NATIONAL UNIVERSITY OF IRELAND, GALWAY

ASSESSMENT OF THE IMPACTS OF FORESTRY ON PEATLANDS ON THE ENVIRONMENT

Joanne Finnegan, B.E.

Research Supervisor:
Dr. Mark G. Healy, Civil Engineering, NUI Galway
Professor of Civil Engineering: Padraic E. O'Donoghue

Thesis submitted in fulfilment of the requirements for the degree of Doctor of Philosophy

September 2012

The National University of Ireland requires the signatures of all persons using or photocopying this thesis. Please sign below, and give the address and date.

“If a tree falls in the forest and no one is there to hear it, does it make a sound?”
Bishop George Berkeley (1685 – 1753)

Abstract

Ireland's forest cover stands at approximately 10 %, or 700,000 ha, of the total surface area of the island and it is estimated that almost 60 % of this forestry is on peat. Forestry on peatland throughout the world is now moving towards a 'progressive management approach', which incorporates sustainable timber production alongside multiple uses such as habitat restoration, ecological regeneration and the minimisation of any potentially negative effects to the surrounding environment. However, the legacy of blanket peatland forestry, planted in the 1950s, must be dealt with, as most of this forestry is now at harvestable age and current and future recommended best management practices (BMPs) for forestry operations must consider soil and water quality, environmental impacts and greenhouse gas (GHG) emissions. The aim of this project was to investigate the short and long-term changes in nutrient and sediment releases, watertable (WT) fluctuations, and GHG emissions arising from harvesting (clearfelling) of forested peatlands in the west of Ireland.

The study was located in three sites: (1) the Altaconey (Altahoney) forest, which comprised a regenerated riparian peatland buffer clearfelled 5 years before the present study, a recently clearfelled coniferous forest, and a standing mature coniferous forest (2) a virgin peat site and (3) a paired catchment study in the Glennamong forest. The Altaconey forest was instrumented with a network of piezometers, one of which was automated, for WT and water quality measurement, a rain gauge, and open-bottomed collars for gas flux measurement. Water, soil and gas measurements, the latter of which were also collected at the VP site, were taken regularly over a 2 ¼ -year study duration (12 months before clearfelling, 15 months after). Two paired catchments in the Glennamong forest, one a study control (no clearfelling) and the other clearfelled, and each with an area of approximately 10 ha, were instrumented for water quality and flow measurement.

Management changes such as drainage, fertilisation, afforestation and subsequent clearfelling of forested peatlands influences WT position, nutrient load transfer to shallow groundwater, and GHG emissions from soil respiration. In the Altaconey forest, there was an immediate rise in the WT after clearfelling, but this had no significant

impact on the concentrations of total oxidized nitrogen (TON), nitrate nitrogen (NO_3^--N) or dissolved reactive phosphorus (DRP), the latter of which was more impacted by degrading logging residues (brash material) than by WT fluctuations. However, fluctuations in WT did influence concentrations of ammonium-nitrogen (NH_4^+-N), which was highest under the standing mature coniferous forest, an area with the deepest WT. Nitrogen (N) and phosphorus (P) discharges to the adjacent watercourse in excess of maximum admissible concentrations were negligible due to the low lateral saturated conductivity and the high inherent natural attenuation capacity of the peat.

Fluctuations in the WT also affected GHG emissions from soil respiration and sequestration, as clearfelling of the forest at Altaconey produced significant increases in carbon dioxide (CO_2) (11±2 kg CO_2-C ha^{-1} d^{-1} before clearfelling to 19±2 kg CO_2-C ha^{-1} d^{-1} after clearfelling) and methane (CH_4) emissions (22±14 g CH_4-C ha^{-1} d^{-1} to 163±99 g CH_4-C ha^{-1} d^{-1}), but a decrease in nitrous oxide (N_2O) emissions (1.7 g N_2O-N ha^{-1} d^{-1} to 0.7 g N_2O-N ha^{-1} d^{-1}).

Elevated levels of nutrients and suspended sediment (SS) in surface waters are frequently associated with forestry clearfelling operations for up to 4 years. Despite significant rises in nutrients and SS at the Glennamong study site and changes to some water parameters, the implementation of BMP, where possible, and the quick execution of a site restoration plan comprising silt traps and water management on extraction racks, appeared to negate excessive nutrients and SS export to the adjoining watercourse.

Declaration

This dissertation is the result of my own work, except where explicit reference is made to the work of others, and has not been submitted for another qualification to this or any other university.

Joanne Finnegan

Acknowledgements

I wish to thank all involved in the preparation of this thesis, especially the staff of NUI Galway, Teagasc (Dr. Gary Lanigan and Dr. Owen Fenton), the Advice and Steering Committees and project partners of HYDROFOR; Coillte for allowing access to state forestry and providing advice on technical issues, all at the Marine Institute for their continued help throughout this project, and the Department of Agriculture, Fisheries and Food and the Environmental Protection Agency for funding the project under the STRIVE program 2007 – 2013.

A special thanks to Dr. John Regan, our dedicated Research Assistant, who gave so freely of his time and patience. Thanks also to Victor, Adrien, Chiara and Chops for keeping me company on site.

I owe my deepest gratitude to my research supervisor, Dr Mark Healy. It was always a great comfort to know you finished your own thesis in less than 3 years! Thanks for the hours of support, advice and chats. My engineering knowledge and movie awareness is greatly improved after knowing you!

Thanks to Nobbles and Gobbles for recycling my paper drafts by the power of chewing with their little rabbit teeth. Your fluffiness kept me entertained in many long days in the office.

Much love to my friends and family who listened to my rambling stories of peat and trees and paper writing, it can’t have been easy! My endless gratitude goes to my parents who raised me to be independent and strong willed enough to see this through.

Finally, thanks must go to my new husband, Tim, who has encouraged, gently, right from the beginning. A man, a plan, a thesis…… ‘til the wheels fall off.

Abbreviations

Al	Aluminium
ANOVA	Analysis of variance
AOD	Above ordnance datum
bgl	Below ground level
BMP	Best management practice
BOD	Biochemical oxygen demand
C	Carbon
CC	Control catchment
CF	Clearfell forest
CH_4	Methane
CO_2	Carbon dioxide
CPR	Cone penetration resistance
DAFM	Department of Agriculture, Fisheries and the Marine
DNRA	Dissimilatory nitrate reduction to ammonia
DO	Dissolved oxygen
DRP	Dissolved reactive phosphorus
EC	Electrical conductivity
EEA	European Environment Agency
EPA	Environmental Protection Agency
EU	European Union
Fe	Iron
FSC	Forest Stewardship Council
FWMC	Flow-weighted mean concentration
GHG	Greenhouse gases
GIS	Geographical information systems
Gt	Gigatonnes
GWP	Global warming potential
ha	Hectare
IDW	Inverse distance weighted
IPCC	Inter-governmental Panel on Climate Change
ISO	International organization for standardization

ITM	Irish Transverse Mercator
k_s	Saturated hydraulic conductivity
LOI	Loss on ignition
LUC	Land-use change
MRP	Molybdate reactive phosphorus
MSS	Mineral suspended sediment
N	Nitrogen
N_2	Nitrogen gas
N_2O	Nitrous oxide
NH_3	Ammonia
NH_4^+	Ammonium
NH_4^+-N	Ammonium nitrogen
NO_2^--N	Nitrite nitrogen
NO_2^-	Nitrite
NO_3^-	Nitrate
NO_3^--N	Nitrate nitrogen
O_2	Oxygen
OS	Ordinance survey
OSS	Organic suspended sediment
P	Phosphorus
Q ratings	Biological Q ratings
RB	Regenerated buffer
RBZ	Riparian buffer zone
SAC	Special Area of Conservation
SC	Study catchment
SF	Standing forest
SFM	Sustainable Forest Management
SI	Statutory instrument
SOC	Soil organic carbon
SOM	Soil organic matter
SS	Suspended sediment
TON	Total oxidised nitrogen
TP	Total phosphorus
UB	Under brash

VA	Vegetated area
VP	Virgin peat site
WEP	Water extractable phosphorus
WFD	Water framework directive
WFPS	Water filled pore space
WT	Watertable
WTH	Whole tree harvesting
yr	Year

Table of contents

List of tables

List of figures

Chapter 1

1.1 Background

Peatlands are found in over 175 countries worldwide, are mostly present in moist, temperate climates in the northern hemisphere (Sjörs, 1980), and cover approximately 3 % of the total landmass in the world (4,000,000 km^2) (Bain et al., 2011). Peatlands produce 10 % of the global freshwater supply and one-third of the world's soil carbon (C) content (Joosten and Clarke, 2002). Approximately 150,000 km^2 of this landmass has been drained for commercial forestry, while the area not commercially drained, but forested, is unknown (Joosten and Clarke, 2002).

Ireland's forest cover stands at 10.15 %, or 700,000 ha, of the total surface area of the island (National Forest Inventory, 2007). The Irish State, under the management of the Forest Service, carried out the majority of the afforestation in the mid-20th century. This was mainly coniferous plantation on non-productive agricultural land (Bacon, 2003). It is estimated that 59.6 % (417,200 ha) of forestry in Ireland is on peat (National Forest Inventory, 2007) and approximately 300,000 ha of afforestation is on upland peat areas (EEA, 2004; Rodgers et al., 2010).

Despite grants from the European Union (EU) and the relatively high productivity of these peatland forests, the economic viability of such plantations on upland peat is limited (Renou and Farrell, 2005) with over 40 % of the forestry having poor production potential (Tierney, 2007). Additionally, the standard forestry practice in Ireland and the UK at the time of afforestation (in the 1950s) was to plant trees in areas adjacent to water courses and to not include riparian buffer zones (RBZs) in the design (Broadmeadow and Nisbet, 2004; Ryder et al., 2011). The absence of RBZs means that there may be nutrient or sediment release into water courses during clearfelling (Carling et al., 2001). As much of the commercial coniferous forestry planted in the 1950s is now at harvesting age, the adoption of current forest practice, which utilises RBZs, may minimise the risk of negative impacts on receiving waters for successive rotation. The decision to either replant or restore these sites needs to be made (Renou-Wilson et al., 2011), but this should be based on empirical knowledge of the response of peatland forests to various activities. Therefore, the aims of this project were to investigate the short- and long-term impacts of forestry activities, such as clearfelling and the use of logging residues and the un-merchantable top part of the tree as ‘brash mats’ for transport of machinery, on water quality and greenhouse gas (GHG) emissions.

1.2 Legislative Drivers

The Water Framework Directive (WFD) (2000/60/EEC) requires all EU member states to achieve good ecological and chemical status for all surface and ground waters by 2015. This directive is the primary driving legislative force directed at improving overall water quality in Ireland today. In 2005, the first risk assessment of the anthropogenic pressures on water resources was undertaken to identify the pressures present in each river basin district in Ireland and the threat they pose to the chemical and ecological status of water bodies. Diffuse pollution from forestry operations, including acidification from afforestation, sedimentation from clearfelling and road construction, and eutrophication from fertilisation and clearfelling, were identified as potential risks to water bodies (Anon, 2005). The WFD proposes to prevent deterioration of water bodies, promote sustainable water use, and ensure “enhanced protection and improvement of the aquatic environment" by limiting and reducing the pressures from various sectors, including forestry.

1.3 Pressures: Source and Processes

Clearfelling of forestry on peat can be challenging due to high soil water contents (gravimetric water contents usually exceed 800 % (Long and Jennings, 2006)), low ground bearing capacities of between 10 and 60 kPa (Owende et al., 2002), and the vulnerable nature of the ecosystem (Forest Research, 2009). Negative effects of clearfelling may include eutrophication (an increase in nutrient levels in a watercourse, causing excessive flora growth (Sharpley, 2003)) and sedimentation (an increase in suspended sediment (SS) release to a watercourse, which may negatively impact water ecology (Rodgers et al., 2011)).

1.3.1 Eutrophication

Nutrients such as nitrogen (N) and phosphorus (P) are often applied to land at the afforestation stage to enhance and promote growth of selected species within ombrotrophic blanket peats (peats which have low nutrient concentrations and poor adsorption capacities) of the west of Ireland (Farrell and Boyle, 1990; Renou and Farrell, 2005). This, combined with N deposition from the atmosphere and ammonification within the peat layers, has led to N saturation, primarily present as ammonium (NH_4^+), in some upland peat catchments in the UK (Daniels et al., 2012). Ammonium can leach from the peat and be converted to nitrate (NO_3^-) by nitrification within the streams (Daniels et al., 2012), leading to toxic environments for aquatic life forms (Stark and Richards, 2008). Similarly, small concentrations of P (> 35 μg L^{-1} molydbate reactive phosphorus (MRP)) can have a negative impact on water quality (Bowman, 2009), leading to restrictions for fisheries, recreation, industry and drinking water (Sharpley, 2003; Elrashidi, 2011). Blanket peat has a poor adsorption capacity for P (O'Driscoll et al., 2011) and during the forest operations of drainage, fertilisation and clearfelling, hydrological losses of P can increase (Cummins and Farrell, 2003; Nieminen, 2003; Väänänen et al., 2008). Phosphorus loss during clearfelling is mainly due to loss from foliage (Paavilainen and Päivänen, 1995), and during clearfelling up to 70 % of P may be lost during high storm events (Rodgers et al., 2010). However, the P levels in receiving waters can return to pre-clearfell levels within 4 years of clearfelling (Rodgers et al., 2010).

1.3.2 Sedimentation

Peat soils are also susceptible to damage by clearfelling machinery traffic and subsequent rutting and compaction (Collins et al., 2000). After clearfelling, SS levels in receiving waters can increase due to soil disturbance, bank erosion and increased flow from the harvested areas, but these impacts are generally not long-term (Rodgers et al., 2011). Variations in concentrations of SS in drainage waters can relate to different site slopes, weather conditions and the rate of vegetation growth after clearfelling (Rodgers et al., 2011). Higher rates of SS loss are associated with steeper slopes (McBroom et al., 2008) and the rapid regeneration of vegetation within clearfelled areas can reduce SS export (Aust and Blinn, 2004). Peat catchments are also susceptible to high rates of runoff. May et al. (2005) found that over 90 % of the total flow in a peat catchment in the west of Ireland originated from precipitation, and was mainly in the form of surface runoff.

1.4 Factors Affecting Transfer of Pressure to the Environment

1.4.1 Watertable fluctuations

Blanket peatlands in the west of Ireland have very slow rates of drainage, low topographical gradients and low hydraulic conductivity (usually averaging at less than 0.01 m d^{-1}) (Farrell and Boyle, 1990). Such soils may pose a risk to groundwater due to extended travel times from source to receptor (Fenton et al., 2009). This slow drainage is compounded by a high annual average rainfall of approximately 2000 mm (Rodgers et al., 2010), leading to prolonged periods of soil saturation and a high watertable (WT) (Byrne and Farrell, 2005). The depth to the WT in peatlands is seen as the key factor to determining the changes in the global C cycle (Erwin, 2009), as it affects soil chemical conditions, soil temperature, and the availability of an aerobic environment (Sottocornola and Kiely, 2010). A fluctuation in the WT position can intensify C mineralization rates by up to three times (Blodau, 2002). A high WT is seen as crucial to obtaining a reduction in emissions of GHGs from peatlands (FAO, 2012). Cultivation of peatlands for forestry results in a lower WT due to increased transpiration from the growing vegetation (Renou and Farrell, 2005).

Many restoration projects have been initiated worldwide (Petrone et al., 2001) and in Ireland (Wilson et al., 2009) on degraded peatlands to raise the level of the WT. This has either increased (Petrone et al., 2001) or decreased (Best and Jacobs, 1997) carbon dioxide (CO_2) emissions. A study on CO_2 fluxes from a restored peatland in Ireland showed a large inter-annual variation, making it difficult to predict restoration effects (Wilson et al., 2007). A reduction in CO_2 emissions is offset by the increase in the flux of methane (CH_4) from the newly re-wetted areas. Despite this increase in CH_4, a re-wetted peatland lowers the global warming potential (GWP) (tonnes of CO_2 equivalent) of a degraded peatland site (Wilson et al., 2009). Globally, it has been found that CH_4 emissions from anaerobic decomposition of peat are small and inconsequential compared to the flux of CO_2 (Jauhiainen et al., 2011). Wilson et al. (2009) modelled various restoration scenarios (cutaway, wetland creation, naturally regenerated deciduous forest, afforestation of conifers and grassland) on degraded cutaway peats in Ireland and found that the natural regeneration of deciduous forestry resulted in the lowest CH_4 flux, while afforestation provided the highest cooling effect.

1.4.2 Greenhouse gas emissions

Land-use change (LUC) from wetland systems to forestry and subsequent deforestation can lead to large-scale changes in ecosystem C and N dynamics (IPCC, 2006). Afforestation assists in both reducing non-methane GHG emissions and sequestering atmospheric CO_2 *via* photosynthesis. Although afforestation of peatlands increases the total amount of C sequestered, this is primarily in the woody biomass, and in the long term, soil C stocks actually decrease (Hargreaves et al., 2003; Byrne and Farrell, 2005). Deforestation, followed by draining of the soil, results in the release of up to 0.5 Gt of CO_2 per year, which is 14 % of the total annual anthropogenic emissions (UNEP, 2009). Carbon dioxide, CH_4 and nitrous oxide (N_2O) are regarded as the most important GHGs, accounting for an estimated 80 % of the total GWP (IPCC, 2001). Land-use change from peatland to forestry generally leads to an increase in CO_2 and N_2O loss and a decrease in CH_4 as the soil dries and the bacterial conditions change (IPS, 2008). The uptake of CO_2 occurs *via* tree photosynthesis, while CO_2 release is principally associated with both autotrophic respiration, the decomposition of organic matter, and the subsequent heterotrophic respiration and combustion of biomass (IPCC, 2006). Globally, forest soils act as sinks for CH_4 and can uptake approximately 30 Tg (30 million tonnes) annually (IPCC, 2001). In contrast, virgin peat soils act as a source of CH_4 due to a higher soil water content, which produces anaerobic conditions and methanogenesis (the formation of CH_4 by microbes) (IPCC, 2006). Within pristine ombrotrophic peatland systems, there is little N_2O efflux due to the fact that both mineral N pools are low and because anaerobic conditions promote total denitrification of any NO_3^- to nitrogen gas (N_2) (Van Beek et al., 2004). Upon drainage, increases in soil redox potential stimulate the biological processes of nitrification and partial denitrification, which results in a flux of N_2O between the soil and atmosphere (IPCC, 2006).

1.4.3 Use of brash mats

Clearfelling of forestry on peat soils can lead to soil disturbance and subsequent rutting due to pressures from clearfelling machinery. Peatland forestry clearfelling in other Boreal countries generally takes place during the winter, when the soil is less susceptible to damage due to the frozen ground. Ireland has a temperate, maritime climate that is heavily influenced by the proximity of the Atlantic Ocean and rarely suffers extremes of temperatures (Met Eireann, 2012). Therefore, in Irish forestry clearfelling practice, soil disturbance is minimised by laying a brash mat (Figure 1.1) ahead of the machinery used for the harvesting and timber removal processes. Previous research has highlighted the negative impacts of soil compaction (i.e. the expulsion of air from the void space) on the use of the soil for future afforestation (Nugent et al., 2003; Gerasimov and Katarov, 2010). The reduced pore space for water movement reduces the growth rate of future tree and plant crops (Antti, 2008), reduces the value of the harvested timber (Eliasson and Wästerlund, 2007), and instances of windthrow and erosion are increased (Nugent et al., 2003). Consolidation (i.e. the expulsion of water when the soil is loaded), caused by clearfelling machinery, also occurs when saturated peats are loaded.

Figure 1.1 Brash mats in use in forestry clearfelling.

Installed brash mats generally remain *in situ* following best management practice (BMP) (Forest Service, 2000) after felling, and can, if timed correctly, fertilise the soil for future crops (Stevens et al., 1995). Peat is considered to be highly vulnerable to a loss in fertility, and the removal of brash material from site following clearfelling can deplete the amount of base cations and reduce available nutrients for the future growth of trees (Forest Research, 2009). Brash can be removed from site once the needle drop period is complete (approximately 6 to 9 months), as up to two-thirds of the total nutrients found in brash material are in the fresh needles (Forest Research, 2009). Nutrient release from decaying brash mats may give rise to excessive nutrient release, which may enter sensitive receiving waters, increasing the eutrophication potential of the aquatic environment. However, the removal of brash material from peat sites may give rise to considerable sediment release to receiving waters.

1.5 Mitigation Measures

An option to mitigate P and SS loss from peat forests to receiving waters is to create RBZs in existing forest stands prior to clearfelling the main coupe behind the buffer zone (Ryder et al., 2011). Current recommended buffer widths in Ireland of 10 – 25 m may not be capable of removing all nutrients from the runoff during high storm events when the majority of the P is transported. This is because the retention time may be too short for uptake of soluble P by vegetation (Rodgers et al., 2010). Vegetation plays an important role in the mitigation of excessive nutrient export to receiving water bodies in forestry clearfelling. Forested buffer zones in the UK have been shown to be successful at allowing sedimentation to occur within a buffer because of a slowing down of the surface runoff due to the well-structured and normally drier character of forest soils (Broadmeadow and Nisbet, 2004) and the increased macroposity from tree roots and soil fauna (Goudie, 2006). This is coupled with the damming effect created by falling debris and protruding roots in the forest buffer, which form sediment traps (Broadmeadow and Nisbet, 2004). Ground vegetation is also an important method of slowing down flow and trapping sediment (Broadmeadow and Nisbet, 2004). It has been noted, however, that there can be a significant time delay in establishing ground vegetation on sites on which brash mats have been left (Broadmeadow and Nisbet, 2004). Ormerod et al. (1993) conducted a study on 11 upland streams in forestry catchments that had been clearfelled from one to seven years prior to their study and noted that the streams had retained some of the characteristics of a forestry catchment stream, even after 7 years of recovery.

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

1.6 Knowledge Gaps and Project Aims

Approximately 60 % of Ireland's forest cover is on peat and the majority of this forestry is now at harvestable age. Similarly, current government policy is to bring the national forest cover to 17 %, or 20,000 hectares per annum, by 2030 (EPA, 2006; Teagasc, 2010). If this target is achieved, 1 % of Ireland's total land cover will be forested every 3 years, which will have a significant impact on the visual and natural environment. It will be imperative to carry out forestry management practices that benefit the environment and fulfil the objectives of the WFD.

There is limited data on the interaction of clearfelling of forests on peatlands with WT fluctuations and shallow ground water nutrient concentrations. It is also relatively unknown which (if any) native deciduous species are likely to survive, if planted in upland peats following clearfelling of coniferous forests. As these trees may be of potential benefit in the mitigation of nutrient loss to receiving waters, empirical data is required for forested western peatlands.

The effects of afforestation on the soil organic carbon (SOC) content (Wellock et al., 2011), carbon (C) stocks and sequestration (Byrne and Milne, 2006) of peatland in Ireland has been studied, but the impacts of deforestation on these areas is relatively unknown. The Environmental Protection Agency (EPA) report on the protocol for sustainable management of peatland in Ireland, BOGLAND (Renou-Wilson et al., 2011), specifically states that research should be carried out in western peatland forests to determine the effects of management options on GHG emissions. The acquisition of such information is vital, due to its role in the overall balance of GHG release nationwide.

To date, there is also little published data on the effects of forest clearfelling on receiving water bodies in Ireland (Rodgers et al., 2011). There is a need to quantify the effects of implementation of BMP (or deviation from BMP) in peatland forestry clearfelling operations on nutrient and sediment release (Coillte, 2008). Little is known about the impact of clearfelling on the pH concentration in upland peat forestry in Ireland. This study aims to provide this knowledge, which may be of potential benefit to the national forestry service, Collite, and the EPA.

Therefore, the objectives of this study were:

1. To investigate the change in soil nutrient concentration in a regenerated buffer zone 5 years after clearfelling has taken place.

2. To access the survival and growth rate of planted saplings in a regenerated buffer zone 5 years after clearfelling took place.

3. To investigate the impacts of forest clearfelling on the hydrology and WT fluctuations of a recently clearfelled site.

4. To investigate GHG emissions from various upland peat sites including (1) a 5-year old riparian regenerated buffer zone (2) a recently clearfelled site (3) a virgin peat site with no forestry activities, and (4) a mature standing forest.

5. To compare the impacts of clearfelling, following best management practice (BMP), on the waters draining an upland blanket peatland forest to a catchment on which no forest operations took place.

1.7 Study Site Description

The study sites were located in the Burrishoole catchment in Co. Mayo, Ireland (ITM reference 495380, 809170) (Figure 1.2). This catchment is situated in the Nephin Beg range and over 31 % of the catchment is fully forested. There is a moderate climate, which is heavily influenced by the proximity of the Atlantic Ocean, with average air temperatures of 13 °C in summer and 4 °C in winter. The catchment receives approximately 2000 mm of rainfall every year. As a result, the area is characterised by upland spate streams and gorged drains. Upland spate streams are very characteristic of peat catchments in the west of Ireland, particularly within the Burrishoole catchment (Allott et al., 2005).

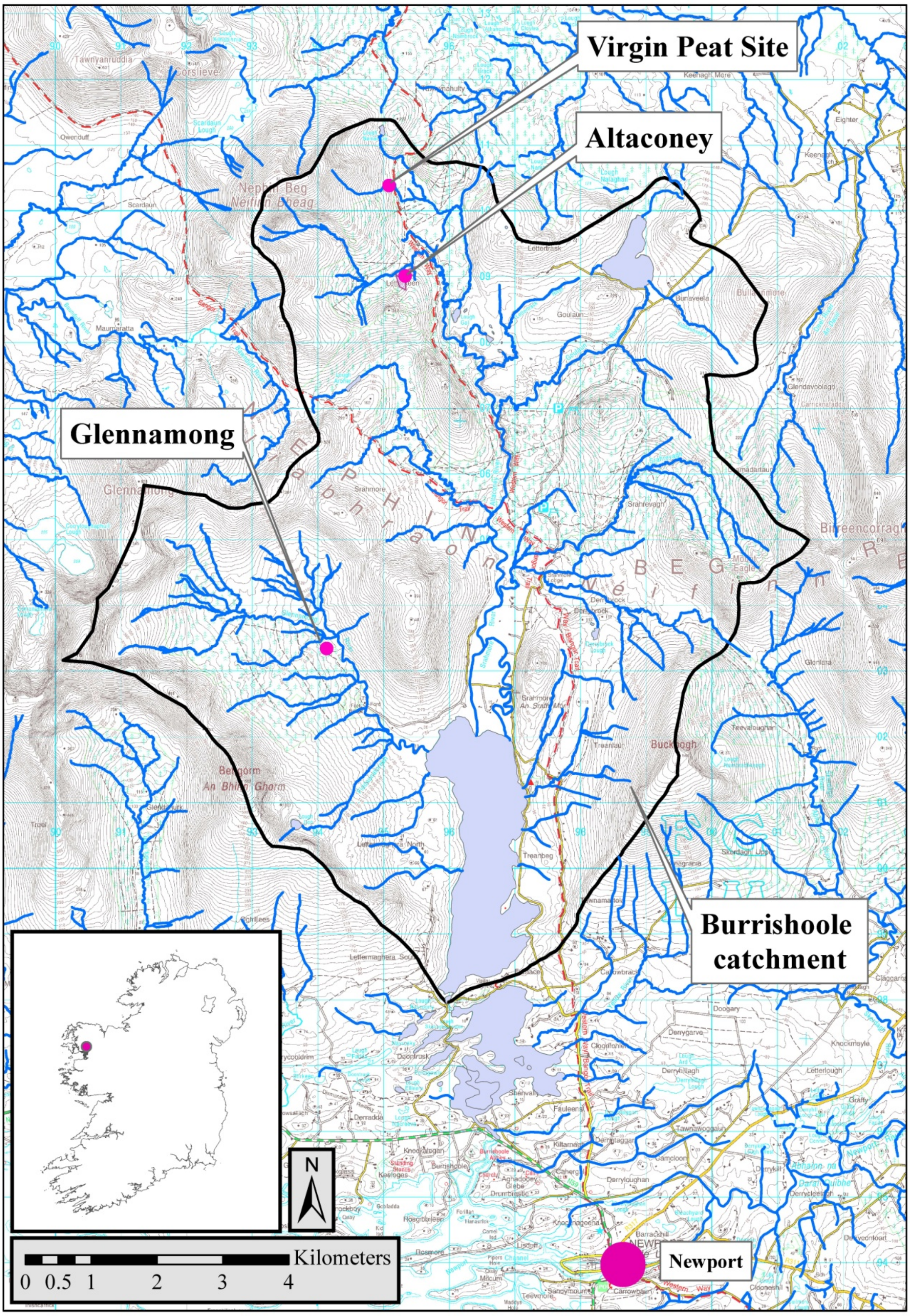

Figure 1.2 Location of the three study sites within the Burrishoole catchment, Co. Mayo, Ireland.

The project focused on three study sites: (1) the Altaconey (also known as Altahoney) forest (2) a virgin peat site, approximately 1.4 km from the Altaconey forest, and (3) the Glennamong forest.

The study site in the Altaconey forest consisted of an area of 2.49 ha 30 m north, 50 m south and 300 m along a stream, which was clearfelled to create a RBZ in May 2006 (Ryder et al., 2011) (Figure 1.3). Research work on this site examined shallow groundwater nutrient concentrations after clearfelling of the RBZ 5 years prior to the present study. This site was also used to investigate planted sapling survival and growth rate, WT fluctuations, nutrient concentrations in the adjacent Altaconey river, and GHG emissions.

Figure 1.3 The Altaconey regenerated buffer zone.

The virgin peat site (Figure 1.4), in close proximity to the Altaconey forest site, was used to investigate the GHG emissions from an upland virgin blanket peat. Soil water content was also measured at the site to elucidate the fluctuations of GHGs.

Figure 1.4 The virgin peat site.

The Glennamong site (Figure 1.5) was a paired catchment study in which one site, the control catchment, was untouched while the adjacent site, the study catchment, was clearfelled in February, 2011. This allowed an investigation of the nutrient, SS, and soil and water quality changes following clearfelling to be quantified against a study control.

Figure 1.5 The Glennamong study site.

1.8 Structure of Dissertation

The PhD thesis structure is as follows:

Chapter 2 comprises a published paper, 'Nutrient dynamics in a peatland forest riparian buffer zone and implications for the establishment of planted saplings' (Ecological Engineering 47: 155 – 164). This paper examines the changes taking place in a riparian regenerated buffer zone 5 years after it has been clearfelled. Analyses include survival and growth rate of planted saplings, shallow groundwater nutrient concentrations, and soil analysis. This chapter addresses the first and second objectives of this study.

In Chapter 3, the impact of brash mats from upland blanket peat forest clearfelling is assessed. This paper, presented at the International Peat Conference 2012, Stockholm, Sweden, investigates the use of brash mats for clearfelling of forestry on peat from an Irish perspective, and focuses on P release to shallow ground waters and water content changes in the peat.

Chapter 4 presents the findings of a paper entitled, 'The effect of management changes on watertable position and nutrients in shallow groundwater in a harvested peatland forest', submitted to Science of the Total Environment. The paper examines the fluctuations in the WT after clearfelling of an upland peat site and the nutrient concentrations found in shallow groundwater for up to 15 months after clearfelling. This chapter addresses the third objective of this study.

Chapter 5 presents the findings of a paper entitled, 'Greenhouse gas emissions from forestry on peatland**',** submitted to Science of the Total Environment. This research looks at the GHG emissions from: (1) a 5-year old riparian regenerated buffer zone (2) a recently clearfelled site (3) a virgin peat site with no forestry activities, and (4) a mature standing forest. This chapter addresses the fourth objective of this study.

Chapter 6 presents the findings of a paper entitled, 'Implications of applied best management practice for peatland forest harvesting', submitted to Forest Ecology and Management. This paper details the effects of applied BMP to a small stream draining a clearfelled site, compared to a control catchment where no forestry operations took place. This chapter addresses the fifth objective of this study.

Finally, Chapter 7 gives the conclusions of the research.

References

Allott, N., P. McGinnity and B. O'Hea (2005). Factors influencing the downstream treansport of sediment in the Lough Feeagh catchment, Burrishoole, Co.Mayo, Ireland. Freshwater Forum: 126-138.

Anon. (2005) The characterisation and analysis of Ireland’s river basin districts In accordance with Section 7(2 & 3) of the European Communities (Water Policy) Regulations 2003 (SI 722 of 2003). National Summary Report (Ireland). (Accessed on 6th September 2012 http://www.wfdireland.ie/Documents/Characterisation%20Report/Ireland_Article_5_WFD.pdf)

Antti, W. (2008). Effect of removal of logging residue on nutrient leaching and nutrient pools in the soil after clearcutting in a Norway spruce stand. Forest Ecol Manag 256(6): 1372-1383.

Ågren, A. and S. Löfgren (2012). pH sensitivity of Swedish forest streams related to catchment characteristics and geographical location – Implications for forest bioenergy harvest and ash return. Forest Ecol Manag 276: 10-23.

Aust, W. M. and C. R. Blinn (2004). Forestry best management practices for timber harvesting and site preparation in the eastern United States: An overview of water quality and productivity research during the past 20 years (1982–2002). Wat. Air Soil Poll. 4(1): 5-36.

Bacon, P. (2003). Forestry: A Growth Industry in Ireland, Wexford. (Accessed 16th June 2010 http://www.coillte.ie/fileadmin/templates/pdfs/BaconReport.pdf)

Bain, C. G., A. Bonn, R. Stoneman, S. Chapman, A. Coupar, M. Evans, B. Gearey, M. Howat, H. Joosten, C. Keenleyside, J. Labadz, R. Lindsay, N. Littlewood, P. Lunt, C. J. Miller, A. Moxey, H. Orr, M. Reed, P. Smith, V. Swales, D. B. A. Thompson, P. S. Thompson, R. Van de Noort, J. D. Wilson and F. Worrall (2011). IUCN UK Commission of Inquiry on Peatlands, IUCN UK Peatland Programme, Edinburgh.

Best, E. P. H. and F. H. H. Jacobs (1997). The influence of raised water table levels on carbon dioxide and methane production in ditch-dissected peat grasslands in the Netherlands. Ecol Eng 8(2): 129-144.

Blodau, C. (2002). Carbon cycling in peatlands - A review of processes and controls. Environ Rev 10(2): 111-134.

Bowman, J. (2009). New Water Framework Directive environmental quality standards and biological and hydromorphological classification systems for surface waters in Ireland. Biology and Environment 109B: 247-260.

Broadmeadow, S. and T. R. Nisbet (2004). The effects of riparian forest management on the freshwater environment: A literature review of best management practice. Hydrol Earth Syst Sci 8(3): 286-305.

Byrne, K. A. and E. P. Farrell (2005). The effect of afforestation on soil carbon dioxide emissions in blanket peatland in Ireland. Forestry 78(3): 217-227.

Byrne, K. A. and R. Milne (2006). Carbon stocks and sequestration in plantation forests in the Republic of Ireland. Forestry 79(4): 361-369.

Carling, P. A., B. J. Irvine, A. Hill and M. Wood (2001). Reducing sediment inputs to Scottish streams: A review of the efficacy of soil conservation practices in upland forestry. Sci Total Environ 265(1-3): 209-227.

Collins, K. D., G. Gallagher, J. J. Gardiner, E. Hendrick and A. McAree (2000). Code of Best Forest Practice - Ireland. Department of Marine and Natural Resources, Dublin.

Coillte, 2008. Technical Final Report for Restoring Active Blanket Bog in Ireland (LIFE02 NAT/IRL/8490). Coillte, Mullingar. (Accesed on 2nd June 2012 http://www.irishbogrestorationproject.ie)

Cummins, T. and E. P. Farrell (2003). Biogeochemical impacts of clearfelling and reforestation on blanket peatland streams I. phosphorus. Forest Ecol Manag 180(1-3): 545-555.

Daniels, S. M., M. G. Evans, C. T. Agnew and T. E. H. Allott (2012). Ammonium release from a blanket peatland into headwater stream systems. Environ Pollut 163: 261-272.

EEA (2004). Revision of the Assessment of Forest Creation and Afforestation in Ireland. Forest Network Newsletter European Environmental Agency's Spatial Analysis Group (Issue 150).

EPA (2006). EPA Viewpoint: Forestry and the Environment. (Accessed on 15th August 2010 http://www.woodlandleague.org/documents/ForestryInIreland/Appendices/4_IrelandsCommitmentsToSFM/documents/Ref16_EPAForestryAndTheEnvironment.pdf)

Eliasson, L. and I. Wästerlund (2007). Effects of slash reinforcement of strip roads on rutting and soil compaction on a moist fine-grained soil. Forest Ecol Manag 252(1-3): 118-123.

Elrashidi, M. A. (2011). Selection of an Appropriate Phosphorus Test for Soils. Soil Survey Laboratory, National Soil Survey Center, USA.

Erwin, K. (2009). Wetlands and global climate change: the role of wetland restoration in a changing world. Wetl Ecol Manag 17(1): 71-84.

FAO (2012). Peatlands – guidance for climate change mitigation by conservation, rehabilitation and sustainable use, Rome, Italy.

Farrell, E. P. and G. Boyle (1990). Peatland forestry in the 1990s. 1. Low-level blanket bog. Irish Forestry 47(2): 69-78.

Fenton, O., K. G. Richards, L. Kirwan, M. I. Khalil and M. G. Healy (2009). Factors affecting nitrate distribution in shallow groundwater under a beef farm in South Eastern Ireland. J Environ Manag 90(10): 3135-3146.

Forest Research (2009). Guidance on site selection for brash removal, The Research Agency of the Forest Commission.

Forest Service (2000). Forest Harvesting and the Environment Guidelines. Department of the Marine and Natural Resources. Dublin. (Accessed on 19th August 2010 http://www.agriculture.gov.ie/media/migration/forestry/publications/harvesting.pdf)

Gerasimov, Y. and V. Katarov (2010). Effect of bogie track and slash reinforcement on sinkage and soil compaction in soft terrains. Croat J For Eng 31(1): 35-45.

Goudie, A. (2006). The human impact on the natural environment: past, present and future, Blackwell Science. 6th Edition, 111 - 112.

Hargreaves, K. J., R. Milne and M. G. R. Cannell (2003). Carbon balance of afforested peatland in Scotland. Forestry 76(3): 299-317.

IPCC (2001). Climate Change 2001: Impacts, Adaptation and Vulnerability, Cambridge University Press, Cambridge, UK.

IPCC (2006). 2006 IPCC Guidelines for National Greenhouse Gas Inventories. IGES, Japan.

IPS (2008). Peatlands and climate change. Finland, International Peat Society.

Jauhiainen, J., A. Hooijer and S. E. Page (2011). Carbon dioxide emissions from an Acacia plantation on peatland in Sumatra, Indonesia. Biogeosciences Discussions 8(4): 8269-8302.

Joosten, H. and D. Clarke (2002). Wise Use Of Mires And Peatlands, A Framework for Decision-making, International Mire Conservation Group & International Peat Society.

Long, M., P. Jennings (2006). Analysis of the peat slide at Pollatomish, County Mayo, Ireland. Landslides 3: 51-61.

May, L., B. O'Hea, M. Lee, M. Dillane and P. McGinnity (2005). Modelling soil erosion and transport in the Burrishoole catchment, Newport, Co. Mayo, Ireland. Freshwater Forum.

Met Eireann (2012). (Accessed 9th September 2012 http://www.met.ie/climate-ireland/climate-of-ireland.asp)

McBroom, M. W., R. S. Beasley, M. Chang and G. G. Ice (2008). Water quality effects of clearcut harvesting and forest fertilization with best management practices. J Environ Qual 37(1): 114-124.

National Forest Inventory (2007). National Forest Inventory Republic of Ireland - Results. Forest Service, Wexford.

Neal, C., B. Reynolds, M. Neal, H. Wickham, L. Hill and B. Williams (1999). The impact of conifer harvesting on stream water quality: the Afon Hafren, mid-Wales. Hydrol. Earth Syst. Sci. 8(3): 503-520.

Nieminen, M. (2003). Effects of clear-cutting and site preparation on water quality from a drained Scots pine mire in southern Finland. Boreal Environ Res 8(1): 53-59.

Nisbet, T., J. Dutch and A. Moffat (1997). Whole-tree harvesting - A guide to good practice. Forestry Commission. Edinburgh, UK, Forestry Commission Practice Guide.

Nugent, C., C. Kanali, P. M. O. Owende, M. Nieuwenhuis and S. Ward (2003). Characteristic site disturbance due to harvesting and extraction machinery traffic on sensitive forest sites with peat soils. Forest Ecol Manag 180(1–3): 85-98.

O’Driscoll, C., M. Rodgers, M. O’Connor, Z.-u.-Z. Asam, E. de Eyto, R. Poole and L. Xiao (2011). A Potential Solution to Mitigate Phosphorus Release Following Clearfelling in Peatland Forest Catchments. Wat Air Soil Pollut: 1-11.

Ormerod, S. J., S. D. Rundle, E. C. Lloyd and A. A. Douglas (1993). The influence of riparian management on the habitat structure and macroinvertebrate communities of upland streams draining plantation forests. J Appl Ecol 30(1): 13-24.

Owende, P. M. O., J. Lyons, R. Haarlaa, A. Peltola, R. Spinelli, J. Molano and S. M. Ward (2002). Operations protocol for eco-efficient wood harvesting on sensitive sites. (Accessed on 3rd September 2010 http://www.ucd.ie/foresteng/.)

Paavilainen, E. and J. Päivänen (1995). Peatland forestry: ecology and principles, Springer.

Petrone, R. M., J. M. Waddington and J. S. Price (2001). Ecosystem scale evapotranspiration and net CO2 exchange from a restored peatland. Hydrol Process 15(14): 2839-2845.

Renou, F. and E. P. Farrell (2005). Reclaiming peatlands for forestry: the Irish Experience. Restoration of Boreal and Temperate Forests. Boca Raton, CRC Press.

Renou-Wilson, F., T. Bolger, C. Bullock, F. Convery, J. Curry, S. Ward, D. Wilson and C. Müller (2011). BOGLAND: Sustainable Management of Peatlands in Ireland. STRIVE 75. EPA., Johnstown Castle, Co. Wexford, Ireland.

Rodgers, M., M. O'Connor, M. G. Healy, C. O'Driscoll, Z.-u.-Z. Asam, M. Nieminen, R. Poole, M. Müller and L. Xiao (2010). Phosphorus release from forest harvesting on an upland blanket peat catchment. Forest Ecol Manag 260(12): 2241-2248.

Rodgers, M., M. O'Connor, M. Robinson, M. Muller, R. Poole and L. Xiao (2011). Suspended solid yield from forest harvesting on upland blanket peat. Hydrol Process 25(2): 207-216.

Ryder, L., E. de Eyto, M. Gormally, M. Sheehy Skeffington, M. Dillane and R. Poole (2011). "Riparian zone creation in established coniferous forests in Irish upland peat catchments: Physical, chemical and biological implications." Biology and Environment 111(1).

Sharpley, A. N., Daniel, T., Sims, T., Lemunyon, T., Stevens, S., Parry, R. (2003). Agricultural Phosphorus and Eutrophication, Agricultural Research Service, United States Department of Agriculture.

Sjörs, H. (1980). Peat on Earth: Multiple use or conservation? Ambio 9(6): 303-308.

Sottocornola, M. and G. Kiely (2010). Energy fluxes and evaporation mechanisms in an Atlantic blanket bog in southwestern Ireland. Water Resour Res 46(11).

Stark, C. H. and K. G. Richards (2008). The continuing challenge of agricultural nitrogen loss to the environment in the context of global change and advancing research. Dynamic Soil, Dynamic Plant 2(1): 1-12.

Stevens, P. A., D. A. Norris, T. G. Williams, S. Hughes, D. W. Durrant, M. A. Anderson, N. S. Weatherley, M. Hornung and C. Woods (1995). Nutrient losses after clearfelling in Beddgelert Forest: A comparison of the effects of conventional and whole-tree harvest on soil water chemistry. Forestry 68(2): 115-131.

Teagasc (2010). A Brief Overview of Forestry in Ireland. (Accessed on 15th August 2010 http://www.teagasc.ie/forestry/technical_info/forestry_history.asp.)

Tierney D (2007). Environmental and social enhancement of forest plantations on western peatlands - a case study. Irish Forestry 64: 5-16.

UNEP, FAO, UNFF (2009). Vital forest graphics, Nairobi.

Väänänen, R., M. Nieminen, M. Vuollekoski, H. Nousiainen, T. Sallantaus, E. S. Tuittila and H. Ilvesniemi (2008). Retention of phosphorus in peatland buffer zones at six forested catchments in southern Finland. Silva Fennica 42(2): 211-231.

Van Beek, C. L., E. W. J. Hummelink, G. L. Velthof and O. Oenema (2004). Denitrification rates in relation to groundwater level in a peat soil under grassland. Bio Fert Soils 39(5): 329-336.

Walker, B. H. (1992). Biodiversity and Ecological Redundancy. Conserv Biol 6(1): 18-23.

Wellock, M. L., B. Reidy, C. M. Laperle, T. Bolger and G. Kiely (2011). Soil organic carbon stocks of afforested peatlands in Ireland. Forestry 84(4): 441-451.

Wilson, D., J. Alm, J. Laine, K. A. Byrne, E. P. Farrell and E. S. Tuittila (2009). Rewetting of cutaway peatlands: Are we re-creating hot spots of methane emissions? Restor Ecol 17(6): 796-806.

Wilson, D. G., E. S. Tuittila, J. Alm, J. Laine, E. P. Farrell and K. A. Byrne (2007). Carbon dioxide dynamics of a restored maritime peatland. Ecoscience 14(1): 71-80.

Chapter 2

The contents of this chapter have been published in Ecological Engineering (2012; 47: 155 - 164). Joanne Finnegan developed the experimental design and collected, analysed and synthesized the experimental data. She is the primary author of this article. Dr. Mark G. Healy contributed to the experimental design and paper writing. Dr. John T. Regan assisted with sampling, the experimental design and paper editing. Dr. Elvira de Eyto assisted with data analysis and paper editing; and Elizabeth Ryder and Dr. Dermot Tiernan assisted with the paper editing.

Nutrient dynamics in a peatland forest riparian buffer zone and implications for the establishment of planted saplings

J. Finnegan[a], J.T. Regan[a], E. De Eyto[b], E. Ryder[c], D. Tiernan[d] and M.G. Healy[a]

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

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

[c] Centre for Freshwater Studies and Department of Applied Sciences, Dundalk Institute of Technology, Dundalk, Co. Louth, Ireland.

[d] Coillte, Cedar House, Moneen Road, Castlebar, Co Mayo, Ireland.

ABSTRACT

Forestry on peatland throughout the world is now focused on minimising destructive effects to the surrounding environment, especially during harvesting. These effects may be mitigated through the use of well-developed riparian buffers zones (RBZs). However, much of the commercial forestry planted in Ireland and the UK in the mid-20th century was planted without adequate RBZs. The creation of new RBZs prior to clearfelling may be a possible mitigation measure in these circumstances. The aim of this paper was to assess the nutrient content and phosphorus (P) adsorption capacity of the soil, and survival of planted saplings in a RBZ, positioned downslope from a standing forest and partly covered with brash mats, five years after its establishment. Dissolved reactive phosphorus (DRP) concentrations were significantly higher under the brash mats in the RBZ when compared to all other areas. The standing forest had the highest concentrations of ammonium nitrogen (NH_4^+–N), while total oxidised nitrogen (TON) was similar for all areas. Water extractable phosphorus and desorption-adsorption testing also confirmed the high concentrations of P under the brash mats, but P did not leach through the peat to the stream. The overall survival rate of the saplings was relatively high, with over half of *Quercus robur* (oak) (57 %), *Sorbus aucuparia* (rowan) (57 %) and *Betula pendula* (birch) (51 %) surviving. *Salix cinerea* (willow) (22 %), *Alnus glutinosa* (alder) (25 %) and *Ilex aquifolium* (holly) (44 %) did not survive as successfully. The RBZ was capable of providing nutrients for the survival of planted saplings, fertilizing the peat with degrading brash material and preventing elevated levels of nutrients entering the adjacent aquatic ecosystem.

2.1 Introduction

Peatlands are found in over 175 countries worldwide, are mostly present in moist temperate climates in the northern hemisphere (Sjörs, 1980), and cover approximately 3 % of the total landmass in the world (4,000,000 km^2) (Bain et al., 2011). These ecosystems produce 10 % of the global freshwater supply and one-third of the world’s soil carbon content (Joosten and Clarke, 2002). Approximately 150,000 km^2 of this landmass has been drained for commercial forestry, while the area not commercially drained, but forested, is unknown (Joosten and Clarke, 2002). Ireland’s forest cover stands at 10.15 %, or 700,000 ha, of the total surface area of the island (National Forest Inventory, 2007). The Irish State, under the management of the Forest Service, carried out the majority of the afforestation in the mid-20th century. This was mainly coniferous plantation on non-productive agricultural land (Bacon, 2003). It is estimated that 59.6 % (417,200 ha) of forestry in Ireland is on peat (National Forest Inventory, 2007) and approximately 300,000 ha of afforestation is on upland peat areas (EEA, 2004; Rodgers et al., 2010). Harvesting of forestry on peat can be challenging due to high soil water contents (gravimetric water contents usually exceed 800 % (Long and Jennings, 2006)), low ground bearing capacities of between 10 and 60 kPa (Owende et al., 2002) and the vulnerable nature of the ecosystem (Forest Research, 2009). In Ireland, forestry harvesting practice (including thinning) minimises soil disturbance by adopting appropriate mitigation measures such as: (1) the use of low ground pressure machines and (2) the laying of brash mats, consisting of small branches and logs under all paths used by the felling and extraction machinery. The scale of soil disturbance to a clearfell site is dependent on a combination of factors, including the number of passes by machinery, soil water content and the effective use of brash mats (Gerasimov and Katarov, 2010).

Forestry on peatland throughout the world is now moving towards a ‘progressive management approach’ (Joosten and Clarke, 2002), which incorporates sustainable timber production alongside multiple uses such as habitat restoration, ecological regeneration and the minimisation of any potentially negative effects to the surrounding environment. These negative effects may include eutrophication (an increase in nutrient levels in a watercourse causing excessive flora growth (Sharpley et al., 2003)), sedimentation (an increase in suspended sediment (SS) release to a watercourse causing damage to water ecology (Rodgers et al., 2011)) and biodiversity loss (a change of species, genetic and ecosystem diversity (Walker, 1992)). Coillte, the Irish State's current forest management company, is certified under the Forest Stewardship Council (FSC) to enforce strict environmental, economic and social criteria for sustainable forest management (Coillte, 2012). This progressive and sustainable management approach includes more effective planning to provide protection to water courses from drainage, fertilisation and afforestation, final harvest and regeneration (Owende et al., 2002). Some of this protection may be provided by riparian buffer zones (RBZs).

The standard forestry practice in Ireland and the UK at the time of afforestation (in the 1950s) led to trees being planted in areas adjacent to water courses with no allowance for a RBZ (Broadmeadow and Nisbet, 2004; Ryder et al., 2011). This lack of a buffer may result in elevated nutrient and SS release into water courses during clearfelling (Carling et al., 2001). Other negative effects in the absence of RBZs are the excessive quantity of shade to the stream provided by the overhanging mature conifer plantations, which leads to a death of the riparian vegetation and leaves the bank sides susceptible to erosion (Broadmeadow and Nisbet, 2004). The presence of commercial conifers close to the edge of a stream is also likely to affect the emergence of invertebrates and the biodiversity in comparison to deciduous trees (Broadmeadow and Nisbet, 2004; Kominoski et al., 2012). Much of the commercial coniferous forestry planted in the 1950s is now at harvesting age and the adoption of current forest practice which creates RBZs will minimise the risk of negative impacts on receiving waters for successive rotations.

Riparian buffer zones are used in forestry worldwide in areas such as Fennoscandia (Syversen and Borch, 2005; Väänänen et al., 2008), the USA and Canada (Aust and Blinn, 2004; Luke et al., 2007), and in New Zealand (Parkyn et al., 2005), to ameliorate the negative impacts of forestry on adjacent water courses. In the UK, forestry planning since the 1990s has allowed for RBZs of native hardwoods to provide shade and shelter for wildlife and the stream inhabitants, and for existing conifer streamside plantations to be felled and restored (Farmer and Nisbet, 2004). Current forest practice in Ireland incorporates the use of buffer zones along waterways, with widths of between 10 m and 25 m depending on slope and soil erodibility (Forest Service, 2000). However, RBZs need to be created in old forest stands on peat soil in the most sustainable method possible.

A RBZ can be created in two ways: (1) by leaving an intact strip of forest adjacent to the stream and clearfelling the main coupe of trees behind it, or (2) by harvesting the trees from a strip beside the stream a number of years prior to clearfelling the main coupe and allowing the area to revegetate, either naturally or artificially (Ryder et al., 2011). Forest buffer zones (option one, with trees left in buffer zone) in the UK have been shown to be successful at allowing sedimentation to occur within the buffer because of a slowing down of the surface runoff due to the well-structured and normally drier character of forest soils (Broadmeadow and Nisbet, 2004) and the increased macroposity from tree roots and soil fauna (Goudie, 2006). This is coupled with the damming effect created by falling debris and protruding roots in the forest buffer, which form sediment traps (Broadmeadow and Nisbet, 2004). However, this option may not be practical in the west of Ireland due to thin soil depths, exposed sites and high winds, leading to the increased chance of wind throw close to the watercourse, resulting in a higher risk of sedimentation and nutrient runoff. The second RBZ creation option has potential to be adopted in Ireland, as it increases the primary production in the stream, provides adequate shade and leaf litter, promotes greater biodiversity and taxon richness, and increases sunlight to the watercourse (Ryder et al., 2011). Ground vegetation is also an important method of slowing down flow and trapping sediment (Broadmeadow and Nisbet, 2004). It has been noted, however, that there can be a significant time delay in establishing ground vegetation on sites on which the logging residues have been left (Broadmeadow and Nisbet, 2004). Ormerod et al. (1993) conducted a study on 11 upland streams in forestry catchments that had been clearfelled from one to seven years prior to their study and noted that the streams had retained some of the characteristics of a forestry catchment stream, even after 7 years of recovery. Ryder et al. (2011) found that the creation of RBZs resulted in increased water discharge and significantly higher SS loads to receiving waters, an elevated stream temperature, and minor changes in the average abundances and taxon richness of macroinvertebrate communities. These effects were consistent with the short-term negative impacts of felling at the time of creation of the RBZs. Nevertheless, the creation of RBZs in this way was not felt to have catastrophic effects on the receiving water course and its inhabitants, and any impacts were short-lived (Ryder et al, 2011).

Coillte's District Strategic Plan 2011 – 2015 specifies a 20 m unplanted strip followed by 10 – 20 m of broadleaf plantation between a permanent water course and conifer forest (Coillte, 2011). This would result in the production of scrub broadleaf cover with a protective function only (Coillte, 2011). The tree species planted in a RBZ are generally recommended to be the native variety and species choice will have an impact on the efficiency of the buffer (Broadmeadow and Nisbet, 2004). Factors such as shade and canopy density need to be taken into consideration, as RBZs are seen to function more efficiently when there is a high level of ground covering plants (Broadmeadow and Nisbet, 2004). Dense planting of species with larger leaf areas, like *Alnus* (alder) or *Quercus* (oak), may provide too much shade for the successful growth of the lower ground covering plants and it is recommended that they are not planted in large groups, but rather dispersed throughout the RBZ with species with lower canopy density such as *Salix* (willow), *Betula (*birch) and *Sorbus* (rowan) (Broadmeadow and Nisbet, 2004). Alder is also suspected of adding to stream acidification due to its ability to fix nitrogen (N) from the atmosphere and, therefore, should be limited in RBZ regeneration projects (Broadmeadow and Nisbet, 2004). It is relatively unknown which (if any) native deciduous species are likely to survive, if planted in upland peats following clearfelling of coniferous forest.

Due to the upland nature of these areas, many of these catchments include headwater streams, which are important salmonid habitats and need to be protected from nutrient enrichment. The phosphorus (P) retention capacity of a soil is partly dependant on its abundance of aluminium (Al) and iron (Fe) compounds (Giesler et al., 2005; Väänänen et al., 2006). Aluminium and Fe are readily available in mineral soil, but are lacking in peat. However, as the mineral layers, where they occur, in riparian peatland buffers aid in retaining higher quantities of P than peat further back from the riparian zone (Väänänen et al., 2006), one option to mitigate P loss from peat forests to receiving waters is to create RBZs in existing forest stands prior to clearfelling the main coupe behind the buffer zone (Ryder et al., 2011). Desorption-adsorption isotherms can indicate the amount of P retained in the soil and show the adsorption properties of the soil, while water extractable phosphorus (WEP) testing measures the readily available fraction of the soil P and is used as an indicator of the amount of P that may be carried from a soil by surface runoff in storm events. Current recommended buffer widths in Ireland of 10 – 25 m may not be capable of removing all nutrients from the runoff during high storm events when the majority of the P is transported, as the retention time may be too short for uptake of soluble P by vegetation (Rodgers et al., 2010). It has been shown that elevated levels of nutrients and sediment are frequently associated with clearfelling operations for up to 4 years (Cummins and Farrell, 2003; Rodgers et al., 2010; Rodgers et al., 2011). Although P can become fixed in the soil and only a small amount may be leached to water courses (Haygarth et al., 1998), even small concentrations (> 35 $\mu g\ L^{-1}$ molydbate reactive phosphorus (MRP)) can have a negative impact on water quality (Bowman, 2009), leading to restrictions of use for fisheries, recreation, industry and drinking water (Elrashidi, 2011; Sharpley et al., 2003). Phosphorus can be found in both dissolved and sediment-bound (minerals and organic matter) forms. Dissolved P is bio-available and is therefore the main cause of eutrophication in freshwater (Elrashidi., 2011; Regan et al., 2010; Sharpley et al., 2003; Väänänen et al., 2006). In Ireland, P is the limiting nutrient for eutrophication (Hutton et al., 2008) and is therefore the nutrient of greatest interest. The limit for MRP, which is similar to dissolved reactive phosphorus (DRP) (Haygarth et al., 1997), concentrations in Irish rivers to maintain 'good ecological status' is 35 $\mu g\ L^{-1}$ and for 'high ecological status' is 25 $\mu g\ L^{-1}$ (Bowman, 2009). A conservative value of 30 $\mu g\ L^{-1}$ has been statistically linked with lower biological Q ratings (biological quality ratings) (EPA, 2005), phytoplankton production (Daniel et al., 1998) and increased algal growth in freshwaters (Haygarth et al., 2005).

The aim of this study was to examine the characteristics of an uncultivated RBZ in an upland peat area in the west of Ireland. The RBZ was clearfelled 5 years previous to the present study and restocked 1 year later with group planted broadleaf species. Specifically, the following characteristics were examined in the 5 year old RBZ: (1) the deciduous species of trees which were able to survive and thrive (2) the soil and surface water nutrient content, and (3) the P adsorbing capacity of the soil in the regenerated zone and in the standing forest. This allowed for an assessment of the function and performance of RBZs to supply nutrients to various native species of growing saplings within peatland forestry, and to provide some protection against nutrient export into receiving waters.

2.2 Materials and Methods

2.2.1 Study site description

The study site was located in the Altaconey (also known as Altahoney) forest in the Burrishoole catchment in Co. Mayo, Ireland (ITM reference 495380, 809170) (Figure 2.1). This catchment is situated in the Nephin Beg range at an approximate elevation of 135 m above sea level. The study stream is a third-order stream (Strahler, 1957) and is located within a subcatchment area of 416.2 ha, of which 176.4 ha is fully forested (Ryder et al., 2011). The site has a north-westerly aspect, while the study stream, which is one of the main tributaries to the Altaconey River, flows in a southwest-to-northeast direction to the north of the site before turning south to join the Altaconey River. There is a moderate climate, which is heavily influenced by the proximity of the Atlantic Ocean, with average air temperatures of 13 °C in summer and 4 °C in winter. The site is subjected to approximately 2400 mm of rainfall every year, with 289 rain days between May 2010 and April 2011. As a result, the area is characterised by upland spate streams and gorged drains. The Altaconey river responds quickly to rainfall events, and discharge frequency curves are characterised by steep amplitudes and extremely fast falling crests, with 75 % of the total runoff in the Altaconey river originating from direct runoff (Muller, 2000). Upland spate streams are very characteristic of peat catchments in the west of Ireland, particularly within the

Burrishoole catchment (Allott et al., 2005). A storm on July 15, 2010 with an intensity of 35 mm hr^{-1} raised the level of the river (Figure 2.2), but the water level reduced back to pre-storm levels within a few days (Figure 2.3).

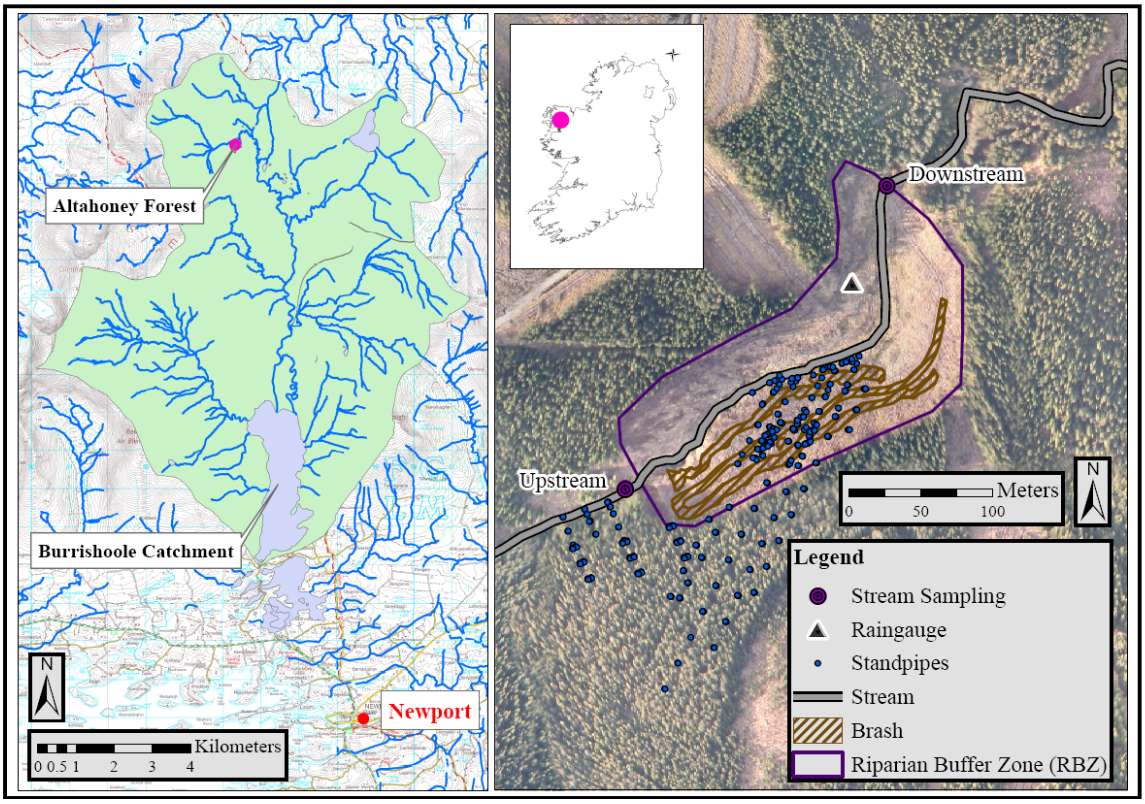

Figure 2.1 Location of Altaconey Riparian Buffer Zone (RBZ) with all piezometers (20, 50 and 100 cm depths), stream sampling locations upstream and downstream of buffer, and rain gauge.

Figure 2.2 River on July 15, 2010.

Figure 2.3 River on July 22, 2010.

The average slope across the buffer zone is 5 % and this increases to 35 % within 10 m of the stream, while the slope down the stream bed is approximately 2.5 %. The stream bed consists of boulders and gravel, while mineral-rich peat is evident along the banks and slopes adjacent to the water course. Blanket peat of varying depth down to 2 m covers the site, which overlays a sand and gravel layer on top of the Cullydoo formation of Srahmore quartzite and schist bedrock (McConnell and Gatley, 2006). This blanket peat is an *in situ* blanket mire with an average gravimetric water content of 85 %, dry bulk density of approximately 0.1 g cm^{-3} and a mineral content of approximately 3 %. Bedrock does not protrude the surface of peat and the minimum peat depth is 0.3 m. Closer to the stream, the mineral-rich peat is at a shallower depth of less than 1 m, has an average gravimetric water content of 35 %, a dry bulk density of approximately 1 g cm^{-3}, and a mineral content of approximately 95 %. During the course of the study, the RBZ had a yearly average water table depth of 0.17 m, while the average water table depth in the standing forest was 0.42 m.

The site was planted in 1966 with Sitka Spruce (*Picea sitchensis*) and Lodgepole Pine (*Pinus contorta*). In May 2006, an area of 2.49 ha 30 m north, 50 m south and 300 m along the stream was clearfelled to create the RBZ (Ryder et al., 2011) (Figure 2.1). This is wider than the current buffer width recommendation of 10 – 25 m. In line with best management practice (BMP), brash mats were used to prevent soil damage by the heavy logging machinery. These mats were created by the harvester, which laid the logging residues of branches and un-merchantable logs in front of the harvester in continuous, slope-dependant strips on which it travelled as it felled the trees. These were left *in situ* on completion of clearfelling. Typical forest practice would normally be to windrow these brash mats into regular rows away from the watercourse when preparing the site for replanting. The direction and position of the brash mats on the southern side of the RBZ are shown in Figure 2.1. No rutting due to brash mat use was noted on site.

In April 2007, one year after felling, the area was replanted with native broadleaved tree species from Coillte nurseries, including *Ilex aquifolium* (holly), *Sorbus aucuparia* (rowan), *Alnus glutinosa* (common alder), *Salix cinerea* (grey willow), *Betula pendula* (common birch) and *Quercus robur* (oak pedunculate). These saplings were all containerised and of varying height ranges: (1) 0.4 – 0.8 m (birch, rowan and willow) (2) 0.3 – 0.5 m (oak and alder) (3) 0.1 – 0.2 m (holly). All saplings were 2 years old, except the birch, which was 3 years old. No fertilizer was applied and the area was not cultivated, but the saplings were pre-treated by dipping in Dimethoate (pyrethroid insecticide) to protect them against the pine weevil (*Hylobus abietis*) (Ryder et al., 2011). This planting was not intended to be productive commercial forestry (expected survival rates of > 90 % after 4 years), but aimed to examine which species of trees had the potential to establish and survive in a hostile peatland environment. The perimeter of the created buffer zone was then fenced off to protect it from grazing by sheep and wild animals, not including deer, as a sufficiently high (exceeding 2.1 m) fence was not installed.

2.2.2. Vegetation

A detailed description of the location and composition of the sapling planting regime in April 2007, post clearfelling, was conducted by Ryder et al. (2011) (Table 2.1). Thirty-three plots in total, 20 on the southern side and 13 on the northern side of the stream, were planted in a 2 x 2 m block pattern with a red stake placed in the centre of the plot for identification (Figure 2.4). Nine trees per plot were planted, totalling 297 saplings of various tree species across the site (birch, alder, rowan, willow, holly and oak; Figure 2.5). No planting took place within 5 m of the stream. In August 2011, as part of the present study, a survey was carried out to determine the percentage survival and increase in height of the surviving saplings. An increase in height was measured as the percentage change from the average original height (obtained from Coillte Nurseries, pers. comm.) to the measured height on site in August 2011. For example, no change in height was denoted as a 0 % change, while an increase in height from 0.4 m to 1 m (a change in height of 0.6 m) was given a 150 % increase.

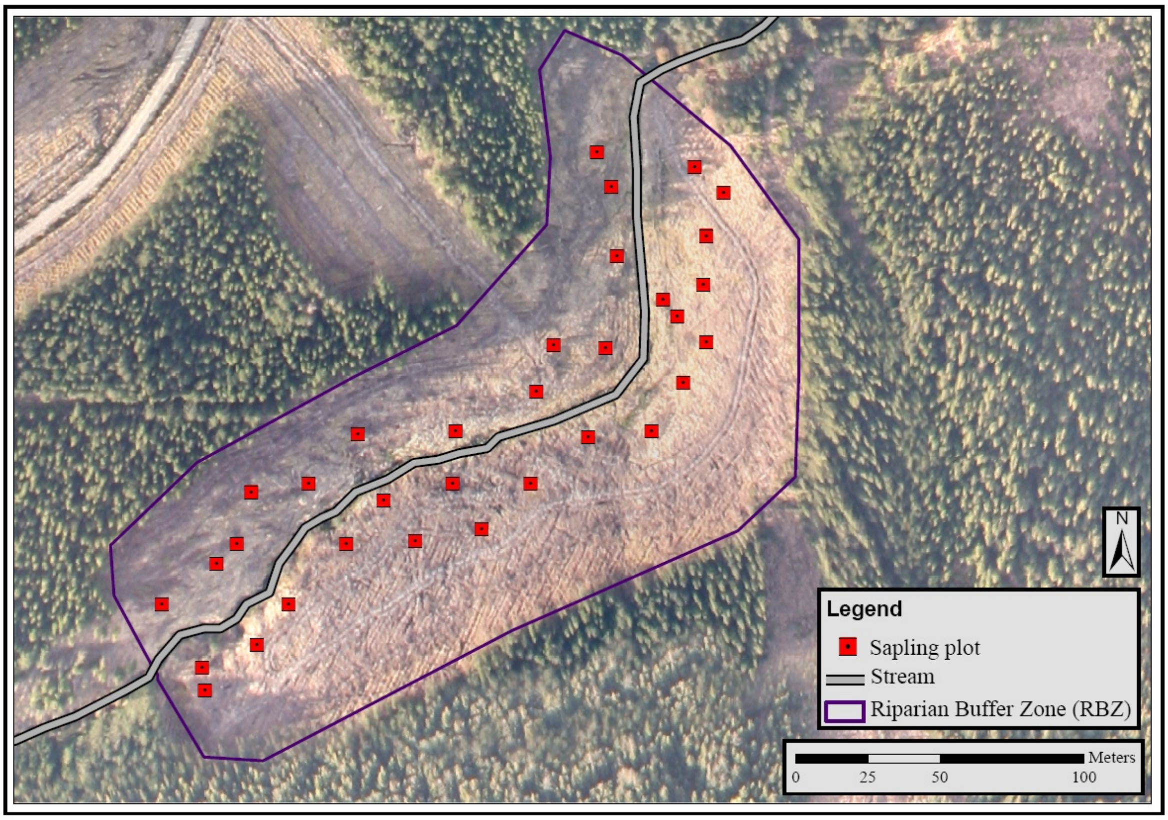

Figure 2.4 Sapling plot planting locations.

Birch *(Betula pendula)*

Alder *(Alnus glutinosa)*

Rowan *(Sorbus aucuparia)*

Willow *(Salix cinerea)*

Holly (*Ilex aquifolium)*

Oak *(Quercus robur)*

Figure 2.5 Sapling species planted in the Altaconey RBZ.

Table 2.1 Description of the location and composition of the sapling planting regime in April 2007, post clearfelling and surviving trees in August 2011.

Plot No.	April 2007 Birch	Oak	Rowan	Willow	Holly	Alder	August 2011 Birch	Oak	Rowan	Willow	Holly	Alder
1	9	-	-	-	-	-	7	-	-	-	-	-
2	-	9	-	-	-	-	-	8	-	-	-	-
3	-	9	-	-	-	-	-	5	-	-	-	-
4	9	-	-	-	-	-	6	-	-	-	-	-
5	-	-	9	-	-	-	-	-	3	-	-	-
6	-	9	-	-	-	-	-	0	-	-	-	-
7	-	9	-	-	-	-	-	8	-	-	-	-
8	9	-	-	-	-	-	2	-	-	-	-	-
9	-	9	-	-	-	-	-	2	-	-	-	-
10	-	9	-	-	-	-	-	4	-	-	-	-
11	9	-	-	-	-	-	9	-	-	-	-	-
12	-	9	-	-	-	-	-	9	-	-	-	-
13	6	-	3	-	-	-	3	-	1	-	-	-
14	9	-	-	-	-	-	3	-	-	-	-	-
15	6	-	3	-	-	-	6	-	1	-	-	-
16	6	-	3	-	-	-	2	-	2	-	-	-
17	-	-	9	-	-	-		-	4	-	-	-
18	9	-	-	-	-	-	9	-	-	-	-	-
19	6	-	3	-	-	-	4	-	2	-	-	-
20	6	-	3	-	-	-	4	-	3	-	-	-
21	9	-	-	-	-	-	3	-	-	-	-	-
22	6	-	-	-	-	3	3	-	-	-	-	1
23	6	-	-	-	-	3	1	-	-	-	-	1
24	-	-	-	9	-	-	-	-	-	2	-	-
25	9	-	-	-	-	-	6	-	-	-	-	-
26	-	-	9	-	-	-	-	-	2	-	-	-
27	6	-	-	-	-	3	2	-	-	-	-	1
28	6	-	-	-	-	3	2	-	-	-	-	-
29	6	-	3	-	-	-	-	-	1	-	-	-
30	-	-	6	-	3	-	-	-	5	-	1	-
31	-	-	6	-	3	-	-	-	6	-	2	-
32	-	-	6	-	3	-	-	-	6	-	1	-
33	9	-	-	-	-	-	0	-	-	-	-	-
Total	**141**	**63**	**63**	**9**	**9**	**12**	**72**	**36**	**36**	**2**	**4**	**3**
					% Survival		**51%**	**57%**	**57%**	**22%**	**44%**	**25%**

2.2.3. Water analysis

Shallow groundwater and surface water samples were collected throughout the site and upstream and downstream of the buffer (Figure 2.1), mainly during storm events from April 2010 to April 2011 (n=5 dates) to examine the movement and concentration of nutrients in the peat and surface runoff. Sampling was focused on the RBZ, but also included the adjacent mature standing forest to allow comparison with the original condition of the buffer prior to clearfelling in 2006. All samples were grouped under four specific locations: (1) 1 m from the stream (within the RBZ) (2) under brash mats (within the RBZ) (3) under the vegetated areas (within the RBZ but not under brash mats) and (4) the standing forest. The standing forest to the left of the RBZ (in Figure 2.1) is at the same topographical location as the RBZ, has a similar slope and peat depth, and has similar mineral-rich peat near the stream. The direction of groundwater flow on site was perpendicular to the stream and the brash mats. Therefore, any vegetated areas within the RBZ, which had no brash directly on them, were still influenced by the decaying brash material.

Piezometers were installed on site for shallow groundwater quality measurement and their locations are illustrated in Figure 2.1. Each sampling location comprised a cluster of 3 sampling piezometers positioned at 20 cm, 50 cm, and 100 cm depths below the soil surface. Each piezometer consisted of a qualpex pipe with an internal diameter of 1.1 cm. Holes were drilled in the lower 10 cm of the piezometer and this was covered with filter sock. A steel rod was inserted into the piezometer for support, as it was hand-pushed into the peat. The top of the piezometers were covered to prevent the ingress of rain water. Any water lodged in the bottom of the piezometer was removed under suction the day before water sampling and the piezometer was allowed to fill overnight. Once extracted from the piezometer, water samples were filtered on site using 0.45 µm filters.

All water samples were returned to the laboratory and tested the following day or frozen for testing at a later date. The water quality parameters measured were: (1) DRP (2) ammonium-N (NH_4^+-N) (3) nitrate-N (NO_3^--N) and (4) total oxidised nitrogen

(TON; NO_3^--N + nitrite-N (NO_2^--N)). All water samples were tested in accordance with standard methods (APHA, 2005) using a nutrient analyser (Konelab 20; Thermo Clinical Labsystems, Finland). Nutrient data were $\log_{10}$ transformed and analysed with ANOVA (analysis of variance) in Datadesk (Data Description Inc., USA), to ascertain the main sources of variation. Date, depth of soil where the sample was taken and the location of the sample site were included as explanatory variables.

Inverse distance weighted (IDW) analysis was carried out on the study area using ArcGIS (Release Version 9.3, Environmental Systems Research Institute (ERSI), California, USA) to show the concentrations of nutrients under the decaying brash mats. Inverse distance weighted analysis is a geospatial analytical tool which interpolates between sampling points, giving a greater weight to values closest to the cell value being interpolated. A 'halo' effect on individual piezometers can be caused where very high concentrations are in close proximity to lower concentrations, giving a shorter distance for interpolation between the points.

2.2.4. Soil analysis

Water extractable phosphorus and desorption-adsorption isotherm testing were carried out on samples of the soil from the RBZ and the adjacent mature standing forest. For both tests, a series of sampling points were selected in three transects parallel to the stream in the RBZ at the following locations: (1) 1 m from the stream (n=10) (2) under the brash mat approximately 35 m from the stream (n=20), and (3) under a vegetated area in-between brash mats approximately 45 m from the stream (n=20). Soil samples (n=10) were also collected from the mature standing forest to represent the contributing area. To select the sampling locations, a grid was laid out on the standing forest, and soil samples were extracted at random locations on the grid. Samples were extracted with a 30 mm-diameter gouge auger after clearing of the Oi horizon, litter layer, which was mainly composed of degrading moss and needles. Väänänen et al. (2007) found that this layer had the lowest P retention capacity and it was therefore omitted from testing in the present study. Samples were then placed in sealed bags on site and were homogenized by hand in the laboratory.

For WEP tests, samples were collected in spring 2011 at two depths, 0 – 0.1 m and 0.1 – 0.5 m, along each transect. Sub-samples of peat (n=5 from each depth), equivalent to 1 g dry weight, were mixed with 30 ml of deionized water and shaken for 30 min at 225 rpm using a rotary shaker after Rodgers et al. (2010). The filtered supernatant water (filtered with 0.45 µm filters) was tested using a nutrient analyser (Konelab 20; Thermo Clinical Labsystems, Finland). The remaining soil sample was used to determine the gravimetric water contents.

For desorption-adsorption isotherm testing, the soil samples were collected at depth increments of 0 – 0.15 m and 0.15 – 0.30 m below the soil surface. Phosphorus solutions were made up to concentrations of 0, 0.2, 0.5, 1, 2, 3.5 and 10 mg P L^{-1}. Sub-samples of peat (n=3 for each depth and each concentration), equivalent to 1 g dry weight, were mixed with 40 ml of the P solution and shaken for 1 hr at 180 rpm using a rotary shaker. The 10 mg P L^{-1} solution was only used for the samples collected 1 m from the stream due to the high mineral content. The solutions were then allowed to stand for 23 hr before being placed in the shaker again for 5 min at 120 rpm after Väänänen et al. (2008). The filtered (0.45 µm) supernatant water was tested using a nutrient analyser (Konelab 20; Thermo Clinical Labsystems, Finland). The remaining soil sample was used to determine the gravimetric water contents and the mineral content, which was determined by loss on ignition (LOI) at 550 °C (BSI, 1990).

2.3. Results and Discussion

2.3.1 Vegetation

The overall survival rate of the planted saplings by 2011 was relatively high, given that the site was not cultivated, no fertiliser was applied and broadleaves were planted in an environment hostile for their survival. Over half of oak (57 %), rowan (57 %) and birch (51 %) saplings survived, suggesting that adequate nutrients were available in the RBZ. However, the survival of willow (22 %), alder (25 %) and holly (44 %) was not as successful, but evidence of surviving saplings was found in some plots. The initial planting rate of the willow, alder and holly saplings was not as high (9, 9 and 12 saplings, respectively) in comparison to the oak, rowan and birch saplings (63, 63 and 141 saplings, respectively). This was based on available saplings at the time of planting and could not be altered by our experimental design. For saplings of an original height and nature similar to that of this study, some degree of protection and care should be afforded to ensure growth (M. Sheehy Skeffington, pers. comm.) and vegetation control is vital for a plant to thrive (Renou-Wilson et al., 2008). However, maintenance and grass removal is generally not performed on saplings planted in peat areas. The study area was fenced off to protect the saplings from grazing (except deer who could possibly jump the fence), but grass and other shading species were not removed from around the saplings. *Sphagnum* spp., *Erica* spp. and *Calluna* spp had naturally regenerated in the RBZ, and were likely to be competing aggressively with the planted saplings for both nutrients and light. The naturally regenerated native vegetation may also have had a large influence on the uptake of nutrients from the RBZ, but this was not quantified in this experiment. It is not known if naturally regenerated ground vegetation alone would suffice for excessive nutrient uptake if the native saplings were not planted in the RBZ. Further research should be carried out to quantify the need for broadleaf plantation on clearfelled RBZs for nutrient uptake.

Figure 2.6 shows the percentage increase in the height of the surviving saplings from the various tree species. Even though 57 % of oak survived after 5 years on site, there was very little growth in the saplings (26 %). In comparison, the 51 % of birch that survived had a percentage increase in height of 70 %. Similarly, Renou-Willson (2008) found that oak seedlings only grew 13.9 cm in 3 years where mounding was employed on a cutaway peatland in the Irish midlands, whereas native birch was reported to grow up to 50 cm per year in the same location (Renou et al., 2007). In the present study, *Calluna vulgaris* was very evident on the site and was close to the oak plots; this may have negatively impacted on the survival and growth of oak. Frost et al. (1997) noted that competition from grass turf led to a significant increase in the seedling mortality of oak species such as *Quercus robur* and observed that *C. vulgaris* had a negative impact on the growth of seedlings of *Q. petraea* due to the hindering of the mycorrhiza development (the mutually beneficial relationship between the fungus and the roots of the plant). Rowan had the greatest increase in height over all other species, with a growth of 73 % in surviving saplings. The willow that survived experienced no growth over the study period (0 %) with the majority of the plants barely progressing beyond seedling stage. Of the 12 alder plants that were originally planted, 3 survived, and only one of these 3 surviving plants experienced growth during the 5 year study period. Holly, at approximately 0.15 m, was the smallest in size to be originally planted and the surviving holly saplings (44 %) saw an overall growth of 22 %.

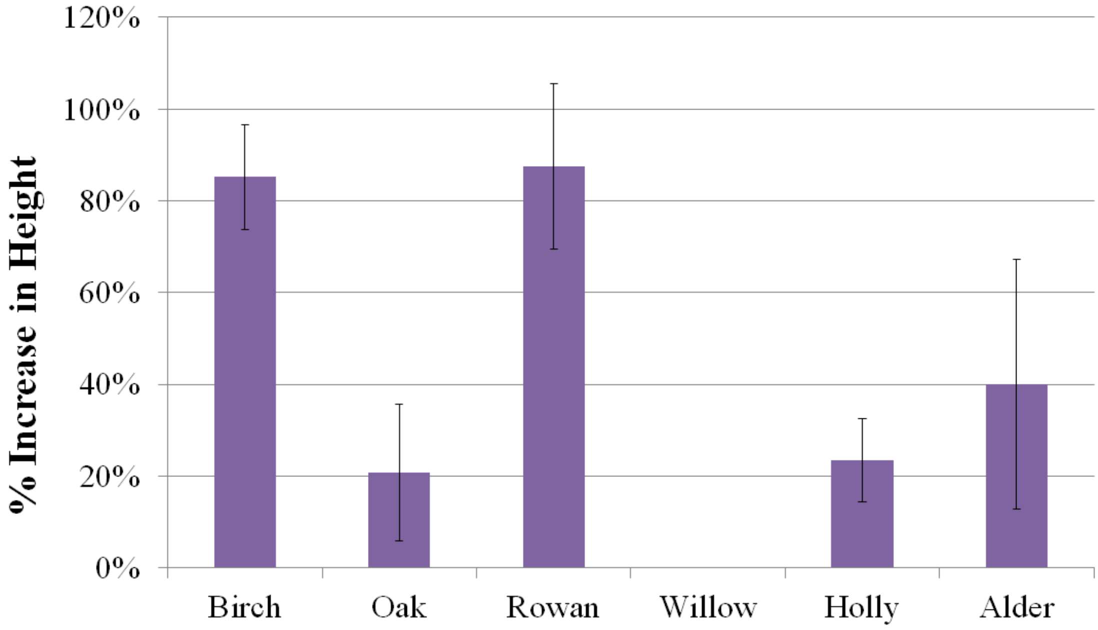

Figure 2.6 Percentage average increase in height of surviving saplings on site from April 2007 to August 2011 per tree species. Error bars indicate standard error.

In the west of Ireland, natural broadleaf regeneration is rare and is largely limited to river banks of higher mineral content than the surrounding infertile peatland (Conaghan, 2007, unpublished report). No planting took place within 5 m of the stream, but it was noted that there was a number of large trees (> 2 m high) of various species (oak, rowan and holly) on the banks of the stream, which were not planted during the saplings' planting regime in 2007. Conaghan (2007, unpublished report) observed that growth of broadleaved trees is more successful in previously forested areas with a peat depth of less than 1 m, and also noted that sites with better drainage and shelter fostered a better environment for the growth and survival of transplants. The exposed nature of the Altaconey site and the depth of peat may have adversely affected the growth rate of the willow, alder and holly species. Conaghan (2007, unpublished report) also observed that the survival of willow cuttings placed directly into the peat was low, but larger willow transplants, which were grown for a time prior to transplanting in more fertile soil, did thrive. In the present study, the height of the willow saplings planted in April 2007 was approximately 60 cm, but this did not appear to survive (22 %) or grow (0 %) very well. Overall, the survival rate of the planted saplings was relatively high, taking into consideration the lack of cultivation, artificial fertilisation and maintenance, possibly grazing by deer and late planting of the saplings. A greater growth rate may have been obtained if any of these management techniques had been employed in the early stages of establishment.

2.3.2. Water analysis

Date and depth were not significant sources of variation for any of the nutrient data, while the location of the sampling site was significant for both DRP and NH_4^+-N (ANOVA, $p<0.05$). Dissolved reactive phosphorus was significantly higher under brash in the RBZ and was significantly lower close to the river ($p<0.05$, LSD post-hoc test) (Figure 2.7). Ammonium-nitrogen in the standing forest was significantly higher than that recorded under brash or in the vegetated parts of the RBZ ($p<0.05$, LSD post-hoc test). Levels of TON were generally similar across all locations sampled (Figure 2.7). Surface water and shallow groundwater concentrations of NO_3^--N were all low and were, in many cases, below the limits of detection with a maximum concentration of 20 μg L^{-1} (results not shown).

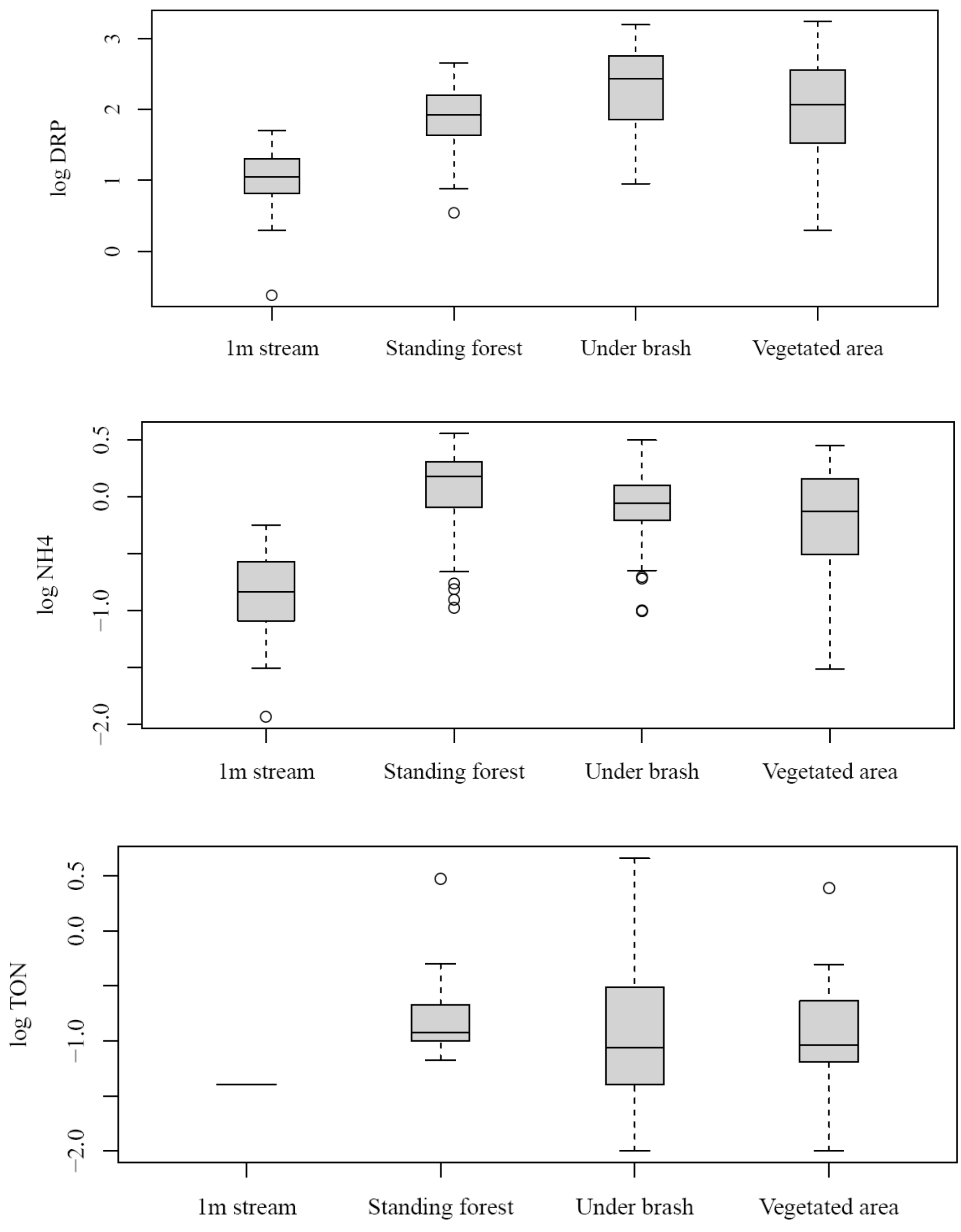

Figure 2.7 Box Plots of dissolved reactive phosphorus (DRP) (top), ammonium-N (NH_4^+-N) (middle) and total oxidized nitrogen (TON) (bottom) for regenerated buffer area (1 m from the stream, under brash mats and under the vegetated area) and standing forest at 20 cm, 50 cm and 100 cm depths from April 2010 – April 2011. All units are µg L^{-1}.

Inverse Distance Weighted images, generated from the shallow groundwater DRP and NH_4 concentrations, show the comparison of the RBZ with the standing forest (Figure 2.8 and 2.9). This illustrates the higher P concentration under the decaying brash mats in the RBZ, which were left on site 5 years before the present study and the higher N concentration in the SF. The DRP concentration reduced close to the stream edge due to the adsorption capacity of the mineral-rich peat near the stream (discussed in Section 2.3.3).

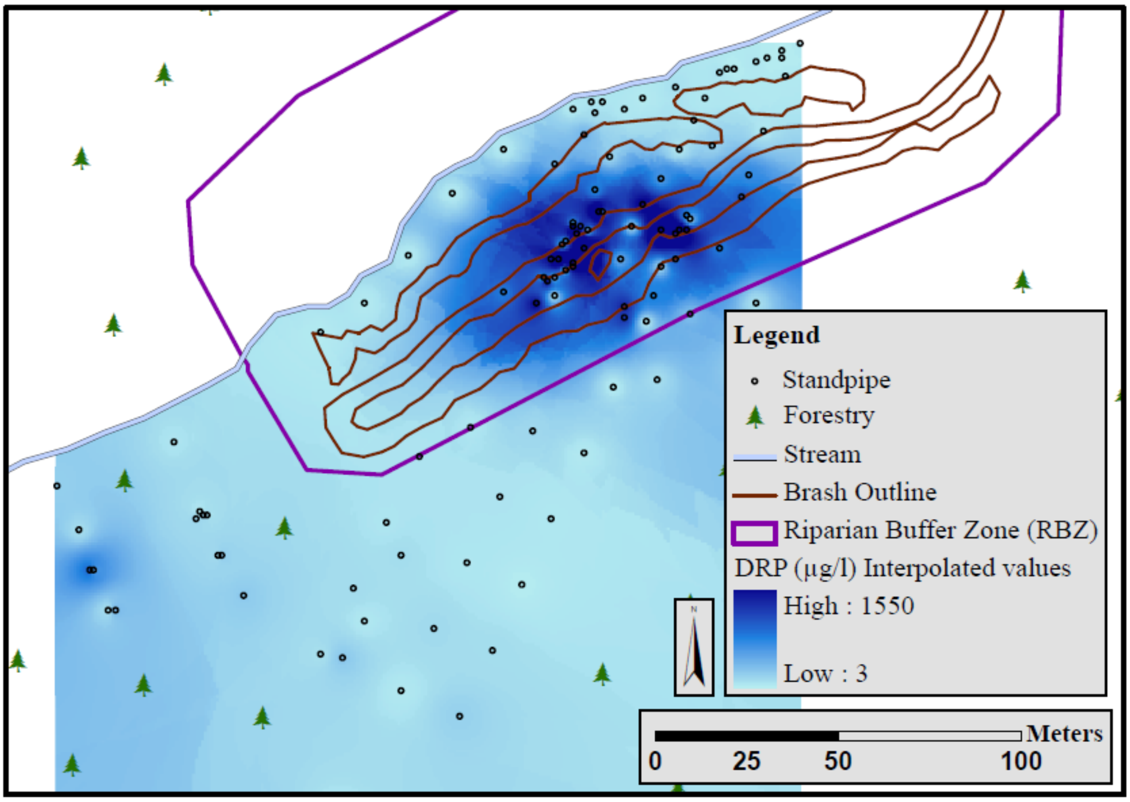

Figure 2.8 Average dissolved reactive phosphorus (DRP) concentration from 20 cm, 50 cm and 100 cm depths below the ground surface measured over a 12 month period (April 2010 – April 2011) and expressed as µg L^{-1}.

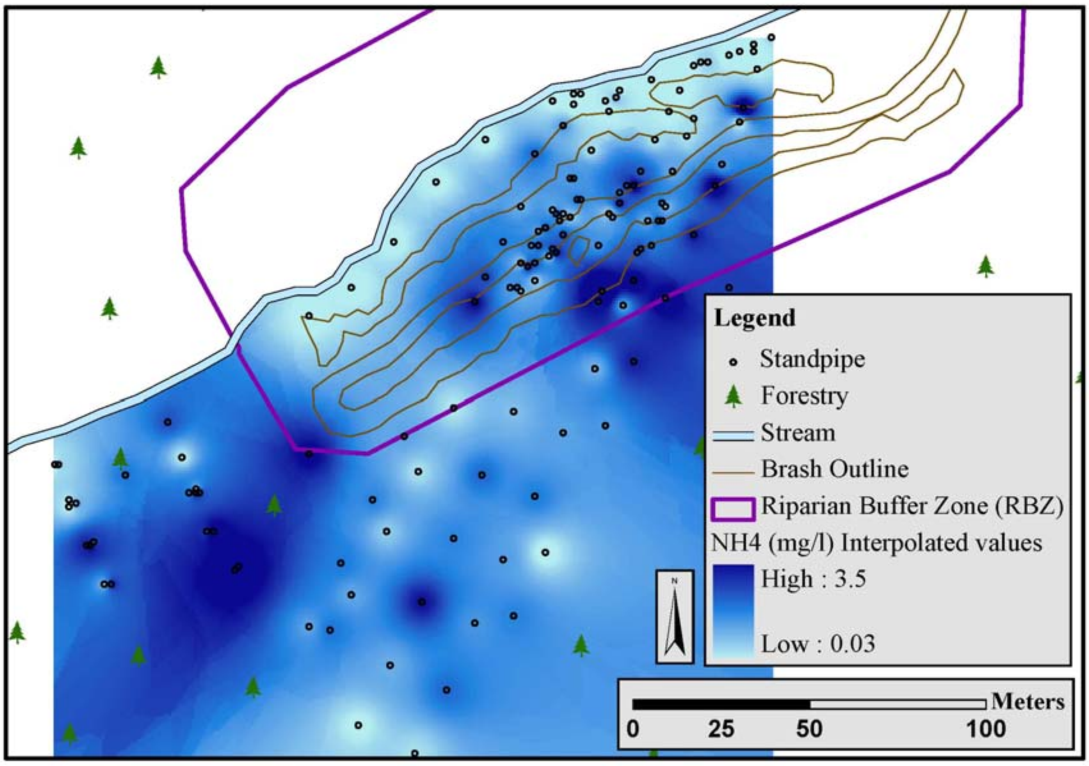

Figure 2.9 Average ammonium (NH_4) concentration from 20 cm, 50 cm and 100 cm depths below the ground surface measured over a 12 month period (April 2010 – April 2011) and expressed as mg L^{-1}.

The high concentrations of DRP in the shallow groundwater under the brash mats and surrounding areas did not leach to the stream, as analysis of stream water upstream and downstream of the buffer showed that it remained at between 4 – 10 µg L^{-1} (Figure 2.10). These values represent no change from buffer creation (May 2006 - January 2007) (Ryder et al., unpublished report), when MRP concentrations were approximately 5 µg L^{-1} in the stream. This is similar to the concentration in the rain water in the area, which was approximately 6 µg L^{-1} (data not shown; figure based on 5 random rain samples analysed by the authors over the study period). However, due to the lack of continuous data of nutrients in the stream, large episodic pluses of P release could have been missed in this stream analysis. Also, the area of the BZB (2.4 ha) is very small in comparison to the over catchment of the Altaconey stream (416.2 ha) and therefore changes may be imperceptible.

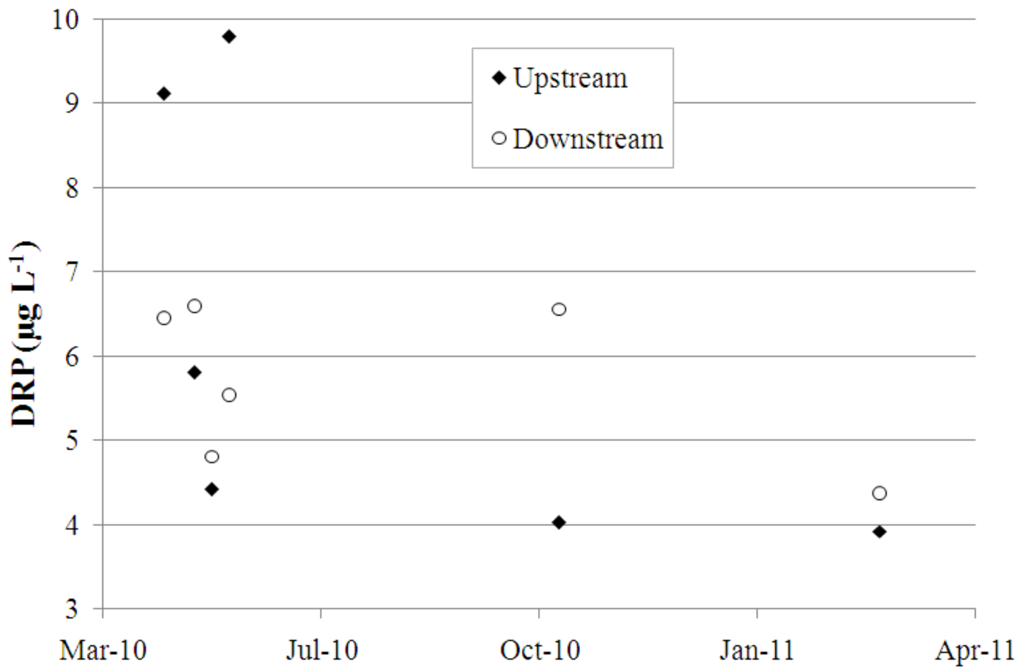

Figure 2.10 Dissolved reactive phosphorus (DRP) concentration measured over a 12 month period (April 2010 – April 2011) and expressed as µg L^{-1} in stream water upstream and downstream of the RBZ.

Brash mats were created on site during clearfelling to protect the peat from consolidation due to heavy machinery. Since creation of the RBZ in May 2006, 5 years prior to the present study, the peat was fertilised by these brash mats, as seen in Figure 2.8, and may easily have been removed from site for commercial purposes (Forest Research, 2009). However, peat is considered highly vulnerable to a loss in fertility and the removal of brash material from site can deplete the amount of base cations and reduce available nutrients for the future growth of trees (Forest Research, 2009). The potential loss of nutrients may be minimised through careful timing of brash removal after the needles have fallen. Needles contain half-to-two thirds of total nutrients of the brash material and the needle drop time period may occur anywhere from 3 to 9 mo following clearfelling, depending on local climate and season (Forest Research, 2009). Hyvönen et al. (2000) showed that after 6 – 8 years of decomposition, needles and twigs provided more nutrients for future tree growth as opposed to larger branches, and the decomposition rate (and therefore nutrient addition) decreased with increasing branch diameter. It was noted by Hyvönen et al. (2000), however, that 16 years after clearfelling, the decomposing branches increased the carbon content of the forest floor by up to 50 – 100 % and woody logging residues provided more N and P release than needles. Nutrient release from decaying brash may enter sensitive receiving waters (Stevens et al., 1995), but this did not occur at the site of the present study. However, if brash was to be removed from peatland sites following fertilisation from needle drop time period, other factors such as sediment release and economic value of degraded brash would need to be considered.

2.3.3. Soil analysis

The WEP was highest in the vegetated areas and under the brash mats in the RBZ (Figure 2.11) due to the leaching of P from the decaying brash mat into the soil in the 5 years since the creation of the buffer zone. Similar results were obtained by Rodgers et al. (2010), who found that WEP under windrows of brash was significantly higher than WEP in a windrow/brash-free area. The impact of the brash on WEP is a function of the length of time it is left on site and the time taken for regeneration of vegetation to occur (Macrea et al., 2005). The role of vegetation in nutrient uptake was investigated by O' Driscoll et al. (2011), who measured WEP concentrations of 6–9 mg P kg^{-1} dry soil from a seeded plot in contrast to 27 mg P kg^{-1} dry soil from an unplanted plot positioned on a recently harvested site. As the brash mats run perpendicular to the flow of water in the present study, any vegetated areas down-slope from the brash mats, but not directly overlain by brash mats, were also affected. Water extractable phosphorus decreased closer to the stream, but this reduction was due to the presence of a mineral-rich peat along the bank of the stream, which protected the water-course from increases in nutrient concentration. As there is a strong correlation between WEP measured in peat and DRP concentration in surface runoff (O' Driscoll et al., 2011), the potential for high concentrations of DRP in surface runoff from under the brash mats is high.

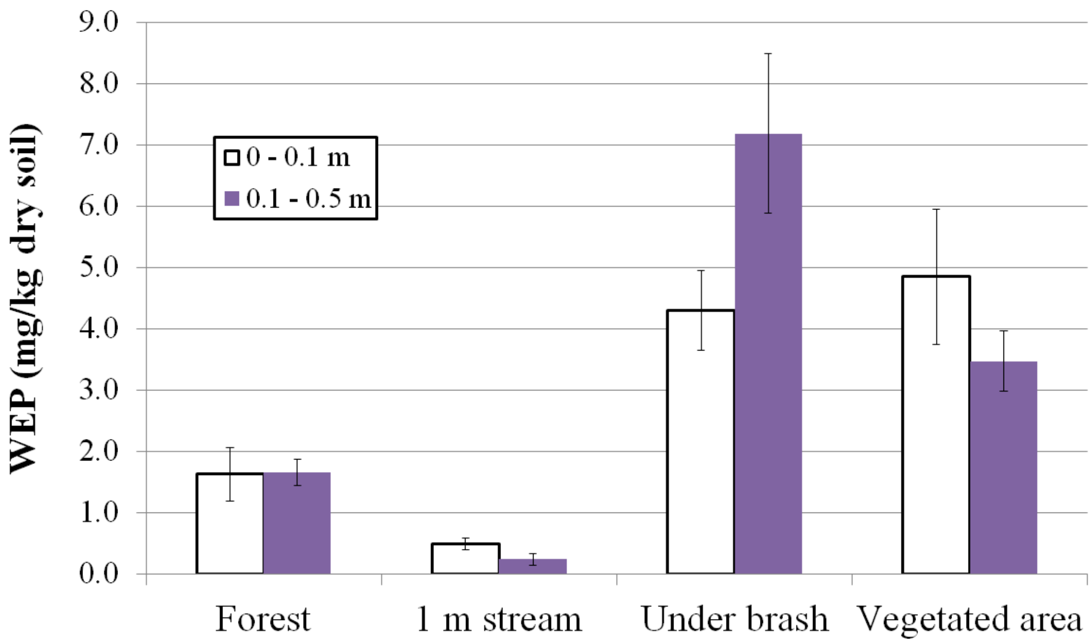

Figure 2.11 Water extractable phosphorus (WEP) concentration (mg kg^{-1} dry soil) in riparian buffer zone (1 m from the stream, under brash and vegetated area) and forest at 0 – 0.1 m and 0.1 – 0.5 m depths. Error bars indicate standard deviation.

The desorption-adsorption isotherms of P adsorption by weight (mg g^{-1} dry material) showed that, while the P adsorption capacity was of the same magnitude in all areas for the 0 – 0.15 m depth examined, more P adsorption took place at a distance of 1 m from the stream at the 0.15 – 0.3 m depth (Figure 2.12). This was due to the mineral-rich peat at this location. At both depths, the peat directly underneath the brash mat appeared to be at P saturation and had little remaining adsorption capacity (reflecting the high DRP concentrations at these points; Figure 2.8). As is typical for soils with a low P retention capacity, the P adsorbed to the peat continued to rise with higher P solutions and a maximum adsorption value was not reached (Väänänen et al., 2007). Desorption of P from the peat in all areas and at both depths occurred when it was overlain with water with a P concentration of 0 mg L^{-1} (Figure 2.12). This was greatest under the brash mats and under the vegetated areas, especially at the 0 – 0.15 m depth.

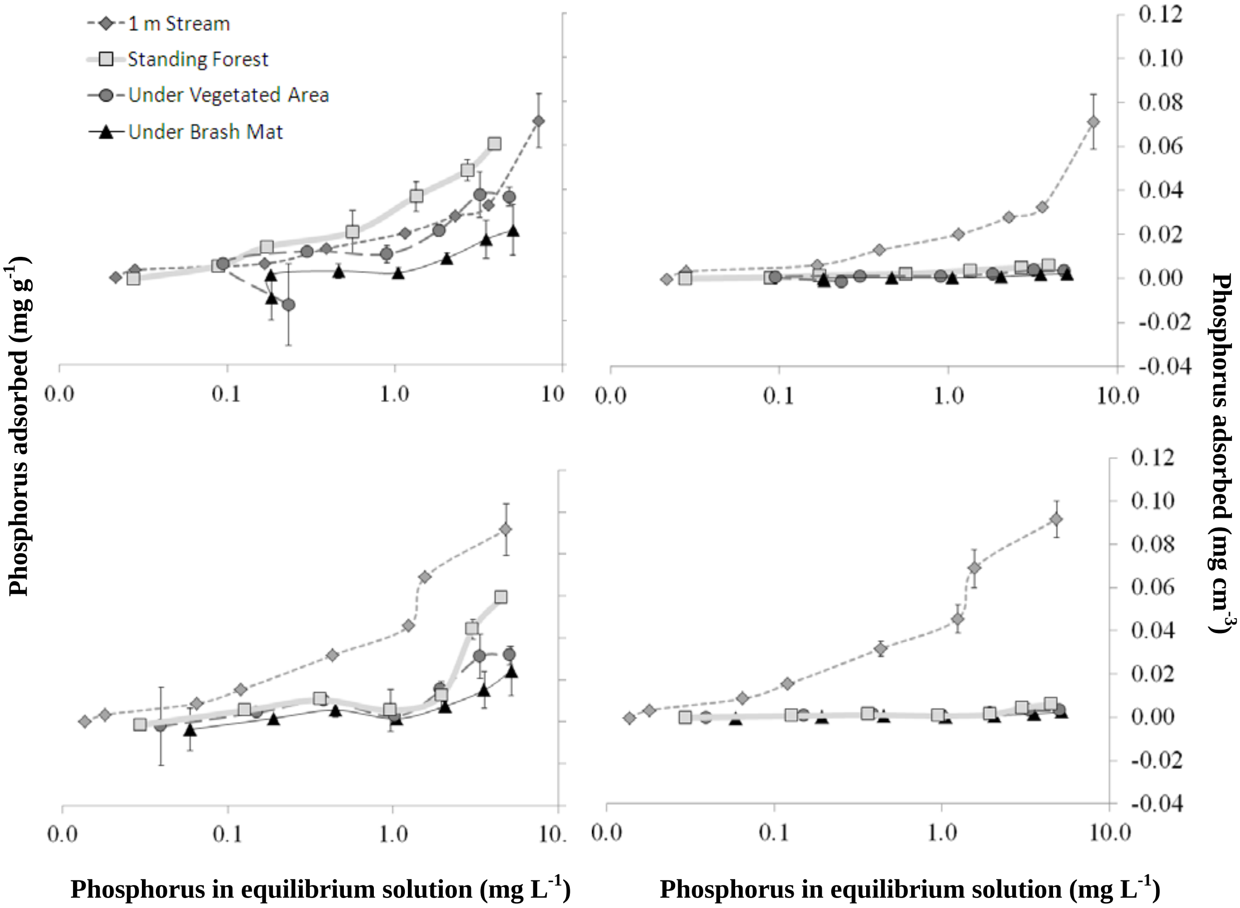

Figure 2.12 Phosphorus (P) adsorption isotherms in riparian buffer zone and forest by weight (mg g^{-1}) on left and by volume (mg cm^{-3}) on right at 0 – 0.15 m (top) and 0.15-0.30 m (bottom) depths. Log scale on X axis for clarity.

When the results were expressed per volume of dry material (mg cm^{-3}) and the bulk densities of the mineral-rich peat soil 1 m from the stream (approximately 1 g cm^{-3}) and the peat further up the buffer (35 – 45 m from the stream; approximately 0.1 g cm^{-3}) were considered, the differences in the adsorption capacity of the soils was more pronounced: the P adsorption capacity of the mineral-rich peat 1 m from the stream were much higher than the peat layers further up the buffer. Loss on ignition analysis showed that there was 90 – 95 % mineral content in the samples 1 m from the stream, while the samples further back from the stream (35 – 45 m) had a 98 – 100 % organic content. This trend was similar to what was found by Väänänen et al. (2006) in their study on peat and mineral soils in Finland.

2.4. Conclusions

1. The created RBZ was capable of providing nutrients to planted saplings, fertilizing the peat with degrading brash material and preventing elevated levels of nutrients entering the adjacent water-course. This indicates that a created RBZ is a realistic management option in peatland forests.
2. The overall survival rate of the planted saplings in the RBZ was relatively high, with over half of oak, rowan and birch saplings surviving after 5 years. The survival of willow, alder and holly was not as successful, possibly due to a number of factors including the exposed nature of the site, peat depth, maintenance, cultivation and fertilization. The low number of planted saplings for these three species also could have had an effect on this outcome.
3. Dissolved reactive phosphorus concentrations were significantly higher under the brash mats in the RBZ compared to all other areas. These high concentrations of DRP were due to the degrading brash mats left on site following clearfelling. However, this did not leach to the stream as the concentration of DRP upstream and downstream of the buffer remained low throughout the study. The standing forest had the highest concentrations of NH_4^+-N, while TON was similar for all areas. It is recommended to leave brash mats on site following clearfelling to fertilise the site and to reduce disturbance to the vulnerable peat sites.
4. Water extractable phosphorus and desorption-adsorption testing also confirmed the high concentrations of P under the brash mats. Water extractable phosphorus was highest in the vegetated areas and under the brash mats in the RBZ. Desorption of P was highest under the brash mats and adsorption was greatest in the mineral-rich peat soil adjacent to the stream.

2.5. Acknowledgements

This work was funded by the Department of Agriculture, Fisheries and Food and the Environmental Protection Agency under the STRIVE program 2007 – 2013. The authors wish to acknowledge the input of Coillte in allowing access to state forestry for this project, and for providing logistical support and advice. A special thanks to Dr. Michael Rodgers and all at the Marine Institute for their continued help throughout this project and for supplying the orthophotography.

References

Allott, N., McGinnity, P., O'Hea, B., 2005. Factors influencing the downstream treansport of sediment in the Lough Feeagh catchment, Burrishoole, Co.Mayo, Ireland. Freshwater Forum. 23, 126-138.

APHA, 1998. Standard methods for examination of water and wastewater. 19th ed. American Public Health Association, Washington.

Aust, W.M., Blinn, C.R., 2004. Forestry best management practices for timber harvesting and site preparation in the eastern United States: An overview of water quality and productivity research during the past 20 years (1982–2002). Wat. Air Soil Poll. 4, 5-36.

Bacon, P., 2003. Forestry: A growth industry in Ireland. (Assessed 17th February 2011) http://www.coillte.ie/fileadmin/templates/pdfs/BaconReport.pdf.

Bain, C.G., Bonn, A., Stoneman, R., Chapman, S., Coupar, A., Evans, M., Gearey, B., Howat, M., Joosten, H., Keenleyside, C., Labadz, J., Lindsay, R., Littlewood, N., Lunt, P., Miller, C.J., Moxey, A., Orr, H., Reed, M., Smith, P., Swales, V., Thompson, D.B.A., Thompson, P.S., Van de Noort, R., Wilson, J.D., Worrall, F., 2011. IUCN UK commission of inquiry on peatlands. (Assessed 11th February 2012) http://www.iucn-uk-peatlandprogramme.org/commission/

Bowman, J., 2009. New Water Framework Directive environmental quality standards and biological and hydromorphological classification systems for surface waters in Ireland. Biol. and Environ: Proc. Roy. Ir. Acad. 109B, 247–60.

British Standards Institution, 1990. Determination by mass-loss on ignition. British standard methods of test for soils for civil engineering purposes. Chemical and electro-chemical tests. BS 1377, 3, BSI, London.

Broadmeadow, S., Nisbet, T.R., 2004. The effects of riparian forest management on the freshwater environment: A literature review of best management practice. Hydrol. Earth Sys. Sci. 8, 286-305.

Carling, P. A., Irvine, B. J., Hill, A., Wood, M., 2001. Reducing sediment inputs to Scottish streams: A review of the efficacy of soil conservation practices in upland forestry. Sci. Tot. Environ. 265: 209-227.

Coillte, 2012. Retrieved 21 May 2012, from http://www.coillte.ie/coillteforest/responsible_forest_management_and_certification/certification_introduction/.

Coillte, 2011. 2011 – 2015 W3 District Strategic Plan. In, Connemara & Mayo. (Assessed 20th May 2012) http://www.coillte.ie/fileadmin/templates/pdfs/dsp/W3-2011-2015-DSP-new2.pdf

Cummins, T., Farrell, E.P., 2003. Biogeochemical impacts of clearfelling and reforestation on blanket peatland streams I. phosphorus. For. Ecol. Man. 180, 545-555.

Daniel, T.C., Sharpley, A.N., Lemunyon, J.L., USDA, A., USDA, N., 1998. Agricultural phosphorus and eutrophication: a symposium overview. J. Environ. Qual. 27, 251-257.

EEA. 2004. Revision of the assessment of forest creation and afforestation in Ireland. Forest Network Newsletter. (Assessed 21st April 2011) http://www.woodlandleague.org/documents/ForestryInIreland/Appendices/4_IrelandsCommitmentsToSFM/documents/Ref20_FNN150.pdf

Elrashidi, M.A., 2011. Selection of an appropriate phosphorus test for Soils. (Assessed 12th June 2011) ftp://ftp-fc.sc.egov.usda.gov/NSSC/Analytical_Soils/phosphor.pdf.

EPA, 2005. Phosphorus regulations national implementation report. (Assessed 14th May 2011) http://www.epa.ie/downloads/pubs/water/phosphorus/EPA_phosphorus_report_2005.pdf.

Farmer, R.A., Nisbet, T.R., 2004. An overview of forest management and change with respect to environmental protection in the UK. Hydrol. Earth Sys. Sci. 8, 279-285.

Forest Research, 2009. Guidance on site selection for brash removal. (Assessed 22nd October 2011)http://www.biomassenergycentre.org.uk/pls/portal/docs/PAGE/BEC_TECHNICAL/BEST%20PRACTICE/BRASH_RESIDUE_PROTOCOL-1.PDF.

Forest Service, 2000. Forest and water quality guidelines. (Assessed 14th November 2011) http://www.agriculture.gov.ie/media/migration/forestry/publications/water_quality.pdf

Frost, I., Rydin, H., 1997. Effects of Competition, Grazing and Cotyledon Nutrient Supply on Growth of Quercus robur Seedlings. Oikos. 79, 53-58.

Gerasimov, Y. and Katarov V., 2010. Effect of bogie track and slash reinforcement on sinkage and soil compaction in soft terrains. Croat. J. For. Eng. 31, 35-45.

Giesler, R., Andersson, T., Lövgren, L., Persson, P., 2005. Phosphate sorption in aluminum- and iron-rich humus soils. Soil Sci. Soc. Am. J. 69, 77-86.

Goudie, A., 2006. The human impact on the natural environment: past, present and future. Sixth ed. Blackwell Sci., New Jersey.

Haygarth, P.M., Condron, L.M., Heathwaite, A.L., Turner, B.L., Harris, G.P., 2005. The phosphorus transfer continuum: Linking source to impact with an interdisciplinary and multi-scaled approach. Sci. Tot. Environ. 344, 5-14.

Haygarth, P.M., Hepworth, L., Jarvis, S.C., 1998. Forms of phosphorus transfer in hydrological pathways from soil under grazed grassland. Euro. J. Soil Sci. 49, 65-72.

Haygarth, P.M., Warwick, M.S., House, W.A., 1997. Size distribution of colloidal molybdate reactive phosphorus in river waters and soil solution. Wat. Res. 30, 439-448.

Hutton, SA., Harrison, SSC., O'Halloran, J., 2008. An evaluation of the role of forests and forest practices in the eutrophication and sedimentation of receiving waters. (Accessed 7th May 2012) http://www.wfdireland.ie/docs/22_ForestAndWater/Forest%20and%20water%20Eutrophication_Sedimentation%20Literature%20review%20.pdf

Hyvönen, R., Olsson, B.A., Lundkvist, H., Staaf, H., 2000. Decomposition and nutrient release from Picea abies (L.) Karst. and Pinus sylvestris L. logging residues. For. Ecol. Man. 126, 97-112.

Joosten, H., Clarke, D., 2002. Wise use of mires and peatlands. A framework for decision-making. (Assessed 9th February 2012). http://www.gret-perg.ulaval.ca/fileadmin/fichiers/fichiersGRET/pdf/Doc_generale/WUMP_Wise_Use_of_Mires_and_Peatlands_book.pdf

Kominoski, J. S., S. LarraÑAga, Richardson, John S., 2012. Invertebrate feeding and emergence timing vary among streams along a gradient of riparian forest composition. Freshwat. Biol. doi: 10.1111/j.1365-2427.2012.02740.x

Luke, S.H., Luckai, N.J., Burke, J.M., Prepas, E.E., 2007. Riparian areas in the Canadian boreal forest and linkages with water quality in streams. Environ. Rev. 15, 79-97.

Long, M., Jennings, P., 2006. Analysis of the peat slide at Pollatomish, County Mayo, Ireland. Landslides. 3, 51-61.

Macrae, M.L., Redding, T.E., Creed, I.F., Bell, W.R., Devito, K.J., 2005. Soil, surface water and ground water phosphorus relationships in a partially harvested Boreal Plain aspen catchment. For. Ecol. Man. 206, 315-329.

McConnell, B. and Gatley, S. 2006. Bedrock Geology of Ireland. Derived from the Geological Survey of Ireland 1:100,000 Bedrock map series and the geological survey of Northern Ireland 1:250,000 Geological Map of Northern Ireland. (Assessed 10th February 2012). http://www.gsi.ie/Programmes/Bedrock/Projects/GSI+publishes+All+Ireland+bedrock+map.htm

Müller, M., 2000. Hydro-geographical studies in the Burrishoole catchment, Newport, Co. Mayo, Ireland: effect of afforestation on the run-off regime of small mountain spate river catchments. SIL. 27, 1146–8.

National Forest Inventory, 2007. National Forest Inventory Republic of Ireland - Results. (Assessed 28th May 2011). http://www.agriculture.gov.ie/media/migration/forestry/nationalforestinventory/nationalforestinventorypublications/4330NFIResults.pdf

O’Driscoll, C., Rodgers, M., O’Connor, M., Asam, Z.-u.-Z., de Eyto, E., Poole, R., Xiao, L., 2011. A potential solution to mitigate phosphorus release following clearfelling in peatland forest catchments. Wat. Air Soil Poll. 221, 1-11.

Ormerod, S.J., Rundle, S.D., Lloyd, E.C., Douglas, A.A., 1993. The influence of riparian management on the habitat structure and macroinvertebrate communities of upland streams draining plantation forests. J. Appl. Ecol. 30, 13-24.

Owende, P.M.O., Lyons, J., Haarlaa, R., Peltola, A., Spinelli, R., Molano, J., Ward, S.M., 2002. Operations protocol for eco-efficient wood harvesting on sensitive sites. (Assessed 6th January 2012). http://www.ucd.ie/foresteng/html/ecowood/op.pdf

Parkyn, S.M., Davies-Colley, R.J., Cooper, A.B., Stroud, M.J., 2005. Predictions of stream nutrient and sediment yield changes following restoration of forested riparian buffers. Ecol. Eng. 24, 551-558.

Regan, J.T., Rodgers, M., Healy, M.G., Kirwan, L., Fenton, O., 2010. Determining phosphorus and sediment release rates from five irish tillage soils. J. Environ. Qual. 39, 185-192.

Renou-Wilson, F., Keane, M., Farrell, E.P., 2008. Establishing oak woodland on cutaway peatlands: Effects of soil preparation and fertilization. For. Ecol. Man. 255, 728-737.

Renou, F., Scallan, Ú., Keane, M., Farrell, E.P., 2007. Early performance of native birch (Betula spp.) planted on cutaway peatlands: Influence of species, stock types and seedlings size. Euro. J. For. Res. 126, 545-554.

Rodgers, M., O'Connor, M., Healy, M.G., O'Driscoll, C., Asam, Z.-u.-Z., Nieminen, M., Poole, R., Müller, M., Xiao, L., 2010. Phosphorus release from forest harvesting on an upland blanket peat catchment. For. Ecol. Man. 260, 2241-2248.

Rodgers, M., O'Connor, M., Robinson, M., Muller, M., Poole, R., Xiao, L., 2011. Suspended solid yield from forest harvesting on upland blanket peat. Hydrol. Proc. 25, 207-216.

Ryder, L., de Eyto, E., Gormally, M., Sheehy Skeffington, M., Dillane, M., Poole, R., 2011. Riparian zone creation in established coniferous forests in Irish upland peat catchments: Physical, chemical and biological implications. Biol. and Environ: Proc. Roy. Ir. Acad. 111B, 1-20.

Sharpley, A.N., Daniel, T., Sims, T., Lemunyon, T., Stevens, S., Parry, R., 2003. Agricultural Phosphorus and Eutrophication. (Assessed 15th January 2011). http://www.ars.usda.gov/is/np/phos&eutro2/agphoseutro2ed.pdf

Sjörs, H., 1980. Peat on Earth: Multiple Use or Conservation? Ambio. 9, 303-308.

Stevens, P.A., Norris, D.A., Williams, T.G., Hughes, S., Durrant, D.W., Anderson, M.A., Weatherley, N.S., Hornung, M., Woods, C., 1995. Nutrient losses after clearfelling in Beddgelert Forest: a comparison of the effects of conventional and whole-tree harvest on soil water chemistry. Forestry 68 (2), 115–131.

Strahler, A.N., 1957. Quantitative analysis of watershed geomorphology. Trans. of the American Geophys. Union 38, 913-920.

Syversen, N., Borch, H., 2005. Retention of soil particle fractions and phosphorus in cold-climate buffer zones. Ecol. Eng. 25, 382-394.

Väänänen, R., Nieminen, M., Vuollekoski, M., Nousiainen, H., Sallantaus, T., Tuittila, E.S., Ilvesniemi, H., 2008. Retention of phosphorus in peatland buffer zones at six forested catchments in southern Finland. Silva Fennica. 42, 211-231.

Väänänen, R., Kenttämies, K., Nieminen, M., Ilvesniemi, H., 2007. Phosphorus retention properties of forest humus layer in buffer zones and clear-cut areas in southern Finland. Boreal Environ. Res. 12, 601-609.

Väänänen, R., Nieminen, M., Vuollekoski, M., Ilvesniemi, H., 2006. Retention of phosphorus in soil and vegetation of a buffer zone area during snowmelt peak flow in southern Finland. Water, Air, and Soil Pollution. 177, 103-118.

Walker, B. H., 1992. Biodiversity and Ecological Redundancy. Conserv. Bio._ 6, 18-23.

Chapter 3

The contents of this chapter were presented at the International Peat Conference 2012, Stockholm, Sweden. Joanne Finnegan developed the experimental design and collected, analysed and synthesized the experimental data. She is the primary author of this article. Dr. Mark G. Healy and Dr. John T. Regan assisted with sampling and contributed to the experimental design and paper writing. Dr. Bryan McCabe assisted with data analysis and paper editing.

Use of brash mats for clearfelling of forestry on peat: Irish experience

J. Finnegan, J.T. Regan, B.A. McCabe, M.G. Healy

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

ABSTRACT

Peat soils are susceptible to damage by clearfelling machinery traffic and subsequent rutting and compaction. In Irish clearfelling practice, a brash mat, consisting of small branches and logs, is laid ahead of the harvester and forwarder traffic to minimise soil disturbance. The aims of this study were to: (1) evaluate the effectiveness of brash mats in preventing peat consolidation and (2) quantify nutrient release to the shallow groundwater. Water content profiles were compared to assess soil consolidation and shallow groundwater samples were analysed for their nutrient concentrations. There was no significant change in water content, indicating that the brash mat was successful in preventing peat volume changes. There was an increase in nutrient concentration under the brash mats, which indicates that they may be a long-term nutrient source if not removed.

3.1 Introduction

It is estimated that 59.6 % of forestry in Ireland is on peat (National Forest Inventory, 2007) and approximately 300,000 hectares of afforestation is on upland peat areas (EEA, 2004; Rodgers et al., 2010). These soils are characterised by high water contents (gravimetric water contents usually exceed 500 %) and typical ground bearing capacities of between 10 and 60 kPa (Owende et al., 2002), making clearfelling (harvesting) with heavy machinery difficult. In Irish forestry clearfelling practice, soil disturbance is minimised by laying a brash mat, consisting of small branches and logs, ahead of the machinery used for the harvesting and timber removal processes. The scale of disturbance to a clearfell site is based on a combination of number of passes by the machinery, water content and the use of brash mats (Gerasimov and Katarov, 2010).

Previous research has highlighted the negative impacts of soil compaction (i.e. the expulsion of air from the void space) on the use of the soil for future afforestation. The reduced pore space for water movement reduces the growth rate of future tree and plant crops (Antti, 2008), the value of the harvested timber is reduced (Eliasson and Wästerlund, 2007), and instances of wind throw and erosion are increased (Nugent et al., 2003). Consolidation (i.e. the expulsion of water when the soil is loaded), caused by clearfelling machinery, also occurs when saturated peats are loaded. As density is not a sufficiently sensitive parameter to assess volume changes induced by consolidation, water content may be a more sensitive gauge of volume changes.

Installed brash mats generally remain *in situ* following best practice guidelines (Forest Service, 2000) after felling and can, if timed correctly, re-fertilise the soil for future crops (Stevens et al., 1995). Peat is considered to be highly vulnerable to a loss in fertility and the removal of brash material from site following clearfelling can deplete the amount of base cations and reduce available nutrients for future growth of trees (Forest Research, 2009). Brash can be removed from site once the needle drop period is complete (approximately 6 to 9 months), as up to two-thirds of the total nutrients found in brash material are in the fresh needles (Forest Research, 2009). However, nutrients released from decaying brash mats may enter sensitive receiving waters.

The aim of this study was to evaluate the efficacy of brash mats in preventing soil consolidation and to quantify nutrient release to shallow groundwater. Water contents were used to assess soil consolidation. The study was conducted in two zones: (1) a naturally revegetated peatland, clearfelled five years prior to the present study, with a brash mat left in place (Zone 1), and (2) a recently clearfelled mature forest (Zone 2) (Figure 3.1). This study also compared the effects of degrading brash mats of varying ages (7 months to 5-years old) on the pore water nutrient content of underlying peat.

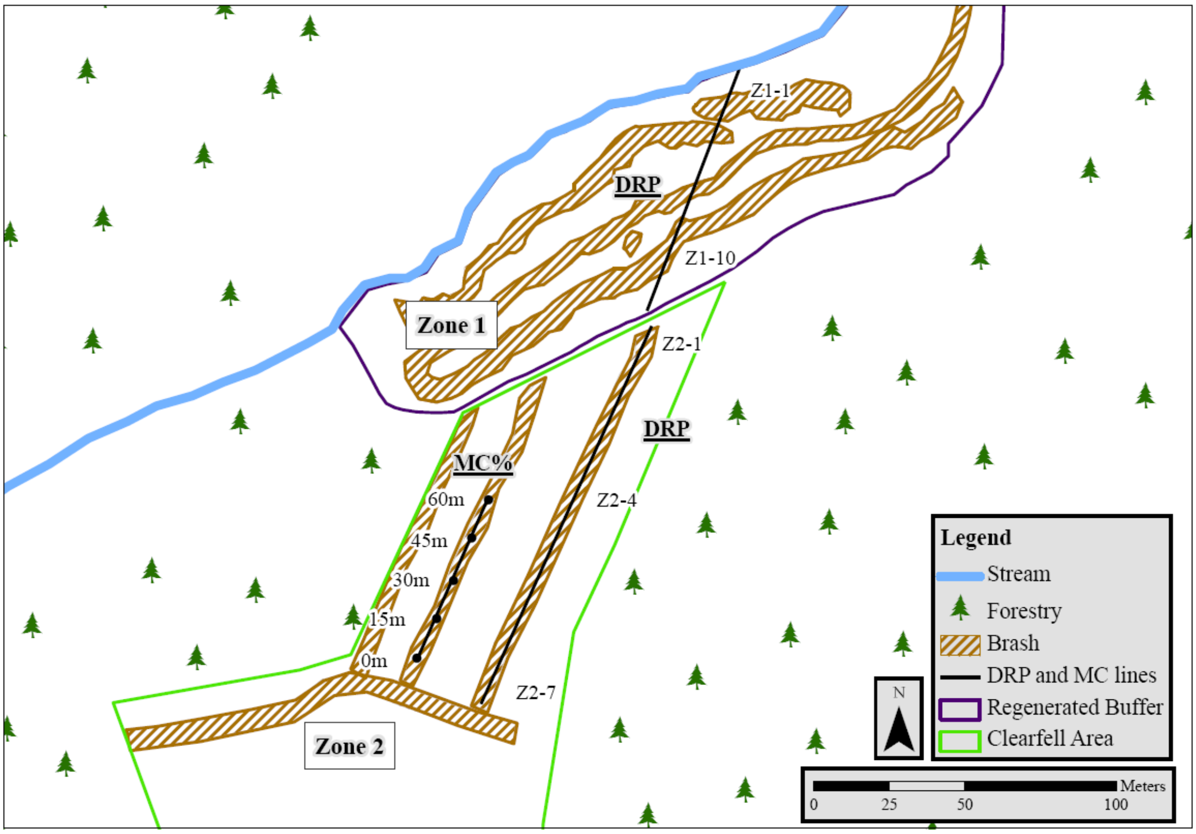

Figure 3.1 Location of Altaconey Forest (Zone 1 and 2) with sampling locations.

3.2 Materials and Methods

The study site was located in the Altaconey forest in the Burrishoole catchment in Co. Mayo, Ireland (ITM reference 495380, 809170). The site has blanket peat to a maximum of 2 m depth and is subjected to approximately 2000 mm of rainfall annually. In 1966, the site was planted with Sitka Spruce (*Picea sitchensis*) and Lodgepole Pine (*Pinus contorta*). In May 2006, a 300 m long strip was clearfelled and allowed to naturally regenerate (Zone 1). The process of bole-only, cut-to-length clearfelling was carried out using a Valmet 921 harvester (wheel width: 650 mm front and 700 mm rear; Figure 3.2). The brash mats were constructed using up to 8 trees and, in line with best management practice (BMP), these brash mats were left *in situ* on completion of clearfelling. Logs were stacked in piles for collection by a Valmet 860 forwarder (wheel width: 600 mm front and rear; Figure 3.2) before being taken off site. In February 2011, the forest upslope of the naturally revegetated peatland was clearfelled in the same manner (Zone 2). A total of 1230 m^3 of timber was removed from Zone 2, of which 58 m^3 pertained to the extraction rack under study.

Figure 3.2 Valmet 921 harvester (on left) and Valmet 860 forwarder (on right).

Piezometers were installed in transects on site for shallow groundwater quality measurement, and each sampling location comprised a cluster of 3 piezometers positioned at 20-cm, 50-cm, and 100-cm-depths below the soil surface. Between April 2010 and October 2011, groundwater samples in Zone 1 were analysed for nutrient concentrations. Similar measurements for nutrient concentrations were conducted in Zone 2 over the same period. Water samples were filtered on site immediately after collection using 0.45 µm filters, returned to the laboratory and tested in accordance with standard methods (APHA, 2005) using a nutrient analyser (Konelab 20; Thermo Clinical Labsystems, Finland). The water quality parameters measured were: (1) dissolved reactive phosphorus (DRP) (2) ammonium-nitrogen (NH_4^+-N) (3) nitrate nitrogen (NO_3^--N) and (4) total oxidised nitrogen (TON).

The initial experimental design included a range of testing on the extractions lines. It was intended to study the consolidation of soil by examining the voids ratio and bulk density using a nuclear gauge (HS-5001EZ Moisture/Density Gauge, Humboldt Manufacturing Company, IL). The nuclear gauge was brought to site for a test period. Confidence in the results was not satisfactory, or repeatable, due to the roots of trees and brash material on site. Testing of the liquid and plastic limits of the peat was also performed with unsatisfactory results and was therefore omitted. It was originally intended to use up to three extraction lines with various numbers of trees and thickness of brash material. However, following discussions with the harvester prior to clearfelling, these plans were not carried out due to their impractically and lack of use in a real life scenario. Therefore, it was decided to test for moisture content, which was used as a proxy for bulk density and changes in consolidation of the peat after clearfelling.

In Zone 2, peat samples were collected from one extraction rack immediately before and after felling for water content determination. This extraction rack was subject to 12 to 20 forwarder passes. Samples to determine soil water content were taken along one extraction rack, 60 m in length, in Zone 2. Peat samples (n=3) were removed every 15 m along the line at 10-cm increments to a depth of approximately 1 m immediately before and after clearfelling of the rack. Peat samples were collected to the same depth in an adjacent standing forest to show changes in water content in the forest soil due to external factors, unrelated to the clearfelling, such as weather or flooding. The soil water content data was analysed both spatially (along the rack and in the controls) and at all depths to identify significant differences using 2 sample t-tests (Minitab, UK).

3.3 Results and Discussion

3.3.1 Water quality tests

There was an increase in nutrient concentration of DRP, NH_4^+-N and TON under the brash mats at all depths in both zones. Results for DRP concentration for both zones at the 50-cm-depth are shown in Figure 3.3. This shows phosphorus (P) leaching to the soil following brash breakdown for two time periods, 5 years (Zone 1) and 7 months (Zone 2) following clearfelling. Median values of NH_4^+-N and TON in Zone 1 at the 50-cm-depth under the brash mat were 940 µg L^{-1} and 30 µg L^{-1}, respectively. The direction of flow in Zone 1 was perpendicular to the brash mats. Therefore, any vegetated areas adjacent to the brash mats, on which no brash was placed, were still influenced by the decaying brash material. This was evident in the median values of NH_4^+-N (710 µg L^{-1}) and TON (110 µg L^{-1}) at the 50 cm-depth under the vegetated areas. In Zone 2, results from up to 7 months post-clearfell showed a gradual increase in concentrations under the newly created brash mats. The post-clearfell results were significantly different from the pre-clearfell results (p=0.031). The highest average concentration of DRP under the new brash mats after 7 months of decay at the 50 cm-depth was 336 µg L^{-1} – just over half the value of 574 µg L^{-1} found under the 5-year old brash mats at the same depth. Yearly data from Stevens et al. (1995) from a Sitka Spruce (*Picea sitchensis*) forest in Wales found that in the first 3 years following bole-only harvesting, approximately one third of the P had leached out of the brash material left on site, leaving a large resource of P in the decaying brash to slowly re-fertilise the soil. Stevens at al. (1995) also noted that this P remained on site rather than leaching to an adjacent stream and was therefore available for the next rotation of plantation.

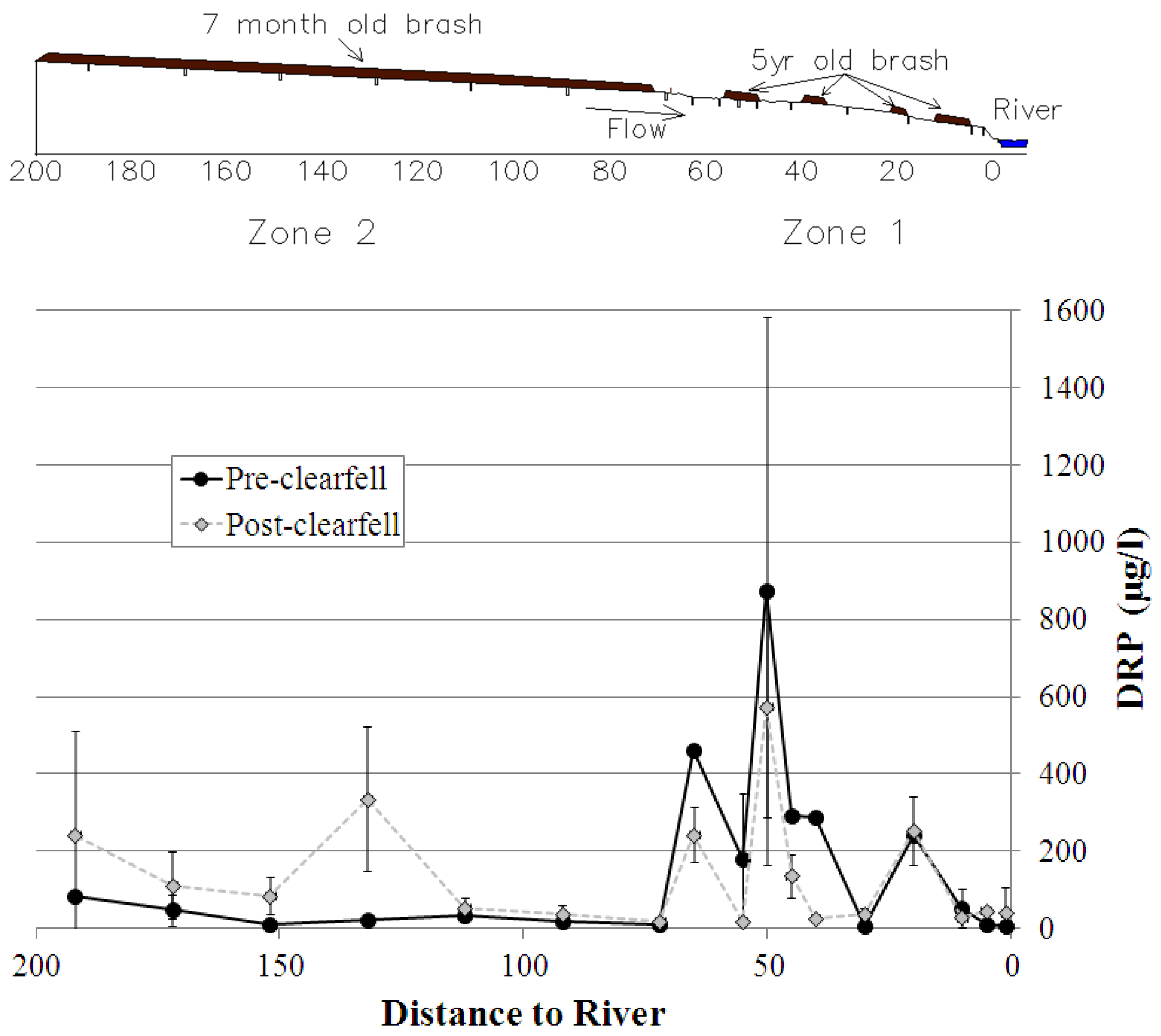

Figure 3.3 Dissolved reactive phosphorus (DRP) (µg L^{-1}) at 50 cm below ground level from transects across site pre- and post- clearfell in Zones 1 and 2.

3.3.2 Consolidation tests

Initial water contents were between 750 and 1000 % across both zones, reflecting the inherent variability of peat. There was no significant change in water content along the entire length of the extraction rack at all depths down to 1 m in Zone 2 before and after clearfelling. This indicated that the brash mat thickness was sufficient in preventing sufficient load transfer and associated volume changes to the peat (Figure 3.4). Controls taken at the same time in an adjacent standing forest showed no change in water content in the forest soil due to external factors, unrelated to the clearfelling, such as weather or flooding (p=0.809). In Zone 2, a slight increase in water content was measured in the top 0 – 30 cm at a distance of 60 m from the river (Figure 3.3), the location which received the least number of forwarder passes (12).

Brash mats have been successful in protecting various underlying soil types in other studies: moist, fine grained soils in northern Sweden (Eliasson and Wästerlund, 2007), sandy soil in the Netherlands (Ampoorter et al., 2007), and a gley soil in the UK (Hutchings et al., 2002). Figures 3.5 and 3.6 show the brash mat just after creation (February 2011) and 15 months after clearfelling (May 2012).

In Ireland, Nugent et al. (2003) quantified typical levels for induced cone penetration resistance (CPR) and rutting depths for heavy machinery traffic associated with forest operations on a peat soil. This was done by studying the pre- and post-thinning processes along 4 extraction racks with brash thicknesses ranging from 8 – 13.8 cm post-harvester travel and 3 – 6.4 cm post-forwarder traffic. They found that the CPR increased in the top 40 cm of the soil following passage of machinery, which indicated that it had become compacted. However, there was no significant rutting caused by the harvester and forwarder traffic. Statistical analysis showed that threshold CPR levels ranged from 594 – 640 kPa for a peat soil with an initial CPR reading of between 524 – 581 kPa (Nugent et al., 2003).

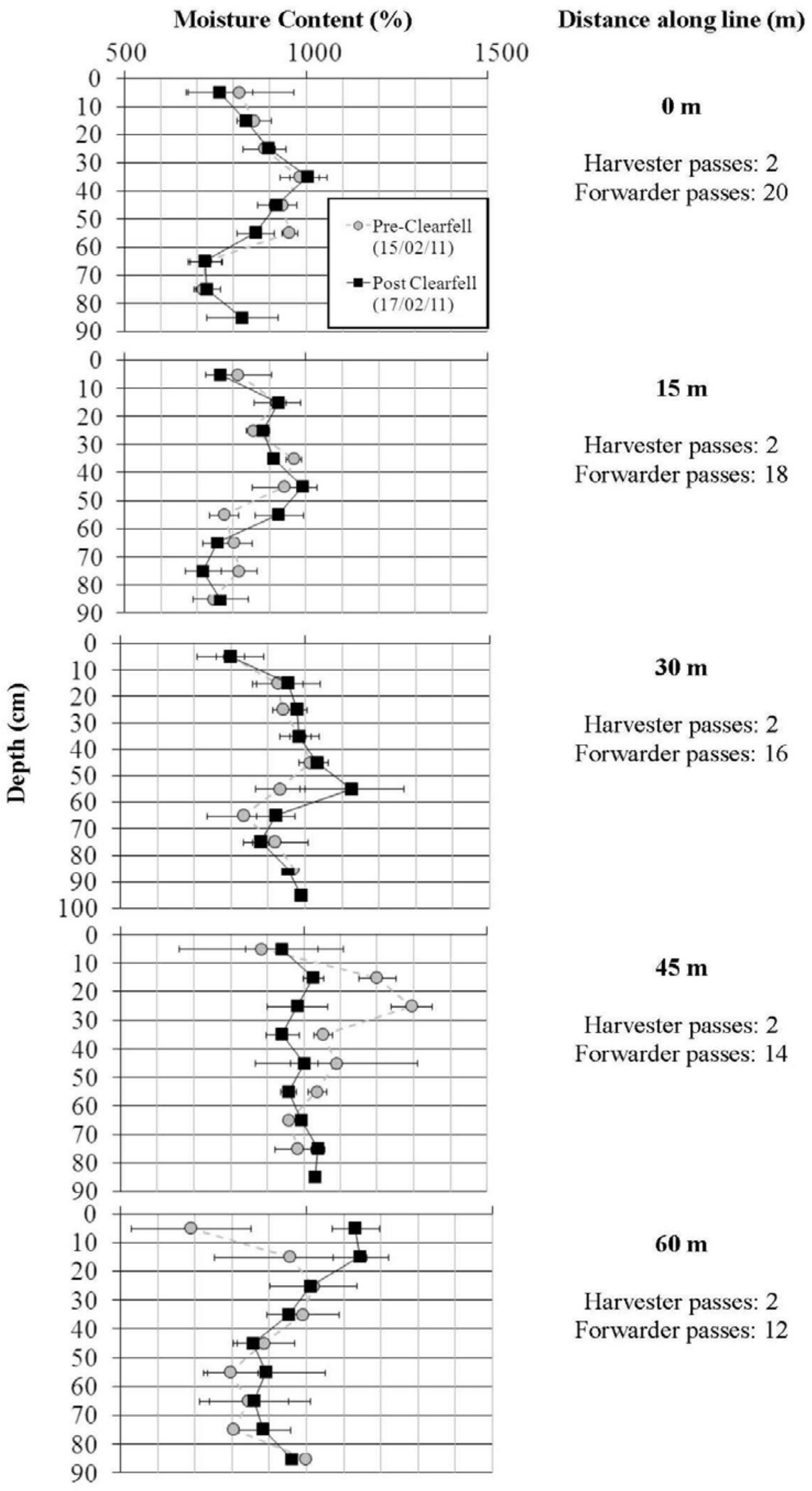

Figure 3.4 Water content (%) and depth below ground level (cm) pre- and post-clearfell at 15 m intervals along a line with number of harvester and forwarder passes. Error bars denote standard deviation of triplicates.

Figure 3.5 February 2011, post-fell.

Figure 3.6 May 2012, 15 months post-fell.

3.4 Conclusion

The use of brash mats of sufficient thickness and quality during clearfelling protects peat from consolidation, minimizes soil disturbance, and re-fertilises the soil with dissolved nutrients. However, best management practices must be implemented to ensure that brash mats are maintained and nutrients from degradation do not reach water courses.

3.5 Acknowledgements

The authors acknowledge the support from the EPA for funding this research under the STRIVE program.

References

Ampoorter, E., R. Goris, W. M. Cornelis and K. Verheyen (2007). "Impact of mechanized logging on compaction status of sandy forest soils." Forest Ecology and Management **241**(1-3): 162-174.

Antti, W. (2008). "Effect of removal of logging residue on nutrient leaching and nutrient pools in the soil after clearcutting in a Norway spruce stand." Forest Ecology and Management **256**(6): 1372-1383.

APHA (2005). Standard methods for the examination of water and wastewater. Washington, American Public Health Association.

EEA (2004). "Revision of the Assessment of Forest Creation and Afforestation in Ireland." Forest Network Newsletter European Environmental Agency's Spatial Analysis Group (Issue 150).

Eliasson, L. and I. Wästerlund (2007). "Effects of slash reinforcement of strip roads on rutting and soil compaction on a moist fine-grained soil." Forest Ecology and Management **252**(1-3): 118-123.

Forest Research (2009). Guidance on site selection for brash removal. The Research Agency of the Forest Commission.

Forest Service (2000). Code of Best Forestry Practice - Ireland. Irish National Forest Standard. Department of the Marine and Natural Resources. Dublin.

Gerasimov, Y. and V. Katarov (2010). "Effect of bogie track and slash reinforcement on sinkage and soil compaction in soft terrains." Croatian Journal of Forest Engineering **31**(1): 35-45.

Hutchings, T. R., A. J. Moffat and C. J. French (2002). "Soil compaction under timber harvesting machinery: a preliminary report on the role of brash mats in its prevention." Soil Use and Management **18**(1): 34-38.

National Forest Inventory (2007). National Forest Inventory Republic of Ireland - Results. Wexford, Forest Service.

Nugent, C., C. Kanali, P. M. O. Owende, M. Nieuwenhuis and S. Ward (2003). "Characteristic site disturbance due to harvesting and extraction machinery traffic on sensitive forest sites with peat soils." Forest Ecology and Management **180**(1–3): 85-98.

Owende, P. M. O., J. Lyons, R. Haarlaa, A. Peltola, R. Spinelli, J. Molano and S. M. Ward (2002). "Operations protocol for eco-efficient wood harvesting on sensitive sites." (Accessed on 12th May 2011) http://www.ucd.ie/foresteng/.

Rodgers, M., M. O'Connor, M. G. Healy, C. O'Driscoll, Z.-u.-Z. Asam, M. Nieminen, R. Poole, M. Müller and L. Xiao (2010). "Phosphorus release from forest harvesting on an upland blanket peat catchment." Forest Ecology and Management **260**(12): 2241-2248.

Stevens, P. A., D. A. Norris, T. G. Williams, S. Hughes, D. W. Durrant, M. A. Anderson, N. S. Weatherley, M. Hornung and C. Woods (1995). "Nutrient losses after clearfelling in Beddgelert Forest: A comparison of the effects of conventional and whole-tree harvest on soil water chemistry." Forestry **68**(2): 115-131.

Chapter 4

The contents of this chapter have been submitted to Science of the Total Environment. Joanne Finnegan developed the experimental design and collected, analysed and synthesized the experimental data. She is the primary author of this article. Dr. Mark G. Healy contributed to the experimental design and paper writing. Dr. John T. Regan assisted with sampling, experimental design and paper editing. Dr. Owen Fenton assisted with the data analysis and paper editing, and Dr. Gary Lanigan assisted with paper editing.

The effect of management changes on watertable position and nutrients in shallow groundwater in a harvested peatland forest

J. Finnegan[1], J.T. Regan[1], O. Fenton[2], G.J. Lanigan[2] and M.G. Healy[1]

[1]*Civil Engineering, National University of Ireland, Galway, Ireland*
[2]*Teagasc, Environmental Research Centre, Johnstown Castle, Wexford, Ireland*

ABSTRACT

Management changes such as drainage, fertilisation, afforestation and subsequent harvesting (clearfelling) of forested peatlands influence watertable (WT) position and nutrient load transfers to shallow groundwater and surface water. This study investigated the impact of clearfelling of a peatland forest on WT and nutrient concentrations. Three areas on the study site were examined: (1) a regenerated riparian peatland buffer (RB) clearfelled 4 years prior to the present study (2) a recently clearfelled coniferous forest (CF) and (3) a standing, mature coniferous forest (SF), on which no harvesting took place and which acted as a study control. The WT remained consistently below 0.3 m during the pre-clearfelling period. Results showed there was an almost immediate rise in the WT post-clearfell on removal of the growing biomass and a rise to 0.15 m below ground level (bgl) within 10 months of clearfelling. The WT depth subsequently fluctuated with dry periods. This rise would indicate that the site has restoration potential, following drain blocking, if reforestation does not take place. Dissolved reactive phosphorus (DRP) concentrations, which were more affected by degrading logging residues (brash material) than WT fluctuations, increased from an average of 28 µg L^{-1} to 230 µg L^{-1} in shallow ground waters after clearfelling. The concentration of ammonium-nitrogen (NH_4^+-N) was highest under the SF (average of 0.82 mg L^{-1}) due to the effect of the fluctuations of the deep WT on decomposing peat. Concentrations of total oxidised nitrogen were generally similar across all locations sampled; very low concentrations of nitrate-nitrogen (NO_3^--N) (< 20 µg L^{-1}) were found across all depths and locations throughout the site and were, in many cases, below the limits of detection. Nutrient discharges to the adjacent watercourse in excess of maximum admissible concentrations were negligible due to the low lateral saturated conductivity and the high inherent natural attenuation capacity of the peat.

4.1 Introduction

Peatlands, with a total area of 4 million km^2, cover approximately 3 % of the total landmass in the world, and are mostly present in boreal regions and tropical zones (Renou-Wilson et al., 2011). These ecosystems produce 10 % of the global freshwater supply (Joosten and Clarke, 2002) and one-third of the world's soil carbon (C) content (Parish et al., 2008). Consequently, peatlands hold an important global position in abating greenhouse gas (GHG) emissions and providing attenuation for fresh water.

The depth to the watertable (WT) in peatlands is seen as the key factor in determining changes in the global C cycle (Erwin, 2009), as it affects soil chemical conditions, soil temperature, and the availability of an aerobic environment (Sottocornola and Kiely, 2010). Any fluctuation in the WT can intensify C mineralization rates by up to three times (Blodau, 2002). Undisturbed blanket peatlands in Ireland have a WT of approximately 0.1 m below ground level (bgl) (Koehler et al., 2011). Restoration of peatlands aims to return the WT back to such a position through management changes (e.g. clearfelling and/or drain blocking). The decision to restore a certain site is based on ecology, hydrology and existing vegetation. Approximately 8 % of the forested peatlands in the west of Ireland are suitable for restoration (Tierney, 2007). For successful bog growth, the depth to the WT must be within 0.1 m of the ground surface for at least 90 % of the year (Conaghan, 2003). Sottocornola and Kiely (2010) found that in undisturbed blanket peatlands in the southwest of Ireland with a 30-year annual average rainfall of 1430 mm, daily average WT levels never fell below 0.16 m from 2002 to 2010.

Drainage of peatlands for agriculture or forestry enterprises lowers the WT (Lewis et al., 2012; Renou and Farrell, 2005), and can increase GHG emissions, peat oxidisation, suspended sediment (SS) losses through surface runoff, and nutrient discharge from shallow groundwater to a surface waterbody (Wichtmann and Wichmann, 2011). Macrae et al. (2012) showed that WT drawdown decreased net nitrification rates in a peat soil, across a varied landscape, and increased extractable nitrate (NO_3^-) concentrations due to an increase in bulk density from compression of the drying peat, but did not affect net nitrogen (N) and phosphorus (P) mineralisation rates, extractable total inorganic N, or ortho-P concentrations. Macrae et al. (2012) also found that a thick capillary fringe above the WT ensured soil water content was near saturation and a fluctuation of 0.2 m was not enough to change the overall nutrient dynamics on site. In the west of Ireland, blanket peatlands exhibit very slow infiltration rates, but vertical saturated hydraulic conductivity (k_s) can be quite high. Laterally, hydraulic gradients are low and lateral k_s is typically <0.01 m d^{-1} (Cummins and Farrell, 2003; Farrell and Boyle, 1990). Such a low lateral k_s compares well with glaciated aquifer sediments in Ireland, which have long residence times, and high nutrient concentrations can pose a risk to a waterbody over long periods of time (Fenton et al., 2011a), although high denitrification potentials may mitigate against losses (Fenton et al., 2009).

Management changes e.g. drainage, fertilisation, afforestation and subsequent clearfelling of peatlands, can lead to increases in nutrients (Jacks and Norrström, 2004; Väänänen et al., 2008; Rodgers et al., 2010) and sediment losses to receiving waters (Rodgers et al., 2011). In Ireland, afforestation generally took place in the 1950s on blanket bogs of ombrotrophic peats (low fertility peat, which takes nutrients from the atmosphere), which have low concentrations of P and N (Farrell and Boyle, 1990). In accordance with the current Code of Best Forest Practice in Ireland (Collins et al., 2000), P is applied at a rate of 42 kg of P ha^{-1} (350 kg of granulated rock phosphate) to peat for forestry growth at the afforestation stage (Renou et al., 2000), although, in the 1950s, a rate of 13.2 kg of P ha^{-1} (110 kg of granulated rock phosphate) was applied (Rodgers et al., 2010). Blanket peat has poor adsorption capacity for P (Renou and Cummins, 2002; O'Driscoll et al., 2011), and hydrological losses of P can increase during clearfelling (Renou and Cummins, 2002; Väänänen et al., 2008). However, P can also be returned to the soil during clearfelling by bole-only harvesting, which involves the removal of only the merchantable timber from site, leaving the branches and logging residue (brash material) to degrade on site (Rodgers et al., 2010). If not correctly managed, this can lead to elevated levels of nutrients in receiving waters due to surface runoff from the clearfelled area (Kaila et al., 2012).

Gaseous nitrogen (N_2) makes up 78 % of the atmosphere, but must first be fixed or chemically processed, bacterially or industrially, to make it available for plants in inorganic forms such as ammonium (NH_4^+), ammonia (NH_3) and NO_3^-, or organic forms such as urea, proteins and nucleic acids (Stark and Richards, 2008). In the 1950s, N fertiliser was only applied during initial afforestation if N was the only deficient nutrient on site (Renou and Farrell, 2005), but was not as widespread or necessary as P fertilisation due to the existing levels of N in the soil. This background N concentration was due to scavenging of N from the atmosphere by the forest canopy, drainage and drying of the peat, and increased microbial activity due to rock phosphate application (Farrell and Boyle, 1990). Daniels et al. (2012) related the presence of NH_4^+ in upland peat catchments in the UK to atmospheric deposition, and the mineralisation (ammonification) of organic N due to fluctuating WTs. Ammonification converts organic N, such as applied urea fertilizer, to NH_3 and NH_4^+, which is then transformed, by nitrification, to nitrite (NO_2^-) and NO_3^- under aerobic conditions in the soil (Vymazal, 2007). Nitrification in peat is low due to the shallow WT (Von Arnold et al., 2005), and low water and soil temperatures (Cabezas et al., 2012). Nitrogen deposition over time can decrease the C:N ratio, which can enhance nitrification and denitrification on some sites (Jacks and Norrström, 2004). Anaerobic conditions in peatlands, provided by the shallow WT and low drainage (Martikainen et al., 1993; Renou and Farrell, 2005), can reduce NO_3^- by denitrification to N_2 gas, which may be emitted to the atmosphere, or reduced back to NH_4^+ by dissimilatory nitrate reduction to ammonium (DNRA), particularly in high C content soils such as peats (Stark and Richards, 2008).

There is limited data on the interaction of harvesting of forests on peatlands with WT fluctuations and shallow ground water nutrient concentrations. Therefore, the aim of this study was to investigate, over 2 years including pre- and post-clearfelling periods, how clearfelling of a forest on a blanket peat soil affects (1) WT fluctuations and (2) P and N concentrations in shallow groundwater.

4.2 Materials and Methods

4.2.1 Study site description and management

The study site was located in the Altaconey (also known as the Altahoney) forest in the Burrishoole catchment in Co. Mayo, Ireland (ITM reference 495380, 809170) (Figure 4.1). The catchment is situated in the Nephin Beg range at an approximate elevation of 150 m above sea level and is located within a sub-catchment area of 416.2 ha, of which

176.4 ha is fully forested (Ryder et al., 2011). Three areas were used in the present study (Figure 4.1): (1) a regenerated riparian peatland buffer (RB), clearfelled 4 years before the present study (2) a recently clearfelled coniferous forest (CF), clearfelled in February 2011, and (3) a standing, mature coniferous forest (SF), acting as a study control and on which no harvesting operations took place (Figure 4.2 and 4.3, photographed from the rain gauge). The site has a north-westerly aspect, while a third-order stream (Strahler, 1964), which is a tributary to the Altaconey River (classed as 'unpolluted' by the EPA (2011)), flows in a north-easterly direction to the north of the site. There is a moderate climate, which is heavily influenced by the proximity of the Atlantic Ocean, with average air temperatures of 13 °C in summer and 4 °C in winter, and a mean annual rainfall of 2000 mm (Rodgers et al., 2010). Blanket peat of varying depth down to 2 m covers the site. Sand and gravel deposits underlay the peat on top of the Cullydoo formation of Srahmore quartzite and schist, which is a poorly unproductive aquifer (McConnell and Gatley, 2006).

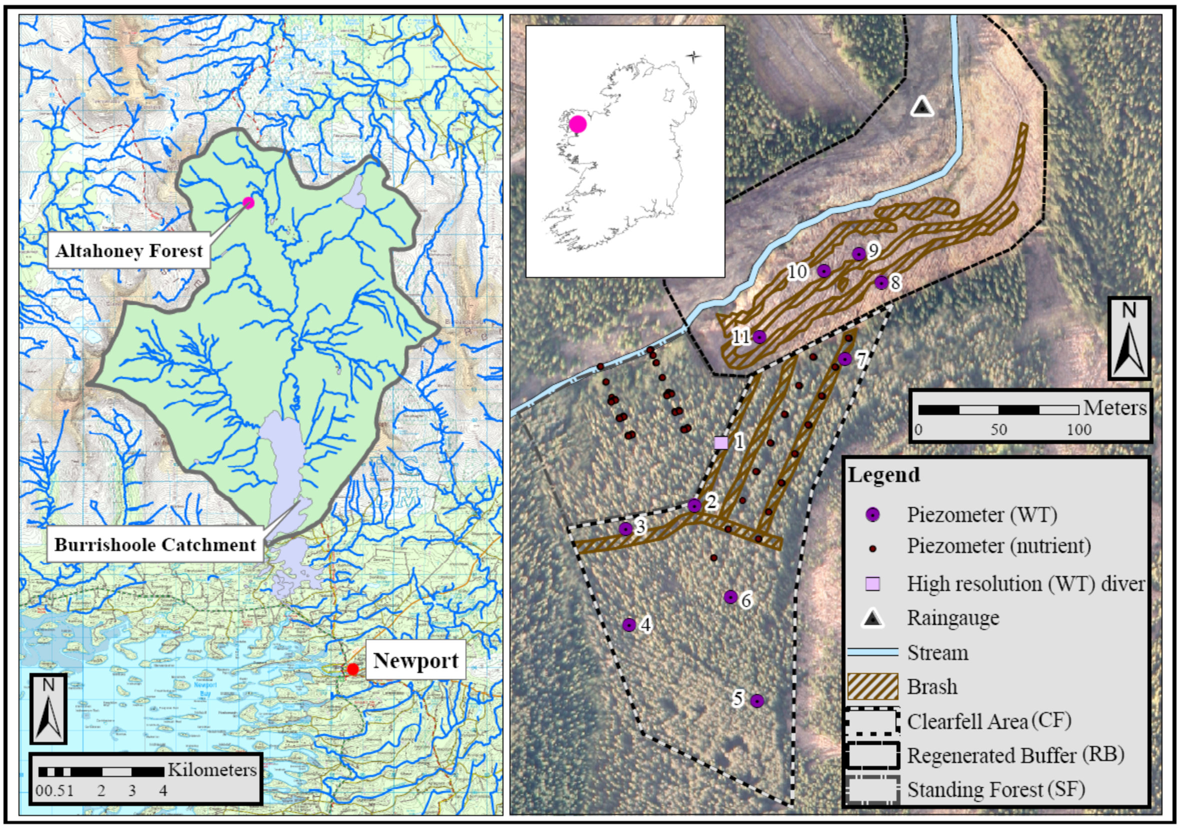

Figure 4.1 Location of the Altaconey Forest with site instrumentation, within the Burrishoole catchment.

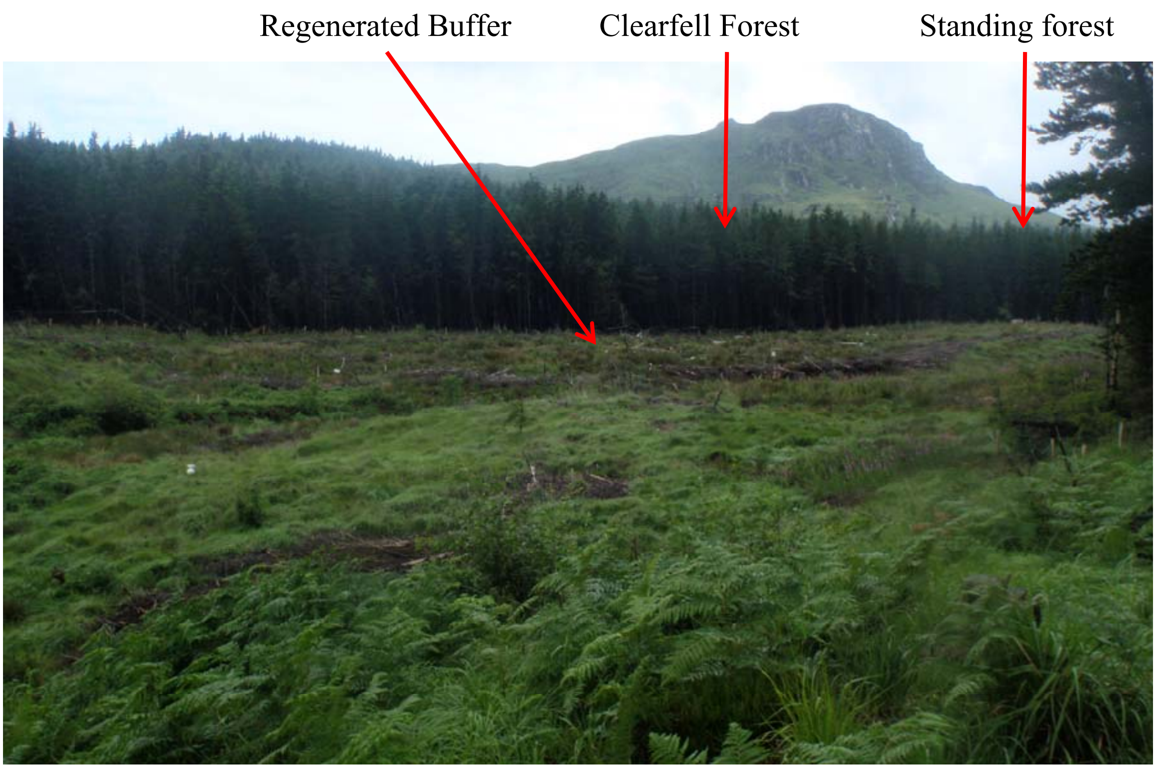

Figure 4.2 View across site from raingauge, taken before clearfelling.

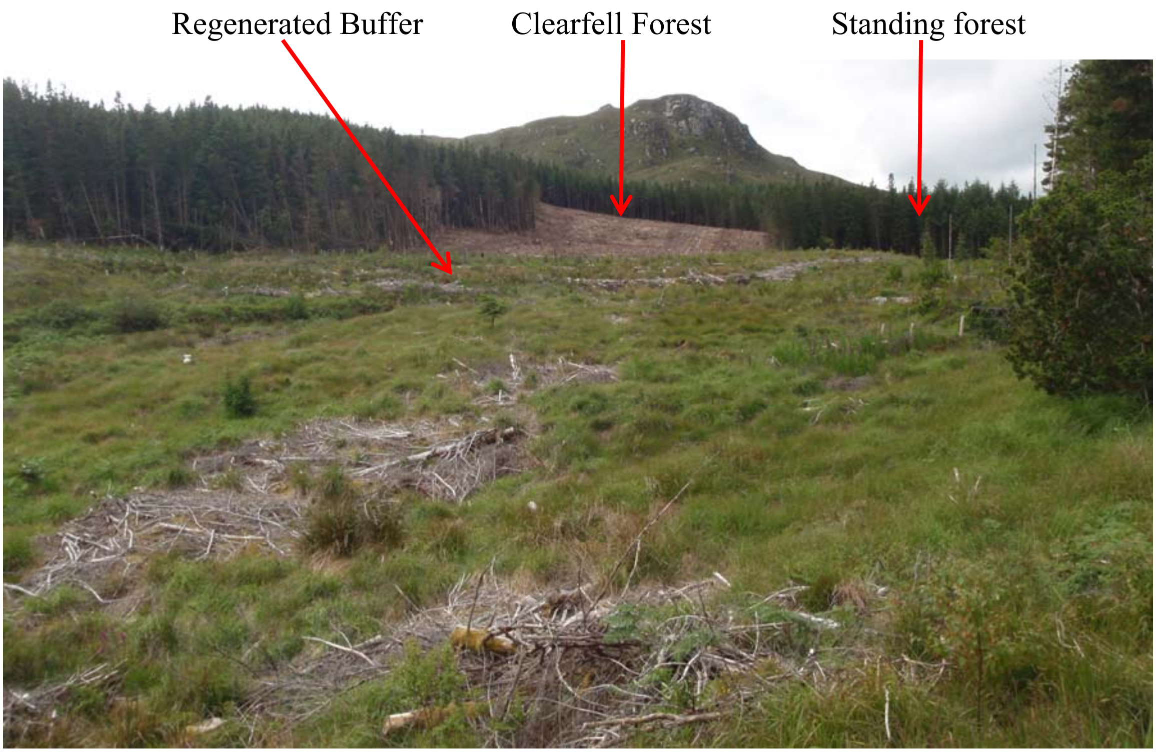

Figure 4.3 View across site from rain gauge, taken after clearfelling.

The site was planted in 1966 with Sitka Spruce (*Picea sitchensis*) and Lodgepole Pine (*Pinus contorta*). In May 2006, an area of 2.49 ha 30 m north, 50 m south and 300 m along the river was clearfelled to create the RB (Ryder et al., 2011) (Figure 4.1, identified as Regenerated Buffer). Bole-only clearfelling was carried out with a harvester and a forwarder with the brash material positioned ahead to create a brash mat on which to drive forward, thus protecting the soil from consolidation. These brash mats remained on site after clearfelling was completed. Typical forest practice would be to arrange these brash mats into regular rows ('windrowing') away from the watercourse when preparing the site for replanting.

In February 2011, clearfelling of an area of 2.61 ha (1230 m^3) of the forest upslope of the RB area began (identified as 'Clearfell Forest' in Figure 4.1). Clearfelling was conducted in a similar manner to the RB. The brash mats in this area also remained *in situ* for the study period. No harvesting took place within the adjacent standing, mature coniferous forest (SF) (identified as 'Standing Forest' in Figure 4.1). The SF is at the same topographical location as the RB, and has a similar slope and peat depth.

4.2.2 Measurement and analysis

The WT and nutrient data for the present study was measured from May 2010 until May 2012. This allowed for examination of two time periods: (1) from 4 to 6 years after clearfelling within the RB (clearfelled in May 2006) and (2) before (9 months) and after (15 months) harvesting of the CF (clearfelled in February 2011). For discussion and analysis, the CF was divided into pre-clearfell (pre-CF) and post-clearfell (post-CF) periods.

Watertable

To investigate the spatial difference in WT depths, eleven piezometers, each with an internal diameter of 40 mm, and an end pipe screen interval of 0.3 m, which was covered with a filter sock, were augured at random locations across 2 sites (4 piezometers in RB and 7 piezometers in CF; Figure 4.1). Contact between the peat and the piezometer was ensured by incorporating a sand infill, with the remainder backfilled with bentonite.

Average depth of installation was approximately 1 m bgl. A high resolution WT mini-diver (OTT Orpheus Mini, Germany), set to record pressure head and water temperature at 30-minute time intervals, was placed at one location in the CF area over the study duration (Figure 4.1). The OTT Orpheus Mini has a pressure equalization capillary tube in the pressure probe cable, which allows the measurement cell to always refer to the current ambient air pressure as a reference. Erroneous measurements due to atmospheric air pressure fluctuations are therefore eliminated. The remaining ten piezometers were manually dipped once-a-month with an electronic water dip meter (Model 101, Solinst, Canada). As the above ordnance datum (m AOD) position of each piezometer was known, WT heights were converted to groundwater heads. Using these data, temporal groundwater flow direction maps were created in ArcGIS (Release Version 9.3, Environmental Systems Research Institute (ESRI), California, USA). The high resolution groundwater head data allowed WT positional trends in all the other piezometers to be inferred.

Rainfall was recorded using a rain gauge (Environmental Measurements Limited, UK) (Figure 4.1). A number of rainfall events and subsequent fluctuations in WT over the study period were tabulated to determine the response rate of the WT to a rainfall event. This analysis was conducted before and after clearfelling, and included initial WT at the beginning of a dry period, volume and timing of daily rainfall, time lag in recovery from deep WTs (to allow elucidation of the recovery time from a deep WT following rain) and relatively shallow WTs (to allow elucidation of the recovery time from a shallow WT following rain when the peat was already saturated). In total, 6 events were analysed, 3 in the pre-CF period (events 1, 2 and 3) and 3 events post-CF (events 4, 5 and 6). These events included dry periods, with associated deep WT, and wetter periods with shallow WTs. The dry period and deep WT at the beginning of the monitoring was not analysed because the initial height of the WT before the dry period began was not known.

Water samples

Another set of multi-level piezometers were installed within the CF (48 piezometers; 24 overlain by brash material with a depth of approximately 0.45 m and 24 in a vegetated area overlain by needles and small branches) and SF (54 piezometers, covering a distance from 45 m to 1 m up-gradient of the river) areas to capture the chemistry of the recently recharged water and that of older shallow groundwater migrating slowly through the aquifer towards the surface waterbody. Each sampling location comprised three multilevel piezometers, each with an internal diameter of 0.011 m, installed to depths of 0.2, 0.5 and 1 m bgl. Each piezometer had a screen interval of 0.1 m at its base and was covered with a filter sock. A steel rod was inserted into the piezometer for support at the installation stage, and the tops of the piezometers were covered to prevent the ingress of rain water.

Water samples from the SF and CF areas were collected during storm events from May 2010 to May 2012 (n=8 dates; 3 pre-CF and 5 post-CF). Any water lodged in the bottom of the piezometers was removed under suction the day before water sampling, and the piezometers were allowed to fill overnight. Shallow groundwater samples were filtered immediately using 0.45 µm filters. Stream water samples were collected within 20 m upstream and downstream of the RB area to determine if nutrient discharge from the site to the stream had occurred. Filtered and un-filtered samples were returned to the laboratory and tested the following day or frozen (-20°C) for testing at a later date. Samples were analysed for (1) dissolved reactive phosphorus (DRP) (2) NH_4^+-N (3) NO_3^--N and (4) total oxidised nitrogen (TON). All water samples were tested in accordance with standard methods (APHA, 2005) using a nutrient analyser (Konelab 20; Thermo Clinical Labsystems, Finland).

Statistical analysis

Depth to the WT and location was analysed with ANOVA (analysis of variance) in Datadesk (Data Description Inc., USA). Nutrient data from shallow groundwater piezometers was $\log_{10}$ transformed and also analysed with ANOVA in Datadesk to ascertain the main sources of variation. Date, depth of soil from which the sample originated, and the location of the sample site were included as explanatory variables.

4.3 Results and Discussion

4.3.1 Watertable

Impact of clearfelling on watertable fluctuations

The WT remained consistently below 0.3 m during the pre-CF period in the CF area. The high resolution WT data within the CF area showed an immediate rise in the WT to within 0.3 m bgl after clearfelling commenced (Figure 4.4). On the first day of clearfelling, the WT rose to 0.27 m bgl after a rainfall event of 41.6 mm. A similar rainfall event (38.4 mm) occurred 2 weeks prior to clearfelling, but the WT only rose to 0.34 m bgl on that occasion. The WT had been at a similar depth for the weeks prior to these events. The final WT recovery position was 0.15 m bgl 10 months after clearfelling began. This rise in WT was despite a lower cumulative rainfall for the time period. Comparing the same number of days pre- and post-CF (277 days), the cumulative rain was greater for the pre-CF time period (2084 mm) compared to the post-CF period (1935 mm) (Figure 4.5 and 4.6). Renou and Farrell (2005) propose that such a change was due to a reduction in transpiration from the trees and increased evapotranspiration from the soil. This rise is also in agreement with other studies (Dubé et al., 1995; Pothier et al., 2003; Jacks and Norrström, 2004; Kaila et al., 2012). The WT fluctuated seasonally between ground level and 0.3 m bgl in the four piezometers (no. 8 – 11; Figure 4.6) positioned in the RB, which had been clearfelled 4 years prior to the present study. The WT in these piezometers did not drop below 0.3 m bgl over the 2-year study period, indicating that the site had reached steady-state WT conditions. This deepest level of 0.3 m bgl, 4 years after clearfelling, may be representative of the WT depth achieved by the clearfelling management change.

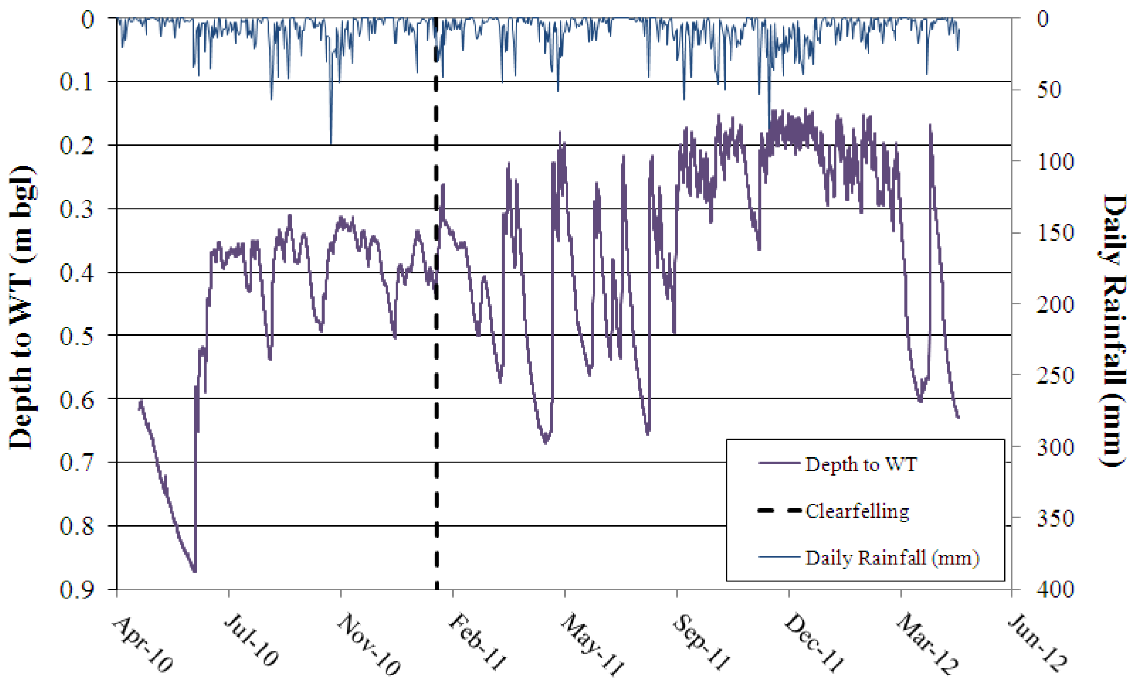

Figure 4.4 Depth to the WT (m bgl) from the high resolution WT diver and daily rainfall (mm) over the two year study period (May 2010 – May 2012) in the Altaconey Forest.

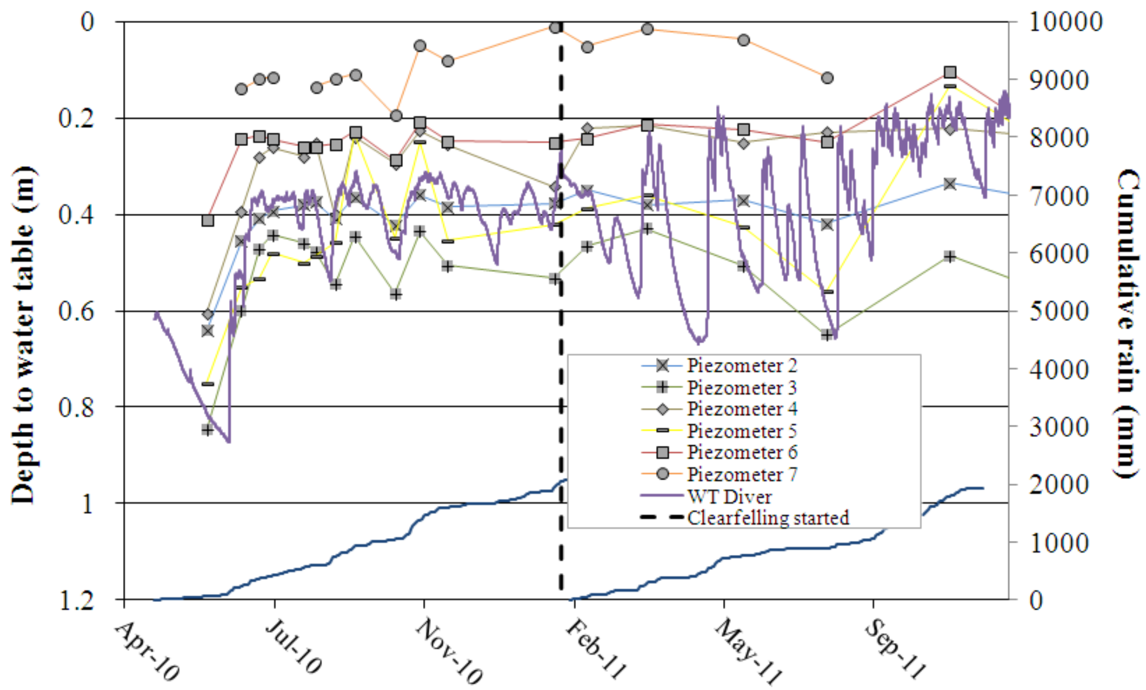

Figure 4.5 Depth to the WT (m bgl) from the high resolution WT diver and the sampling piezometers no. 2 - 7 from the CF over the same time period of 277 days pre- and post-CF in the Altaconey forest. Cumulative rainfall (mm) is on the secondary axis.

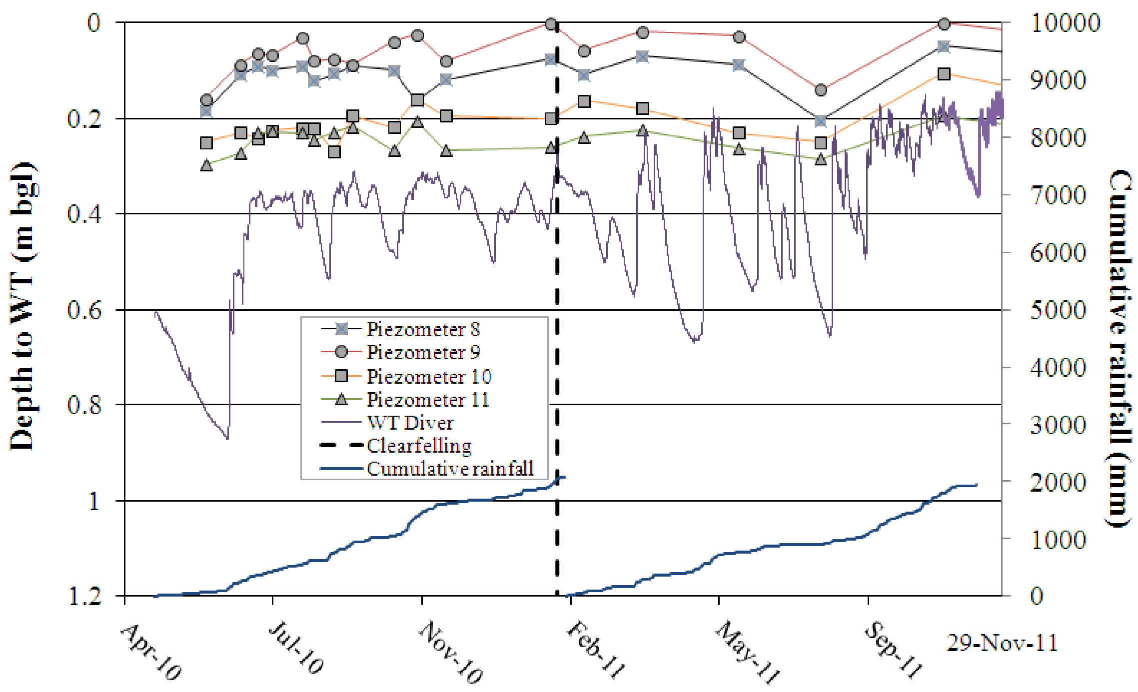

Figure 4.6 Depth to the WT (m bgl) from the high resolution WT diver and the sampling piezometers no. 8 - 11 from the RB over the same time period of 277 days pre- and post-CF in the Altaconey forest. Cumulative rainfall (mm) is on the secondary axis.

Minimum, median and maximum depths to WT pre- and post-CF from the high resolution WT diver and the sampling piezometers (with corresponding fluctuations in WT) are shown in Table 4.1. The minimum depths to WT (bgl) from piezometers in the CF area were all post-CF, with the exception of piezometer 7, which was closest to the RB (Figure 4.1) and was most probably influenced by the WT drawdown caused by the removal of the trees in May 2006.

Date of sampling and location were significant sources of variation for the depth to the WT (ANOVA; $p<0.05$): the depth to the WT in the RB and CF was significantly higher (shallower), while the SF was significantly lower (deeper WT) (ANOVA; $p<0.05$). The temporal groundwater flow direction map (Figure 4.7) also showed that there was a deeper WT in the SF and a higher WT in the RB and CF. Ecosystem respiration measured on site (Chapter 5) showed a higher release of methane (CH_4) from the RB and post-CF due to the proximity of the WT to the ground level and a lower emission of CH_4 from the SF because of the deeper WT. As a result of the larger aerobic zone in the SF, greater quantities of nitrous oxide (N_2O) were produced by partial denitrification compared to the waterlogged RB or post-CF. Under the waterlogged anaerobic conditions, N_2O flux was negligible because denitrification to N_2 was most likely the predominant gaseous N loss pathway. Nitrous oxide production by nitrification and/or partial denitrification is optimum at a water filled pore space (WFPS) of less than 50-70 % (Silvan et al., 2002). High N_2O emissions from the deeper WT in the SF, coupled with low NO_3^--N concentrations throughout the site (Section 4.3.2), indicated that both conditions were present in the study site.

Table 4.1 Minimum, median and maximum depths (m bgl) to the WT, with subsequent fluctuations in the WT, from the high resolution WT diver (Piezometer no.1) and the sampling piezometers no. 2 – 11 in the RB and the CF for the periods pre- and post-clearfelling over the two year study period (May 2010 – May 2012) in the Altaconey Forest.

Piezometer No.	Area	WT depth (m bgl) Pre-CF				WT depth (m bgl) Post-CF			
		Min	Median	Max	Fluctuation	Min	Median	Max	Fluctuation
1*	CF	0.310	0.393	0.872	0.562	0.142	0.312	0.669	0.527
2	CF	0.360	0.389	0.642	0.282	0.335	0.380	0.430	0.095
3	CF	0.435	0.492	0.847	0.412	0.430	0.546	0.680	0.250
4	CF	0.226	0.280	0.605	0.379	0.215	0.238	0.420	0.205
5	CF	0.234	0.469	0.750	0.516	0.133	0.407	0.560	0.427
6	CF	0.210	0.250	0.412	0.202	0.105	0.246	0.280	0.175
7	CF	0.010	0.118	0.195	0.185	0.015	0.052	0.140	0.125
8	RB	0.075	0.104	0.185	0.110	0.048	0.084	0.205	0.157
9	RB	0.000	0.068	0.088	0.088	0.000	0.040	0.140	0.140
10	RB	0.160	0.220	0.270	0.110	0.105	0.175	0.250	0.145
11	RB	0.205	0.238	0.298	0.093	0.194	0.244	0.285	0.091

* high resolution WT diver

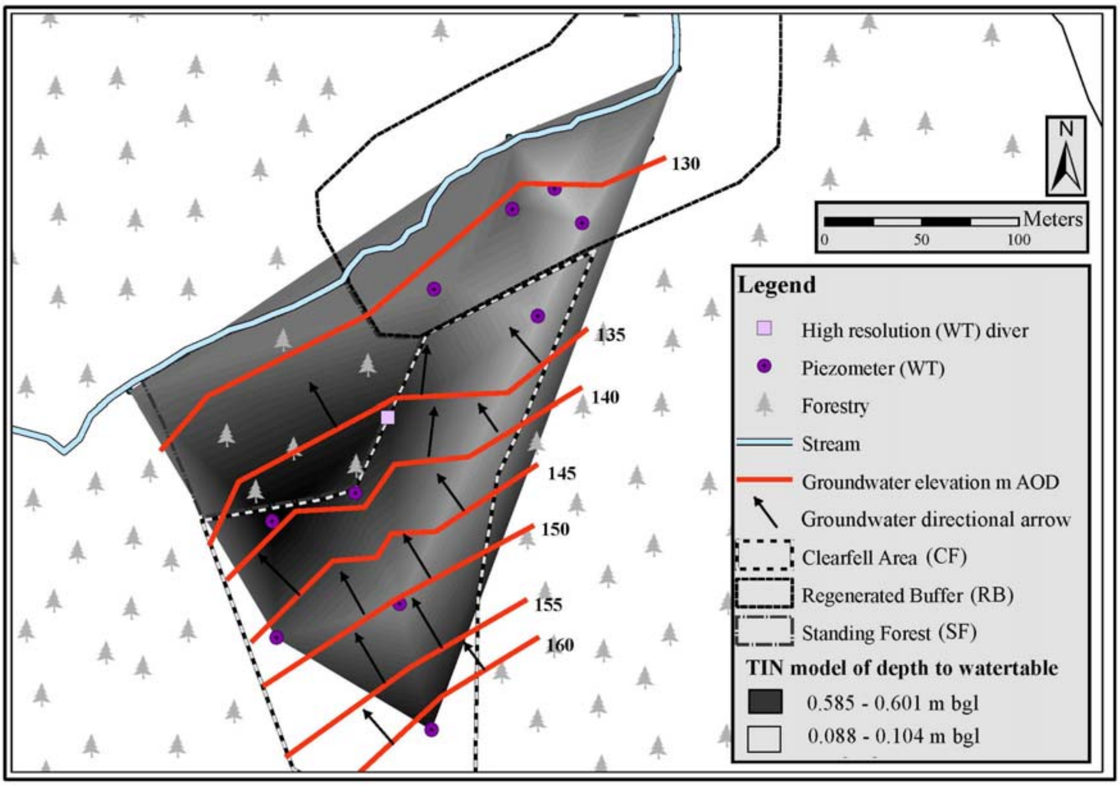

Figure 4.7 On site depth to the WT (m bgl) (taken on 8th July 2010, 6 months prior to clearfelling) in the Altaconey Forest with groundwater flow direction (arrows) based on groundwater heads (m AOD).

Watertable fluctuations with rainfall

The study site received 5611 mm of rainfall over 586 rain days for the duration of the present study (May 2010 to May 2012; 730 days in total). In 2011 alone, there was 3203 mm of rainfall – 1203 mm above the average amount for the catchment. The deepest WT (0.872 m bgl on June 30, 2010) was associated with the driest period of weather (98 mm of rain in the previous 57 days). A number of rainfall events and subsequent fluctuations in WT are shown in Figure 4.8 and Table 4.2. Following prolonged dry periods, there was a time lag between the occurrence of a rainfall event and a rise in WT. This lag was due to the time required to re-saturate the peat. Once the deepest WT was reached and the peat had become recharged from the rainfall, a rise in the WT was measured over a number of days of rainfall. Pre-CF, the three deepest WTs (event 1, 2 and 3) were followed by three large storms of 38.4 mm, 56.8 mm and 88 mm of daily rainfall. These rainfall events and subsequent fluctuations in WT resulted in a drop in WT of approximately 0.131 m over 12 days (average of event 1, 2 and 3). Following this decrease in the WT, it took a number of days of rainfall (average of 20 mm over 4 days) for the WT to reach a minimum bgl position before commencement of recovery. Once a minimum bgl position was reached, an average of 207.8 mm of rainfall over 14 days was required to increase the WT. When the peat was already saturated and the level of the WT was approximately 0.3 m bgl, large rainfall events were not as influential. The antecedent conditions resulted in the considerably different capacity of the peat to receive infiltrating rainfall and subsequent WT fluctuations. A daily rainfall of 45.4 mm fell during a storm just after event 2 (November 7, 2010), but had little impact on the WT. Over 16 days, 245.6 mm of rain fell on the study site and the WT level only fluctuated < 0.001 m. This may be due to the uptake of water by the trees above 0.3 m bgl. Other explanations were examined such as depth of furrows for drainage, pipes or depressions, but no obvious conclusion was reached.

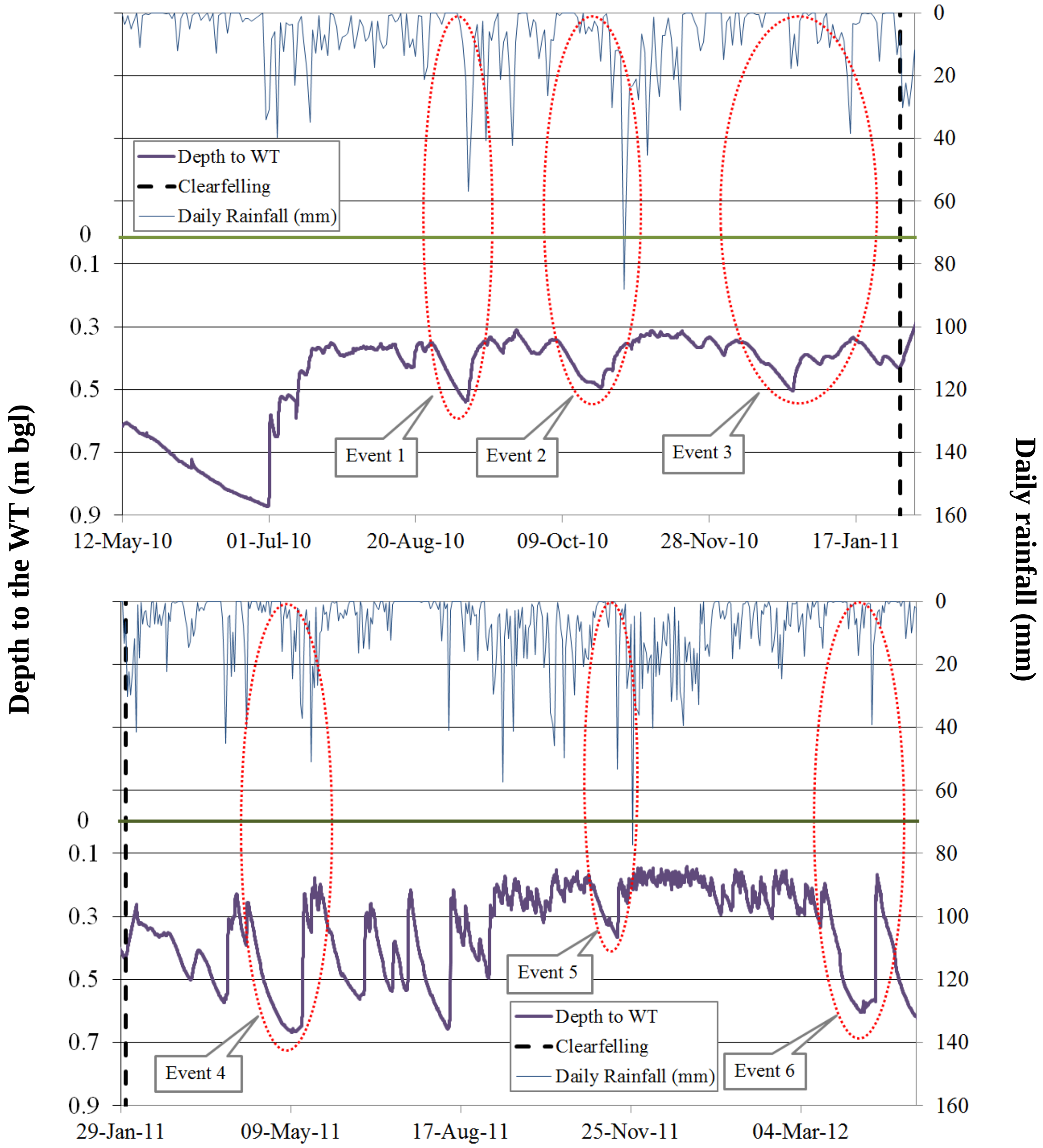

Figure 4.8 Depth to the WT (m bgl) from the high resolution WT diver from the CF area for pre-CF (top) and post-CF (bottom) in the Altaconey forest with highlighted rainfall events and associated fluctuations in the WT. Event magnitude in Table 4.2. Daily rainfall (mm) is on the secondary axis.

Table 4.2 Six rainfall events (n=3 Pre-CF; Event No.1 - 3, n=3 Post-CF; Event No.4 - 6) including duration, volume of rain and subsequent fluctuations in WT in the high resolution WT diver over the two year study period (May 2010 – May 2012) in the Altaconey Forest.

Event No.	Dates	Duration (Days)	Change in WT depth (m)	Total Change (m)	Rain volume (mm)	Description*
1	Aug 25 to Sept 12, 2010	11	0.351 - 0.498	-0.147	6.2	☼ ▼
		2	0.498 - 0.539	-0.041	11.8	☁ ▼
		6	0.539 - 0.345	0.194	146.4	☁ ▲
2	Oct 6 to Nov 4, 2010	8	0.34 - 0.442	-0.102	3	☼ ▼
		7	0.442 - 0.493	-0.051	30.6	☁ ▼
		14	0.493 - 0.324	0.169	320.6	☁ ▲
3	Dec 9 to Dec 30, 2010	16	0.35 - 0.495	-0.145	8	☼ ▼
		2	0.495 - 0.504	-0.009	17.8	☁ ▼
		21	0.504 - 0.334	0.17	156.4	☁ ▲
4	Apr 13 to May 17, 2011	21	0.256 - 0.642	-0.386	18.2	☼ ▼
		6	0.642 - 0.669	-0.027	43.8	☁ ▼
		7	0.669 - 0.227	0.442	122.6	☁ ▲
5	Nov 2 to Nov 18, 2011	14	0.209 - 0.359	-0.15	24.8	☼ ▼
		-	-	-	-	☁ ▼
		2	0.359 - 0.197	0.162	76.8	☁ ▲
6	July 18 to Aug 11, 2011	14	0.196 - 0.548	-0.352	6.4	☼ ▼
		7	0.548 - 0.604	-0.056	50.4	☁ ▼
		10	0.604 - 0.167	0.437	79	☁ ▲

*☼ ▼: Dry weather and drop in WT, ☁ ▼: Rainfall but continued drop in WT, ☁ ▲: Rainfall and rise in WT

Analysis of rainfall events post-CF showed a similar pattern to pre-CF and WT fluctuations (Figure 4.8). The deepest WT was associated with the driest periods, but these periods of little rain were longer and therefore produced deeper WTs, which resulted in a greater drop in WT of approximately 0.296 m over 16 days (average of event 4, 5 and 6). The same pattern of delay in recovery time due to recharge of the unsaturated upper peat layers occurred, with the deepest WTs noted after an average of 47.1 mm of rainfall over 7 days. This was almost twice the rainfall and time period of the pre-CF minimum depth (20 mm over 4 days), but was also twice the overall change in level (pre-CF overall change of 0.131 m and post-CF overall change of 0.296 m). Once the deepest WT was reached and recovery began, it took half the time and rainfall (92.8 mm over 6 days) compared to pre-CF (207.8 mm over 14 days). Equally, following recharge of the WT to above 0.2 m bgl, subsequent high rainfall events had little influence on WT fluctuation. Following event 5, there were 50 days of rain on site, which totalled 965 mm. This amount of rainfall only caused the WT to fluctuate by ± 0.03 m.

Comparisons of storm events with WT fluctuations showed that the WT responded quickly to a rainfall event, if the peat was already saturated (Figure 4.8). Fenton et al. (2011b) showed that time lag of drainage and nutrients through the unsaturated zone can be estimated using three parameters (effective rainfall, effective porosity and depth to WT). Although infiltration rates on peat are low, once water enters an unsaturated zone it can travel quickly. This allows for the rapid vertical response of the WT to a rainfall event, but the slow lateral movement in peatland systems ensures high residence times with correspondingly high denitrification potentials before discharge to nearby surface water bodies. In the current study, high concentrations of both DRP and NH_4^+-N existed in shallow groundwater (Section 4.3.2), but low concentrations exited in the adjoining stream (Chapter 2). This impeded lateral migration may allow for long-term storage of nutrient concentrations within the shallow groundwater, as seen by Fenton et al. (2009), leading to natural attenuation of nutrient plumes before discharge to surface water bodies. Furthermore, the high denitrification potential of forested riparian peatlands, especially in surface peats, further reduces the export of excessive N components from site (Hayakawa et al., 2012).

Restoration potential

In Ireland, peat soils cover up to 20.6 % of the landmass and contain over 75 % of the soil organic C stock (Renou-Wilson et al., 2011). Ireland's forest cover stands at 10 % (698,000 ha) of the total surface area of the island, and 59.6 % of total afforestation is on peat (National Forest Inventory, 2007). The current trend for forestry is plantation on more suitable soil types, leading to increased productivity and enhanced environmental quality (EPA, 2012). However, the legacy of blanket peatland forestry, planted in the 1950s, must be dealt with as most of this forestry is now at harvestable age and the decision to either replant or restore these sites needs to be made (Renou-Wilson et al., 2011). Despite grants from the European Union (EU) and the relatively high productivity of these peatland forests today, the economic viability of such plantations on upland peat is limited (Renou and Farrell, 2005) and over 40 % of the forests have poor production potential (Tierney, 2007).

Coillte, the Irish State's current forest management company, carried out a number of restoration projects on various types of peatlands (raised bogs (Coillte, 2008a), active blanket bogs (Coillte, 2008b) and priority woodlands (Coillte, 2010)) between 2004 and 2009. The removal of conifers on raised bogs elevated the WT from 0.55 m bgl to within 0.1 m bgl over a time period of approximately 3 months after clearfelling, and the installation of drain blocking further elevated it to greater than 0.05 m bgl within 2 months (Coillte, 2008a). Restoration of some active blanket bogs and the removal of non-native conifers at bog woodlands, without the use of drain blocking, have raised WTs to original conditions (Coillte, 2008b; Coillte, 2010).

Results from the present study would indicate that drain blocking may be necessary to restore the WT to approximately 0.1 m bgl, as the steady-state position of the WT in the RB 4 years post-CF was 0.3 m bgl. In the UK, drain blocking was only found to be effective in lowland raised bogs and whole tree harvesting (i.e. the complete removal of all trees and brash material from site) was most effective at raising the WT (Forestry Commission, 2010).

4.3.2 Water samples

Pre- and post-clearfelling

The date of sampling (pre- or post-CF) was a significant source of variation for the nutrient data in the CF (ANOVA; $p<0.05$). The nutrient data from the SF showed no significant difference with date, indicating that there was no change in the study control due to the clearfelling operations (Figures 4.9 and 4.10). Both DRP and NH_4^+-N concentrations were significantly lower before clearfelling and significantly higher after clearfelling in the newly clearfelled area ($p<0.05$, LSD post-hoc test, Figures 4.9 and 4.10). Concentrations of TON were generally similar across all locations sampled, and had a median concentration of 0.12 mg L^{-1} and a maximum concentration of 0.50 mg L^{-1} (results not shown). Similarly, very low values of NO_3^--N (<20 µg L^{-1}) were found across all depths and locations throughout the site and were, in many cases, below the limits of detection.

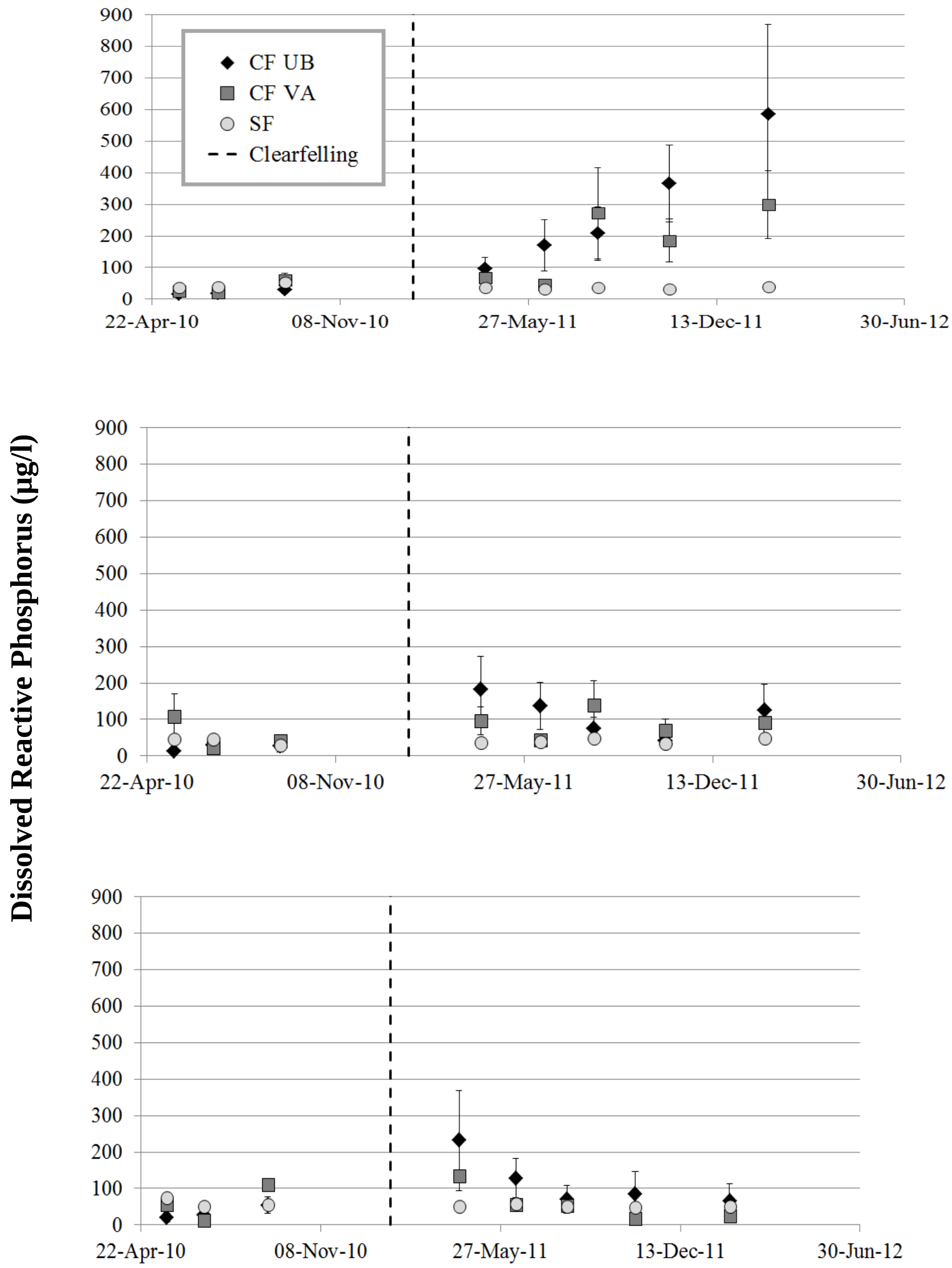

Figure 4.9 Dissolved reactive phosphorus (DRP) (µg L^{-1}) in shallow groundwater at 0.2 m (top), 0.5 m (middle) and 1 m (bottom) depths from the CF area (clearfell forest under brash (CF UB) (n=24) and CF vegetated area (CF VA) (n=24)) and the standing forest (SF) (n=54) over the 2-year study period (May 2010 – May 2012) in the Altaconey Forest. Standard error shown by error bars.

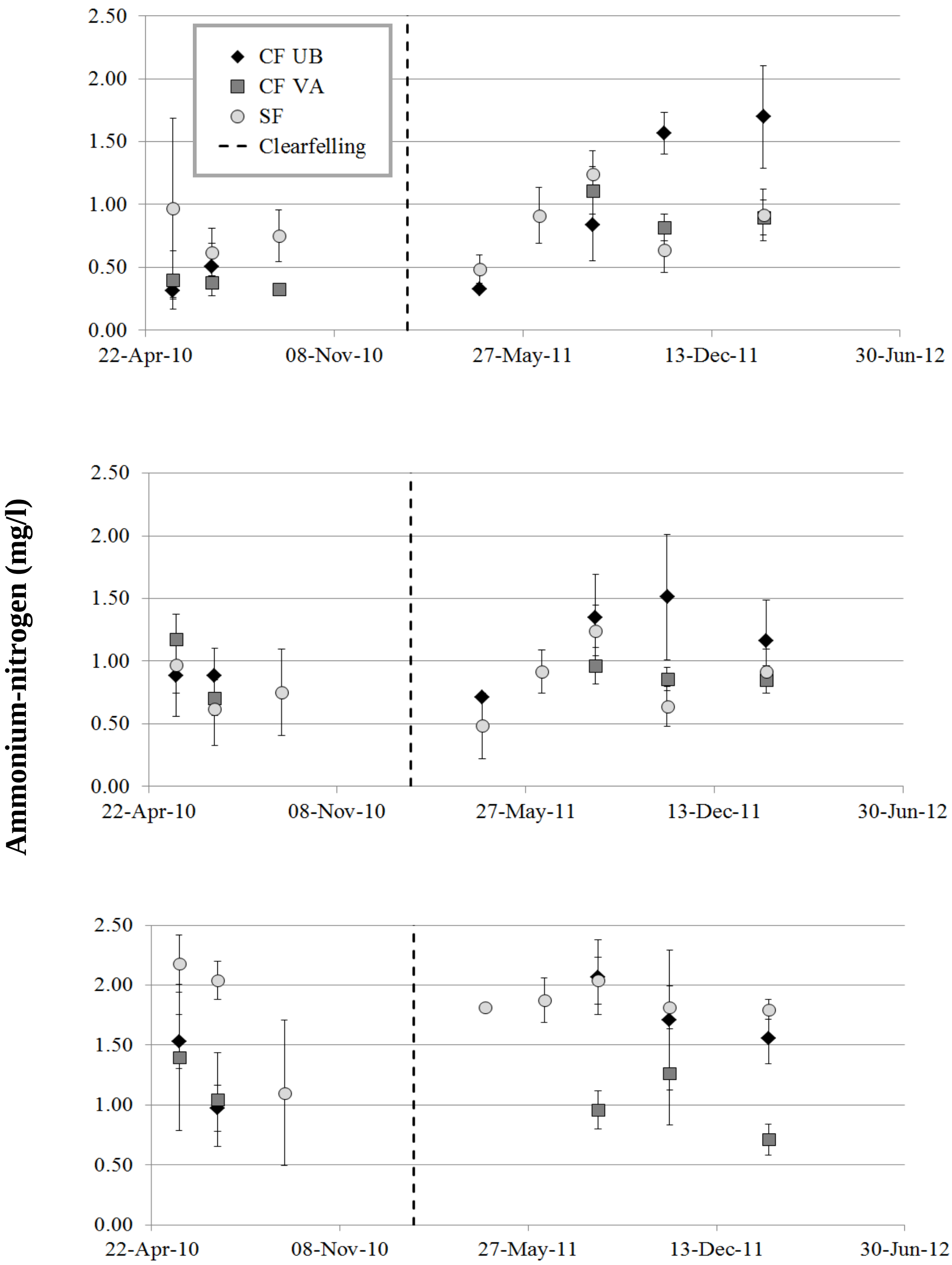

Figure 4.10 Ammonium (NH_4^+-N) (mg L^{-1}) in shallow groundwater at 0.2 m (top), 0.5 m (middle) and 1 m (bottom) depths from the CF area (clearfell forest under brash (CF UB) (n=24) and CF vegetated area (CF VA) (n=24)) and the standing forest (SF) (n=54) over the 2-year study period (May 2010 – May 2012) in the Altaconey Forest. Standard error shown by error bars.

During clearfelling, nutrients can be returned to the soil by practicing bole-only harvesting (Rodgers et al., 2010) and leaving the brash to degrade on site (Chapter 2). Stevens et al. (1995) found that the brash material from a site in North Wales, which was planted with Sitka spruce, contained 291 kg ha^{-1} of N and 32 kg ha^{-1} of P. After clearfelling, 14 kg ha^{-1} of inorganic N (within four years of clearfelling), with the majority of this being NH_4^+-N, and up to 10 kg ha^{-1} of P (within one year after felling) was found in the brash throughfall (Stevens et al., 1995). Large amounts of P can be easily released from brash material, but N varies from small releases (approximately 5 % of N content of brash material) (Stevens et al., 1995) to no obvious gain or loss (Kaila et al., 2012). This is due to the high C:N ratio of the brash material and subsequent high initial N immobilization as opposed to mineralization (Nieminen, 1998). Stevens et al. (1995) found that brash material was a sink of inorganic-N in the first three years after clearfelling, with values in the brash throughfall lower than that in the rain. Needles contain half-to-two thirds of total nutrients of the brash material and the needle drop time period may occur anywhere from 3 to 9 months following clearfelling, depending on local climate and season (Forest Research, 2009). If not correctly managed, bole-only harvesting can lead to elevated concentrations of nutrients in the receiving waters (Rodgers et al., 2010). To prevent this, whole tree harvesting is practiced, but this may lead to the removal of base cations and nutrients from the site, resulting in a lack of adequate supply for the next rotation of forestry or vegetation (Nisbet et al., 1997).

Comparing clearfell forest and standing forest areas

The location of the sampling site, either CF or SF, was a significant source of variation for nutrient concentrations, with DRP concentrations significantly higher in the CF and the NH_4^+-N concentrations significantly higher in the SF ($p<0.05$, LSD post-hoc test).

The higher DRP concentrations in the newly clearfelled forest indicated DRP leaching from the degrading brash material. A similar finding was made in other studies (Stevens et al., 1995; Väänänen et al., 2007; Rodgers et al., 2010). No significant difference was noted in DRP concentrations at different sampling locations of either under brash (CF UB; Figure 4.9) or under vegetation (CF VA; Figure 4.9) in the newly clearfelled area, which indicated that nutrients were leached from the residues over the entire clearfelled area and not just from under the brash mats.

Higher concentrations of water extractable phosphorus (WEP) are found under windrows compared to under vegetated areas with no brash material (Macrae et al., 2005). Water extractable phosphorus testing measures the readily available fraction of the soil P and is used as an indicator of the amount of P that may be carried from a soil by surface runoff in storm events. As there is a strong correlation between WEP measured in peat and DRP concentration in surface runoff (O'Driscoll et al., 2011), high levels of WEP close to the soil surface under windrows of brash material can be indicative of high potential for leaching of P to watercourses. Leaching of P to the stream did not occur on the current study site, as frequent analysis of stream water upstream and downstream of the RB showed that stream concentrations of DRP remained between 4 - 10 µg L^{-1} (Chapter 2), but this may be due to dilution by the receiving waters. Typical forest practice would be to arrange these brash mats into regular rows ('windrowing') away from the watercourse when preparing the site for replanting to prevent excessive nutrient runoff (Collins et al., 2000).

Dissolved reactive phosphorus was significantly higher in the 0.2 m piezometer than in piezometers at other depths in the post-CF area and significantly lower at the 0.2 m depth than at other depths in the SF (p<0.05, LSD post-hoc test). On the same study site, the DRP concentrations were significantly higher under 5 year-old brash mats in the RB than in the adjacent SF (Chapter 2), showing the degradation of the logging residue and leaching of DRP to the soil following clearfelling. Lower concentrations at depths of 1 m could be due to adsorption to mineral layers close to the bedrock or a lower vertical conductivity at this depth.

The concentration of NH_4^+-N was highest in the SF, an area which had a deeper WT than the CF (Figure 4.9). Adamson et al. (2001) found high concentrations of NH_4^+-N in soil water with a deep WT due to the microbes involved in ammonification benefiting from the larger aerobic zone above the lowered WT. Daniels et al. (2012) reported high levels of NH_4^+-N in naturally acidic upland peat catchments due to the NH_4^+-N being incorporated into the microbial biomass and therefore not leaching from the site. This appears to be the case in the current study site, as even though high concentrations, sometimes exceeding 2 mg L^{-1} of NH_4^+-N, were found in the shallow groundwater, the concentration in the adjacent stream did not exceed 0.14 mg L^{-1} (results not shown).

The movement and build-up of NH_4^+-N concentrations in the CF and SF were similar in the CF and SF (ANOVA, p<0.05), but varied in magnitude pre- and post-CF. This highlighted the importance of the fluctuation of the WT, as opposed to N leaching from degrading brash material, in the N cycle. Negligible, or very low, N releases from degrading brash material have been reported from other peatland sites in Finland (Kaila et al., 2012) and Wales (Stevens et al., 1995), indicating that N export after clearfelling is most likely not from logging residues (Nieminen, 2004).

Concentrations of NH_4^+-N were significantly lower at the 0.2 m depth and significantly higher at the 1 m depth (p<0.05, LSD post-hoc test) in both CF and SF areas. The presence of NH_4^+-N and low concentrations of NO_3^--N in the aerobic upper peat layers was most likely due to the nitrification of NH_4^+-N to NO_3^--N. The low levels of NO_3^--N indicated that this process was likely followed by denitrification of NO_3^--N to N_2 or N_2O during occasional water logging of the soil. A high N_2O release was measured from ecosystem respiration sampling carried out in the SF (Section 4.3.1 and Chapter 5), which supports this hypothesis. Jacks and Norrström (2004) also found that more NO_3^- reduction occurred in the upper 0.15 – 0.20 m of peat in forested wetland buffers.

Higher NH_4^+-N concentrations were present at the 1 m depth, possibly due to the process of DNRA, which is thought to require a nitrate-limited environment with excessive labile carbon (Stark and Richards, 2008), provided here by the organic peat. The elevated NH_4^+-N, coupled with limited NO_3^--N concentrations and site conditions, indicated that DNRA, as opposed to denitrification, was occurring at this depth (Necpalova et al., 2012). Indeed, the C:NO_3^- ratio, rather than the redox potential of the soil *per se,* is considered to be the principle factor regulating nitrate partitioning between denitrification and DNRA, with C:NO_3^- ratios greater than 12 considered to be required for substantial DNRA (Fazzolari et al., 1998). Dissimilatory nitrate reduction to ammonium, carried out by strictly anaerobic bacteria (Necpalova et al., 2012), results in the production of NH_4^+-N (Scott et al., 2008). This NH_4^+-N is incorporated into the microbial biomass in the peat and does not leach to receiving waters (Daniels et al., 2012), providing potential long-term attenuation of the pollutant (Fenton et al., 2009). Similar concentrations and patterns in inorganic-N to the current study site (but mostly as NO_3^--N) were observed by Stevens et al. (1995) in a freely draining ferric stagnopodzol, and were attributed to active nitrification in the freely draining soil. In contrast, Titus and Malcolm (1991) found that, due to a lack of nitrification following a rise in the WT after clearfelling, NH_4^+-N dominated the inorganic concentration in shallow groundwater.

4.4 Conclusions

Clearfelling of a standing forest on blanket peat raised the WT, and was due to a reduction in transpiration from the trees and increased evapotranspiration from the soil. By the end of the study, the WT recovered to 0.15 m bgl, which indicated that there may be potential for restoration on the site to a pre-forestation state, if reforestation does not take place.

Different transformational processes occurred in clearfelled and standing forest areas, which resulted in different nutrient speciation. An elevated nutrient concentration in shallow groundwater was measured in the clearfelled area compared to the undisturbed standing forest. Phosphorus concentrations, measured in this study as DRP, were more affected by the decay from the logging residue, particularly from the upper 0.2 m of the peat layer, than by fluctuations in WT. The fluctuations in WT in the SF, which had the deepest WT of all areas, led to changes in microbial activity, which produced an average NH_4^+-N concentration of 0.82 mg L^{-1}. Lower NH_4^--N concentrations were measured in the upper 0.2 m, due to nitrification, but low NO_3^--N concentrations were also measured, indicating denitrification during periods of shallow WT. Dissimilatory nitrate reduction to ammonium was suspected to be occurring at deeper depths due to the high concentrations of NH_4^+-N, low NO_3^--N and an excessive labile carbon source from the organic peat. Nutrient discharge to the adjacent watercourse was negligible due to the low hydraulic conductivity of the blanket peat on site.

4.5 Acknowledgements

This work was funded by the Department of Agriculture, Fisheries and Food and the Environmental Protection Agency under the STRIVE program 2007 – 2013. The authors wish to acknowledge the input of Coillte in allowing access to state forestry for this project. A special thanks to Dr. Michael Rodgers and all at the Marine Institute for their continued help throughout this project and for supplying the orthophotography.

References

Adamson JK, Scott WA, Rowland AP, Beard GR. Ionic concentrations in a blanket peat bog in northern England and correlations with deposition and climate variables. Euro J Soil Sci 2001; 52: 69-79.

APHA. Standard methods for the examination of water and wastewater. American Public Health Association, Washington, 2005.

Blodau C. Carbon cycling in peatlands - A review of processes and controls. Environ Rev 2002; 10: 111-134.

Cabezas A, Gelbrecht J, Zwirnmann E, Barth M, Zak D. Effects of degree of peat decomposition, loading rate and temperature on dissolved nitrogen turnover in rewetted fens. Soil Biol Biochem 2012; 48: 182-191.

Coillte. Project Results Booklet - Restoring Raised Bogs in Ireland (LIFE04 NAT/IE/000121). Coillte, Newtownmountkennedy, 2008a. (Accessed on 2nd June 2012) http://www.raisedbogrestoration.ie

Coillte. Technical Final Report for Restoring Active Blanket Bog in Ireland (LIFE02 NAT/IRL/8490). Coillte, Mullingar, 2008b. . (Accessed on 2nd June 2012) http://www.irishbogrestorationproject.ie

Coillte. Technical Final Report for Restoring Priority Woodland Habitats in Ireland (LIFE05 NAT/IRL/000182). Coillte, Mullingar, 2010. . (Accessed on 2nd June 2012) http://www.woodlandrestoration.ie/publications.php

Collins KD, Gallagher G, Gardiner JJ, Hendrick E, McAree A. Code of Best Forest Practice - Ireland. Forest Service, Dublin, 2000.

Conaghan J. The restoration of blanket bogs in Ireland. Irish Timber and Forestry 2003; 12:5

Cummins T, Farrell EP. Biogeochemical impacts of clearfelling and reforestation on blanket peatland streams I. phosphorus. For Ecol Manag 2003; 180: 545-555.

Daniels SM, Evans MG, Agnew CT, Allott TEH. Ammonium release from a blanket peatland into headwater stream systems. Environ Poll 2012; 163: 261-272.

Dubé S, Plamondon AP, Rothwell RL. Watering up After Clear-Cutting on Forested Wetlands of the St. Lawrence Lowland. Water Resour Res 1995; 31: 1741-1750.

EPA. Water Framework Status Update based on Monitoring Results 2007-2009, Johnstown Castle Estate, Ireland., 2011.

EPA. Irelands Environment. An Assessment, Johnstown Castle Estate, Ireland., 2012.

Erwin K. Wetlands and global climate change: the role of wetland restoration in a changing world. Wetl Ecol Manag 2009; 17: 71-84.

Farrell EP, Boyle G. Peatland forestry in the 1990s. 1. Low-level blanket bog. Irish Forestry 1990; 47: 69-78.

Fazzolari E, Nicolardot B, and Germon JC. Simultaneous effects of increasing levels of glucose and oxygen partial pressures on denitrification and dissimilatory reduction to ammonium in repacked soil cores. Eur J Soil Biol 1998; 34; 47–52.

Fenton O, Healy MG, Henry T, Khalil MI, Grant J, Baily A, et al. Exploring the relationship between groundwater geochemical factors and denitrification potentials on a dairy farm in southeast Ireland. Ecol Eng 2011a; 37: 1304-1313.

Fenton O, Richards KG, Kirwan L, Khalil MI, Healy MG. Factors affecting nitrate distribution in shallow groundwater under a beef farm in South Eastern Ireland. J Environ Manag 2009; 90: 3135-3146.

Fenton O, Schulte RPO, Jordan P, Lalor STJ, Richards KG. Time lag: a methodology for the estimation of vertical and horizontal travel and flushing timescales to nitrate threshold concentrations in Irish aquifers. Environ Sci Policy 2011b; 14: 419-431.

Forest Research. Guidance on site selection for brash removal. The Research Agency of the Forest Commission, 2009.

Forestry Commission. Restoring afforested peat bogs: results of current research, Roslin, UK, 2010.

Hayakawa A, Nakata M, Jiang R, Kuramochi K, Hatano R. Spatial variation of denitrification potential of grassland, windbreak forest, and riparian forest soils in an agricultural catchment in eastern Hokkaido, Japan. Ecol Eng 2012; 47: 92-100.

Jacks G, Norrström AC. Hydrochemistry and hydrology of forest riparian wetlands. For Ecol Manag 2004; 196: 187-197.

Joosten H, Clarke D. Wise Use Of Mires And Peatlands, A Framework for Decision-making: International Mire Conservation Group & International Peat Society, 2002.

Kaila A, Asam ZUZ, Sarkkola S, Xiao L, Laurén A, Vasander H, et al. Decomposition of harvest residue needles on peatlands drained for forestry - Implications for nutrient and heavy metal dynamics. For Ecol Manag 2012; 277: 141-149.

Koehler AK, Sottocornola M, Kiely G. How strong is the current carbon sequestration of an Atlantic blanket bog? Global Change Biol 2011; 17: 309-319.

Lewis C, Albertson J, Zi T, Xu X, Kiely G. Hydrological response of afforested peat catchment. Hydrol Process 2012: n/a-n/a. (DOI 10.1002/hyp.9486)

Macrae ML, Devito KJ, Strack M, Waddington JM. Effect of water table drawdown on peatland nutrient dynamics: implications for climate change. Biogeochemistry 2012: 1-16.

Macrae ML, Redding TE, Creed IF, Bell WR, Devito KJ. Soil, surface water and ground water phosphorus relationships in a partially harvested Boreal Plain aspen catchment. For Ecol and Manag 2005; 206: 315-329.

Martikainen PJ, Nykänen H, Crill P, Silvola J. Effect of a lowered water table on nitrous oxide fluxes from northern peatlands. Nature 1993; 366: 51-53.

McConnell B, Gatley S. Bedrock Geology of Ireland. Derived from the Geological Survey of Ireland 1:100,000 Bedrock Map Series and the Geological Survey of Northern Ireland 1:250,000 Geological Map of Northern Ireland. 2006.

National Forest Inventory. National Forest Inventory Republic of Ireland - Results. In: Service F, editor. Forest Service, Wexford, 2007.

Necpalova M, Fenton O, Casey I, Humphreys J. N leaching to groundwater from dairy production involving grazing over the winter on a clay-loam soil. Sci Tot Environ 2012; 432: 159-172.

Nieminen M. Changes in nitrogen cycling following the clearcutting of drained peatland forests in southern Finland. Boreal Environ Res 1998; 3: 9-21.

Nieminen M. Export of dissolved organic carbon, nitrogen and phosphorus following clear-cutting of three Norway spruce forests growing on drained peatlands in southern Finland. Silva Fennica 2004; 38: 123-132.

Nisbet T, Dutch J, Moffat A. Whole-tree harvesting - A guide to good practice. In: Forestry Commission, editor. Forestry Commission Practice Guide, Edinburgh, UK, 1997.

O'Driscoll C, Rodgers M, O'Connor M, Asam Z-u-Z, de Eyto E, Poole R, et al. A Potential Solution to Mitigate Phosphorus Release Following Clearfelling in Peatland Forest Catchments. Wat Air Soil Poll 2011: 1-11.

Parish F, Sirin A, Charman D, Joosten H, Minayeva T, Silvius M, et al. Assessment on Peatlands, Biodiversity and Climate change: Main Report, Global Environment Centre; Kuala Lumpur and Wetlands International; Wageningen, 2008. (Accessed on 15th June 2012) http://www.imcg.net/media/download_gallery/books/assessment_peatland.pdf

Pothier D, Prévost M, Auger I. Using the shelterwood method to mitigate water table rise after forest harvesting. For Ecol and Manag 2003; 179: 573-583.

Renou-Wilson F, Bolger T, Bullock C, Convery F, Curry J, Ward S, et al. BOGLAND: Sustainable Management of Peatlands in Ireland. STRIVE 75. EPA., Johnstown Castle, Co. Wexford, Ireland, 2011. (Accessed on 14th June 2012) http://www.epa.ie/downloads/pubs/research/land/name,31495,en.html

Renou F, Cummins T. Soil as a key to sustainable forest management. In: Convery F, Feehan J, editors. Achievement and Challenge Rio + 10 in Ireland. The Environmental Institute, University College Dublin, 2002, pp. 85-90. (Accessed on 14th March 2012) http://www.ucd.ie/ferg/Research/Projects/BOGFOR/Renou%20%20Cummins%20Rio10_Dublin_2002.pdf

Renou F, Farrell EP. Reclaiming peatlands for forestry: the Irish Experience. Restoration of Boreal and Temperate Forests. CRC Press, Boca Raton, 2005.

Renou F, Jones SM, Farrell EP. Leaching of phosphorus fertiliser applied on cutaway peatland forests recently established in central Ireland. In: Rochefort L, Daigle JY, editors. Sustaining our Peatlands: 11th International Peat Congress, Québec, 2000, pp. 984–990.

Rodgers M, O'Connor M, Healy MG, O'Driscoll C, Asam Z-u-Z, Nieminen M, et al. Phosphorus release from forest harvesting on an upland blanket peat catchment. For Ecol Manag 2010; 260: 2241-2248.

Rodgers M, O'Connor M, Robinson M, Muller M, Poole R, Xiao L. Suspended solid yield from forest harvesting on upland blanket peat. Hydrol Process 2011; 25: 207-216.

Ryder L, de Eyto E, Gormally M, Sheehy Skeffington M, Dillane M, Poole R. Riparian zone creation in established coniferous forests in Irish upland peat catchments: Physical, chemical and biological implications. Biol and Environ 2011; 111.

Scott JT, McCarthy MJ, Gardner WS, Doyle RD. Denitrification, dissimilatory nitrate reduction to ammonium, and nitrogen fixation along a nitrate concentration gradient in a created freshwater wetland. Biogeochemistry 2008; 87: 99-111.

Silvan N, Regina K, Kitunen V, Vasander H, Laine J. Gaseous nitrogen loss from a restored peatland buffer zone. Soil Biol Biochem 2002; 34: 721-728.

Sottocornola M, Kiely G. Energy fluxes and evaporation mechanisms in an Atlantic blanket bog in southwestern Ireland. Wat Resour Res 2010; 46.

Stark CH, Richards KG. The continuing challenge of agricultural nitrogen loss to the environment in the context of global change and advancing research. Dynamic Soil, Dynamic Plant 2008; 2: 1-12.

Stevens PA, Norris DA, Williams TG, Hughes S, Durrant DW, Anderson MA, et al. Nutrient losses after clearfelling in Beddgelert Forest: A comparison of the effects of conventional and whole-tree harvest on soil water chemistry. Forestry 1995; 68: 115-131.

Strahler AN. Quantitative geomorphology of drainage basin and channel networks. Handbook of Applied Hydrology 1964.

Tierney D. Environmental and social enhancement of forest plantations on western peatlands - a case study. Irish Forestry 2007; 64: 5-16.

Titus BD, Malcolm DC. Nutrient changes in peaty gley soils after clearfelling of sitka spruce stands. Forestry 1991; 64: 251-270.

Väänänen R, Kenttämies K, Nieminen M, Ilvesniemi H. Phosphorus retention properties of forest humus layer in buffer zones and clear-cut areas in southern Finland. Boreal Environ Res 2007; 12: 601-609.

Väänänen R, Nieminen M, Vuollekoski M, Nousiainen H, Sallantaus T, Tuittila ES, et al. Retention of phosphorus in peatland buffer zones at six forested catchments in southern Finland. Silva Fennica 2008; 42: 211-231.

Von Arnold K, Weslien P, Nilsson M, Svensson BH, Klemedtsson L. Fluxes of CO2, CH4 and N2O from drained coniferous forests on organic soils. For Ecol Manag 2005; 210: 239-254.

Vymazal J. Removal of nutrients in various types of constructed wetlands. Sci Tot Environ 2007; 380: 48-65.

Wichtmann W, Wichmann S. Environmental, Social and Economic Aspects of a Sustainable Biomass Production. J Sustain Energ Environ 2011; Special Issue: 77-81.

Chapter 5

The contents of this chapter have been submitted to Science of the Total Environment. Joanne Finnegan developed the experimental design and collected, analysed and synthesized the experimental data. She is the primary author of this article. Dr. Mark G. Healy contributed to the experimental design and paper writing. Dr. John T. Regan assisted with sampling and paper editing. Dr. Gary Lanigan assisted with the experimental design, data analysis and paper editing. Dr. Owen Fenton assisted with paper editing.

Greenhouse gas emissions from forestry on peatland

J. Finnegan[1], J.T. Regan[1], G.J. Lanigan[2], O. Fenton[2], M.G. Healy[1]

[1]*Civil Engineering, National University of Ireland, Galway, Ireland*
[2]*Teagasc, Environmental Research Centre, Johnstown Castle, Wexford, Ireland*

ABSTRACT

Land-use change (LUC) from peatland to forestry and subsequent deforestation can lead to large-scale changes in ecosystem carbon (C) and nitrogen (N) dynamics, and generally leads to an increase in carbon dioxide (CO_2) and nitrous oxide (N_2O) loss and a decrease in methane (CH_4) as the soil dries and the bacterial conditions change. The aim of this study was to examine, over a period of one year, changes to greenhouse gas (GHG) fluxes (measured as soil respiration flux) across four different areas of forestry and peatland. These areas were: (1) a regenerated riparian peatland buffer (RB) clearfelled 5 years before the present study (2) a recently clearfelled coniferous forest (CF) (3) a virgin peat site (VP) and (4) a standing mature coniferous forest (SF). Results show a high quantity of CO_2 being emitted from the RB (average flux of 30.8 kg C ha^{-1} d^{-1}), a high flux of CH_4 from both the VP (average flux of 106±30 g CH_4-C ha^{-1} d^{-1}) and the CF after clearfelling (average flux of 173±57 g CH_4-C ha^{-1} d^{-1}), and a low efflux of N_2O (average flux of 1.07±0.52 g N_20-N ha^{-1} d^{-1}) from all sites. The effect of clearfelling in the CF could be observed within 6 months, with significant increases in soil respiration from 11±2 kg CO_2-C ha^{-1} d^{-1} to 19±2 kg CO_2-C ha^{-1} d^{-1}. The rise in the watertable post-clearfell (0.18 m within 4 months of the beginning of clearfelling), produced the highest cumulative CH_4-C release of 65±40 kg CH_4-C ha^{-1} y^{-1} and caused a decrease in average N_2O flux from 1.7 g N_2O-N ha^{-1} d^{-1} to 0.7 g N_2O-N ha^{-1} d^{-1}. Considerable variation was observed in fluxes due to variation in watertable depth and the presence/absence of a brash layer. Ultimately, watertable depth was the controlling factor governing GHG release and sequestration within the ecosystem.

5.1 Introduction

Forests and forest soils worldwide contain one trillion tonnes of carbon (C) (Watson et al., 2000) and can sequester 2.6 Gt (gigatonnes) of C every year (UNEP, 2009). The majority of the world's C stocks are in forest ecosystems and wetlands (UNEP, 2009). These wetlands contain organic-rich peat soil, cover an approximate area of 4 million km^2 (3 %) of the world's landmass (Bain et al., 2011), and are believed to contain one-third of the world's soil C content and 10 % of the global freshwater supply (Joosten and Clarke, 2002).

Land-use change (LUC) from wetland systems to forestry and subsequent deforestation can lead to large-scale changes in ecosystem C and nitrogen (N) dynamics (IPCC, 2006). Afforestation assists in both reducing non-methane greenhouse gas (GHG) emissions and sequestering atmospheric carbon dioxide (CO_2) via photosynthesis. Although afforestation of peatlands increases the total amount of C sequestered, this is primarily in the woody biomass, and in the long term, soil C stocks actually decrease (Hargreaves et al., 2003; Byrne and Farrell, 2005). Deforestation, followed by draining of the soil, results in the release of up to 0.5 Gt of CO_2 per year, which is 14 % of the total annual anthropogenic emissions (UNEP, 2009). The Inter-governmental Panel on Climate Change (IPCC) estimated that between 0.5 and 2.7 Gt of C was emitted annually and was mainly due to deforestation in the tropics (IPCC, 2007).

Peat soils in Ireland cover up to 20.6 % of the landmass and contain over 75 % of the soil organic C stock (Renou-Wilson et al., 2011). Forestry activities, including afforestation and deforestation, have a significant impact on emissions, and account for just under 0.5 Mt CO_2 yr^{-1} (O'Brien, 2007). Afforestation and the promotion of sustainable forest management practices were identified in the Kyoto protocol as methods to reduce net emissions of GHG (Kyoto Protocol, 1997). An estimated 59.6 % (417,200 ha) of Irish forestry is on peat (National Forest Inventory, 2007) and approximately 300,000 ha of this is in upland peat areas (Rodgers et al., 2010). The effects of afforestation on the soil organic carbon (SOC) content (Wellock et al., 2011), C stocks and sequestration (Byrne and Milne, 2006) of peatland in Ireland has been studied, but the impacts of deforestation on these areas is relatively unknown. The Environmental Protection Agency (EPA) report on the protocol for sustainable management of peatland in Ireland, BOGLAND (Renou-Wilson et al., 2011), specifically states that research should be carried out in western peatland forests to determine the effects of management options on GHG emissions. The acquisition of such information is vital, due to its role in the overall balance of GHG release nationwide.

Carbon dioxide, methane (CH_4) and nitrous oxide (N_2O) are regarded as the most important GHGs, accounting for an estimated 80 % of the total global warming potential (IPCC, 2001). Land-use change from peatland to forestry generally leads to an increase in CO_2 and N_2O loss and a decrease in CH_4 as the soil dries and the bacterial conditions change (IPS, 2008). The uptake of CO_2 occurs via tree photosynthesis, while CO_2 release is principally associated with autotrophic respiration (emission by plants), the decomposition of organic matter, and the subsequent heterotrophic respiration and combustion of biomass (IPCC, 2006). Globally, forest soils act as sinks for CH_4 and can uptake approximately 30 Tg (30 million tonnes) annually (IPCC, 2001). In contrast, virgin peat soils act as a source of CH_4 due to a higher soil water content, which produces anaerobic conditions and methanogenesis (the formation of CH_4 by microbes) (IPCC, 2006). Within pristine ombrotrophic peatland systems, there is little N_2O efflux due to the fact that both mineral N pools are low and because anaerobic conditions promote total denitrification of any nitrate (NO_3^-) to N_2 (Van Beek et al., 2004). Upon drainage, increases in soil redox potential stimulate the biological processes of nitrification and partial denitrification, which results in a flux of N_2O between the soil and atmosphere (IPCC, 2006).

Greenhouse gas fluxes from forestry depends primary on soil redox potential and the pools of available C and N for mobilisation. These, in turn, depend on a number of environmental factors including: (1) drainage or effective rainfall (soil water content, depth of unsaturated zones, and fluctuations in the watertable (WT)) (2) air and soil temperature, and (3) amount of brash material (logging residues) remaining on site. Factors such as removal of litter and brash, disturbance of roots, and the exposure, or mixing, of soil horizons have an impact on the extent and direction of GHG fluxes due to both alteration in soil water content and in the amount of labile C and N inputted into the system (Fernandes et al., 2010). The use of heavy machinery during clearfelling can impact on the depth and fluctuations of the WT (Sottocornola and Kiely, 2010). It can also lead to compaction of the soil (Zerva and Mencuccini, 2005), which may reduce soil macroporosity, air diffusion and water infiltration, but may increase soil water content, leading to increased anaerobic conditions. Soil water content can either increase after harvesting due to lack of trees to uptake the water from the soil (Von Arnold et al., 2005), or, in mainly tropical forests, can decrease due to elevated evapotranspiration caused by the removal of the tree canopy and exposure of the soils to sunlight (Hashimoto et al., 2004). These changes in water content in the soil can have a large impact on the yield of the trees and the amount of biomass produced, all of which impact on the GHG emissions (Von Arnold et al., 2005). The soil water content and depth of the vadose zone also ultimately affects the balance between both aerobic and anaerobic respiratory processes, as well as the ratio of partial to total de-nitrification, and can result in large shifts in microbial and fungal dynamics (Von Arnold et al., 2005; Mäkiranta et al., 2009, 2010). With drainage of the soil, there is an increase in the void space, which allows for aerobic digestion and enhanced CO_2 and N_2O release (IPS, 2008). In Sweden, Joosten and Clarke (2002) found that CO_2 and N_2O emissions were significantly higher from drained sites compared to un-drained mire (peatland in which peat is formed), while the opposite was true for CH_4 emissions, with higher releases of CH_4 from un-drained mire. Increased soil bulk density by soil compaction may also decrease CH_4 consumption (Bradford et al., 2000).

The CH_4 flux for well-drained soils is as a result of CH_4 oxidation and production within the soil profile, and the decline of CH_4 oxidation, subsequent to harvesting, may be due to elevated levels of inorganic N in the soil (Bradford et al., 2000), or a rising of the WT in peat soils (Renou-Wilson et al., 2011). Methane uptake decreases with increasing soil water content as a result of limited diffusion of CH_4 from the soil surface to the atmosphere (Morishita et al., 2005). The production of CH_4 requires an anoxic environment and therefore generally occurs within the saturated zone of the soil (Von Arnold et al., 2005). The amount of CH_4 oxidised depends on the size of the anoxic environment and the quantity of CH_4 released to the atmosphere.

Soil respiration responds to changes in either soil temperature or soil water content. Zerva and Mencuccini (2005) noted that soil temperature alone could explain the variability of CO_2 fluxes prior to clearfelling. However, after clearfelling, they found that CO_2 fluxes were more correlated with soil water content, the presence of fine roots, and the disturbance of the soil due to clearfelling, rather than soil temperature. The C balance of boreal forests (forests of the northern temporal zone) is sensitive to changes in temperature and photosynthesis (Lindroth et al., 1998). These changes affect the net C balance, so that the forests may act as a source of C at certain times of the year. Methane emissions are generally weakly correlated with soil temperature and more strongly correlated with the soil water content (Morishita et al., 2005).

Zerva and Mencuccini (2005) found that CH_4 fluxes were weakly correlated with soil temperature at 0.01 m and 0.05 m depths below the soil surface, but less so at a depth of 0.1 m. This indicates that fluxes were more dependent on substrate availability rather than environmental factors. A study, carried out in laboratory conditions (Dunfield et al., 1993), showed that CH_4 production was optimum at a temperature range of 25 – 30˚C, while optimum consumption of CH_4 occurred at 20 - 25˚C. There was extremely low activity in the temperature range of 0 - 15˚C – the range of soil temperature found in Ireland. Nitrous oxide production can increase after clearfelling with an increase in soil temperature. This increase is enhanced by the large increase in harvesting residues and decomposing roots, which provide substrate to microbes (Zerva and Mencuccini, 2005) and cause a higher N_2O efflux.

The aim of this study was to examine, over a period of one year, changes to GHG fluxes across four different areas of forestry and peatland, which represent a range of forest land uses (forested, riparian buffer, recently clearfelled, and a virgin peat site). This allowed for assessment of gaseous emissions from peatland over its lifespan from un-forested virginal peat site to coniferous plantation to subsequent clearfelling. To elucidate the drivers of change, high resolution WT data, soil water content, and air and soil temperature were examined.

5.2 Materials and Methods

5.2.1 Study site description

The study areas were located in the Burrishoole catchment in Co. Mayo, Ireland (Figure 5.1) (ITM reference 495380, 809170), at an approximate elevation of 135 m above sea level. Four peatland uses were examined in this study: (1) a regenerated riparian peatland buffer (RB), clearfelled 5 years before the present study (2) a recently clearfelled coniferous forest (CF) (3) a virgin peat site (VP), and (4) a standing mature coniferous forest (SF). The RB, CF and SF were located in the Altaconey forest, while the VP was approximately 1.4 km from the Altaconey site (Figure 5.1). The Altaconey site has a north-westerly aspect, while a third-order stream (Strahler, 1964), which is a tributary to the Altaconey River (classed as ‘unpolluted’ by the EPA (2011)), flows in a southwest-to-northeast direction to the north of the site. The Altaconey site has a north-westerly aspect and is within a subcatchment area of 416.2 ha, of which 176.4 ha is fully forested (Ryder et al., 2011). There is a moderate climate, which is heavily influenced by the proximity of the Atlantic Ocean, with average air temperatures of 13°C in summer and 4°C in winter. The Altaconey site is subjected to approximately 2400 mm (Figure 5.2) of rainfall every year, and had 300 rain days over the duration of the present study (July 2010 to June 2011). As a result, the area is characterised by upland spate streams. Blanket peat of varying depth down to 2 m covered both sites (Altaconey and VP). Sand and gravel deposits underlay the peat on top of the Cullydoo formation of Srahmore quartzite and schist, which is a poorly unproductive aquifer (McConnell and Gatley, 2006). Bedrock does not protrude the surface of the peat and the minimum peat depth is 0.3 m. Site characteristics are shown in Table 5.1.

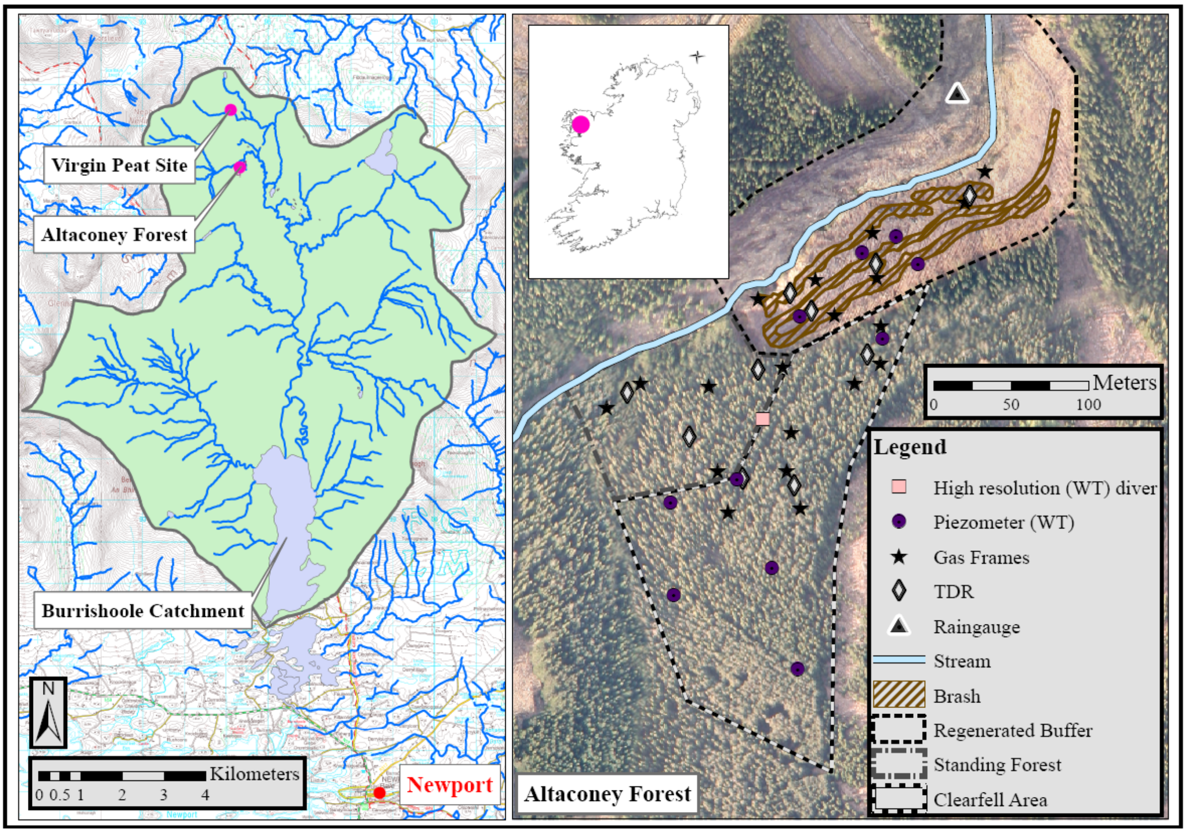

Figure 5.1 Location of the Virgin Peat Site and Altaconey Forest with site instrumentation, within the Burrishoole catchment.

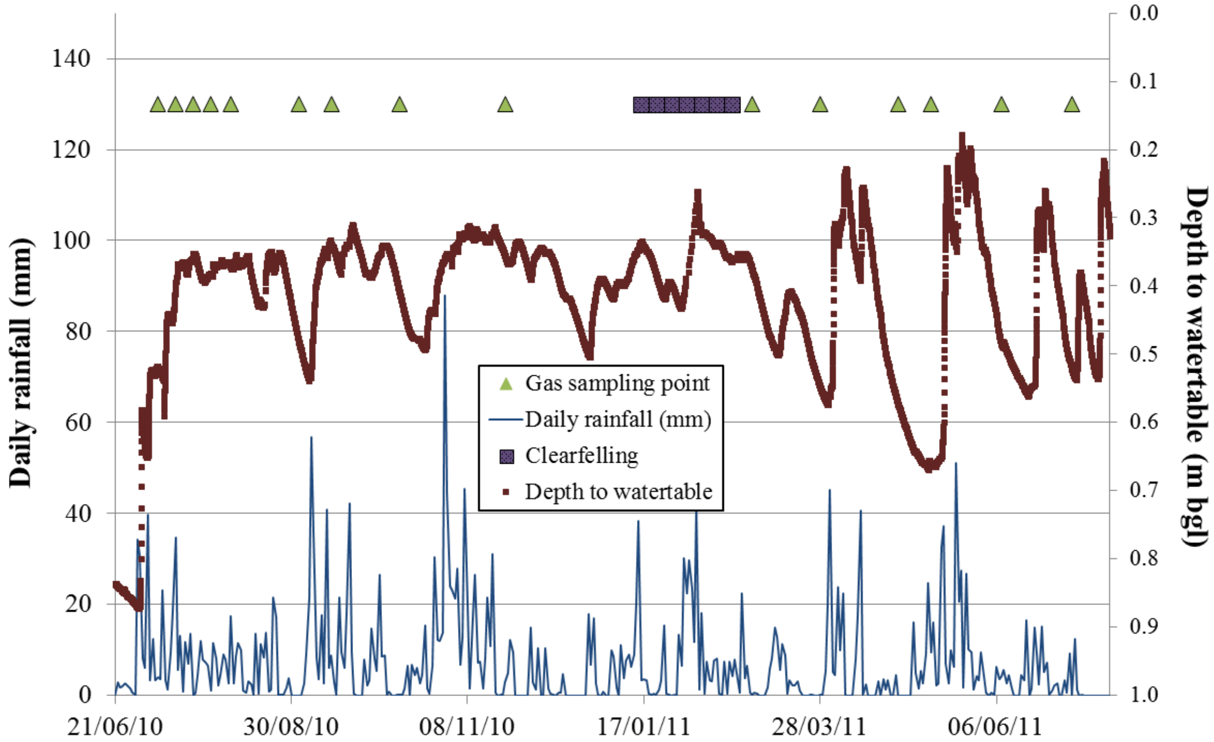

Figure 5.2 Average daily rainfall (mm) and depth to watertable (m bgl) in the Altaconey Forest with gas sampling regime and clearfelling dates.

Table 5.1 Site characteristics for Altaconey Forest and Virgin Peat Site.

Site	Regenerated Buffer	Pre Clearfell Forest	Virgin Peat	Standing Forest
Abbreviation	RB	CF	VP	SF
Area (ha)	2.49	2.61	1.23	1.06
Avg tree height (m)	n.a.	17.7 (0.94) Pine 17.36 (0.70) Spruce	n.a.	n.d.
Avg air temp (°C)	13	11 (14°C post-CF)	12	11
Avg soil temp (°C)	11	11 (10°C post-CF)	12	9
Avg water content ($m\ m^{-3}$)	0.12	0.11 (0.08 post-CF)	0.15	0.11
Avg depth to watertable (m)	0.31	0.34	n.d.	0.43
Avg pH in soil	2.44 (0.06)	2.32 (0.05)	n.d.	2.47 (0.12)
Avg bulk density (g/cm^3)	0.13 (0.01) 0-10 cm 0.12 (0.01) 10-20 cm	0.12 (0.01) 0-10 cm 0.11 (0.01) 10-20 cm	0.10 (0.01) 0-10 cm 0.11 (0.01) 10-20 cm	0.11 (0.01) 0-10 cm 0.10 (0.01) 10-20 cm
Soil inorganic nitrogen (µg N / g dry soil)	29.59 (7.89) 0-10 cm 55.08 (13.96) 10-20 cm	50.16 (5.39) 0-10 cm 64.39 (8.22) 10-20 cm	28.45 (1.58) 0-10 cm 26.45 (1.23) 10-20 cm	n.d.
Carbon content (%)	46.68 (0.72) 0-10 cm 43.59 (0.72) 10-20 cm	45.41 (0.68) 0-10 cm 46.76 (0.71) 10-20 cm	44.23 (0.44) 0-10 cm 47.01 (0.35) 10-20 cm	n.d.
Nitrogen content (%)	2.02 (0.10) 0-10 cm 1.98 (0.13) 10-20 cm	2.29 (0.05) 0-10 cm 2.30 (0.06) 10-20 cm	2.37 (0.06) 0-10 cm 2.48 (0.05) 10-20 cm	n.d.
C:N ratio	23 (0-10 cm) 22 (10-20 cm)	20 (0-10 cm) 20 (10-20 cm)	19 (0-10 cm) 19 (10-20 cm)	n.d.

Numbers in brackets indicates standard error from mean. n.a. – not applicable. n.d. – not determined

The Altaconey site was planted in 1966 with Sitka Spruce (*Picea sitchensis*) and Lodgepole Pine (*Pinus contorta*). No understorey vegetation survived following canopy closure. On May 26, 2006, an area of 2.49 ha 30 m north, 50 m south and 300 m along the river was clearfelled (Ryder et al., 2011) (Figure 5.1, identified as Regenerated Buffer, RB). Bole-only clearfelling was carried out with a harvester and a forwarder (Rodgers et al., 2010). The harvester machine had a cutting mechanism on the end of a hydraulic arm, which cut the tree at the base and allowed controlled falling to the ground. The branches were sheared off by pulling the tree through the rotating blades, while the log itself was cut into various lengths, depending on the quality of the wood. Logs were laid to the left in piles for collection by the forwarder, which put them on the rear frame of the machine and carried them off site. The brash material (logging residues and un-merchantable part of the log) was laid ahead to create a mat on which to drive forward, thus protecting the soil from consolidation. These brash mats were left *in situ* on completion of clearfelling. Typical forest practice would be to arrange these brash mats into regular rows (‘windrowing’) away from the watercourse when preparing the site for replanting.

In April 2007, one year after felling, the area was replanted with native broadleaved tree species from nurseries, including *Ilex aquifolium* (holly), *Sorbus aucuparia* (rowan), *Alnus glutinosa* (common alder), *Salix cinerea* (grey willow), *Betula pendula* (common birch) and *Quercus robur* (oak pedunculate). No fertilizer was applied, but the saplings were pre-treated by dipping in Dimethoate (pyrethroid insecticide) to protect them against the pine weevil (*Hylobus abietis*) (Ryder et al., 2011). The perimeter of the newly created buffer zone was then fenced off to protect from grazing from sheep and wild animals.

In February 2011, clearfelling of an area of 2.61 ha (1230 m^3) of the forest upslope of the regenerated buffer area began (Figure 5.1, identified as Clearfell Forest, CF). Clearfelling was conducted in a similar manner to the RB, described above, with a harvester (Valmet 921) and a forwarder (Valmet 860). The brash mats in this area also remained *in situ* for the study period.

5.2.2 Measurement and analysis

Testing on site began in July 2010 (approximately 6 months before clearfelling began) and included measurements of (1) air and soil temperature (2) soil volumetric water content (3) bulk density (4) soil and water pH (5) depth to WT (6) C and N content of the soil (7) soil inorganic N content (8) rainfall, and (9) gas fluxes (CO_2, CH_4 and N_2O). Sampling continued until July 2011 (approximately 6 months after clearfelling) to obtain a full year of data. This was to fully assess seasonal effects, as Shrestha et al. (2009) observed peak CO_2 fluxes in spring and summer, and peak N_2O and CH_4 fluxes in summer in a forested site in Ohio, USA. The sampling regime is illustrated in Figure 5.2.

Environmental and physical parameters

Rainfall was recorded by a rain gauge (Figure 5.3) installed on site (Environmental Measurements Limited, UK) (Figure 5.1). Volumetric soil water content (m^3 water m^{-3} soil) was measured at a depth of approximately 0.15 m below the soil surface with a PR1 profile probe and a HH2 handheld device (Figure 5.4) (Delta-T Devices, Burwell, UK) on each sampling date. Air temperature, recorded at 1 m above ground level, and soil temperature, recorded 0.1 m below the soil surface, were measured on site with a thermometer at the time of gas sampling. Soil bulk density at two depths, 0 – 0.1 m and 0.1 – 0.2 m, was measured using 0.05-m-diameter × 0.05-m-deep (total volume 9.8×10^{-5} m^3) steel rings (Eijelkamp, The Netherlands) (n=20 for each of the 4 study areas).

Figure 5.3 Rain gauge installed on site.

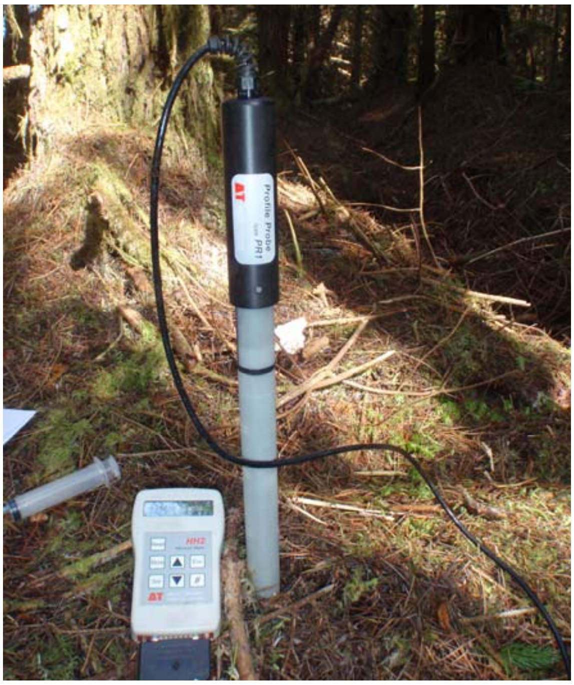

Figure 5.4 Soil volumetric water content equipment.

Soil pH (n=10 for each of the 3 areas, RB, CF and SF) was determined by mixing 15 g of peat with 25 ml 1M KCl, while water pH (n=10 for each of the 3 areas, RB, CF and SF) was measured using a pH probe (WTW SenTix 41 probe with a pH 330 m, WTW, Germany).

To investigate the spatial difference in WT heights, eleven piezometers (each covered with a filter sock) with internal diameter of 0.04 m with an end pipe screen interval of 0.3 m, were augured at random locations across the 3 sites, RB, CF and SF (Figure 5.1). Contact between the peat and each piezometer was ensured by a sand infill, which was backfilled to a depth of 0.3 m below the ground surface. This was then overlain by bentonite to prevent against the ingress of rainwater. Average depth of installation was approximately 1 m below ground level (bgl). A high resolution mini-diver (OTT Orpheus Mini, Germany) set to record pressure head at 5-minute intervals was in place at one location over the study duration (Figure 5.1). Soil and water pH, and depth to WT measurements were not analysed in the VP site.

As the depth above ordnance datum (AOD) position of each piezometer was known, WT heights were converted to groundwater heads (m AOD). From this data, temporal groundwater flow direction, WT relief and slope maps were created in ArcGIS (Release Version 9.3, Environmental Systems Research Institute (ESRI), California, USA). The high resolution groundwater head data allowed WT positional trends and flow rates in all the other piezometers to be inferred.

To determine the C and N content of the peat and the inorganic N soil content, only 3 areas were tested: RB, the CF prior to clearfelling (pre-CF) and VP. The SF was assumed to have similar characteristics as the pre-CF. To determine the C and N content of the soil, soil cores (n=20 for each of the 3 areas, RB, pre-CF and VP) were taken at depths of 0 – 0.1 m and 0.1 – 0.2 m, air dried in the laboratory for over a month, milled and tested using a thermal conductivity detector, following combustion and separation in a chromatographic column. The potentially mineralisable inorganic N content (n=20 for each of the 3 areas, RB, pre-CF and VP) of the soil at two depths, 0-0.1 m and 0.1-0.2 m, was determined by KCl extraction. Sub-samples of peat (equivalent to 1 g dry weight) were mixed with 40 ml of 1M KCl and shaken for 1 hour at 225 rpm using a rotary shaker. The filtered (0.45 μm) supernatant water was tested using a nutrient analyser (Konelab 20; Thermo Clinical Labsystems, Finland). The remaining soil sample was used to determine the gravimetric water content.

Gas fluxes

Gas fluxes were measured using a closed chamber technique (Hutchinson and Mosier, 1981). Varying numbers of open-bottomed collars (Figure 5.5) were installed in the four study areas: 7 collars were installed in both the RB and in the CF, and 5 collars in both the VP and the SF, giving a total of 24 collars. For analysis, the CF was divided into pre-clearfell (pre-CF) and post-clearfell (post-CF). During harvesting, the gas collars were removed from the CF to prevent against accidental damage (after Castro et al., 2000). To select the sampling locations, a grid was laid out and the collars were inserted at random locations on the grid. Each stainless steel collar had a surface area of 0.17 m^2, a depth of between 0.09 m and 0.18 m, and contained a trough for holding water. The static chambers were also stainless steel with a volume of 0.016 m^3 (Figure 5.6). After installation, collars were allowed to settle for two weeks in order to prevent against the inadvertent sampling of gas emissions arising from root disturbance. When not in use, all collars were open to the atmosphere to prevent damage to the vegetation underneath. Two butyl rubber septa (SW-050, Soil Measurement Systems LLC, USA) were inserted into the top of each chamber for gas extraction. Before each measurement, the air was manually mixed in the chamber by pumping the syringe 3 times. Each chamber was sampled four times on a given sampling date at time intervals (measured from immediately after chamber placement) of 0, 2, 10 and 60 min. Pre-evacuated 7 ml, vacuum sealed, glass vials were filled from a 25 ml luer-lock syringe (SY3 20L CET, Nipro, Japan). Samples were returned to the laboratory and tested with a gas chromatograph (Varian GC 450; The Netherlands) and automatic sampler (Combi-PAL autosampler; CTC Analytics, Zwingen, Switzerland).

Figure 5.5 Open-bottomed collar.

Figure 5.6 Static chamber during testing.

5.2.3 Statistics

Statistical analysis was performed using Datadesk (Data Description Inc., USA). Environmental (air and soil temperature) and physical parameters (soil volumetric water content and depth to WT) were analysed with ANOVA (analysis of variance) to ascertain the main sources of variation. Date and the location of the sample site were included as explanatory variables. Cumulative CO_2-C, CH_4-C and N_2O-N losses were log-transformed and subsequently analysed using a General Linear Model. In order to assess differences within treatments, Fishers Least Significant Difference test was performed.

5.3 Results and Discussion

5.3.1 Air and soil temperature

Date and location were significant sources of variation for both the air and soil temperatures (ANOVA, $p<0.05$). The air temperature in the post-CF was significantly higher than other areas, while the pre-CF and the SF had significantly lower values than all other areas (ANOVA, $p<0.05$). Similarly, the soil temperature in the RB was significantly higher, while the pre-CF and the SF had significantly lower values than all other areas (ANOVA, $p<0.05$). Clearfelling of the forestry raised the average air temperature by 3 °C (average value post-CF of 14°C), while the other areas remained at the pre-harvest temperature. No average difference in soil temperature pre- and post-CF was recorded.

5.3.2 Soil volumetric water content and depth to watertable

Date of sampling was not a significant source of variation for the volumetric water content, but the location of the sampling point was significant (ANOVA, $p<0.05$). The volumetric water content in the RB and the VP was significantly higher than the other study areas, while the post-CF forest had significantly lower values than all other areas (ANOVA, $p<0.05$). The volumetric water contents in the VP were higher than the SF (average of VP, 0.14 m^3 m^{-3}; average of SF, 0.11 m^3 m^{-3}). Date of sampling and location were significant sources of variation for the depth to the WT (ANOVA, $p<0.05$). The WT in the RB and the post-CF (Figure 5.2) had a significantly higher elevation, while the SF had significantly lower WT elevation than all other areas (ANOVA, $p<0.05$).

5.3.3 Average and cumulative gas fluxes

The average (Figure 5.7) and cumulative (Table 5.2) results for the year show a high quantity of CO_2 being emitted from the RB following clearfelling in May 2006, a high flux of CH_4 - from both the VP and the post-CF, reflecting the high volumetric water content and WT of these areas and a low level of efflux of N_2O from all sites. These results are soil respiration fluxes and are discussed in the following sections.

Carbon dioxide (CO_2)

Within 6 months of clearfelling, significant increases in soil respiration from 11±2 kg CO_2-C ha^{-1} d^{-1} to 19±2 kg CO_2-C ha^{-1} d^{-1} were measured in CF (Figure 5.7). The highest average rates of emissions (30.8 kg C ha^{-1} d^{-1}) were observed in the RB, 5 years after clearfelling (Figure 5.7). This was equal to a cumulative yearly emission of 6488±1313 kg CO_2-C ha^{-1} y^{-1} (Table 5.2). Low fluxes were recorded for the SF, which had average daily emissions of 13 kg CO_2-C ha^{-1} d^{-1} (Figure 5.7). This was equal to an annual cumulative flux of 3396±281 kg CO_2-C ha^{-1} y^{-1}, although these were not significantly different to rates from clearfelled areas (post-CF) and undisturbed peatlands (VP) (Figure 5.7; Table 5.2). A similar range in values has been reported for afforested peatland in Southeast China, where soil respiration increased from 2900 kg CO_2-C ha^{-1} y^{-1} to over 7200 kg CO_2-C ha^{-1} y^{-1} after clearfelling (Tamai and Hsia, 2012). The high efflux of CO_2 in the RB was possibly due to decomposition of material and roots following clearfelling 5 years prior to the present study. This high flux could also be due to the high autotrophic respiration from the growing vegetation in the RB. Carbon dioxide efflux at these sites also exhibited a strong seasonal response, with a trebling in rates from April 2011 onwards compared to winter fluxes (Figure 5.8).

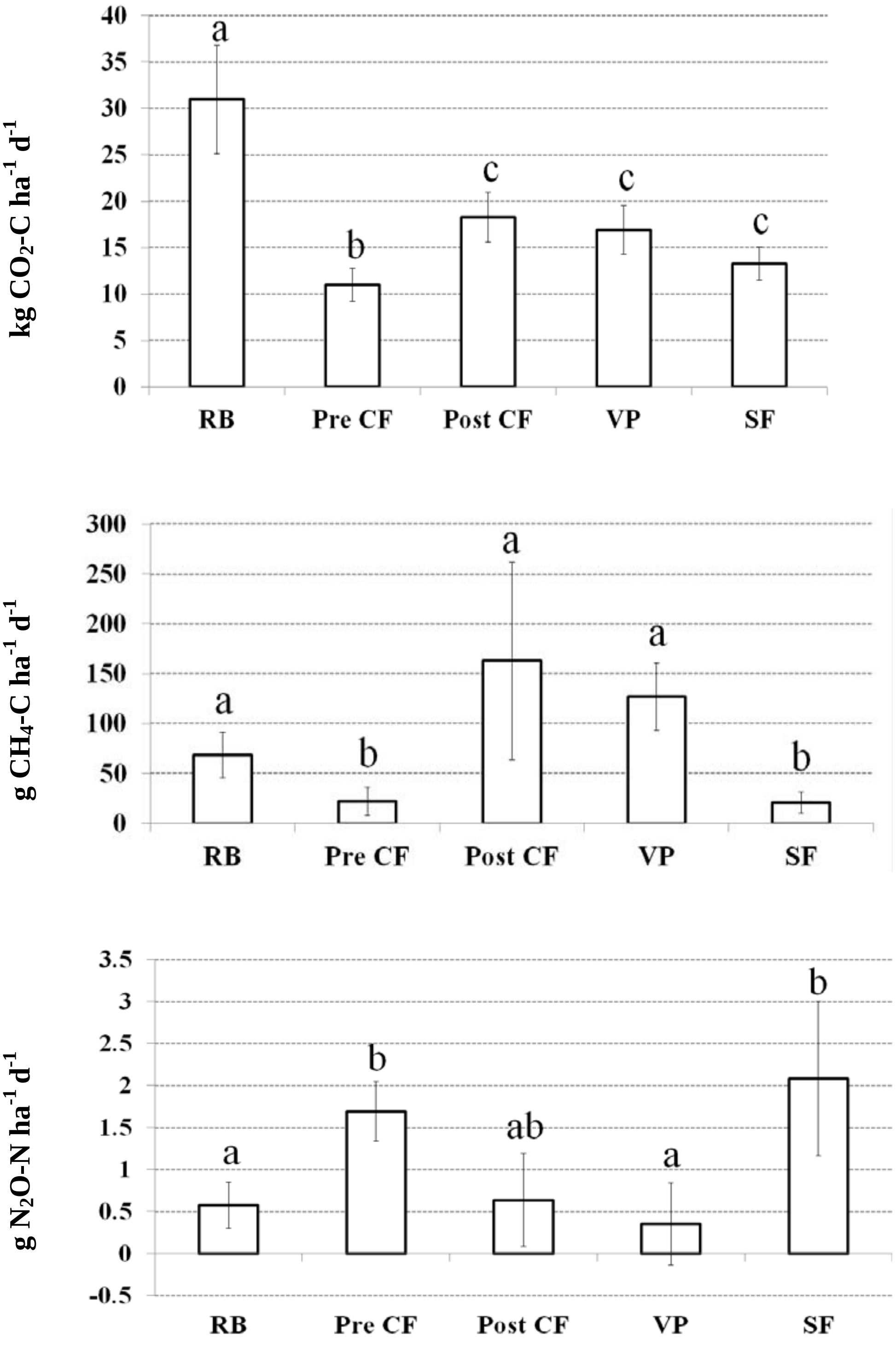

Figure 5.7 Average emissions of carbon dioxide (top), methane (middle), and nitrous oxide (bottom) over (July 2010 to July 2011) for all areas (RB – Regenerated Buffer, Pre-CF – Pre Clearfell Forest, Post-CF – Post Clearfell Forest, VP – Virgin Peat and SF – Standing Forest). Bars denote standard error of the mean and letters indicate significant differences (p<0.05).

Table 5.2 Cumulative fluxes of ecosystem respiration of a) carbon dioxide, b) methane and c) nitrous oxide over a one year period (July 2010 to July 2011) for all areas (RB – Regenerated Buffer, Pre-CF – Pre Clearfell Forest, Post-CF – Post Clearfell Forest, VP – Virgin Peat and SF – Standing Forest) in the Altaconey forest. Numbers in brackets indicates standard error from mean.

GHG	RB	Pre-CF	Post-CF	VP	SF
CO_2 ($kg\ CO_2\text{-}C\ ha^{-1}\ y^{-1}$)	6488 (1313)	4227 (612)	3796 (602)	4960 (928)	3396 (281)
CH_4 ($kg\ CH_4\text{-}C\ ha^{-1}\ y^{-1}$)	25 (9)	20 (11)	65 (40)	42 (11)	9 (5)
N_2O ($g\ N_20\text{-}N\ ha^{-1}\ y^{-1}$)	39 (172)	257 (217)	262 (217)	-15 (193)	1173 (422)

Within peatland systems, WT is a principle driver of soil respiration as it determines the proportion of oxic conditions available for aerobic respiration (Silins and Rothwell, 1999). Under anaerobic conditions, phenol oxidase is inactive, resulting in the accumulation of chemically labile soil organic matter (SOM) (Freeman et al., 2001). Whilst the relationship between WT and soil microbial activity has been clearly demonstrated in laboratory studies (Blodau et al., 2004), under field conditions, considerable variability in fluxes has been shown, with the relationship between lowering of the WT and increases in CO_2 emissions occurring only to a certain depth (Silvola et al., 1996; Chimner and Cooper, 2003; Flanagan and Syed, 2011). The low efflux reported in this study for the SF may be due to the deep WT in the sitka plantation, with WT decreasing to 0.6 m bgl over the study duration. Indeed, the WT may occasionally fluctuate to a depth at which drying of the peat surface may start to limit decomposition rates (Lieffers, 1988; Laiho et al., 2004). In Finnish afforested peatlands, for example, an excessive WT drawdown (> 0.61 m) inhibited peat soil respiration (Mäkiranta et al., 2009, 2010).

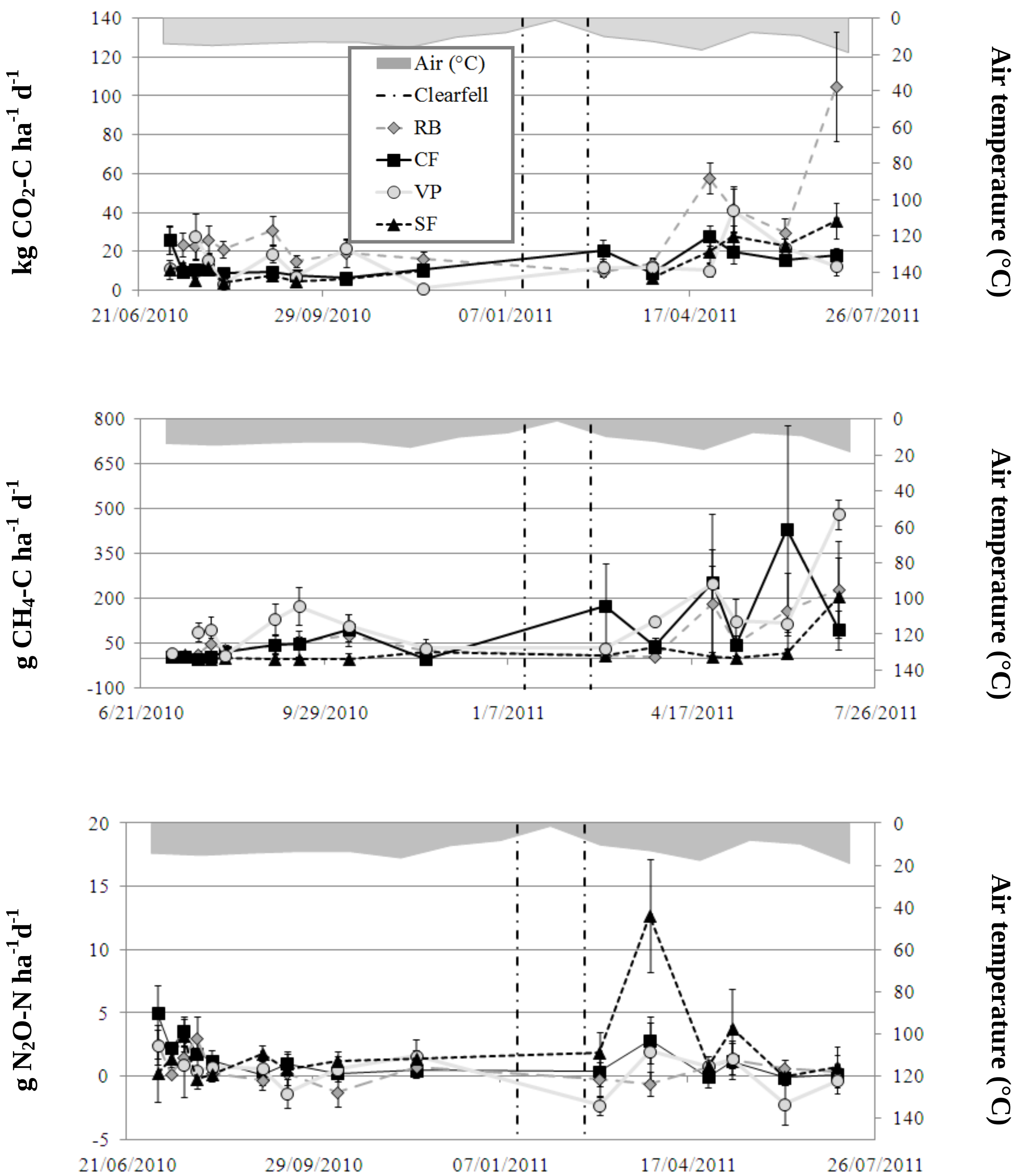

Figure 5.8 Temporal traces of carbon dioxide (top), methane (middle) and nitrous oxide (bottom) over a one year period (July 2010 to July 2011) for all areas (RB – Regenerated Buffer, Pre-CF – Pre Clearfell Forest, Post-CF – Post Clearfell Forest, VP – Virgin Peat and SF – Standing Forest) and air temperature (°C) on inverted secondary axis. Bars denote standard error of the mean.

In general, CO_2 respiration rates increased in response to changes in WT (Figure 5.9). The WT in the post-CF rose to above 0.3 m bgl, while the pre-CF WT constantly stayed below 0.3 m. This result was despite the reduced quantity of rain in the post-CF time period. Comparing the same number of days pre- and post-CF, the cumulative rain was greater for the pre-CF time period (1764 mm) compared to the post-CF total (1267 mm). This rise in the WT created a larger saturated zone, and therefore increased the production of CH_4 and reduced the N_2O flux (Figure 5.9). There was considerable heterogeneity of response of CO_2 to WT at the CF site. There was a strong correlation between pre- and post-CF volumetric water contents and CO_2 emissions in individual frames (range of r^2 = 0.24 and 0.72 over 4 collars, Figure 5.10). The same 4 frames correlated well with CO_2 flux and depth to WT (range of r^2 = 0.24 and 0.44 over 4 frames). Over 31 % of CO_2 emissions in the SF could be related to WT depth. A 'pumping' effect that enhances soil CO_2 efflux – especially when the soil is wet – is more likely to take place in a clearfelled forest than under the canopy closure of an intact forest due to near surface turbulence (Hollinger et al., 1998).

The contribution of brash material in newly-clearfelled areas may also be considerable and may add to this heterogeneity within the CF and RB sites. When left on site post-CF, this brash can act as a source of nutrients (Rodgers et al., 2010) and gas (Zerva and Mencuccini, 2005) as it is decomposing. The amount of CO_2 efflux from the soil may be very small compared to the flux of GHG from a decomposing heap of brash (Zerva and Mencuccini, 2005). A study in Northern England showed a relatively small flux of CO_2 measured by collars inserted into the soil when compared with the values from a nearby eddy covariance system (Zerva and Mencuccini, 2005).

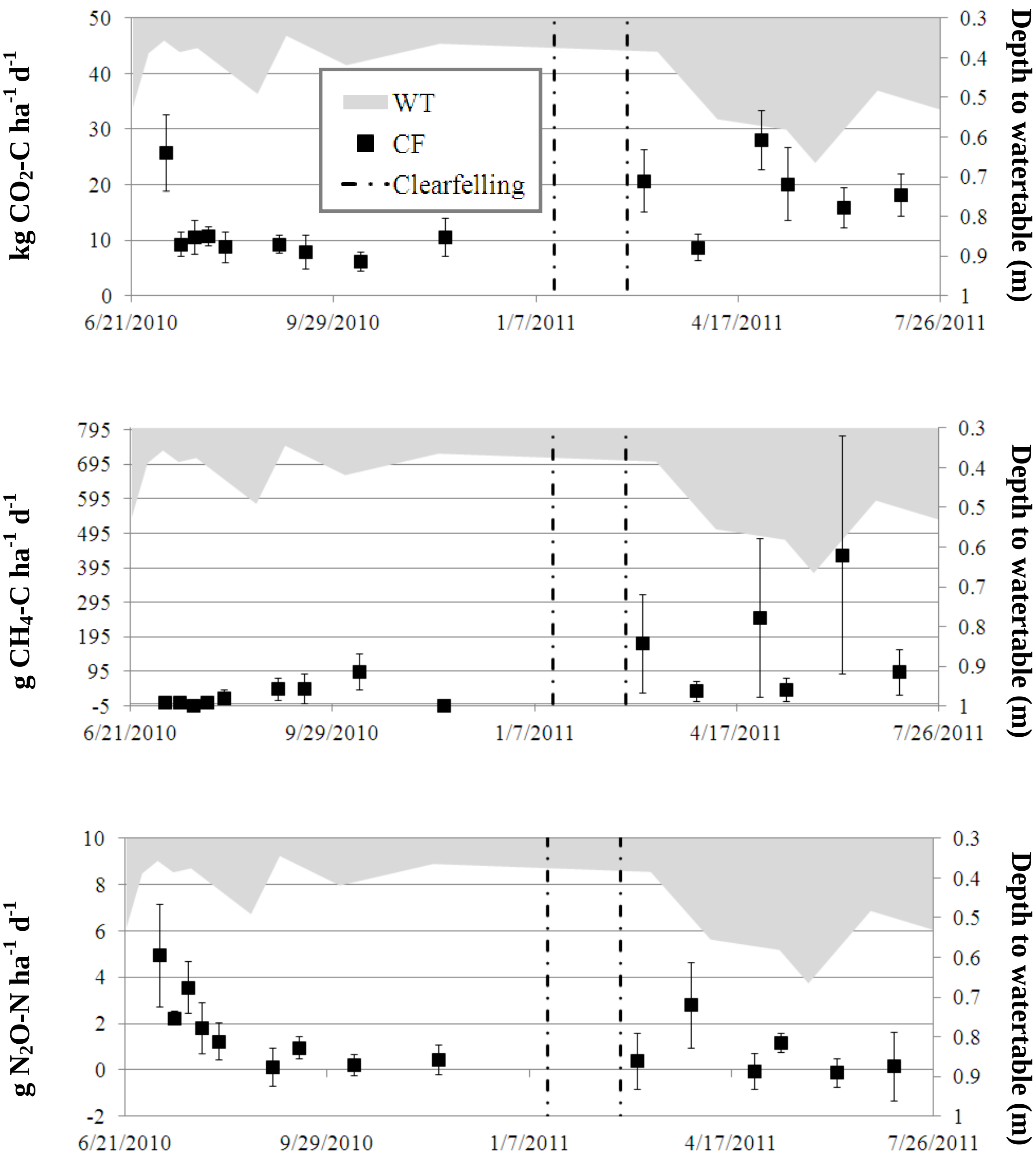

Figure 5.9 Average carbon dioxide (top), methane (middle) and nitrous oxide (bottom) flux over a one year period (July 2010 to July 2011) for the pre- and post clearfell forest (CF) in Altaconey. Depth to watertable (WT) (m bgl) on inverted secondary axis. Bars denote standard error of the mean.

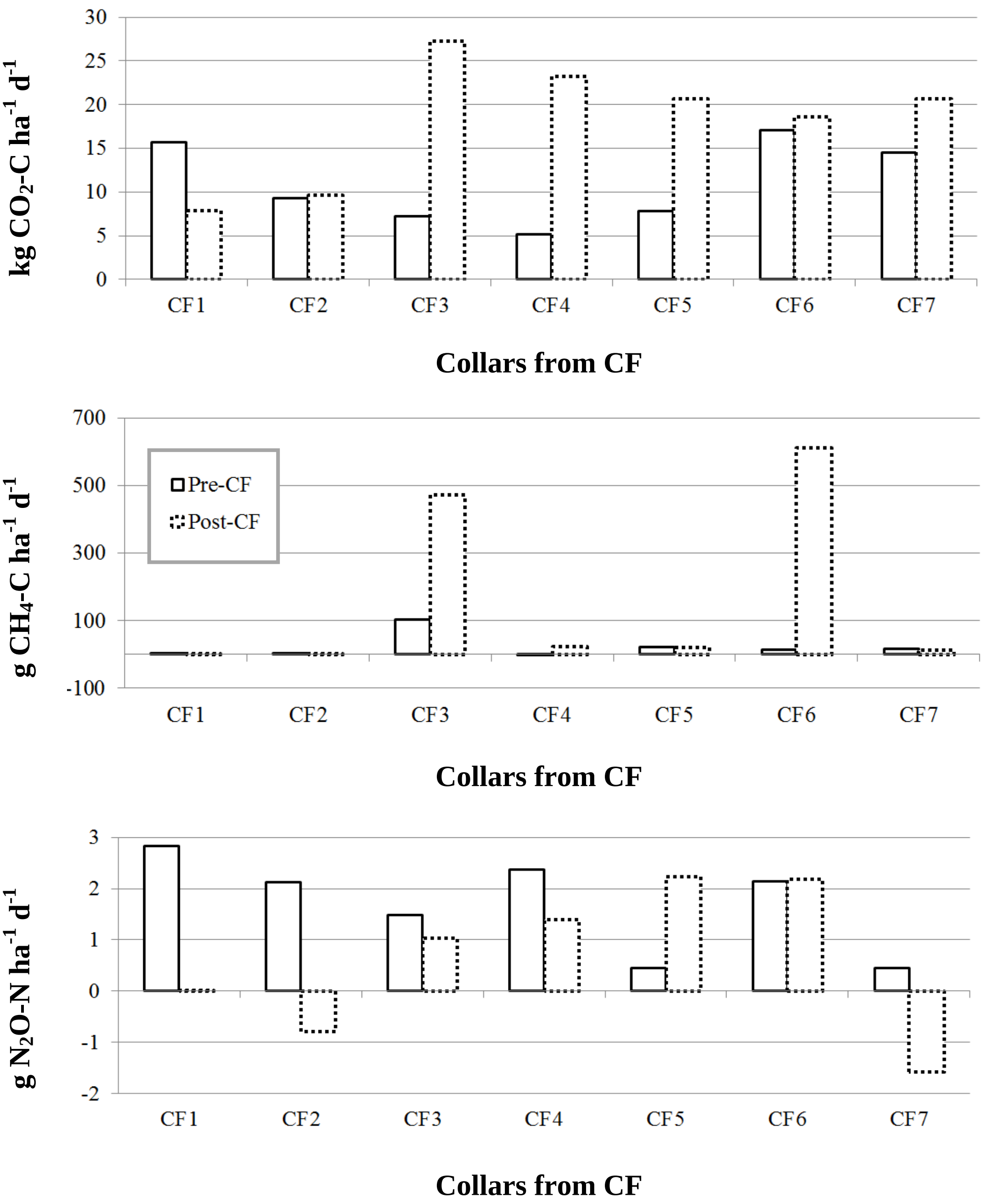

Figure 5.10 Average carbon dioxide (top), methane (middle) and nitrous oxide (bottom) flux over a one year period (July 2010 to July 2011) from each collar from the pre- and post clearfell forest (CF) in Altaconey.

Soil temperature is generally considered to be the main factor governing temporal variations in soil CO_2 respiration, with the relationship between efflux and soil temperature described by the Lloyd and Taylor (1994) exponential response function. Increases in soil temperature have been previously found to induce a 60 % increase in heterotrophic respiration (Dorrepaal et al., 2009). In this study, the relationship with temperature was more tenuous and was confounded both by spatial heterogeneity within sites and the effect of WT (Figures 5.8 and 5.9). Over half of the large CO_2 flux in the RB can be correlated with the air and soil temperature (range of r^2 = 0.23 and 0.67 over 6 frames for air temperature and r^2 = 0.34 and 0.48 over 6 frames for soil temperature). In contrast, the CO_2 emission in SF had very weak negative correlations with soil (r^2 = 0.003) and air (r^2 = 0) temperature.

Methane (*CH_4*)

The planting of forestry on peatland results in a lowering of the elevation of the WT (Alm et al., 1999; Moore, 2002) and a subsequent drying out of the soil. This, in turn, reduces the volume of the anaerobic zone, which leads to lower CH_4 emissions, as seen in the present study. Following clearfelling, the maximum WT position in the clearfelled site rose to depths of 0.18 m bgl (Figure 5.2). Pre-CF maximum WT positions were 0.31 m at the same location. The wetter conditions at the post-CF site produced the highest cumulative CH_4-C release from all areas (65±40 kg CH_4-C ha^{-1} y^{-1}) (Table 5.2). Similar values have been reported by Morishita et al. (2005). The lowest emissions of CH_4-C came from the SF (cumulative flux of 9±5 kg CH_4-C ha^{-1} y^{-1}) and the pre-CF (cumulative flux of 20±11 kg CH_4-C ha^{-1} y^{-1}) (Table 5.2). Uptake of CH_4 generally occurs in forests on mineral soil (Morishita et al., 2005), but very little research has been carried out in Ireland on the CH_4 sink or source capacity of peatland (Kiely et al., 2009). Morishita et al. (2005) found that a standing forest on a loamy, sandy soil had an uptake rate of approximately 4 g CH_4-C ha^{-1} d^{-1}, while a forest clearfelled 2 years prior to the study on the same soil type emitted over 2 g CH_4-C ha^{-1} d^{-1}. Zerva and Mencuccini (2005) found that a forest on a peaty, gley soil was a sink of CH_4-C prior to clearfelling (-101 g CH_4-C ha^{-1} d^{-1}), but turned into a CH_4-C source (548 g CH_4-C ha^{-1} d^{-1}) within a few months after clearfelling. The average daily flux of CH_4 from the RB was 63±27 g CH_4-C ha^{-1} d^{-1}, compared to the average daily flux from the post-CF area of 173±57 g CH_4-C ha^{-1} d^{-1} (Figure 5.7). These high CH_4 emission rates soon after clearfelling (Figure 5.7 - 5.10) were unusual, when rates of CH_4 efflux have been previously observed to take 1 - 4 years post-restoration (Waddington and Day, 2007). This delay is due to the fact that anaerobic decomposition of labile plant material is relatively slow (Lay, 2009). The high emission rates soon after clearfelling were observed primarily in two locations (CF3 and CF6; Figure 5.10). The high values in these areas may indicate that adequate labile C reserves were available for immediate decomposition or may indicate a large contribution of inputs from brash in these two areas to CH_4 production.

Kiely et al. (2009) measured an emission rate of 137 g CH_4-C ha^{-1} d^{-1} from a pristine peatland site in the south-west of Ireland, which is comparable to the average values obtained in the VP site of the present study (106±30 g CH_4-C ha^{-1} d^{-1}). Results from all sites show no clear correlations between soil and air temperatures and CH_4 fluxes (Figure 5.8). Similarly, heterogeneity within sites confounded correlation of CH_4 flux and volumetric water content and depth to WT on site (Figure 5.9). Macrea et al. (2012) found in a peatland in Canada, that a thick capillary fringe above the WT limited the effect of the fluctuating WT on the microbial activity due to sufficient water remaining in the peat, despite a drop in the WT. This capillary fringe could be a possible reason for limited correlation of flux on the current site and WT position.

Nitrous oxide (N_2O)

The drier soils in the SF produced the highest cumulative N_2O-N release (1173±422 g N_2O-N ha^{-1} y^{-1}), while the lowest N_2O-N release was from the VP, an area which had an annual cumulative uptake of 15±193 g N_2O-N ha^{-1} y^{-1} (Table 5.2). A study in Northern England on a virgin peaty gley also found low emissions of N_2O (3.7 g N_2O-N ha^{-1} d^{-1}) and high variability in the results (Zerva and Mencuccini, 2005). Clearfelling of the forest caused a decrease in average N_2O flux (pre-CF average flux of 1.7 g N_2O-N ha^{-1} d^{-1}; post-CF average flux of 0.7 g N_2O-N ha^{-1} d^{-1}), while the RB had the lowest values at 0.6±0.42 g N_2O-N ha^{-1} d^{-1}, most likely due to a rise in WT above 0.3 m bgl post-clearfelling in 2006 (Figure 5.7). In contrast to the findings of this study, clearfelling of forestry on free-draining brown earths can cause increases in N_2O emissions (Bradford et al., 2000). Production of N_2O is governed by rates of nitrification and denitrification in the soil. While nitrification is controlled by available soil ammonium and oxygen (O_2) availability, subsequent partial denitrification to N_2O is linked to anoxic soil conditions, NO_3^- levels in the shallow groundwater and C levels (which act as a reductant) (Bradford et al., 2000). Total denitrification to nitrogen gas (N_2) needs anaerobic conditions and although peatlands do have a high denitrification potential, NO_3^- availability is a limiting factor (Firestone and Davison, 1989; Aerts, 1997). Indeed, the concentration of NO_3^--N on this site was below 20 µg L^{-1} (Chapter 2) which, combined with anaerobic conditions, were responsible for the overall low efflux of N_2O from the site. The temporal profile of N_2O efflux revealed that, for certain periods, net N_2O uptake was observed, particularly from VP (undisturbed) sites (Figures 5.8 and 5.9). Previous studies reported a net uptake of 0.5 μg m^{-2} h^{-1} in an ombrotrophic peatland spruce forest ecosystem (Butterbach-Bahl et al., 1998). Uptake may be due to the fact that N_2O is the only acceptor available for denitrification. Alternatively, dissimilatory nitrate reduction to ammonia (DNRA), which occurs under anaerobic, carbon-rich conditions, may consume available N_2O (Matheson, 2002).

5.4 Conclusions

This study showed:

- A high quantity of CO_2 being emitted from the RB (average flux of 30.8 kg C ha^{-1} d^{-1}), clearfelled 5 years before the present study,
- A high flux of CH_4 from both the VP (average flux of 106±30 g CH_4-C ha^{-1} d^{-1}) and the CF after clearfelling (average flux of 173±57 g CH_4-C ha^{-1} d^{-1}) due to the shallow WT and high soil water content in these areas,
- A low efflux of N_2O (average flux of 1.07±0.52 g N_20-N ha^{-1} d^{-1}) from all sites, with the greatest efflux from the SF (average flux of 2.1±0.9 g N_2O-N ha^{-1} d^{-1}).

Clearfelling of afforested peatland resulted in an increase in both soil respiration and methane emissions and a decrease in nitrous oxide emissions, though considerable variation was observed due to (1) variation in the WT depth and (2) the presence/absence of a brash layer. Ultimately, the position of the WT is the controlling factor governing GHG release and sequestration within the ecosystems.

5.5 Acknowledgements

This work was funded by the Department of Agriculture, Fisheries and Food and the Environmental Protection Agency under the STRIVE program 2007 – 2013. The authors wish to acknowledge the input of Coillte in allowing access to state forestry for this project. A special thanks to Dr. Michael Rodgers and all at the Marine Institute for their continued help throughout this project and for supplying the orthophotography.

References

Aerts R. Atmospheric nitrogen deposition affects potential denitrification and N2O emission from peat soils in the Netherlands. Soil Biol Biochem 1997; 29(7): 1153-1156

Alm J. Schulman L. Walden J. Nykänen H. Martikainen PJ. Silvola J. Carbon balance of a boreal bog during a year with an exceptionally dry summer. Ecology 1999; 80: 161–174.

Bain CG, Bonn A, Stoneman R, Chapman S, Coupar A, Evans M, et al. IUCN UK Commission of Inquiry on Peatlands. IUCN UK Peatland Programme, Edinburgh, 2011.

Blodau C. Carbon cycling in peatlands - A review of processes and controls. Environ Rev 2002; 10: 111-134.

Bradford MA, Ineson P, Wookey PA, Lappin-Scott HM. Soil CH4 oxidation: Response to forest clearcutting and thinning. Soil Biol Biochem 2000; 32: 1035-1038.

Butterbach-Bahl K, Gasche R, Huber C, Kreutzer K, Papen H. Impact of N-input by wet deposition on N-trace gas fluxes and CH 4-oxidation in spruce forest ecosystems of the temperate zone in Europe. Atmos Environ 1998; 32: 559-564.

Byrne KA, Farrell EP. The effect of afforestation on soil carbon dioxide emissions in blanket peatland in Ireland. Forestry 2005; 78: 217–227.

Byrne KA, Milne R. Carbon stocks and sequestration in plantation forests in the Republic of Ireland. Forestry 2006; 79: 361-369.

Castro MS, Gholz HL, Clark KL, Steudler PA. Effects of forest harvesting on soil methane fluxes in Florida slash pine plantations. Can J Forest Res 2000; 30: 1534-1542.

Chimner RA, Cooper DJ. Influence of water table levels on CO_2 emissions in a Colorado subalpine fen: an *in situ* microcosm study. Soil Biol Biochem 2003; 35: 345–351.

Dorrepaal E, Toet S, Van Logtestijn RSP, Swart E, Van De Weg MJ, Callaghan TV, et al. Carbon respiration from subsurface peat accelerated by climate warming in the subarctic. Nature 2009; 460: 616-619.

Dunfield P, knowles R, Dumont R, Moore T.R. Methane production and consumption in temperate and subarctic peat soils: Response to temperature and pH. Soil Biol Biochem 1993; 25: 321-326.

EPA. Water Framework Status Update based on Monitoring Results 2007-2009, Johnstown Castle Estate, Ireland., 2011.

Fernandes AN, Almeida CAP, Debacher NA, Sierra MMdS. Isotherm and thermodynamic data of adsorption of methylene blue from aqueous solution onto peat. J Mol Struct 2010; 982: 62-65.

Finnegan J, Regan JT, De Eyto E, Ryder E, Tiernan D, Healy MG. Nutrient dynamics in a peatland forest riparian buffer zone and implications for the establishment of planted saplings. Ecol Eng 2012; 47: 155– 164.

Firestone MK, Davidson EA. Microbiological Basis of NO and N_2O Production and Consumption in Soil. In: Andreae MO, Shimel DS. (eds) Exchange of Trace Gases between Terrestrial Ecosystems and the Atmosphere. John Wiley & Sons, Chichester: 1989; 7-22

Flanagan LB, Syed KH. Stimulation of both photosynthesis and respiration in response to warmer and drier conditions in a boreal peatland ecosystem. Glob Change Biol 2011; 17: 2271–2287.

Freeman C, Ostle N, Kang H. An enzymatic 'latch' on a global carbon store. Nature 2001; 409: 149.

Hargreaves K, Milne R, Cannell M. Carbon balance of afforested peatland in Scotland. Forestry 2003; 76: 299–317.

Hashimoto S, Tanaka N, Suzuki M, Inoue A, Takizawa H, Kosaka I, et al. Soil respiration and soil CO_2 concentration in a tropical forest, Thailand. J Forest Res-JPN 2004; 9: 75-79.

Hollinger DY, Kelliher FM, Schulze ED, Bauer G, Arneth A, Byers JN, et al. Forest-atmosphere carbon dioxide exchange in eastern Siberia. Agr Forest Meteorol 1998; 90: 291-306.

Hutchinson GL, Mosier AR. Improved Soil Cover Method for Field Measurement of Nitrous Oxide Fluxes. Soil Sci. Soc. Am. J. 1981; 45: 311-316.

IPCC. Climate Change 2001: Impacts, Adaptation and Vulnerability: Cambridge University Press, Cambridge, UK., 2001.

IPCC. Guidelines for National Greenhouse Gas Inventories. In: Prepared by the National Greenhouse Gas Inventories Programme: Eggleston H.S. BL, Miwa K., Ngara T. Tanabe K. (eds) Japan, 2006.

IPCC. Climate Change 2007: Mitigation. Contribution of Working Group III to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change. B. Metz, O.R. Davidson, P.R. Bosch, R. Dave, L.A. Meyer (eds). Cambridge United Kingdom and New York, NY, USA, 2007.

IPS. Peatlands and climate change. In: Dr. Maria Strack UoC, Canada, editor. International Peat Society, Finland, 2008.

Joosten H, Clarke D. Wise Use Of Mires And Peatlands, A Framework for Decision-making: International Mire Conservation Group & International Peat Society, 2002.

Kiely G, Xu X, Hsieh C, Leahy P, Sottocornola M, Laine A, et al. Measurement and Modeling of GHG Fluxes from Grasslands and a Peatland in Ireland. In: CarboEastAsia Workshop 2009, Tsukuba, Japan, 2009.

Kyoto Protocol. Kyoto Protocol to the United Nations Framework Convention on Climate Change, FCCC/CP/1997/7/Add.1, Decision 1/CP.3, Annex 7, UN. 1997

Laiho R. Laine J. Trettin C. Finer L. Scots pine litter decomposition along drainage succession and soil nutrient gradients in peatland forests, and the effects of inter-annual weather variation. Soil Biol Biochem 2004; 36: 1095–1109.

Lay DYF. Methane dynamics in northern peatlands: A review. Pedosphere 2009; 19: 409-421.

Lieffers VJ. Sphagnum and cellulose decomposition in drained and natural “areas of an Alberta peatland. Can J Soil Sci 1988; 68: 755–761.

Lindroth A, Grelle A, MorÉN A-S. Long-term measurements of boreal forest carbon balance reveal large temperature sensitivity. Glob Change Biol 1998; 4: 443-450.

Lloyd J. Taylor JA. On the temperature dependence of soil respiration. Funct Ecol 1994; 8: 315–323.

Macrae ML, Devito KJ, Strack M, Waddington JM. Effect of water table drawdown on peatland nutrient dynamics: implications for climate change. Biogeochemistry 2012: 1-16.

Mäkiranta P. Laiho R. Fritze H. Hytönen J. Laine J. Minkkinen K. Indirect regulation of heterotrophic peat soil respiration by water level via microbial community structure and temperature sensitivity. Soil Biol Biochem 2009; 41: 695–703.

Mäkiranta P. Riutta T. Penttilä T. Minkkinen K. Dynamics of net ecosystem CO_2 exchange and heterotrophic soil respiration following clearfelling in a drained peatland forest. Agr Forest Meteorol 2010; 150: 1585–1596.

Matheson FE. Nguyen ML. Cooper AB. Burt TP. Bull DC. Fate of 15N-nitrate in unplanted, planted and harvested riparian wetland soil microcosms. Ecol Eng 2002; 19: 249-264

McConnell B, Gatley S. Bedrock Geology of Ireland. Derived from the Geological Survey of Ireland 1:100,000 Bedrock Map Series and the Geological Survey of Northern Ireland 1:250,000 Geological Map of Northern Ireland. 2006.

Moore PD. The future of cool temperate bogs. Environ Conserv 2002; 29: 3–20.

Morishita T, Hatano R, Takahashi K, Kondrashov LG. Effect of deforestation on CH_4 uptake in Khabarovsk, Far East, Russia. Phyton - Annales Rei Botanicae 2005; 45: 267-274.

National Forest Inventory. National Forest Inventory Republic of Ireland - Results. In: Service F, editor. Forest Service, Wexford, 2007.

O'Brien P. Data Analysis and Estimation of Greenhouse gas emissions and removal for the IPCC sector land use, land use change and forestry sectors in Ireland. In: Environmental Protection Agency, Wexford, 2007, pp. 61.

Renou-Wilson F, Bolger T, Bullock C, Convery F, Curry J, Ward S, et al. BOGLAND: Sustainable Management of Peatlands in Ireland. STRIVE 75. EPA., Johnstown Castle, Co. Wexford, Ireland, 2011.

Rodgers M, O'Connor M, Healy MG, O'Driscoll C, Asam Z-u-Z, Nieminen M, et al. Phosphorus release from forest harvesting on an upland blanket peat catchment. For Ecol Manag 2010; 260: 2241-2248.

Ryder L, de Eyto E, Gormally M, Sheehy Skeffington M, Dillane M, Poole R. Riparian zone creation in established coniferous forests in Irish upland peat catchments: Physical, chemical and biological implications. Biol and Environ 2011; 111.

Shrestha RK, Lal R, Penrose C. Greenhouse Gas Emissions and Global Warming Potential of Reclaimed Forest and Grassland Soils. J Environ Qual 2009; 38: 426-436.

Silins U. Rothwell RL. Spatial patterns of aerobic limit depth and oxygen diffusion rate at two peatlands drained for forestry in Alberta. Can J Forest Res 1999; 29: 53–61.

Silvola J. Alm J. Ahlholm U. Nykänen H. Martikainen PJ. CO_2 fluxes from peat in boreal mires under varying temperature and moisture conditions. J Ecol 1996; 84: 219–228.

Sottocornola M, Kiely G. Energy fluxes and evaporation mechanisms in an Atlantic blanket bog in southwestern Ireland. Wat Resour Res 2010; 46.

Strahler AN. Quantitative geomorphology of drainage basin and channel networks. Handbook of Applied Hydrology 1964.

Tamai, K, Hsia, Y-J. CO2 flux observation in various forests of Monsoon-Asia J Forest Res 2012; 17:225–226.

UNEP F, UNFF. Vital forest graphics. In: Arendal UG, editor. UNEP, Nairobi, 2009.

Van Beek CL. Hummelink EW. Veltof GL. Oenema O. Denitrification rates in relation to groundwater level in a peat soil under grassland. Biol Fert Soils 2004; 39: 329–336

Von Arnold K, Weslien P, Nilsson M, Svensson BH, Klemedtsson L. Fluxes of CO_2, CH_4 and N_2O from drained coniferous forests on organic soils. For Ecol Manag 2005; 210: 239-254.

Waddington J, Day S. Methane emissions from a peatland following restoration. J Geophys Res 2007; 112: G03018.

Watson RT, Noble IR, Bolin B, Ravindranath NH, Verardo DJ, Dokken DJ. Land Use, Land-Use Change and Forestry. A Special Report of the IPCC. Cambridge, UK: Cambridge University Press, 2000.

Wellock ML, Reidy B, Laperle CM, Bolger T, Kiely G. Soil organic carbon stocks of afforested peatlands in Ireland. Forestry 2011; 84: 441-451.

Zerva A, Mencuccini M. Short-term effects of clearfelling on soil CO2, CH4, and N2O fluxes in a Sitka spruce plantation. Soil Biol Biochem 2005; 37: 2025-2036.

Chapter 6

The contents of this chapter have been submitted to Forest Ecology and Management. Joanne Finnegan developed the experimental design and collected, analysed and synthesized the experimental data. She is the primary author of this article. Dr. Mark G. Healy contributed to the experimental design and paper writing. Dr. John T. Regan and Mark O'Connor assisted with sampling, the experimental design and paper editing.

Implications of applied best management practice for peatland forest harvesting

J. Finnegan[1], J.T. Regan[1], M. O'Connor[1, 2], and M.G. Healy[1]

[1]*Civil Engineering, National University of Ireland, Galway, Ireland*
[2] *Marine Institute, Newport, County Mayo, Ireland*

ABSTRACT

Elevated levels of nutrients and suspended sediment (SS), and changes to other environmental parameters, are frequently associated with forestry harvesting (clearfelling) operations for up to 4 years, and are indicative of the potentially complex changing environment associated with clearfelling. Current and future recommended best management practices (BMPs) for forestry clearfelling on upland peat catchments must provide for a healthy soil and good water quality. The aim of this study was to quantify the effects of implementation, or violation, of BMP in the clearfelling of an upland peat conifer forest. Over periods of 12 months prior to clearfelling and 15 months after clearfelling, two peatland forests, comprising a study control (no clearfelling) and a study site (clearfelling), were monitored for the release of phosphorus (P) and nitrogen (N) species (dissolved reactive phosphorus (DRP), total phosphorus (TP), total oxidised nitrogen (TON) and ammonium nitrogen (NH_4^+-N)), SS, dissolved oxygen (DO), electrical conductivity (EC), pH and stream water temperature. Clearfelling was conducted during poor weather conditions and a watercourse, which drained the study site, was not protected. The maximum recorded concentrations exported from the study site after clearfelling was 133 μg L^{-1} for DRP, 368 μg L^{-1} for TP, 226 μg L^{-1} for NH_4^+-N, and 194 μg L^{-1} for TON. Concentrations of SS exiting the study site increased only once in samples taken during clearfelling (maximum release of 481 mg L^{-1}, with 68 % of this organic) and returned to pre-clearfelling levels, or below, within 6 months of the commencement of clearfelling. Despite significant rises in nutrients and SS on site, and changes to some water parameters, the implementation of BMP, where possible, and the quick implementation of a site restoration plan comprising silt traps and water management on extraction racks, appeared to negate excessive nutrients and SS export to the adjoining watercourse.

6.1 Introduction

Ireland’s forest cover stands at 10 % (698,000 ha) of the total surface area of the island and 59.6 % of total afforestation is on peat (National Forest Inventory, 2007). Most of this forestry is now at harvestable age (Renou-Wilson et al., 2011) and clearfelling (harvesting) of forestry on peat is seen as particularly sensitive to soil erosion (Forest Service, 2000a). Clearfelling of this forestry may cause elevated levels of nutrients (Cummins and Farrell, 2003a; Rodgers et al., 2010) and suspended sediment (SS) (Rodgers et al., 2011) in adjacent waterways for up to 4 years after it has taken place (Adamson and Hornung, 1990; Neal et al., 1999). Therefore, current and future recommended best management practices (BMPs) for forestry clearfelling on upland peat catchments must consider soil and water quality (Collins et al., 2000).

Forestry operations on peatland throughout the world are now moving towards a ‘progressive management approach’ (Joosten and Clarke, 2002), which aims to reduce the potentially negative effects to the surrounding environment. Coillte, the Irish State’s current forest management company, is certified under the Forest Stewardship Council (FSC) to enforce strict environmental, economic and social criteria for sustainable forest management (Coillte, 2012). These criteria advocate detailed planning (prior to the commencement of clearfelling) to provide protection to watercourses from drainage, fertilisation and afforestation, final harvest and regeneration (Owende et al., 2002). The ‘Code of Best Forest Practice – Ireland’ (Collins et al., 2000), and the associated guidance documents (Forest Service, 2000a,b,c,d,e,f), which are based on the principles of Sustainable Forest Management (SFM), contain BMPs for all forestry operations, including nursery practices, planting, thinning and transport of materials (Collins et al., 2000). Under present BMP, management of final harvest needs to include consideration of fell coupe size and shape, road construction, soil type and sensitivities, local watercourses, extraction routes and landing areas (Collins et al., 2000) (Table 6.1). In particular, the practice of clearfelling in dry weather, the use of brash mats (logging residues used for machinery traffic) and ancillary structures such as silt traps, are recommended (Forest Service, 2000a). Harvest site restoration guidelines include provisions for drain and road repair, and water management on extraction routes (Forest Service, 2000a) in order to prevent, or reduce, excessive loss of nutrients and sediment to receiving watercourses.

Table 6.1 Best management practice (BMP) from 'Forest Harvesting and the Environmental Guidelines' (Forest Service, 2000c) and 'Forest and Water Quality Guidelines' (Forest Service, 2000a) with applied BMP at the Glennamong study site.

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

Nutrients such as nitrogen (N) and phosphorus (P) are often applied to land at the afforestation stage to enhance and promote growth of selected species within ombrotrophic blanket peats (peats which have low nutrient concentrations and poor adsorption capacities) of the west of Ireland (Farrell and Boyle, 1990; Renou and Farrell, 2005). This, combined with N deposition from the atmosphere and ammonification within the peat layers, has led to N saturation, primarily present as ammonium (NH_4^+), in some upland peat catchments in the UK (Daniels et al., 2012). Ammonium can leach from the peat and be converted to nitrate (NO_3^-) by nitrification within the streams (Daniels et al., 2012), leading to toxic environments for aquatic life forms (Stark and Richards, 2008). Similarly, small concentrations of P (> 35 $\mu g\ L^{-1}$ molydbate reactive phosphorus (MRP)) can have a negative impact on water quality (Bowman, 2009), leading to restrictions for fisheries, recreation, industry and drinking water (Sharpley, 2003; Elrashidi, 2011). Blanket peat has a poor adsorption capacity for P (O'Driscoll et al., 2011) and during the forest operations of drainage, fertilisation and clearfelling, hydrological losses of P can increase (Cummins and Farrell, 2003a; Nieminen, 2003; Väänänen et al., 2008). Phosphorus loss during clearfelling is mainly due to loss from foliage (Paavilainen and Päivänen, 1995), and during clearfelling up to 70 % of P may be lost during high storm events (Rodgers et al., 2010). However, the P levels in receiving waters can return to pre-clearfell levels within 4 years of clearfelling (Rodgers et al., 2010). Peat soils are also susceptible to damage by clearfelling machinery traffic and subsequent rutting and compaction (Collins et al., 2000). After clearfelling, SS levels in receiving waters can increase due to soil disturbance, bank erosion and increased flow from the harvested areas, but these impacts are generally not long-term (Rodgers et al., 2011).

Other environmental parameters, such as dissolved oxygen (DO) (Ensign and Mallin, 2001), electrical conductivity (EC) (Cummins and Farrell, 2003b), pH (Neal et al., 1992) and stream water temperature (Stott and Marks, 2000), may be impacted by clearfelling, and are indicative of the potentially complex changing environment associated with forestry harvesting (Rodgers et al., 2008). An increase in biochemical oxygen demand (BOD) from increased organic material and algal blooms can decrease the DO within waterbodies downstream of clearfelled areas (Ensign and Mallin, 2001). Similarly, DO is lower in lakes influenced by afforestation compared to unforested blanket bog lakes (Drinan et al., 2012).
Stream water temperature is seen as one of the best indicators of stream vitality, and can be affected by forestry operations such as afforestation and deforestation (Stott and Marks, 2000; Quinn and Wright-Stow, 2008). Studies in the UK have shown that a decrease in stream water

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

The upland peat catchments of the west of Ireland are classified as acid sensitive with the main sources of pressures on rivers coming from forestry and peat degradation (O' Driscoll et al., 2012). The typical low pH values (approximately 4) of these catchment streams is assumed to arise from the high runoff from low permeability, acidic soils, with little interaction with groundwater to neutralise the acidity, as seen in similar sites in the UK (Neal et al., 2004). Forests may exacerbate the existing acid conditions both indirectly, through canopy interception of atmospheric pollutants, and directly, by the uptake of base cations and nutrients during biomass growth and subsequent removal from site during clearfelling (Johnson et al., 2008). Little is known about the impact of clearfelling on the pH concentration in upland peat forestry in Ireland.

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

6.2 Materials and Methods

6.2.1 Study site description

The study area was located in the Glennamong forest in the Burrishoole catchment in Co. Mayo, Ireland (ITM reference 494252, 803180) (Figure 6.1). Two adjacent sub-catchments were studied: (1) a control catchment (CC), in which no clearfelling or forestry operations took place and (2) a study catchment (SC), in which clearfelling of the catchment took place (identified as ‘Control’ and ‘Study’ in Figure 6.1). The CC and SC are each approximately 10 ha in area, and each is drained by a small stream instrumented with sampling equipment (identified as ‘Steams’ and ‘Sampling’ in Figure 6.1). These streams flow into the Glennamong River, which is a fourth-order river at the point of entry of the streams (Strahler, 1964). The study area is situated at an approximate elevation of 95 m above ordnance datum (AOD) and there is a moderate climate, which is heavily influenced by the proximity of the Atlantic Ocean. The average air temperature is 13 °C in summer and 4 °C in winter, while the mean annual rainfall for the catchment is 2000 mm (Rodgers et al., 2011). The catchment has a low buffering capacity and has been classified as acid oligotrophic (O'Driscoll et al., 2012). Blanket peat of varying depth down to 1 m covers the site, which overlays an Anaffrin formation of quartzite and schist bedrock (McConnell and Gatley, 2006). The blanket peat is an *in situ* blanket mire with an average gravimetric water content of 85 % and a dry bulk density of approximately 0.1 g cm^{-3}.

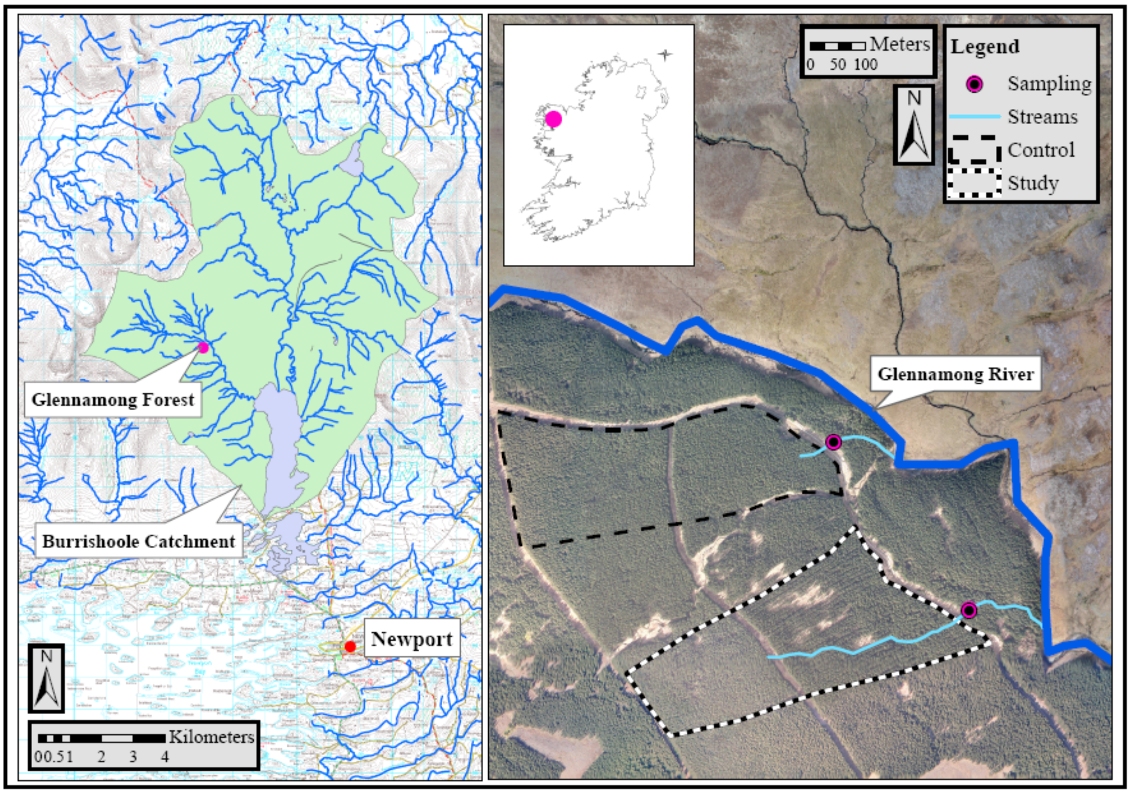

Figure 6.1 Location of the Glennamong Forest control catchment (CC) and study catchment (SC).

The site was planted with Lodgepole Pine (*Pinus contorta*) in 1972. The CC is at the same topographical location as the SC, and has a similar slope and peat depth. Clearfelling of the SC commenced on February 8, 2011. Bole-only clearfelling, which involves the removal of only the merchantable timber from site, leaving the branches and logging residue (brash material) to degrade on site, was carried out with a harvester (Timberjack 1470D) and a forwarder (8-wheeled, Timberjack 1110). A total of 14.8 ha of forestry was clearfelled, of which 9.4 ha drained into the small stream in the SC. This stream has a mineral bed, no buffer strip due to its size, occasionally goes underground and is not identified on Ordinance Survey (OS) maps (Figure 6.2). Therefore, very little care was afforded to the stream during clearfelling, and occasionally brash mats were laid over the stream and parallel to the path of the stream (Figure 6.3).

Figure 6.2 Small stream (in low flow) in SC with mineral bed, pre-CF.

Figure 6.3 Same location as Figure 6.2, post-CF.

Operations continued during heavy rainfall (Figure 6.4) and resulted in deep rutting (up to 1.5 m) on the main extraction racks (Figure 6.5). Timber was removed from the site *via* extraction racks running parallel to the slope of the site, and was deposited at a timber landing area adjacent to the road. Harvesting finished at the end of March 2011 and forwarding continued until the middle of April 2011. Temporary silt traps were installed on completion of forwarding and extra brash was placed on the rutted extraction racks for water management control (Figure 6.6 and 6.7). Three permanent silt traps, preceded upslope by settling ponds, were constructed with filter stone and geotherm at the end of April 2011 (Figure 6.8 and 6.9). No drain cleaning took place on site and, to date, no maintenance of silt traps has been conducted. Windrowing (arranging the brash mats into piles) and replanting is planned for the site in 2013.

Figure 6.4 Ponding of water following heavy rainfall during clearfelling.

Figure 6.5 Deep rutting on extraction racks.

Figure 6.6 During clearfelling ponding on extraction rack.

Figure 6.7 Post clearfelling with brash placement for water management.

Figure 6.8 Permanent silt trap, preceded upslope by a settling pond.

Figure 6.9 Permanent silt trap on road side drain.

6.2.2 Measurement and analysis

Installation of H-flumes (or open channel flow nozzles) and water level recorders for flow measurement (OTT SE200, Germany), data sondes (Hydrolab, USA) for continuous measurement (every 5 minutes) of environmental parameters (DO, EC, pH and temperature) and ISCO samplers (Teledyne ISCO, USA) for stream water collection in the two streams in the CC and SC (identified as 'Sampling' in Figure 6.1) began in February 2010. The upper flow limit of the H-flumes was 148 L s^{-1}. The sondes were removed for calibration every 8-10 weeks. For analysis, the SC was divided into pre-clearfell (pre-CF) and post-clearfell (post-CF) periods. Pre-CF data collection was from February 2010 to February 2011 (12 months pre-CF data) and post-CF data collection was from February 2011 to May 2012 (15 months post-CF data). The nutrient and sediment release during a total of 18 storm events (n=8 pre-CF, n=2 during CF and n=8 post-CF; and n=24 samples within each storm event), over 24- or 48-hour time periods, were monitored using the ISCO samplers in the CC and SC streams (Figure 6.10). A weather station (Vantage Pro 2, Davis, USA) was positioned at the study site.

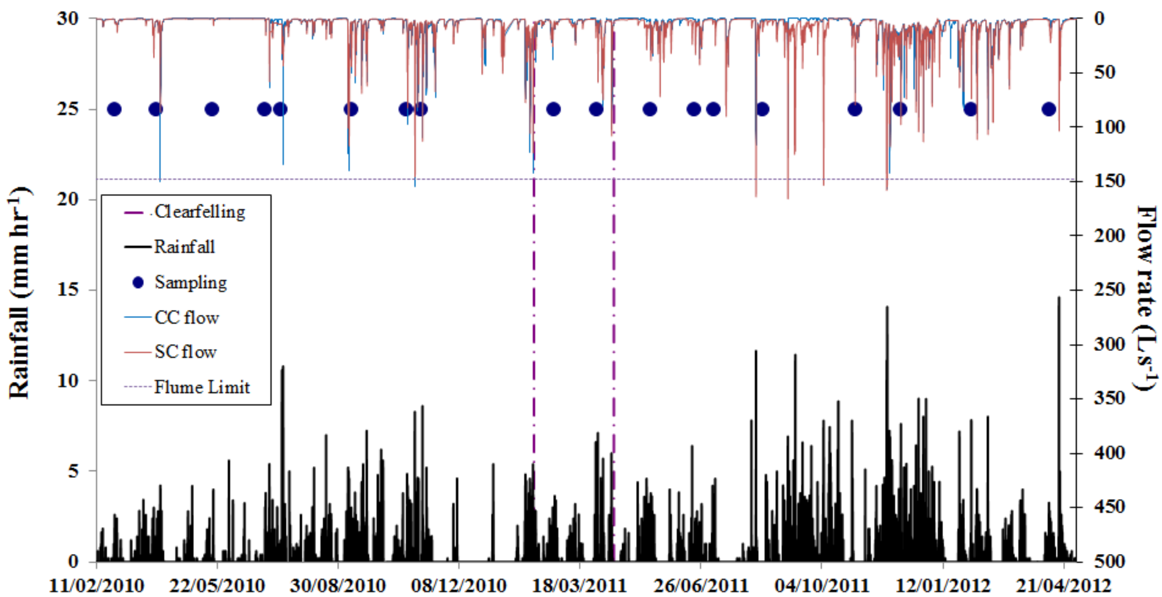

Figure 6.10 Rainfall (mm hr^{-1}) from the Glennamong weather station and stream water sampling dates from February 2010 to May 2012. Flow rates (L s^{-1}) from the control catchment (CC) and study catchment (SC) are on the inverted secondary axis.

After collection, all water samples were returned to the laboratory and tested the following day or frozen (at -20°C) for testing at a later date. The water quality parameters measured were: (1) dissolved reactive phosphorus (DRP) (2) total phosphorus (TP) (3) NH_4^+-N (4) total oxidised nitrogen (TON = NO_3^- + nitrate (NO_2^-)) and (5) SS. The SS component was also broken into organic suspended sediment (OSS) and mineral suspended sediment (MSS). All water samples were tested in accordance with standard methods (APHA, 1998) using a nutrient analyser (Konelab 20; Thermo Clinical Labsystems, Finland). Suspended sediment testing was carried out by passing a known volume of water through a pre-dried and weighed 1.2 µm GF/C filter disc (Whatman, England) under suction. The filter was then dried at 105 °C for 24 hr and reweighed for OSS (APHA, 1998) and the MSS was determined by loss on ignition (LOI) at 550 °C (BSI, 1990).

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

Nutrient data were $\log_{10}$ transformed and analysed with ANOVA (analysis of variance) in Datadesk (Data Description Inc., USA) to determine the main sources of variation. Date and the location of the sample site were included as explanatory variables.

6.3 Results and Discussion

6.3.1 Best management practice

Best management practice in forestry clearfelling has been shown to be effective at decreasing non-point source pollution (Aust and Blinn, 2004; McBroom et al., 2008; Rodgers et al., 2011). A comparison of the actual clearfelling practice at the study site, in comparison to BMP (Forest Service, 2000a, b), is shown in Table 6.1. As far as practicable, clearfelling at the study site was carried out in accordance with BMP, with harvesting plans, coupe size, timber landing areas, use of brash mats, site restoration and machine servicing being conducted. Road planning and construction was not necessary due to the existing road on site. A number of deviations in the implementation of BMP were encountered due to the specific site conditions, which included lack of dry weather and avoidance of watercourses. The study site received a total of 5250 mm of rain on 625 rain days over the duration of the present study (February 2010 to May 2012; 821 days in total). This allowed for only 196 days for clearfelling operations. In 2011 alone, there was 3037 mm of rainfall recorded from the rain gauge situated in the SC. During the clearfelling operations, which lasted approximately 80 days, there were 59 days of rainfall (387 mm in total), with 44 wet days (rainfall over 1 mm of rain). Due to time constraints and availability of the machines, clearfelling was conducted during poor weather conditions.

The guidelines (Forest Service, 2000b) define an aquatic zone ‘as a permanent or seasonal river, stream, or lake shown on an Ordnance Survey 6 inch map’. The stream draining the SC is not on the Ordnance Survey 6 inch map, as it is little more than a drainage channel and occasionally goes underground. Upland spate streams are very characteristic of peat catchments in the west of Ireland, particularly within the study catchment (Allott et al., 2005) and during periods of high rainfall, this small stream carried large volumes of water from the catchment to the receiving river. Due to the sensitive nature of peatland sites, these small streams, despite their lack of order number, should be protected during clearfelling operations. In the present study, temporary silt traps were installed at the end of clearfelling, but this may have been too late to prevent SS export during the clearfelling process (Section 6.3.2). However, the impacts of clearfelling would appear to be negated by the installation of permanent silt traps and water management on extraction racks, which entailed the placement of extra brash on rutted extraction racks to reduce the runoff from these disturbed areas (Section 6.3.2). A rapid site restoration plan, undertaken in the present study, prevented excessive nutrient and SS release to the study stream.

6.3.2 Nutrient and SS concentration

Throughout the entire study period, there was no significant difference between the pre-CF and post-CF nutrient concentrations in the CC, and prior to clearfelling, there was no significant difference between the CC and SC for any of the nutrient concentrations measured. A summary of nutrient concentrations post-CF is shown in Table 6.2.

Table 6.2 Maximum concentrations (μg L^{-1}) pre- and post- clearfelling for dissolved reactive phosphorus (DRP), total phosphorus (TP), total oxidised nitrogen (TON) and ammonium-nitrogen (NH_4^+-N) from the current study site and comparable study sites worldwide.

Reference	Location	Area of CF (ha)	Type of harvesting	Soil type	Average Annual Rainfall	Max concentrations pre-clearfelling (μg L^{-1})				Max concentrations post-clearfelling (μg L^{-1})			
						TP	DRP	TON	NH_4-N	TP	DRP	TON	NH_4-N
Cummins and Farrell (2003 a, b)	Galway, Ireland	1	Bole only clearfelling	Peat	1600	-	13	≈ 400	≈ 300	-	4164	≈ 3000	≈ 1800
Ensign and Mallin (2001)	Northern Carolina, USA	52.6	Clearcut with track cutter and shovel logger	Swamp soils	1270	188	47[a]	581[b]	146	427	297[a]	191[b]	440
Neal (2004)	Plynlimon, Mid-Wales	< 1	Bole only clearfelling	Peaty gley	2500	-	30[a]	-	160	-	550[a]	-	1120
Nieminen (2003)	Southern Finland	7	Bole only clearfelling	Peat	600	-	< 10[a]	< 20[b]	< 25	-	100[a]	< 20[b]	< 15
Rodgers et al. (2010)	Mayo, Ireland	25.3	Bole only clearfelling	Peat	2000	28	-	-	-	201	-	-	-
Present study	Mayo, Ireland	9.4	Bole only clearfelling	Peat	3000	80	33	128	162	368	133	194	226

[a] measured as orthophosphate in these studies.

[b] measured as nitrate in these studies.

Dissolved Reactive Phosphorus and Total Phosphorus

There were significant increases in DRP (Figure 6.11) and TP (Figure 6.12) (ANOVA, $p<0.05$) in the SC during/after clearfelling, and both parameters were significantly higher in the SC than the CC after clearfelling. The limit for MRP, which is similar to DRP (Haygarth et al., 1997), for good status of surface water bodies is ≤ 35 µg L^{-1} (S.I. No. 272 of 2009). The FWMC of DRP pre-CF were well below this limit for both sites, and the concentration from the post-CF SC only exceeded this limit on one of the ten sampling dates (FWMC of 39 µg L^{-1} P on October 31, 2011). Concentrations of TP measured in the SC and CC streams in the period prior to the start of clearfelling in the SC were below the EPA critical threshold limit for TP of 62 µg L^{-1} (Coillte, 2008). During and after clearfelling of the SC, the FWMC of TP exceeded this limit on six of the ten sampling dates, but returned to within the critical limit for the final two sampling dates. The maximum concentrations exported from the SC post-CF were 133 µg L^{-1} for DRP and 368 µg L^{-1} for TP. Similar P concentrations were released from a similar sized catchment (20 ha) during the restoration (clearfelling of conifers followed by drain blocking) of a blanket bog in the southwest of Ireland (Coillte, 2008). Increases in DRP and TP of greater magnitude than the present study were measured after clearfelling of a 1-km^2 and a 1-ha peat catchment in the west of Ireland by Cummins and Farrell (2003a). They found that maximum values of MRP increased from 9 µg L^{-1} (1 km^2 catchment) and 93 µg L^{-1} (1 ha catchment) to 256 µg L^{-1} and 3530 µg L^{-1}, respectively, within a few weeks of clearfelling, and the median values obtained were just over 100 µg L^{-1} (1 km^2 catchment) and 1000 µg L^{-1} (1 ha catchment). However, dissimilar to the present study, these values were not flow weighted. Also, unlike the present study, which has mineral content in its stream bed, the stream and drain beds of the Cummins and Farrell (2003a) study consisted of purely peat-based matter and the flowing water had no interaction with mineral material, therefore giving little opportunity for adsorption of P to mineral layers. Similar values to the current study were obtained on a clearfell site in the east of Ireland (Machava et al., 2005), possibly due to the mineral content of the river bed.

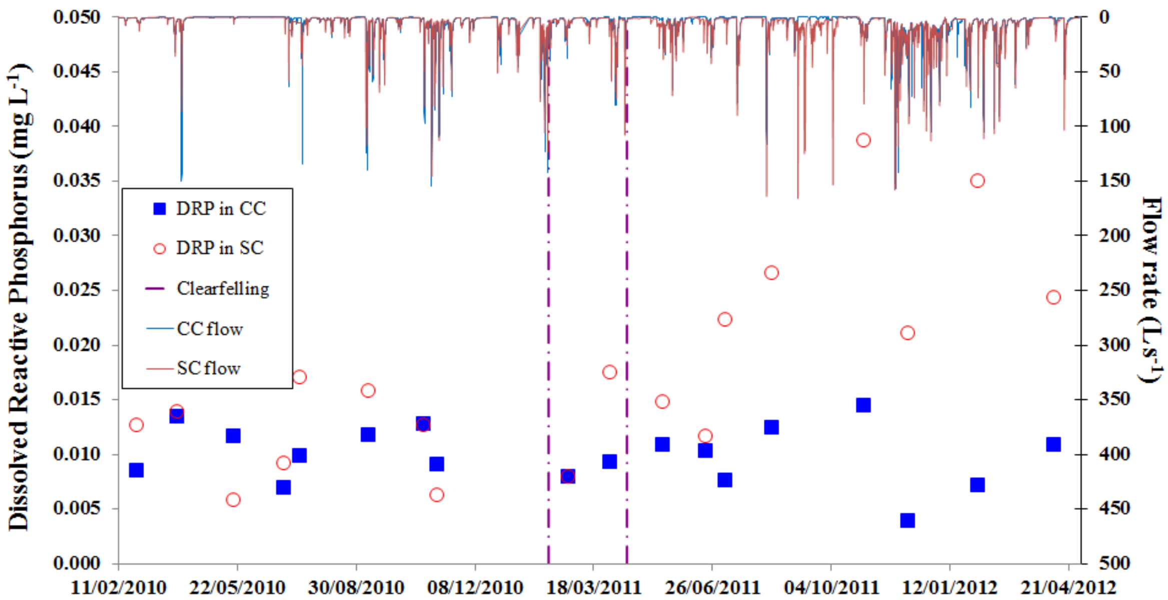

Figure 6.11 Flow-weighted mean concentrations of dissolved reactive phosphorus (DRP) (mg L^{-1}) measured in the control catchment (CC) and the study catchment (SC) from February 2010 to May 2012. Flow rate (L s^{-1}) is on the inverted secondary axis.

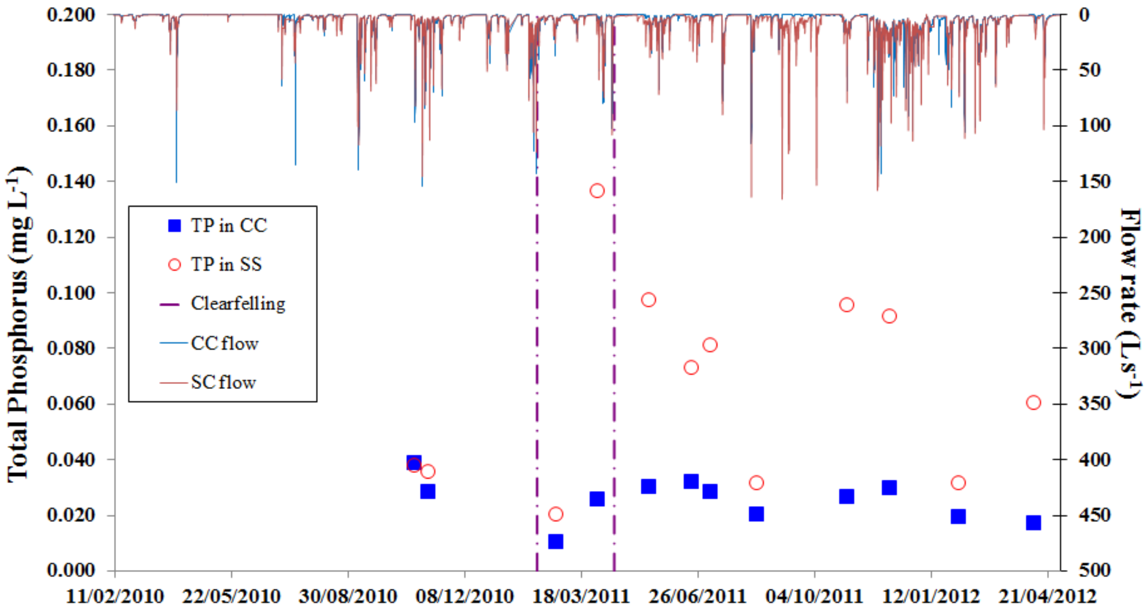

Figure 6.12 Flow-weighted mean concentrations of total phosphorus (TP) (mg L^{-1}) measured in the control catchment (CC) and the study catchment (SC) from February 2010 to May 2012. Flow rate (L s^{-1}) is on the inverted secondary axis.

Rodgers et al. (2010) measured average FWMC of TP in the receiving waters of 14 ± 10 μg L^{-1} prior to the clearfelling of a peatland study site, using BMP. A peak in the TP concentration of 201 µg L^{-1} was reached 5 weeks after the end of clearfelling, but this concentration had reduced back to pre-clearfelling concentrations 10 weeks after felling. The differences in the concentrations of P could be due to weather conditions at the time of felling, depth of peat, or the slope of the site. A second peak in P concentrations within the first year of clearfelling, noted in other studies (Cummins and Farrell, 2003a; Rodgers et al., 2010), was not recorded during the current study. This could be due to the very wet weather conditions experienced during and after clearfelling. The concentrations of P in the receiving waters can return to pre-clearfelling levels within 4 years of harvesting (Rodgers et al., 2010).

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

Ammonium-Nitrogen and Total Oxidised Nitrogen

There were significant increases in NH_4^+-N (Figure 6.13) and TON (Figure 6.14) (ANOVA, p<0.05) in the SC during/after clearfelling, and concentrations were significantly higher in the SC than the CC after clearfelling. The FWMC of NH_4^+-N and TON in the CC and SC before clearfelling was below 0.1 mg L^{-1}. Post-CF, the FWMC of NH_4^+-N and TON rose to a maximum of 0.17 and 0.18 mg L^{-1}, respectively. These values are similar to concentrations found by Coillte on a blanket bog restoration site of a Special Area of Conservation (SAC) in the southwest of Ireland (Coillte, 2008), where discrete concentrations of NO_3^--N were between 0.02 - 0.04 mg L^{-1}.

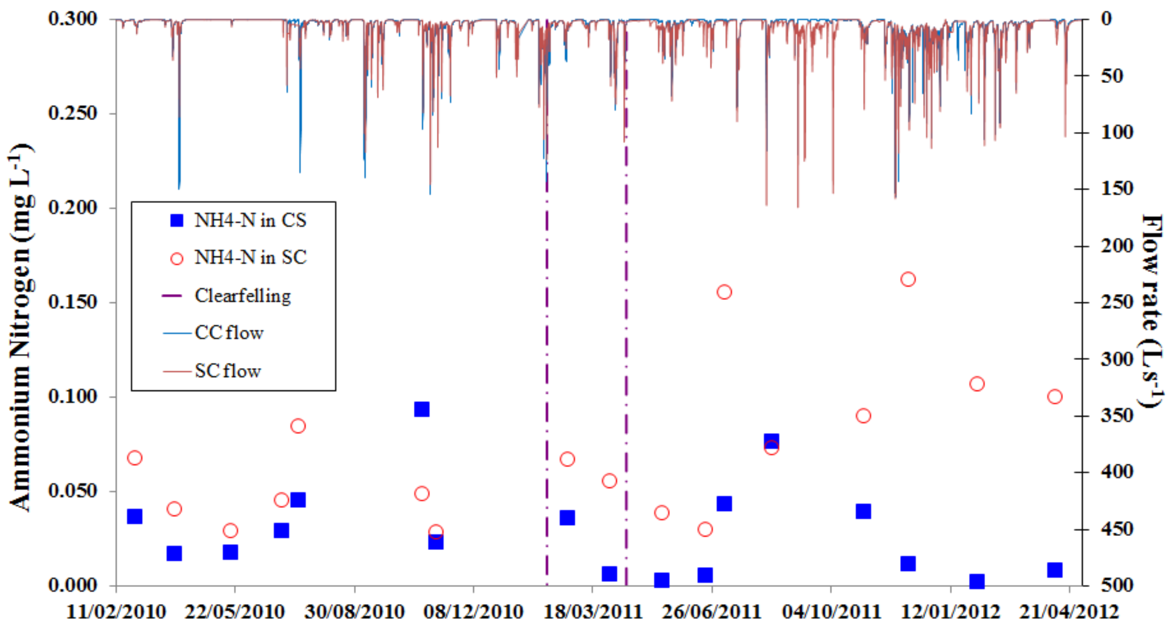

Figure 6.13 Flow-weighted mean concentrations of ammonium-nitrogen (NH_4^+-N) (mg L^{-1}) measured in the control catchment (CC) and the study catchment (SC) from February 2010 to May 2012. Flow rate (L s^{-1}) is on the inverted secondary axis.

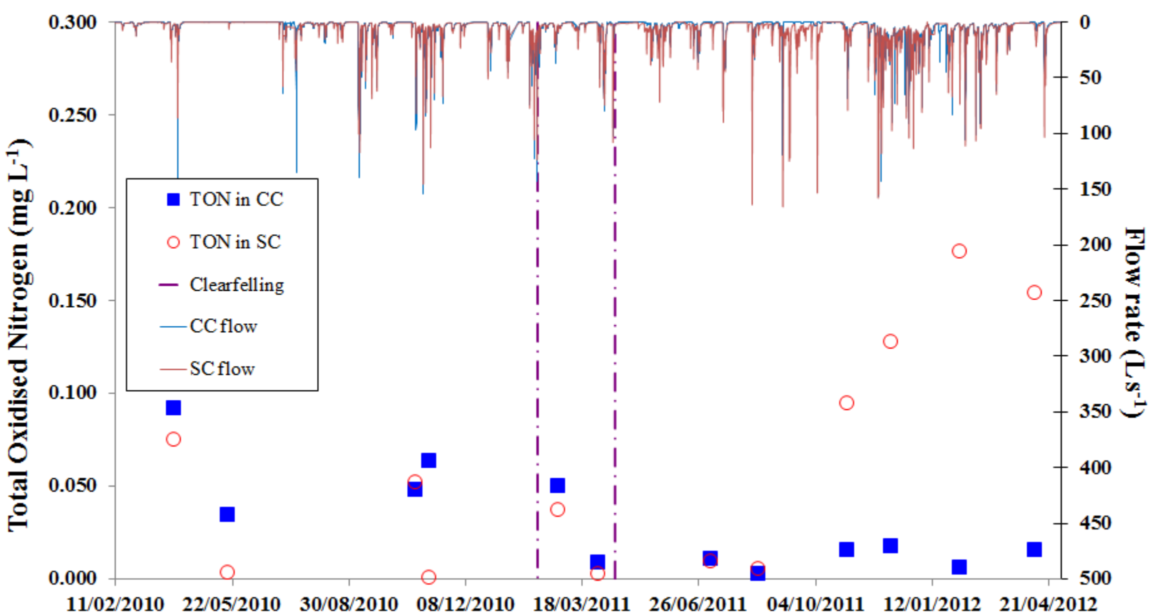

Figure 6.14 Flow-weighted mean concentrations of total oxidised nitrogen (TON) (mg L^{-1}) measured in the control catchment (CC) and the study catchment (SC) from February 2010 to May 2012. Flow rate (L s^{-1}) is on the inverted secondary axis.

There are no critical limits for TON and NH_4^+-N for river water bodies in Ireland. As a proxy value for TON, the critical limits for NO_3^- and nitrite (NO_2^-) are used. There is also no critical limit for NO_3^- or NO_2^- for surface waters, so the critical limits for groundwater are used – 37.5 mg L^{-1} for NO_3^- and 0.375 mg L^{-1} for NO_2^- (S.I. No. 9 of 2010). The concentrations of TON post-CF were below these limits. As a proxy value for NH_4^+-N, the critical limit for total ammonia (ionic-NH_4 + un-ionic NH_3) is used, which has a mean value of 0.065 mg L^{-1}, or 0.14 mg L^{-1} 95 % of the time, for good status of river water bodies (S.I. No. 272 of 2009). The concentration of NH_4^+-N in the SC post-CF exceeded this value. The maximum threshold for NH_4^+-N in groundwater is 0.175 mg L^{-1} (S.I. No. 9 of 2010), and the concentration in the SC post-CF was below this threshold.

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

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

Suspended Sediment

Concentrations of SS in the SC increased only once in samples taken during clearfelling and returned to pre-CF levels, or below pre-CF levels, within 6 months of the commencement of clearfelling (Figure 6.15). This rise during clearfelling was not significant for SS, and there was no significant difference in date or location of sampling for OSS (Figure 6.16) or MSS (Figure 6.17).

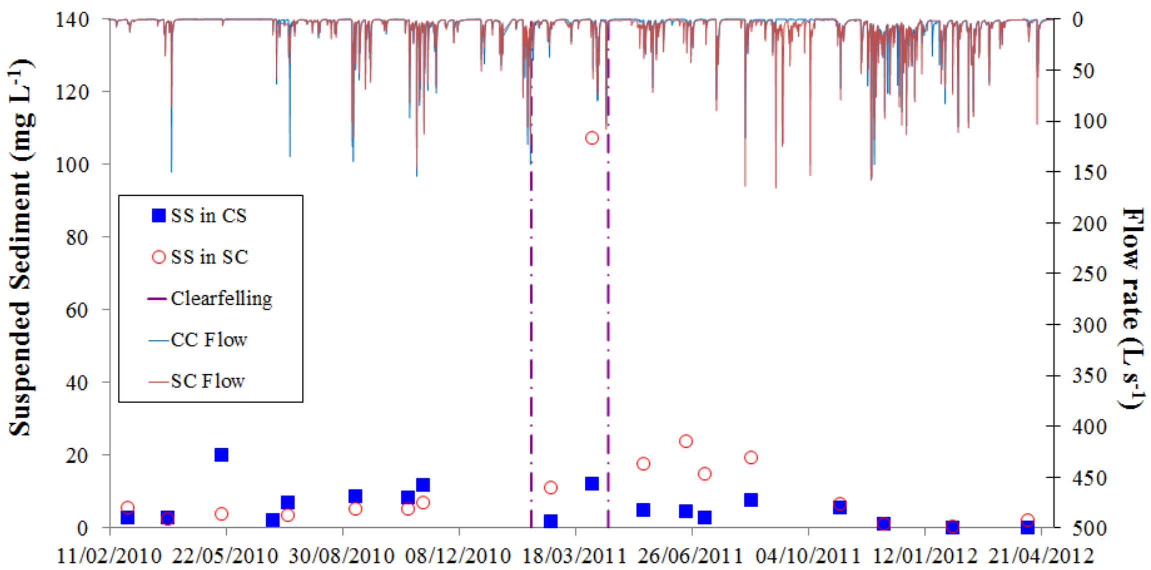

Figure 6.15 Flow-weighted mean concentrations of suspended sediment (SS) (mg L^{-1}) measured in the control catchment (CC) and the study catchment (SC) from February 2010 to May 2012. Flow rate (L s^{-1}) is on the inverted secondary axis.

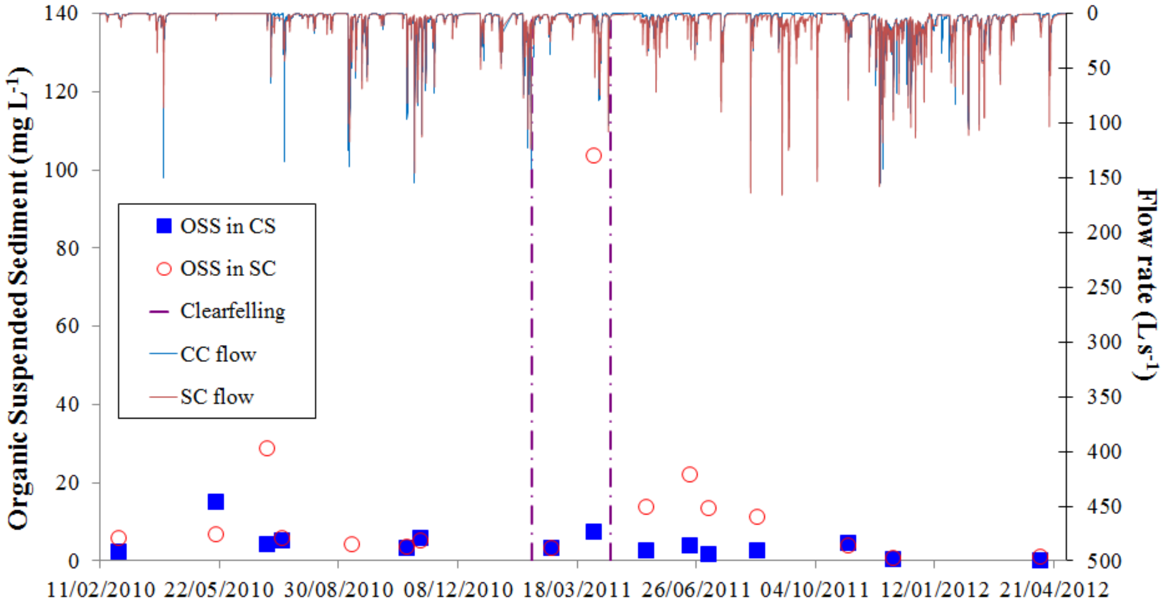

Figure 6.16 Flow-weighted mean concentrations of organic suspended sediment (OSS) (mg L^{-1}) measured in the control catchment (CC) and the study catchment (SC) from February 2010 to May 2012. Flow rate (L s^{-1}) is on the inverted secondary axis.

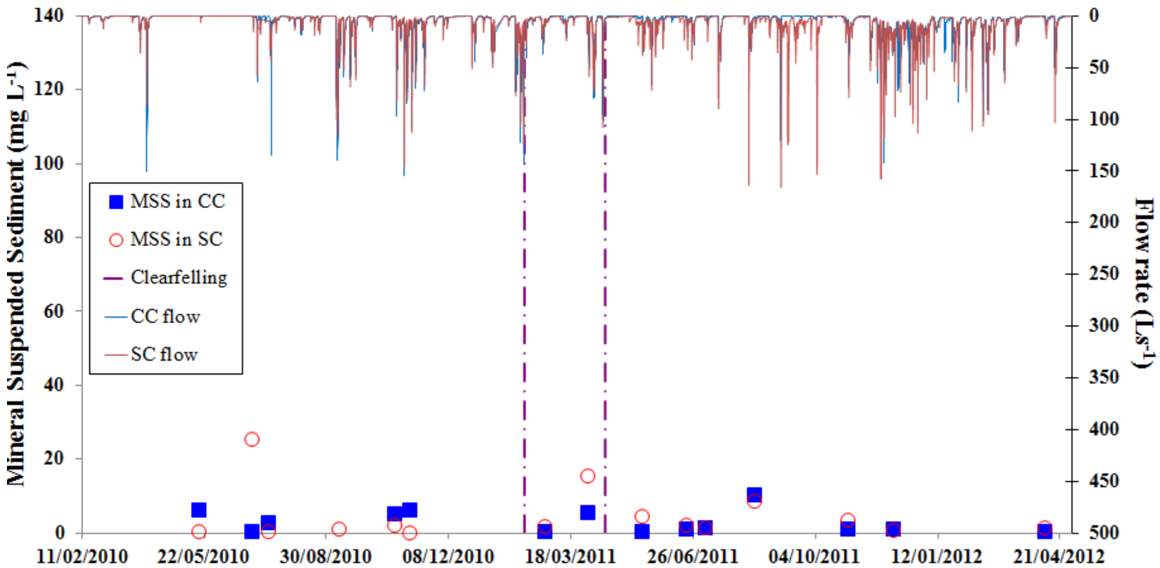

Figure 6.17 Flow-weighted mean concentrations of mineral suspended sediment (MSS) (mg L^{-1}) measured in the control catchment (CC) and the study catchment (SC) from February 2010 to May 2012. Flow rate (L s^{-1}) is on the inverted secondary axis.

Large increases in SS were only noted during one storm, which occurred during the end of the clearfelling period in late April 2011. Over a period of 12 hours, 18.4 mm of rain fell, producing an average flow in the stream of 21.3 L s^{-1} with an associated median SS concentration of 35.4 mg L^{-1} (the maximum release at the peak of the storm was 481 mg L^{-1} SS and 68 % of this was organic material). The recommended level of SS in salmonoid waters is 25 mg L^{-1} (European Community, 1988), therefore the release at peak storm levels was over 19 times greater than the recommended level. However, this was the only time the recommended level was exceeded in all sampling occasions. Following installation of silt traps and extra brash placement on rutted extraction racks at the end of clearfelling, the concentrations of SS returned to pre-CF levels or below pre-CF levels. Similar patterns in SS concentrations was noted by Nieminen (2003) on a peatland clearfell site in southern Finland, with the only significant increase from the most productive, highly fertile mire. Rodgers et al. (2011) also found that clearfelling, following BMPs, on a peat catchment did not significantly increase SS concentrations after clearfelling and that no adverse impacts on the receiving waters were noted.

Increased sediment export after clearfelling, following implementation of BMP, has been reported by other studies (Kirby et al., 1991; Ensign and Mallin, 2001; Aust and Blinn, 2004; McBroom et al., 2008; Ryder et al., 2011). Variations in results can relate to different site slopes, weather conditions and the rate of vegetation growth post-CF (Rodgers et al., 2011). Higher rates of sediment loss are associated with steeper slopes (McBroom et al., 2008) and the rapid regeneration of vegetation within clearfelled areas can reduce SS export (Aust and Blinn, 2004). However, establishing ground vegetation can be slow on sites where brash material has not been removed (Broadmeadow and Nisbet, 2004).

6.3.3 Water parameters: DO, EC, pH and temperature

In the SC, there was a change in DO, EC, pH and stream water temperature during and post-CF. Prior to clearfelling, there was no significant difference between CC and SC for EC or stream water temperature. However, the pH and the DO from both sites were significantly different from each other pre-CF (ANOVA, $p<0.05$), but followed the same pattern, with the CC having significantly higher values and the SC significantly lower values for both parameters. Post-CF, the CC had significantly higher values of DO (Figure 6.18) and EC (Figure 6.19), while the pH (Figure 6.20) and temperature (Figure 6.21) was significantly lower (ANOVA, $p<0.05$). These results highlight the importance of continuous data logging, which allows changes in levels over time to be easily identified.

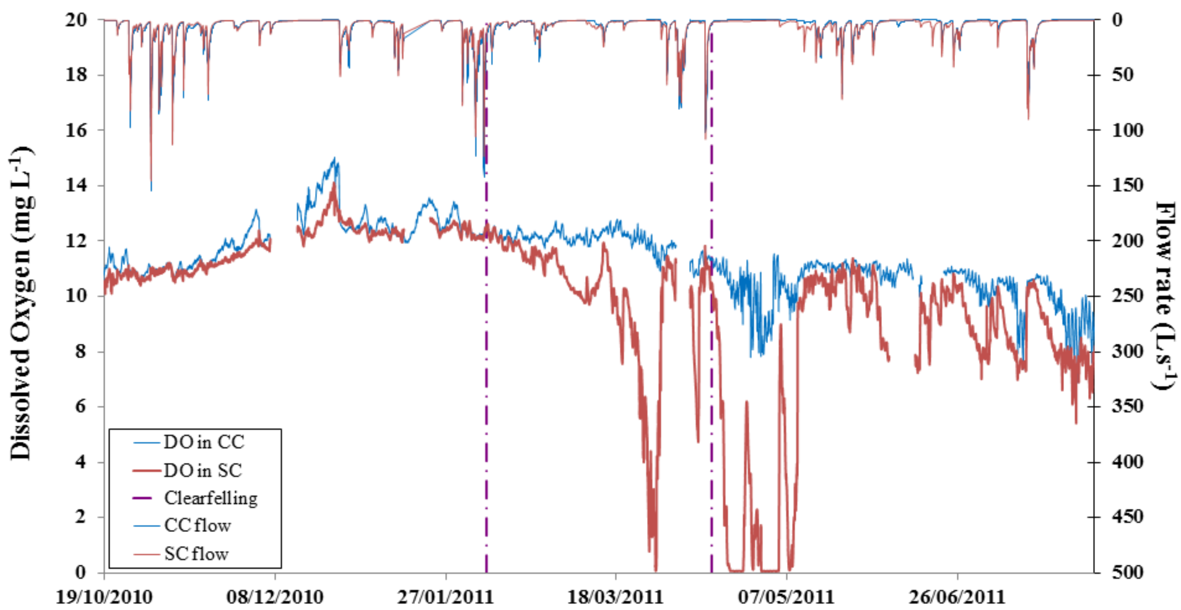

Figure 6.18 Dissolved oxygen (DO) (mg L^{-1}) at 5-minute intervals measured in the control catchment (CC) and the study catchment (SC) from October 2010 to July 2011. Flow rate (L s^{-1}) is on the inverted secondary axis.

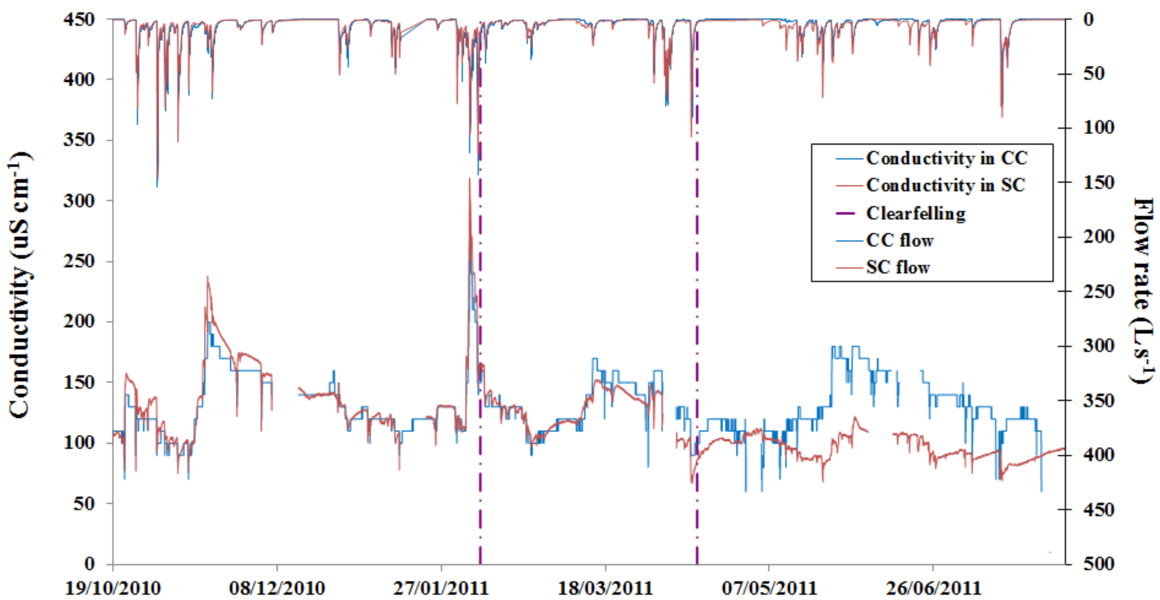

Figure 6.19 Electrical conductivity (EC) (µS cm^{-1}) at 5-minute intervals measured in the control catchment (CC) and the study catchment (SC) from October 2010 to July 2011. Flow rate (L s^{-1}) is on the inverted secondary axis.

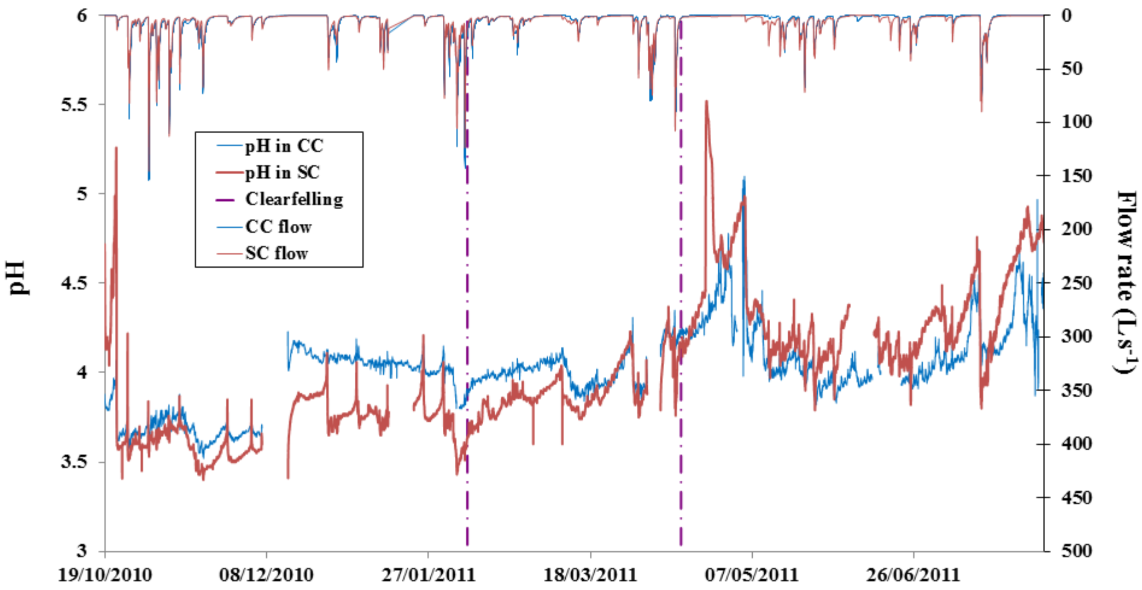

Figure 6.20 pH at 5-minute intervals measured in the control catchment (CC) and the study catchment (SC) from October 2010 to July 2011. Flow rate (L s^{-1}) is on the inverted secondary axis.

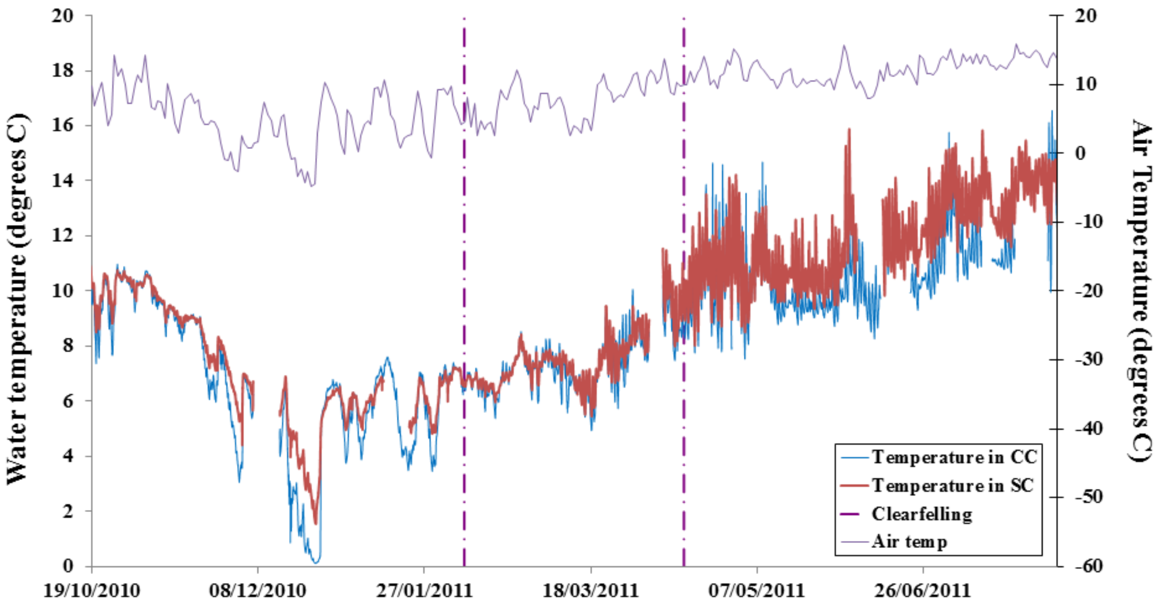

Figure 6.21 Stream water temperatures (°C) at 5-minute intervals measured in the control catchment (CC) and the study catchment (SC) from October 2010 to July 2011. Air temperatures (°C) from the weather station are on the inverted secondary axis.

Dissolved Oxygen

Prior to clearfelling, the DO levels at both sites were significantly different from each other (ANOVA, $p<0.05$), with the CC having significantly higher values than the SC (Figure 6.18). During clearfelling, the DO dropped to zero in the SC, and continued to fluctuate for up to one month after the end of felling. This may have been due to the development of an algal bloom arising from nutrients produced by the logging residue in the SC, although no evidence of this was noted on site during visits. It is more plausible that the higher concentrations of OSS measured during clearfelling affected the DO concentration within the receiving waters due to the organic component being biologically active and thus utilising oxygen during decomposition (Rodgers et al., 2011). Extra light to the stream, provided by the removal of the tree canopy, may also have enhanced algal blooms in the stream of the SC, if these occurred. A similar pattern was noted by Ensign and Mallin (2001) on a wetland clearfell site in the eastern US, which they attributed to an increased BOD load from logging residues and algal blooms.

Electrical Conductivity

There was no significant difference in EC of both streams pre-CF, and the EC of the SC was generally above, or the same as, that of the CC. During periods of low flow or dry weather, the EC dropped to zero due to the sonde being exposed to the atmosphere (these values have been removed from the graphs for clarity). During clearfelling, the EC dropped in the SC and stayed below that of the CC for the remainder of the study (Figure 6.19; ANOVA, $p<0.05$). An identical pattern was found in the restoration of a blanket bog in the southwest of Ireland (Coillte, 2008), and Cummins and Farrell (2003a) found, on a clearfelled peatland site, that values of EC reduced after clearfelling. This reduction in EC, despite an overall increase in nutrient concentration in the stream, was most likely due to the dilution effect from an increase in water discharge post-CF.

pH

The pH was consistently higher in the CC than the SC pre-CF. By the end of clearfelling, this pattern swapped, with the SC having a significantly higher pH (Figure 6.20; ANOVA, $p<0.05$). The pH measured in a stream during restoration (clearfelling of conifers followed by drain blocking) of a blanket bog in the southwest of Ireland (Coillte, 2008) varied from 7.5 during low flow to approximately 4.3 during peak storm events, which is characteristic of acid sensitive blanket bogs. Similarly, in the present study, the initial high pH (seen at the end of October 2010) and the observed peaks in April 2011 followed dry periods when the pH was elevated due to more interaction with the bedrock in the stream. Rodgers et al. (2008) attributed the higher values of pH during low flow to a greater residence time within their study site, and interaction with an aquifer located above their sampling point.

There are few other long-term data in Ireland on changes in pH levels following harvesting (Johnson et al., 2008). Long-term studies in the UK (Neal et al., 1992) all show a slight decrease, or no change, in the pH after clearfelling. Dissimilar to these studies, Cummins and Farrell (2003b) observed an elevated pH immediately after clearfelling on a peatland site, and attributed this to the road side location of the sampling point. However, elevated pH levels have been reported by other researchers (Ryder et al., 2011) on peatland sites, which were not attributable to surface runoff from roads. Road runoff and drainage were not suspected to be the major causes of increased pH in the present study, as the influence of the road and drainage was identical on both sites pre- and post-CF. The increase in pH post-CF could be due to the decomposition of brash material on site (Staaf and Olsson, 1991), which allowed the return of base cations to the soil (Thiffault et al., 2011).

Temperature

The temperature of the stream water on both sites pre-CF was not significantly different from each other, and both sites responded well to the air temperature changes. Post-CF saw a significant rise in the stream water temperature in the SC and was likely due to the removal of the tree canopy and more light and solar radiation entering the stream (Rodgers et al., 2008) (Figure 6.21; ANOVA, $p<0.05$). A rise in stream water temperature was also noted by Stott and Marks (2000) in a forest clearfell study of a similar size (20 ha) on a peaty gley catchment in mid-Wales, and by Rodgers at al. (2008) in a clearfell study in Ireland. Changes to stream water temperature impacts most on the aquatic fauna of a waterbody (Mellina et al., 2002), and studies have shown results ranging from little recovery of invertebrates after clearfelling (Gee and Smith, 1997) to an increase in the number of mayflies (Kirby et al., 1991). The influence of the increase in stream temperature on the aquatic fauna of the Glennamong catchment was not investigated in the current study.

6.3.4 Outlook for implementation of best management practices

Best management practices in clearfelling operations, as recommended by the forest management organisation in Ireland (Coillte), and the Forest Service guidelines (Forest Service, 2000a,b,c,d,e,f), were mostly followed in this study. The P and N concentrations from the clearfelled site were within the limits for good status of surface water bodies within 15 months of the end of clearfelling, but other water parameters, such as DO, EC, pH and temperature, were affected by clearfelling. Whole tree harvesting (WTH) may reduce the export of nutrients from harvested sites, but this technique leads to the removal of base cations and may have consequences for future rotations (Nisbet et al., 1997). In addition, WTH may further compound the acidification of peatland forested catchments (Ågren and Löfgren, 2012) and therefore is unadvisable in the acid sensitive catchments of the west of Ireland. The leaching of cations from degrading foliage may reverse the effect of acidification in low N-releasing sites (Neal et al., 1999). Nutrient export from nutrient-poor peat, similar to that in the current study, is less likely than from highly productive mires (Nieminen, 2003).

The implementation of BMPs in forestry clearfelling has been shown to be effective at decreasing non-point source pollution to receiving watercourses (Ensign and Mallin, 2001; Wallbrink and Croke, 2002; Aust and Blinn, 2004). Similarly, in an Irish context, any major changes to stream nutrient content, SS concentration, or water parameters are ameliorated by the implementation of BMP such as phased felling and selective coupe sizes (Johnson et al., 2008). On the current study site, the application of these BMPs, despite a number of deviations in their implementation, and the restoration of the site after clearfelling, which included silt trap installation and the laying of extra brash on extraction racks, resulted in limited post-CF export of nutrients and SS by the end of the monitoring period.

6.4 Conclusions

The main conclusions from this study are:

- Following implementation, where possible, of BMP, clearfelling of an upland peat forested catchment in the west of Ireland resulted in limited export of nutrients and SS from site. Maximum concentrations of DRP (133 μg L^{-1}), TP (368 μg L^{-1}) and SS (481 mg L^{-1}) returned to EPA critical threshold limits, or below, within 6 months of the commencement of clearfelling. Site-specific parameters, such as the depth of peat or the slope of a site, and other potential confounding factors, such as the time of felling and weather conditions at the time of felling, may impact on nutrient and sediment release rates, and cognisance should be taken of these factors when drafting a harvest plan.

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

6.4 Acknowledgements

This work was funded by the Department of Agriculture, Fisheries and the Marine (DAFM) and the Environmental Protection Agency (EPA) under the STRIVE program 2007 – 2013. The authors wish to acknowledge the input of Coillte in allowing access to state forestry for this project, and for providing logistical support and advice. A special thanks to Dr. Michael Rodgers and the Marine Institute for their help throughout this project and for supplying the orthophotography.

References

Adamson, J.K., Hornung, M., 1990. The effect of clearfelling a Sitka spruce (Picea sitchensis) plantation on solute concentrations in drainage water. J Hydrol 116, 287-297.

Ågren, A., Löfgren, S., 2012. pH sensitivity of Swedish forest streams related to catchment characteristics and geographical location – Implications for forest bioenergy harvest and ash return. Forest Ecol Manag 276, 10-23.

Allott, N., McGinnity, P., O'Hea, B., 2005. Factors influencing the downstream treansport of sediment in the Lough Feeagh catchment, Burrishoole, Co.Mayo, Ireland. Freshwater Forum, pp. 126-138.

APHA, 1998. Standard methods for the examination of water and wastewater. American Public Health Association, Washington.

Aust, W.M., Blinn, C.R., 2004. Forestry best management practices for timber harvesting and site preparation in the eastern United States: An overview of water quality and productivity research during the past 20 years (1982–2002). Wat. Air Soil Poll. 4, 5-36.

Bowman, J., 2009. New Water Framework Directive environmental quality standards and biological and hydromorphological classification systems for surface waters in Ireland. Biology and Environment 109B, 247-260.

Broadmeadow, S., Nisbet, T.R., 2004. The effects of riparian forest management on the freshwater environment: A literature review of best management practice. Hydrol Earth Syst Sc 8, 286-305.

BSI, 1990. Determination by mass-loss on ignition. British standard methods of test for soils for civil engineering purposes. Chemical and electro-chemical tests. BSI., London.

Coillte, 2008. Technical Final Report for Restoring Active Blanket Bog in Ireland (LIFE02 NAT/IRL/8490). Coillte, Mullingar. (Accesed on 2nd June 2012) http://www.irishbogrestorationproject.ie

Coillte, 2012. (Accesed on 12th May 2012) http://www.coillte.ie/coillteforest/responsible_forest_management_and_certification/certification_introduction/

Collins, K.D., Gallagher, G., Gardiner, J.J., Hendrick, E., McAree, A., 2000. Code of Best Forest Practice - Ireland. Forest Service, Dublin.

Cummins, T., Farrell, E.P., 2003a. Biogeochemical impacts of clearfelling and reforestation on blanket peatland streams I. phosphorus. Forest Ecol Manag 180, 545-555.

Cummins, T., Farrell, E.P., 2003b. Biogeochemical impacts of clearfelling and reforestation on blanket-peatland streams II. Major ions and dissolved organic carbon. Forest Ecol Manag 180, 557-570.

Daniels, S.M., Evans, M.G., Agnew, C.T., Allott, T.E.H., 2012. Ammonium release from a blanket peatland into headwater stream systems. Environ Pollut 163, 261-272.

Drinan, T., Graham, C., O'Halloran, J., Harrison, S., 2012. The impact of conifer plantation forestry on the Chydoridae (Cladocera) communities of peatland lakes. Hydrobiologia, 1-17.

Elrashidi, M.A., 2011. Selection of an appropriate phosphorus test for Soils. (Assessed 12th June 2011) ftp://ftpfc.sc.egov.usda.gov/NSSC/Analytical_Soils/phosphor.pdf.

Ensign, S.H., Mallin, M.A., 2001. Stream water quality changes following timber harvest in a coastal plain swamp forest. Wat Res 35, 3381-3390.

European Community, 1988. EEC of the European Parliament and of the Council of 18 July 1978, on the quality of Salmonid Waters Regulation (the quality of fresh waters needing protection or improvement in order to support fish life). In: Official Journal of the European Communities (Ed.), Directive 78/659/, EEC.

Farrell, E.P., Boyle, G., 1990. Peatland forestry in the 1990s. 1. Low-level blanket bog. Irish Forestry 47, 69-78.

Finnegan, J., Regan, J.T., De Eyto, E., Ryder, E., Tiernan, D., Healy, M.G., 2012. Nutrient dynamics in a peatland forest riparian buffer zone and implications for the establishment of planted saplings. Ecol Eng 47, 155– 164.

Forest Service, 2000a. Forest Harvesting and the Environment Guidelines. In: Department of the Marine and Natural Resources (Ed.), Dublin.

Forest Service, 2000b. Forest and Water Quality Guidelines. In: Department of the Marine and Natural Resources (Ed.), Dublin.

Forest Service, 2000c. Forest Biodiversity Guidelines. In: Department of the Marine and Natural Resources (Ed.), Dublin.

Forest Service, 2000d. Forest Protection Guidelines. In: Department of the Marine and Natural Resources (Ed.), Dublin.

Forest Service, 2000e. Forestry and Archaeology Guidelines. In: Department of the Marine and Natural Resources (Ed.), Dublin.

Forest Service, 2000f. Forestry and the Landscape Guidelines. In: Department of the Marine and Natural Resources (Ed.), Dublin.

Gee, J.H.R., Smith, B.D., 1997. Benthic invertebrates in the headwaters of the Wye and Severn: effects of forestry and clear-felling. Hydrol Earth Syst Sci 1, 549-556.

Haygarth, P.M., Warwick, M.S., House, W.A., 1997. Size distribution of colloidal molybdate reactive phosphorus in river waters and soil solution. Wat Res 31, 439-448.

Johnson, J., Farrell, E.P., Barrs, J.-R., Cruikshanks, R., Matson. R, Kelly Quinn. M, 2008. Forests and surface waters acidification literature review. In: Water Framework Directive, Water Framework Directive, Western River Basin District (Accessed 25th June 2012) http://www.wfdireland.ie/docs/22_ForestAndWater/Forests%20and%20Surface%20Water%20Acidification_literature%20Review.pdf

Joosten, H., Clarke, D., 2002. Wise Use Of Mires And Peatlands, A Framework for Decision-making. International Mire Conservation Group & International Peat Society.

Kirby, C., Newson, M., Gilman, K., 1991. Plynlimon research: the first two decades. Institute of Hydrology Wallingford,UK.

Kreutzweiser, D.P., Hazlett, P.W., Gunn, J.M., 2008. Logging impacts on the biogeochemistry of boreal forest soils and nutrient export to aquatic systems: A review. Environ Rev 16, 157-179.

Machava, J.O., McCabe, O., O`Dea, P., Farrell, E.P., 2005. The impact of forest operations on P levels in surface water. In, Agricultural research forum, Tullamore.

Macrae, M.L., Redding, T.E., Creed, I.F., Bell, W.R., Devito, K.J., 2005. Soil, surface water and ground water phosphorus relationships in a partially harvested Boreal Plain aspen catchment. Forest Ecol Manag 206, 315-329.

McBroom, M.W., Beasley, R.S., Chang, M., Ice, G.G., 2008. Water quality effects of clearcut harvesting and forest fertilization with best management practices. J. Environ. Qual. 37, 114-124.

McConnell, B., Gatley, S., 2006. Bedrock Geology of Ireland. Derived from the Geological Survey of Ireland 1:100,000 Bedrock Map Series and the Geological Survey of Northern Ireland 1:250,000 Geological Map of Northern Ireland.

Mellina, E., Moore, R.D., Hinch, S.G., Macdonald, J.S., Pearson, G., 2002. Stream temperature responses to clearcut logging in British Columbia: the moderating influences of groundwater and headwater lakes. Can J Fish Aquat Sci 59, 1886-1900.

National Forest Inventory, 2007. National Forest Inventory Republic of Ireland - Results. In: Service, F. (Ed.). Forest Service, Wexford.

Neal, C., Reynolds, B., Neal, M., Wickham, H., Hill, L., Williams, B., 1999. The impact of conifer harvesting on stream water quality: the Afon Hafren, mid-Wales. Hydrol. Earth Syst. Sci. 8, 503-520.

Neal, C., Reynolds, B., Neal, M., Wickham, H., Hill, L., Williams, B., 2004. The water quality of streams draining a plantation forest on gley soils: The Nant Tanllwyth, Plynlimon mid-Wales. Hydrol Earth Syst Sci 8, 485-502.

Neal, C., Smith, C.J., Hills, S., 1992. Forestry impact on upland water quality. Institute of Hydrology.

Nieminen, M., 1998. Changes in nitrogen cycling following the clearcutting of drained peatland forests in southern Finland. Boreal Environment Research 3, 9-21.

Nieminen, M., 2003. Effects of clear-cutting and site preparation on water quality from a drained Scots pine mire in southern Finland. Boreal Environ Res 8, 53-59.

Nisbet, T., Dutch, J., Moffat, A., 1997. Whole-tree harvesting - A guide to good practice. In: Forestry Commission (Ed.). Forestry Commission Practice Guide, Edinburgh, UK.

O'Driscoll, C., De Eyto, E., Rodgers, M., O'Connor, M., Asam, Z.U.Z., Xiao, L., 2012. Diatom assemblages and their associated environmental factors in upland peat forest rivers. Ecol Indic 18, 443-451.

O'Driscoll, C., Rodgers, M., O'Connor, M., Asam, Z.-u.-Z., de Eyto, E., Poole, R., Xiao, L., 2011. A Potential Solution to Mitigate Phosphorus Release Following Clearfelling in Peatland Forest Catchments. Wat Air Soil Poll, 1-11.

Owende, P.M.O., Lyons, J., Haarlaa, R., Peltola, A., Spinelli, R., Molano, J., Ward, S.M., 2002. Operations protocol for eco-efficient wood harvesting on sensitive sites. In. ECOWOOD Partnership. (Accessed on 14th April 2012) http://www.ucd.ie/foresteng/

Paavilainen, E., Päivänen, J., 1995. Peatland forestry: ecology and principles. Springer.

Piirainen, S., Finér, L., Mannerkoski, H., Starr, M., 2004. Effects of forest clear-cutting on the sulphur, phosphorus and base cations fluxes through podzolic soil horizons. Biogeochemistry 69, 405-424.

Quinn, J.M., Wright-Stow, A.E., 2008. Stream size influences stream temperature impacts and recovery rates after clearfell logging. Forest Ecol Manag 256, 2101-2109.

Renou-Wilson, F., Bolger, T., Bullock, C., Convery, F., Curry, J., Ward, S., Wilson, D., Müller, C., 2011. BOGLAND: Sustainable Management of Peatlands in Ireland. STRIVE 75. In. EPA., Johnstown Castle, Co. Wexford, Ireland. (Accessed on 14th June 2012) http://www.epa.ie/downloads/pubs/research/land/name,31495,en.html

Renou, F., Farrell, E.P., 2005. Reclaiming peatlands for forestry: the Irish Experience. In, Restoration of Boreal and Temperate Forests. CRC Press, Boca Raton.

Rodgers, M., O'Connor, M., Healy, M.G., O'Driscoll, C., Asam, Z.-u.-Z., Nieminen, M., Poole, R., Müller, M., Xiao, L., 2010. Phosphorus release from forest harvesting on an upland blanket peat catchment. Forest Ecol Manag 260, 2241-2248.

Rodgers, M., O'Connor, M., Robinson, M., Muller, M., Poole, R., Xiao, L., 2011. Suspended solid yield from forest harvesting on upland blanket peat. Hydrol Process 25, 207-216.

Rodgers, M., Xiao, L., Müller, M., O'Connor, M., E., D.E., Poole, R., Robinson, M., Healy, M.G., 2008. Quantification of Erosion and Phosphorus Release from a Peat Soil Forest Catchment. STRIVE Report Series No. 8. In. EPA., Johnstown Castle, Co. Wexford, Ireland. (Accessed on 14th July 2012) http://www.epa.ie/downloads/pubs/research/land/name,25212,en.html

Ryder, L., de Eyto, E., Gormally, M., Sheehy Skeffington, M., Dillane, M., Poole, R., 2011. Riparian zone creation in established coniferous forests in Irish upland peat catchments: Physical, chemical and biological implications. Biology and Environment 111.

S.I. No. 272 of 2009. European Communities Environmental Objectives (Surface Waters) Regulations In, Ireland.

S.I. No. 9 of 2010. European Communities Environmental Objectives (Groundwater) Regulations. In, Ireland.

Sharpley, A.N., Daniel, T., Sims, T., Lemunyon, T., Stevens, S., Parry, R., 2003. Agricultural Phosphorus and Eutrophication. Agricultural Research Service, United States Department of Agriculture.

Staaf, H., Olsson, B.A., 1991. Acidity in four coniferous forest soils after different harvesting regimes of logging slash. Scandinavian Journal of Forest Research 6, 19-29.

Stark, C.H., Richards, K.G., 2008. The continuing challenge of agricultural nitrogen loss to the environment in the context of global change and advancing research. Dynamic Soil, Dynamic Plant 2, 1-12.

Stevens, P.A., Norris, D.A., Williams, T.G., Hughes, S., Durrant, D.W., Anderson, M.A., Weatherley, N.S., Hornung, M., Woods, C., 1995. Nutrient losses after clearfelling in Beddgelert Forest: A comparison of the effects of conventional and whole-tree harvest on soil water chemistry. Forestry 68, 115-131.

Stott, T., Marks, S., 2000. Effects of plantation forest clearfelling on stream temperatures in the Plynlimon experimental catchments, mid-Wales. Hydrol Earth Syst Sci 4, 95-104.

Strahler, A.N., 1964. Quantitative geomorphology of drainage basin and channel networks. Handbook of Applied Hydrology.

Thiffault, E., Hannam, K.D., Paré, D., Titus, B.D., Hazlett, P.W., Maynard, D.G., Brais, S., 2011. Effects of forest biomass harvesting on soil productivity in boreal and temperate forests—A review. Environ Rev 19, 278-309.

Väänänen, R., Nieminen, M., Vuollekoski, M., Nousiainen, H., Sallantaus, T., Tuittila, E.S., Ilvesniemi, H., 2008. Retention of phosphorus in peatland buffer zones at six forested catchments in southern Finland. Silva Fennica 42, 211-231.

Von Arnold, K., Weslien, P., Nilsson, M., Svensson, B.H., Klemedtsson, L., 2005. Fluxes of CO2, CH4 and N2O from drained coniferous forests on organic soils. Forest Ecol Manag 210, 239-254.

Wallbrink, P., Croke, J., 2002. A combined rainfall simulator and tracer approach to assess the role of Best Management Practices in minimising sediment redistribution and loss in forests after harvesting. Forest Ecol Manag 170, 217-232.

Weatherley, N.S., Ormerod, S.J., 1990. Forests and the temperature of upland streams in Wales: a modelling exploration of the biological effects. Freshwater Biol 24, 109-122.

Chapter 7

7.1 Background

Approximately 60 % of Ireland's forest cover is on peat and the majority of this forestry is now at harvestable age. Decisions need to be made on the future management of forestry operations on the upland peat areas of the west of Ireland to limit their impact on the surrounding environment. The objectives of this study were to investigate the short and long-term impact of clearfelling on nutrient concentrations, water table (WT) fluctuations and greenhouse gas (GHG) emissions from soil respiration.

7.2 Conclusions

The main conclusions from this study were:

1. Clearfelling of forests influenced the WT position, the nutrient concentration of shallow groundwater and GHG emissions. In the Altaconey forest, there was a rise in WT elevation to within 0.15 m bgl 10 months after clearfelling took place. The WT depth subsequently fluctuated with dry periods. The rise in the WT affected the concentration of the N species in the shallow groundwater, but did not affect the P species.

2. The brash mats used for clearfelling in the Altaconey forest were successful at preventing soil water content changes beneath the mats and, because of their degradation over time, fertilised the peat. High P concentrations leached into the shallow groundwater beneath the brash mats and even though the peat has a low lateral saturated conductivity, there is the potential for leaching to nearby water courses over time. High water extractable phosphorus concentrations in the peat beneath the decaying brash mats also indicated a potential for high runoff concentrations of P to the adjacent surface waterbody. However, nutrient discharges to the stream in excess of maximum admissible concentrations were negligible due to the high inherent natural attenuation capacity of the peat, the adsorption of P to mineral layers adjacent to the watercourse, and dilution within the stream due to the relative size of the RBZ compared to the overall catchment.

3. The rise in the WT after clearfelling also affected GHG emissions from soil respiration due to the changes in the bacterial community. The shallow WT produced the greatest amounts of CH_4 (yearly cumulative flux of 65±40 kg CH_4-C ha^{-1} y^{-1}) and decreased the average daily N_2O emission (before clearfelling, the average flux was 1.7 g N_2O-N ha^{-1} d^{-1}; after clearfelling, 0.7 g N_2O-N ha^{-1} d^{-1}). Clearfelling also increased the average daily flux of CO_2 (11±2 kg CO_2-C ha^{-1} d^{-1} to 19±2 kg CO_2-C ha^{-1} d^{-1}) possibly due to the decomposition of brash material and roots or the autotrophic respiration from the native vegetation.

4. Elevated levels of nutrients and SS in surface waters are frequently associated with forestry clearfelling operations for up to 4 years. Following implementation, where possible, of best management practices (BMP), clearfelling of the upland peat forested Glennamong catchment resulted in limited export of nutrients and SS. Maximum concentrations of DRP (133 µg L^{-1}), TP (368 µg L^{-1}) and SS (481 mg L^{-1}) returned to the EPA critical threshold limits within 6 months of clearfelling. Site-specific parameters, such as the depth of peat or the slope of a site, and other potential confounding factors, such as the time of felling and weather conditions at the time of felling, may impact on nutrient and sediment release rates, and cognisance should be taken of these factors when drafting a harvest plan.

7.3 Recommendations

The main recommendations from this study are:

1. In areas that were forested to the water edge, the creation of riparian buffer zones (RBZs) prior to clearfelling larger coupes of forestry behind the RBZs is a possible mitigation measure for future forestry practice. Riparian buffer zones are capable of providing nutrients to planted saplings, fertilizing the peat with degrading brash material and preventing elevated concentrations of nutrients entering adjacent water courses - but only if a layer of mineral material exists in the peat close to the receiving watercourse.

2. The overall survival rate of native broadleaf planted saplings in a RBZ is relatively high. Five years following plantation, over half of oak, rowan and birch saplings had survived at the study site. The survival of planted willow, alder and holly saplings was not as successful, possibly due to a number of factors including the exposed nature of the upland peat sites, peat depth, maintenance, cultivation, fertilization and the low number of saplings initially planted. Survival and growth of native broadleaf species should be studied in other peatlands for plantation at lower stocking densities.

3. The rise in WT after clearfelling is due to reduced transpiration from the growing biomass and increased evapotranspiration from the soil. This indicates that the sites have restoration potential, following drain blocking, if reforestation does not take place.

4. In general, there is a greater flux of CO_2 from clearfelled areas and higher N_2O emission from mature standing forests, mainly due to the deeper WT. Virgin peatlands have high emissions of CH_4 due to the WT being almost at ground level. Globally, it has been found that CH_4 emissions from anaerobic decomposition of peat are small and inconsequential relative to the flux of CO_2 and therefore forest operations on peatlands may be detrimental to the overall GHG emission budget in the future. Considerable variation was observed in fluxes of GHG emissions from various peatland sites due to (1) variation in the WT depth and (2) the presence/absence of a brash layer. Ultimately, the position of the WT is the controlling factor governing GHG release and sequestration within the ecosystems.

5. Despite significant rises in nutrients and SS concentrations in drainage water following clearfelling and changes to some water parameters, the implementation of BMP, where possible, and the quick execution of site restoration plans, comprising silt traps and water management on extraction racks, negates excessive nutrients and SS export to adjoining water courses.