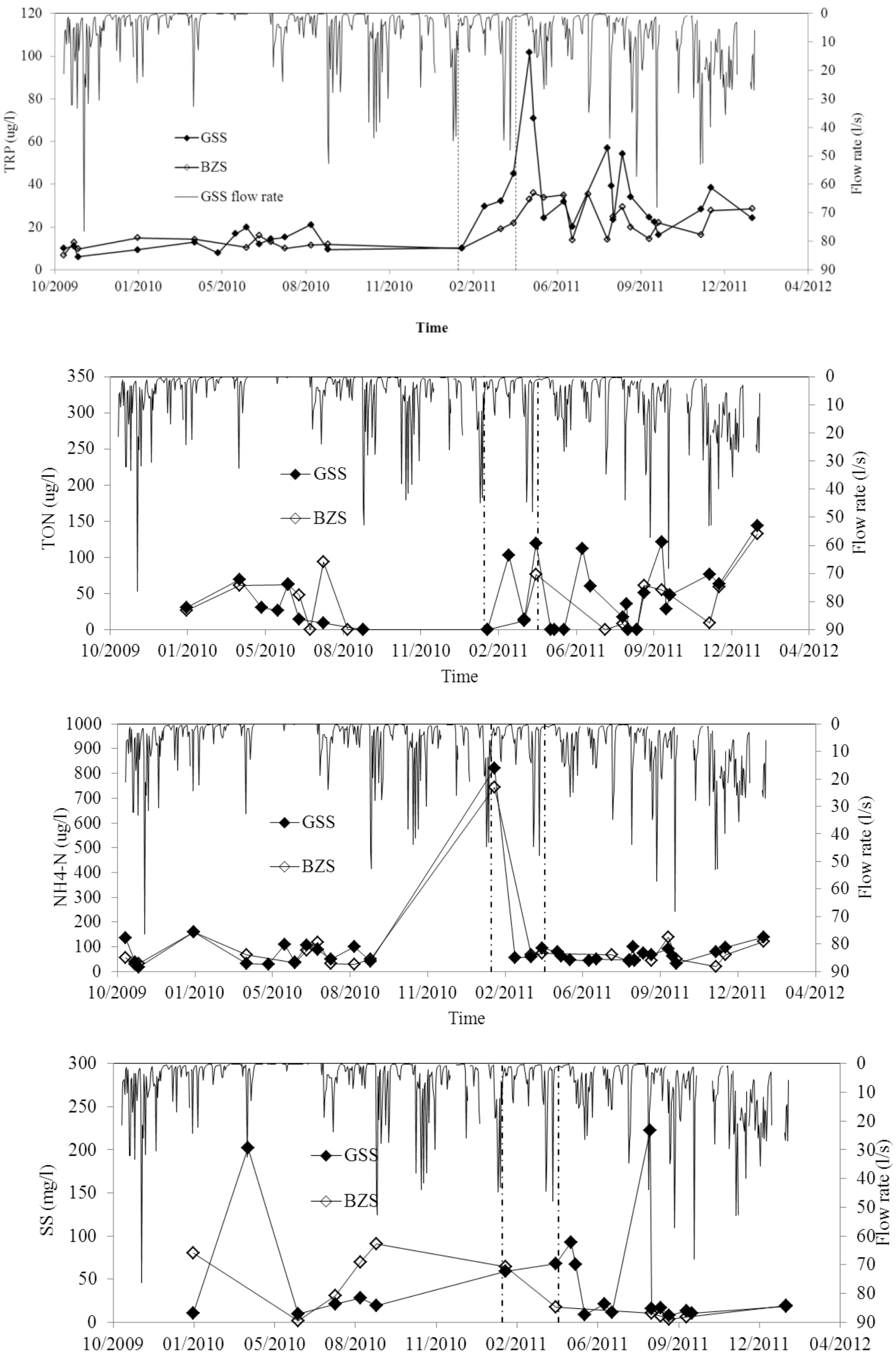

Figure 6. The daily discharge-weighted mean TRP, NH4-N, TON and SS concentrations at the GSS and the BZS during the study period. Dash lines indicate the harvesting period.

Table1 Input and output loads on the buffer zone (0.11 ha) from Feb 2011 to Jan 2012

	Input loading rate ($kg\ ha^{-1}\ yr^{-1}$)	Output loading rate ($kg\ ha^{-1}\ yr^{-1}$)	Removal efficiency (%)
TRP	55.07	45.36	17.6
TON	112.66	74.67	33.7
NH_4-N	137.73	135.6	1.5
SS	47638	26072	45.3

During the one year period from February 2011 to Jan 2012, a total of 6.17 kg TRP, 12.62 kg TON, 15.43 kg NH_4-N and 5335 kg SS were released from the GSS. These values were comparable with that reported by Nieminen (2003), Cummins and Farrell (2003) and Rodgers et al. (2010 and 2011). The input loading rates of TRP, TON, NH_4-N and SS to the buffer zone were extremely high (Table 1), compared with the studies reviewed by Hoffmann et al. (2009). The high loading rates were mainly due to the high runoff from the upstream study site, high nutrient and SS concentrations and the relative small buffer zone area (1.1 % of the study site). The BZ retained the buffer zone retained significant amounts of SS (21566 kg ha^{-1}), TON (38 kg ha^{-1}) and TRP (9.7 kg ha^{-1}) (Table 1) from through flow water from February 2011 to January 2012. However, whilst the retention efficiency was good for SS (45%) and TON (34%) it was relatively low for P (18%). The probable reason for low P retention is the high hydraulic loading during storm events (Väänänen et al., 2008) when the water residence time is low (Koskiaho et al., 2003), which is disadvantageous for efficient retention of nutrients.

Buffer zone performance in storm events

A total of 20 storm events (24 samples per event) were analysed during the harvesting and post-harvesting period. Higher PO_4-P and SS loading rates resulted in lower retention percentages (Figure 7). In storm events with a loading rate of more than 28 g P ha^{-1}, the buffer zone could become a PO_4-P release source (Figure 7). This has also been observed by Väänänen et al. (2008) who reported that high hydrological loads and the associated creation of preferential flow paths were particularly disadvantageous to buffer zone efficiency. With the increase of loading rates, the NH_4-N and TON retention slightly increased (Figure 7).

While no obvious trends were observed for NH_4-N and TON, logarithmic equations could be used to describe the relationship between the P retention and inlet flow rates and concentrations (Figure 8). The buffer zone started to release P when the flow rate was higher than 88.5 L s^{-1}, and when the inlet P concentration was lower than 17 µg L^{-1} (Figure 8). A logarithmic equation could also be used to describe the relationship between SS retention and inlet concentrations.

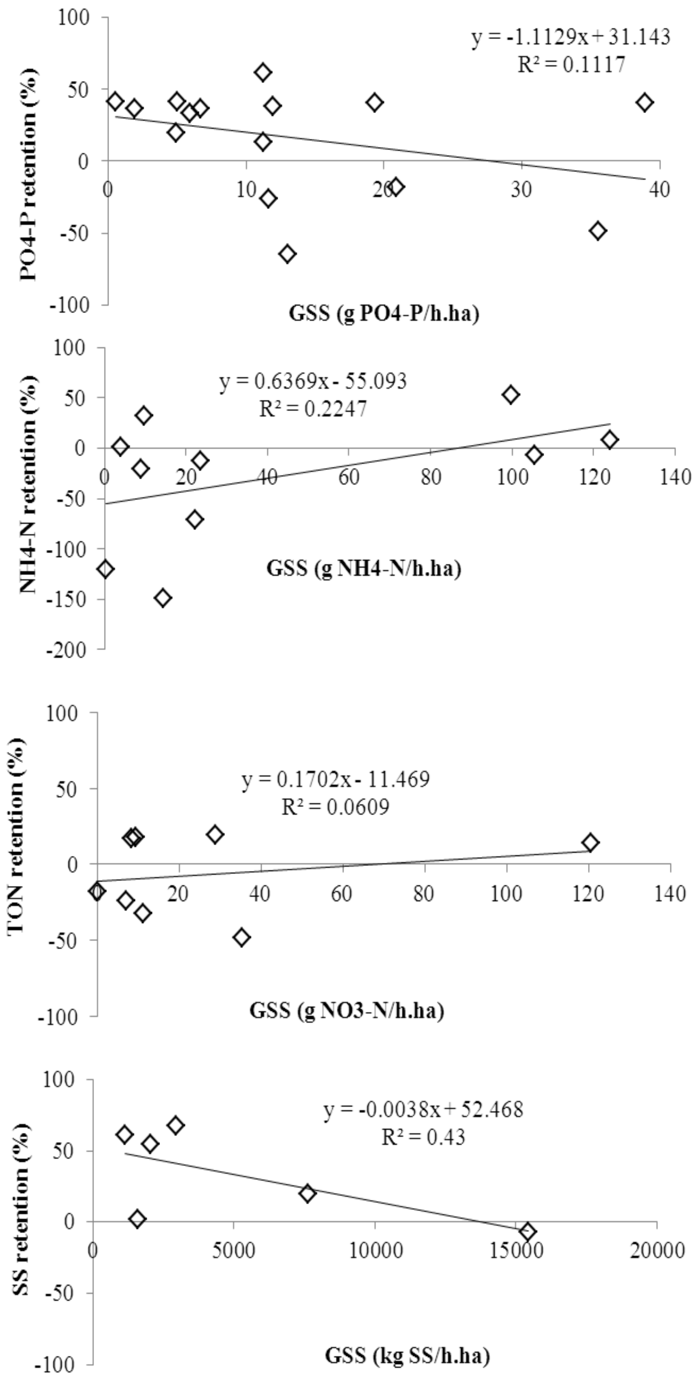

Figure 7. The relationships between TRP, TON, NH4-N and SS retention and loads during storm events

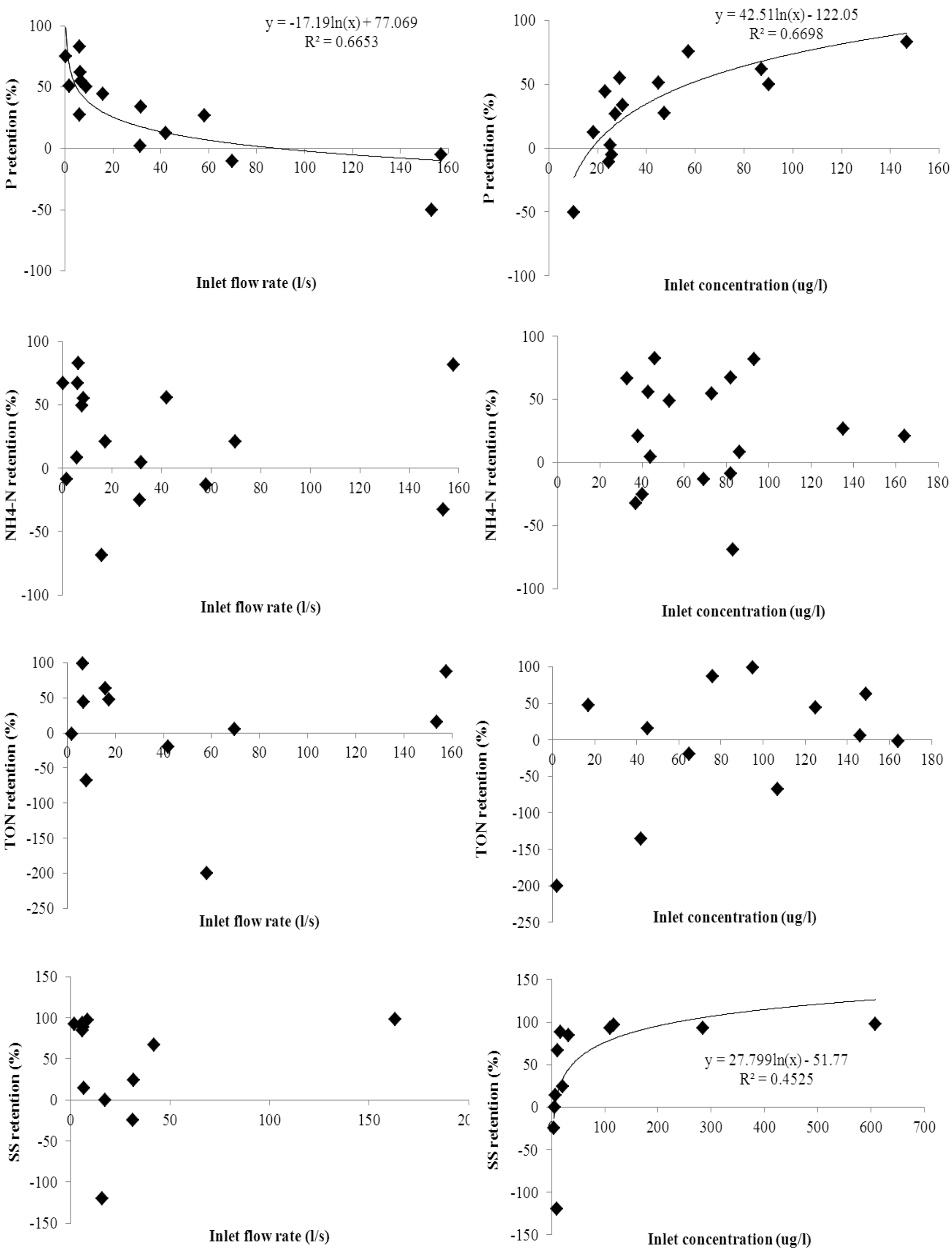

Figure 8 Relationship between instantaneous % retention and inlet flow rate and inlet concentrations.

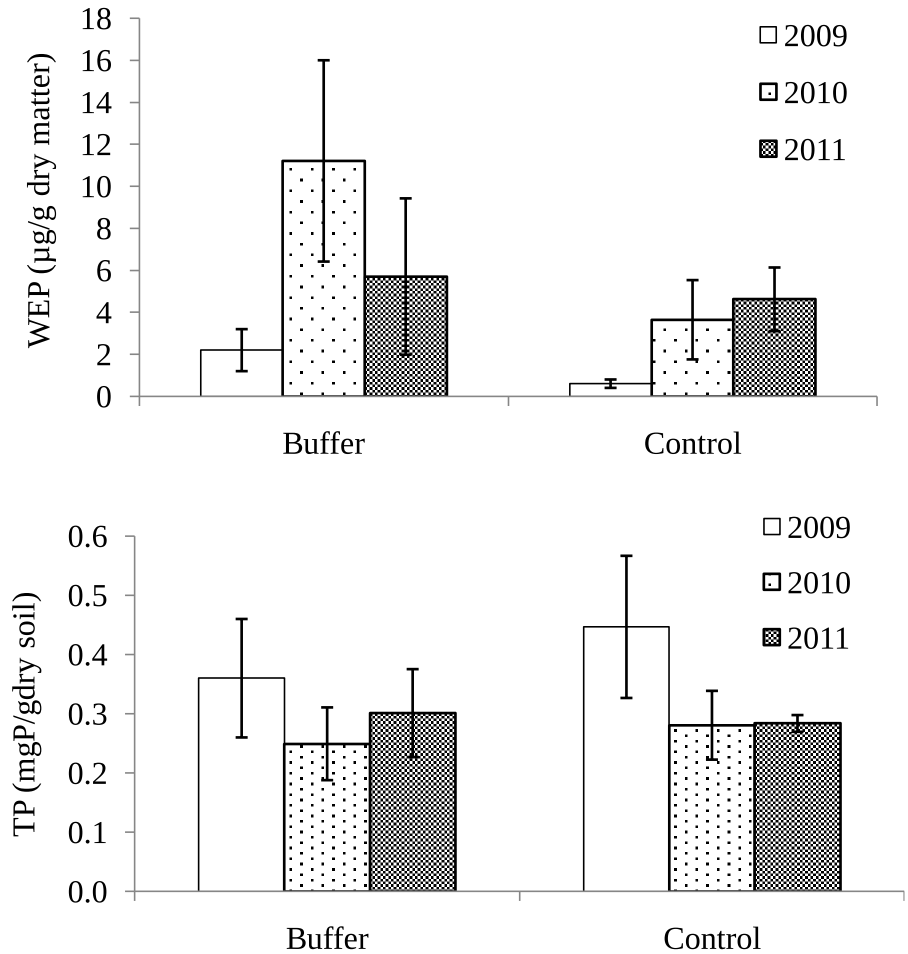

Figure 9 Soil water extractable (WEP) and total (TP) phosphorus in the buffer zone and control area.

Water extractable (WEP) and total (TP) P concentrations of the soil in the buffer zone after harvesting

In 2009, before the clearfelling of the buffer zone, the WEP was low (2.21 ± 1.00 µg L^{-1} and 0.61 ± 0.21 µg L^{-1}) and the TP was high (0.36 ± 0.10 µg L^{-1} and 0.45 ± 0.12 µg L^{-1}) in both the buffer and control plots respectively (Figure 9), as the standing forest immobilises the P (Walbridge and Lockaby, 1994; Herz, 1996). In 2010 during the establishment of the buffer zone, the TP was leached out of the soil and the values were reduced to 0.25 ± 0.06 µg L^{-1} and 0.28 ± 0.06 µg L^{-1} in the buffer and control plots, respectively (Figure 9). The WEP increased during the same period to 11.22 ± 4.79 µg L^{-1} and 3.65 ± 1.89 µg L^{-1} (Figure 9), reflecting the leaching P and the decomposition of logging residues (i.e. needles, twigs, and roots) (Hyvönen et al., 2000; Piirainen et al., 2004) and lowered plant uptake. In 2011, the P from the receiving water was retained by the buffer plots and the TP increased in the buffer zone (0.30 ± 0.07 μg L^{-1}) but not in the control (0.28 ± 0.01 μg L^{-1}). The WEP decreased in the buffer plot (5.71 ± 3.72) μg L^{-1} reflecting the uptake by the seeded grass.

Biomass production and P concentration of the vegetation in the buffer zone after harvesting

Seeding of *Holcus lanatus* L. and *Agrostis capillaris* L. increased the aboveground vegetation biomass and P content 1 year after grass seeding (Figure 10). One year later after the study site had been felled and stream water passed through the buffer zone, the vegetation in the buffer zone had again increased in the study plots and the P content was 9.73 ± 1.14 μg L^{-1}, 7.80 ± 0.72 μg L^{-1}, 7.61 ± 0.87 μg L^{-1} and 1.00 ± 0.1 μg L^{-1} in plot 1, 2, 3 and the control plot, respectively. The novel native grass seeding method (NGSM) practice resulted in an increased amount of aboveground vegetation biomass in the buffer zone plots compared to the control in 2010. This increased 5-fold in 2011 when the above ground biomass contained 6043, 7438 and 9992 kg ha^{-1} in the buffer plots (1, 2 and 3) compared to 1200 kg ha^{-1} in the control plot. This highlights these chosen species as successful for buffer zone function. Compared with 2010, an additional 8 kg P was uptaken by the grass. This value is close to the value of 9.48 kg TRP retained by the buffer zone.

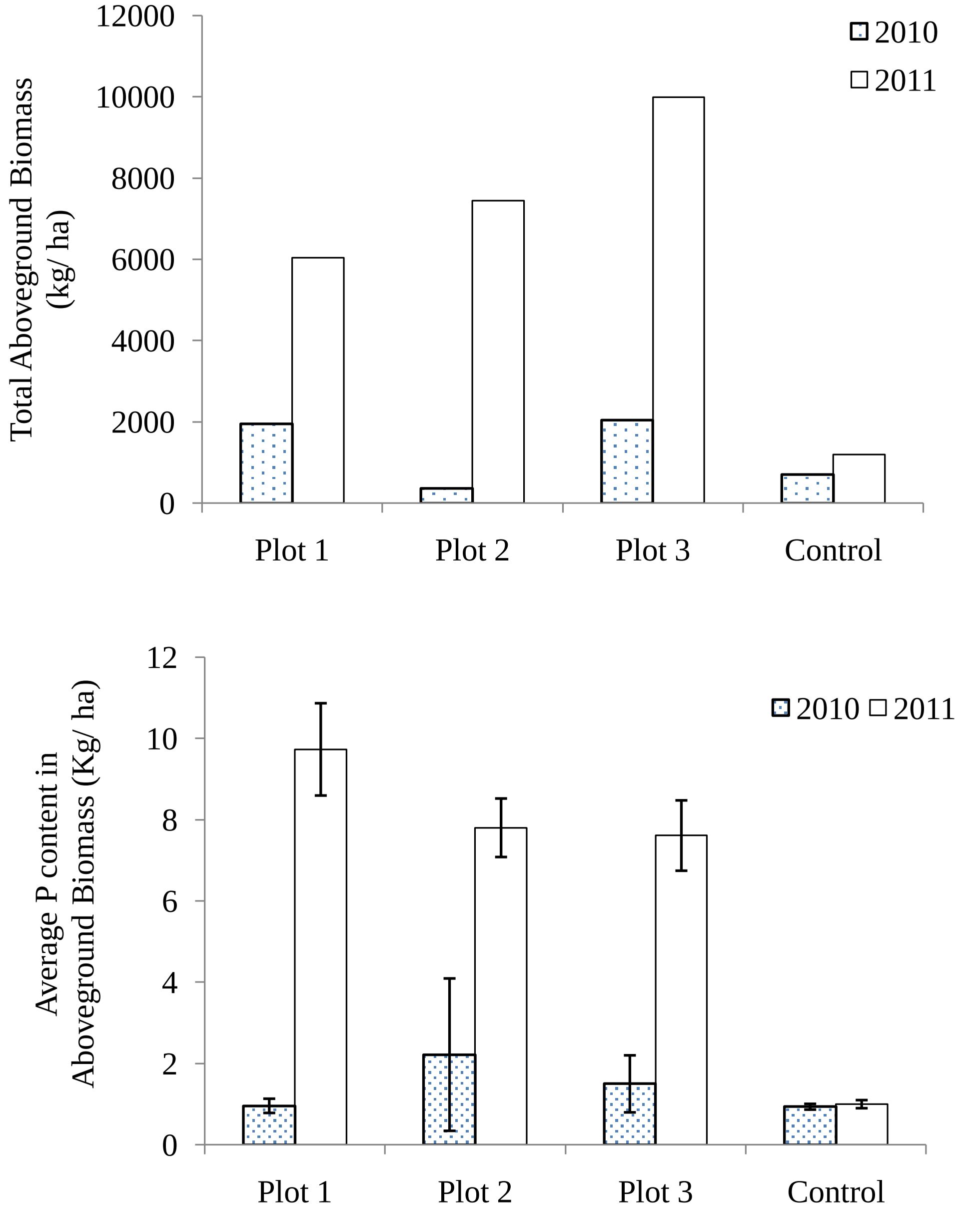

Figure 10 Above ground biomass and P in the buffer zone in 2010 and 2011.

Impact of buffer zone on flora and fauna

A total of 133 taxa were encountered during the study. Of these taxa, 74 were invertebrates, 15 were birds, 2 were mammals and 42 were plants. The results show that the buffer zone was significantly more diverse than the mature plantation (Table 2). No significant difference was found between the buffer zone and the area of natural vegetation. Small mammals (*Sorex minitus* and *Apodemus sylvaticus*) were present in the buffer zone but were not encountered on either control site. There was no significant difference in bird species richness between the buffer zone and the control sites. The results indicated that, relative to the control sites, buffer zones are an important area for terrestrial invertebrates and small mammals.

Table 2 Biodiversity of invertebrates in the three sites

Site	Average Species Richness	Shannon - Weiner	Effective no. of species
Mature conifer plantation	19	1.63	4.89
Buffer zone	24.66	3.362	10.27
Naturally vegetated area	18.33	3.04	8.22

Of particular interest is the diversity within the ground beetle (Carabid) community. The buffer zone in the Glennamong catchment supported 12 species of Carabid, including the rare/threatened *Carbus clatratus*. The conifer plantation and blanket bog sites supported 4 and 9 Carabid species respectively.

In a national survey, Williams and Gormally (2010), found that *C. clatratus* only occurred in 10 out of 129 localities sampled in Ireland. *C. clatratus* is a powerful predator that requires areas of bare ground for basking, vegetated areas for cover and areas of standing water for hunting. In this study, *C. clatratus* was not encountered in the lodgepole pine plantation but was found in the blanket bog site, supporting the view that afforestation of such sites is a potential threat to this species, as concluded by Williams and Gormally (2010). However, the presence of a buffer zone offers a potential refuge for this threatened species within plantation forests. Structural diversity appears to be a key habitat requirement for the small mammals encountered in this survey (*Sorex minitus* and *Apodemus sylvaticus*). The presence of BMs, tree stumps and areas of mixed vegetation in the buffer zone adds to the structural diversity of the site and improves the suitability of the site for these two species.

Conclusion

The newly established small peatland forest buffer zone, with the area of about 1.12 % of the upstream study site, has the capacity to reduce SS, TON and TRP loads to water courses, with the removal rate of 45.3 %, 33.7 % and 17.6 %, respectively. In addition to protecting aquatic systems, the buffer zone also provided a habitat for flora and fauna that would not normally be associated with conifer plantations. The buffer zone was significantly more diverse than the mature plantation. No significant difference was found between the buffer zone and the area of natural vegetation in terms of biodiversity.

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Bibliography

See Chapter Six pp. 187 - 222.

Chapter Five

Conclusions and Recommendations

5.1 Overview

This study examined the impacts of current best management practices (BMPs) in forest harvesting on hydrology, nutrient release and receiving stream biota, and investigated possible mitigation measures, including the use of buffer zones, a novel grass seeding method, and whole tree harvesting. It also aimed to make recommendations on potential mitigation measures that may be undertaken.

5.2 Main conclusions

The main conclusions from this study are:

1. Forest harvesting increased the monthly water yield and base flow significantly in receiving first order streams. However, this had no impact on flood risk downstream. This could be due to the implementation of the good management practices such as (1) use of brash mats and (2) harvesting only in dry weather minimising soil surface disturbance.
2. Harvesting of the blanket peat forest increased the TRP export in the study stream, and this impact could last for more than four years. More than 70 % of the P release occurred during storm events. Due to the dilution capacity of the main river, the P concentrations in the river were low during the study period, indicating that rational sizing of the harvesting coupe could be an efficient practice to limit the P concentration in the receiving waters following harvesting.
3. Alkalinity and conductivity were the main physicochemical drivers of the diatom assemblages in upland peatland rivers typical of the west of Ireland. This highlights the importance of the underlying geology in determining diatom assemblage composition.

Multivariate analysis indicated that nutrient enrichment from forestry activities did not appear to have an influence on the diatom assemblages. Therefore, these upland peatland rivers represent reference conditions with respect to nutrient status. This could be due to the spatey nature of the rivers and the frequent flushing of the rivers systems.

4. In this study significant spatial trends were evident in both macroinvertebrate and diatom assemblages relating to downstream increases in alkalinity. These trends were reflected in the biotic indices calculated for both groups. Significant seasonal changes were confirmed in the invertebrate assemblages relating to water temperature and life-cycles. Despite this, ecological quality classes were consistent throughout seasons and years. However for these upland blanket peat catchments, sampling season did not significantly affect the diatom related biotic indices.
5. Forest harvesting had a short-term impact on the water quality and macroinvertebrates, but had little impact on the diatom assemblages in acidic headwater streams of peatland catchments. This could be due to the implementation of BMPs, such as the use of brash mats and harvesting only in dry weather, which may have minimised soil disturbance. However, it is also questionable as to whether nutrient tolerant diatom species could tolerate the low acidic nature of these sites; and in which case, the high ecological status observed is a reflection of this rather than a true nutrient encrichment detection. Harvesting strategies, such as phased felling in acid sensitive catchments, are effective in protecting larger salmonid rivers against the additional input of nutrient and sediments after forestry harvesting on watersheds.
6. The results of this study indicate that (1) *Holcus lanatus* and *Agrostis capillaris* can be established quickly in blanket peat forest areas after harvesting and (2) sowing *Holcus lanatus* and *Agrostis capillaris* immediately after harvesting has the potential to immobilize the P that would otherwise be available for leaching.
7. The research carried out in this study has highlighted that buffer zones established on forest harvested peat sites have the potential to uptake P. The study demonstrated that during high flow storm events when the hydrological load is high and preferential flow occurs, there can be some leaching of P from the BZ. When the P coming from the study site returns to baseline levels the BZ could be a source of P in the future.

8. The most effective method for reducing the P export from harvested sites to receiving water is a combination of the whole tree harvesting method along with the grass seeding method. This practice should be applied to sensitive sites such as freshwater pearl mussel catchments.

5.3 Main recommendations for future research and water quality managers

1. Further work needs to be carried out to determine if the acidic nature of the sites is a response to anthropogenic impacts or natural acidity. The results of this study could be applied to similar upland peat forest catchments and used as a benchmark to assess the impact of ongoing forest harvesting on ecological status. The impact of a major flood event on diatom assemblage structure is evident at the lower sites; however, this had no bearing on the EQR status. Future work needs to be carried out to determine how long (if at all) it takes for these species to return. Many ecological patterns appeared in the analysis of this biological data and further study using controlled experiments is recommended to disentangle the ecological variables driving these trends. The sampling strategy of using one sampling point in a river may not be deemed sufficient to provide a reliable estimate of the measured biotic index score for a waterbody. It is important to consider the pressures on these upland blanket peat catchments when choosing a sampling site. With regard to site selection both invertebrates and diatoms contribute to increases in acidity indices downstream and away from the constraining influence of the peat. The work presented here provides valuable information which will enhance the dialogue between ecological research and biomonitoring programmes.
2. Further research into the feasibility of grass seeding as a potential new BMP is warranted. Sowing the right grass species at appropriate rates should diminish the deleterious effects of forest harvesting on surface water quality and facilitate the forest regeneration.

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