DRAFT REPORT

Forests and Surface Water Eutrophication and Sedimentation

FORWATER

FINAL DRAFT REPORT

Authors:

Stephen Hutton, Simon Harrison

and John O'Halloran

Table of Contents

List of Tables

LIST OF FIGURES

EXECUTIVE SUMMARY

Conifer plantation forests in Ireland are recognised as a potential source of diffuse pollution, in conjunction with many others, to water courses and represents a risk to the ecological integrity of running waters. Although current planting tends to occur on better quality land, the majority of Irish forests have historically been planted on agriculturally unproductive land. Much of this unproductive land is in the uplands, such that many Irish rivers either rise in or receive drainage from these upland catchments. While recognising other landuse/catchment pressures, this study aimed to assess the risks of forestry-derived eutrophication and sedimentation to running waters and to identify and quantify the factors that affect these risks.

Sampling occurred in a total of 460 sites, spanning a large geographical area and incorporating the major soil types and geology typical of forest plantations in Ireland. Samples were collected for hydrochemistry, macroinvertebrates, diatoms, bedload sediment and fish.

Site catchments that exhibited a high percentage of forest cover combined with a high percentage of felling, within the past 5 years, had very high levels of mean SRP and total ammonia. However these sites only represented 3.7% and 5.6% respectively of the total number of sites that failed 'good' water quality status. When the sites were divided further, by soil type, the impacts were most evident on the unproductive peat soils with 16.7% of sites failing 'good' status in relation to mean SRP, and 22.2% of sites failing 'good' status in relation to mean total ammonia.

Further extended sampling of two river systems, the Ballycorban and Corra Rivers, was undertaken to show how the levels of SRP and total ammonia change going downstream within a catchment. Three samples taken from the tributaries of the Corra River showed high levels of mean SRP, which all failed 'good' status. Sampling in the main river immediately after where the tributary entered the main stretch, the SRP values were reduced/diluted to such an extent that the water quality was of 'high' status. In the

Ballycorban River, of the 8 samples collected, five had mean SRP levels that failed ‘good’ status, with the two sites adjacent to felling giving the highest recorded values. Two of the 8 sites had elevated total ammonia levels that failed ‘good’ water quality status, with these sites again being adjacent to the felling. It was also shown that the levels of both SRP and ammonia decrease with an increase in stream width (i.e. dilution). The data showed the importance at sampling at a multitude of sample points to obtain a better overall picture of the hydrochemistry of a particular catchment.

Although there was no significant difference detected between sites in relation to bedload sediment, catchments that had some felling activity tended to have higher bedload sediment levels than catchments that had no felling activities. As seen with the hydrochemistry results, a potential risk exists in highly forested catchments combined with felling activities, but further work is required to assess the settlement patterns of bedload sediment at a multitude of downstream sites from the felling activity.

There was a trend for macroinvertebrate abundance to increase with decreasing percentage forest on sedimentary geology, with a significantly greater abundance detected between the 5-25% forestry sites compared to the > 50% forestry sites. No such trend was observed for the igneous-metamorphic geology sites. To assess the potential impact of eutrophication on macroinvertebrate metrics, sites that had a pH less than 6 were removed from the dataset, to attempt to detect impacts due to eutrophication rather than pH. There were six significant correlations between the macroinvertebrate metrics and sites with pH greater than 6, with the most significant relationship being with Plecopteran richness, where the richness decreased with increased levels of SRP, TP, Ammonia and DTOC. Plecopterans are known to be sensitive to eutrophication and this impact would appear to be the case in this study on peat sites with high forestry and presence of felling activity. Another eutrophication tool, the SSRS, was calculated for all sites. With an increase in percentage forestry within a catchment there was a greater risk of failing the SSRS, particularly on peat soils where 60-63% of sites were classified ‘At risk’.

The salmon data gave the most prominent results between control and forest sites. For each parameter related to salmon (biomass, total abundance, adult abundance and young-of-year abundance) there were significantly greater numbers found in the control sites compared to forest sites. An observation of high importance, and one that needs further research, is the absence of salmon at many of the forest sites, in particular where salmon are found in the paired control site. Again, peat sites on forestry showed the most striking results. Salmon fry were found in 39% of control streams on peat catchments, but only in 6% of forest sites. On non-peat catchments the percentage occurrence of salmon fry was 25%, with 50% found in control sites. There was also a significant difference in salmon fry length between the control sites and both forestry with felling and forestry without felling sites. The impact of combined high percentage forestry and percentage felling would appear to be detrimental to salmon growth and could potentially be attributed to stress from acid-sensitive conditions or to possible changes in the trophic status.

The results of this project emphasise the complexity of forest-site interactions. The indications from these results are that there is a potential risk of elevated nutrient and bedload sediment in catchments with high percentage forestry within the catchment combined with high percentage of felling on peat soils, which in turn is potentially impacting on the biology of the stream. Further studies should include more intensive studies in order to elucidate ecosystem processes controlling the retention and release of nutrients and sediment in forested catchments.

1. Introduction

The process of 'risk assessment' entails identification of environmental conditions, design of monitoring programmes, and the formulation of appropriate, cost effective protection and improvement measures (European Union, 2003). The concept of 'pressure-pathway-receptor' is used as the framework in risk assessment and requires an understanding of the relationships between a pressure (such as physical or chemical habitat degradation) and the mechanism (or pathway) by which the pressure exerts its impact on an ecosystem (the receptor) (European Union, 2003). Pressures that impact on freshwaters in Western Europe include 1) direct inputs of organic and inorganic pollutants from point sources 2) physical habitat alteration and 3) diffuse input of pollutants from surrounding catchments.

Conifer plantation forests in Ireland is recognised as a potential source of diffuse pollution, in conjunction with many others, to water courses and represents a risk to the ecological integrity of running waters (Duggan *et al.*, 2000; Gallagher *et al.*, 2000; Johnson *et al.*, 2000; Giller *et al.*, 1997). The European Water Framework Directive (WFD) is driving national programmes to monitor, assess and improve the quality of streams, rivers and lakes. Quantitative data on the risk posed by afforestation to receiving waters is needed both to develop effective mitigation measures for future plantations and to assess the relative impact of forestry on water quality, in relation to other land uses. As outlined in the WFD, the principle of the assessment procedure is to measure the deviation of the ecological situation of any observed site from (type-specific) reference conditions. Thereby, reference conditions represent a status with no or only minor human alterations of all quality elements included in the monitoring.

Forests currently cover approximately 10.15% of the Irish landscape, of which 77% is predominantly coniferous, 13% broadleaved, 4% mixed forest and 6% other wooded areas (ITGA, 2006). There has been a shift away from conifer planting in recent years with approximately 30% of new forest planting in 2005 being of broadleaved forest (ITGA, 2007). Although the land now being planted/afforested is generally of better quality and located at lower altitudes, the majority of Irish forests have historically been planted on agriculturally unproductive land. Forestry on unproductive peat soils reached a peak in the early to mid-70's, reaching almost 70%. This level remained high for a period of approximately 10 years (in the region of 64% forest on peat soils between 1981-1985). A move away from planting on peat to gley soils was evident in the mid-80's until the present day with peats accounting for approximately 40% of forest soils today and steadily decreasing.

Much of this “unproductive land” is in the uplands, such that many Irish rivers either rise in or receive drainage from these upland catchments. While recognising other landuse/catchment pressures, this study focuses on the concern of forests and forest operations and the pressures of eutrophication and sedimentation.

During forest operations, including harvesting and clearfelling, soil surface disturbance can result in increases in soil erosion, and suspended sediment (SS) and nutrients in run-off waters. Enriched receiving waters can occur as a result of these nutrient and sediment increases, posing a risk to the chemical and biological integrity of these streams if careful management practices are not implemented.

Potential eutrophication, the process whereby a body of water becomes over-enriched with nutrients (in particular phosphorus (P) and nitrogen (N)), of forest streams in Ireland is mainly concerned with the levels of P in the catchment as N is applied in insubstantial amounts compared to other European countries. This over-enrichment can result in accelerated growth of algae and other plant life which in turn can deplete oxygen levels in the water, leading to the loss of aquatic fauna.

Recent single catchment studies in Ireland have produced contrasting results in relation to the potential impacts of P and sediment from forestry and related forest operations (Gallagher *et al.*, 2000; Cummins and Farrell, 2003; Machava *et al.*, 2007; Rodgers *et al.*, 2008) even when the studies have purposely focussed on soil types that would expect to have a greater potential risk of eutrophication to water bodies. Nevertheless, there are still concerns that forestry can contribute to eutrophication, particularly as many plantations have been established on peat and peaty soils, which have a low capacity to retain free phosphate. Peat soils are particularly vulnerable to phosphate loss, as their concentrations of clay, iron and aluminium oxides, which reduce the solubility of phosphate, are very low. Low hydraulic conductivity, leading to increased overland flow also contributes to the potential for P loss.

Although all these effects have been documented at local scales, there have been few attempts to assess the wider scale risk of commercial conifer forestry to the quality of receiving waters. The impact of a particular plantation will depend on many factors including parent rock type and soil type, percentage catchment afforestation and felling operations, slope, altitude, geographical position, and small scale 'random' factors such as the nature of a particular planting or felling operation, rainfall patterns, and water drainage from a site.

1.1. Objectives

The aims of this study were to (i) assess the risks of forestry-derived eutrophication and sedimentation to running waters and to (ii) identify and quantify the factors that affect these risks.

2. SITE SELECTION AND METHODOLOGY

2.1 SITE SELECTION

Sites were selected with a view to examining potential eutrophication and sedimentation impacts due to coniferous plantation forests and forest operations in Irish streams. Consultation with Steering Committee members was included in designating site selection criteria (Cóillte, Irish Forest Service, EPA, Marine Institute, Central Fisheries Board and National Parks and Wildlife). The main principal of site selection was to locate sites encompassing various forestry activities (i.e. mature phase, felling and replanting) and neighbouring control sites. Large areas of forestry were initially targeted to increase the probability of obtaining a high sampling number of each forest activity. The sites selected were 1st and 2nd order streams with a stream width of approximately 2-4m. Selected forest streams were chosen on the basis that the majority (> 60%) of the watershed above the sampling point was draining from forest. This would better indicate what impact forestry was having on watercourses than if the wider landscape, and hence other land-uses, was used. 'Control' sites were selected in neighbouring catchments with little (<5%) or no forestry.

A total of 460 sites were selected (383 of which were delineated using GIS – 144 control and 239 forested sites) across igneous-metamorphic and sedimentary geology types and a range of soils types (well drained mineral, poorly drained gleys, podzolic-lithosolic and peats). As a result of GIS mapping, the forest sites showed a variable percentage of forestry in their catchments. The 239 delineated forested sites were designated into three forest bands (54 sites with 5-25% forest, 64 sites with 25-50% forest and 121 sites with >50% forest). A further 60 sites were also sampled for water chemistry analysis as a subset of the 460 total sites. Areas studied by UCC were located in Co. Cork, Kerry, Waterford, Limerick, Tipperary, Clare, Laois, Galway and Mayo. Additionally, UCD sampling occurred in Co. Cork, Kerry, Tipperary, Donegal, Galway, Mayo and Wicklow. Sites were selected on a regional basis, representing Irish geological types in counties.

Similar watersheds in each region were selected in terms of slope, elevation and aspect, in order to limit physical variation between sites. Sampled sites were designated as 50m stretches. Physical data was collected at all sites including stream width, depth, substrate and habitat cover, flow condition and GPS location. From information supplied by Cóillte, the percentage of the catchment afforested and the percentage of the catchment felled within the last 5 years were also derived for each forest site. The site distributions are shown in Figure 1.

The sites used for the majority of the analysis in this report, and their respective soil type and forest cover, are shown in Table 1. As there were few poorly drained gley soils in the final dataset (n = <15), these samples were removed from further analysis.

Table 1. Sampling sites used for analysis. Sites are divided into soil type and the percentage forestry within the catchment.

	< 5%	**5-25%**	**25-50%**	**> 50%**	**Total**
Peat	33	13	12	43	101
Podzolic/Lithosolic	29	7	16	36	88
Well Drained Mineral	2	7	8	8	25
Total	64	27	36	87	214

2.2 Hydrochemistry

Water samples were collected from all sites in one-litre and 250ml polypropylene bottles. Conductivity readings (µS cm^{-1}) were collected on site using field meters. All samples were sent to the Aquatic Services Unit at the Environmental Research Institute (ERI, UCC) for analysis within 24-hours of sampling. A full suite of hydrochemical parameters were routinely carried out for all samples and these parameters and their corresponding methodologies are listed in Table 2. The one-litre bottle was used for the majority of the analyses, with the 250ml bottle being utilised for pH and inorganic aluminium fractionation analysis. Water samples were taken on three sampling occasions (April/May 2007; Nov/Dec 2007; Mar/April 2008).

An additional study on water chemistry was undertaken in the southern Slieve Aughty mountains, Co. Clare. The objective of this study was to investigate any potential dilution effects of SRP (mg P/l) and Ammonia (mg N/l) when sampling down a particular catchment. A longitudinal sampling design was carried out on the Corra and Ballycorban Rivers where samples were taken from tributaries and the main channel. These rivers were selected as they had the highest values of SRP recorded in earlier water sampling runs. The water samples were collected in the same way as above.

Fig 1. Distribution of sampling sites.

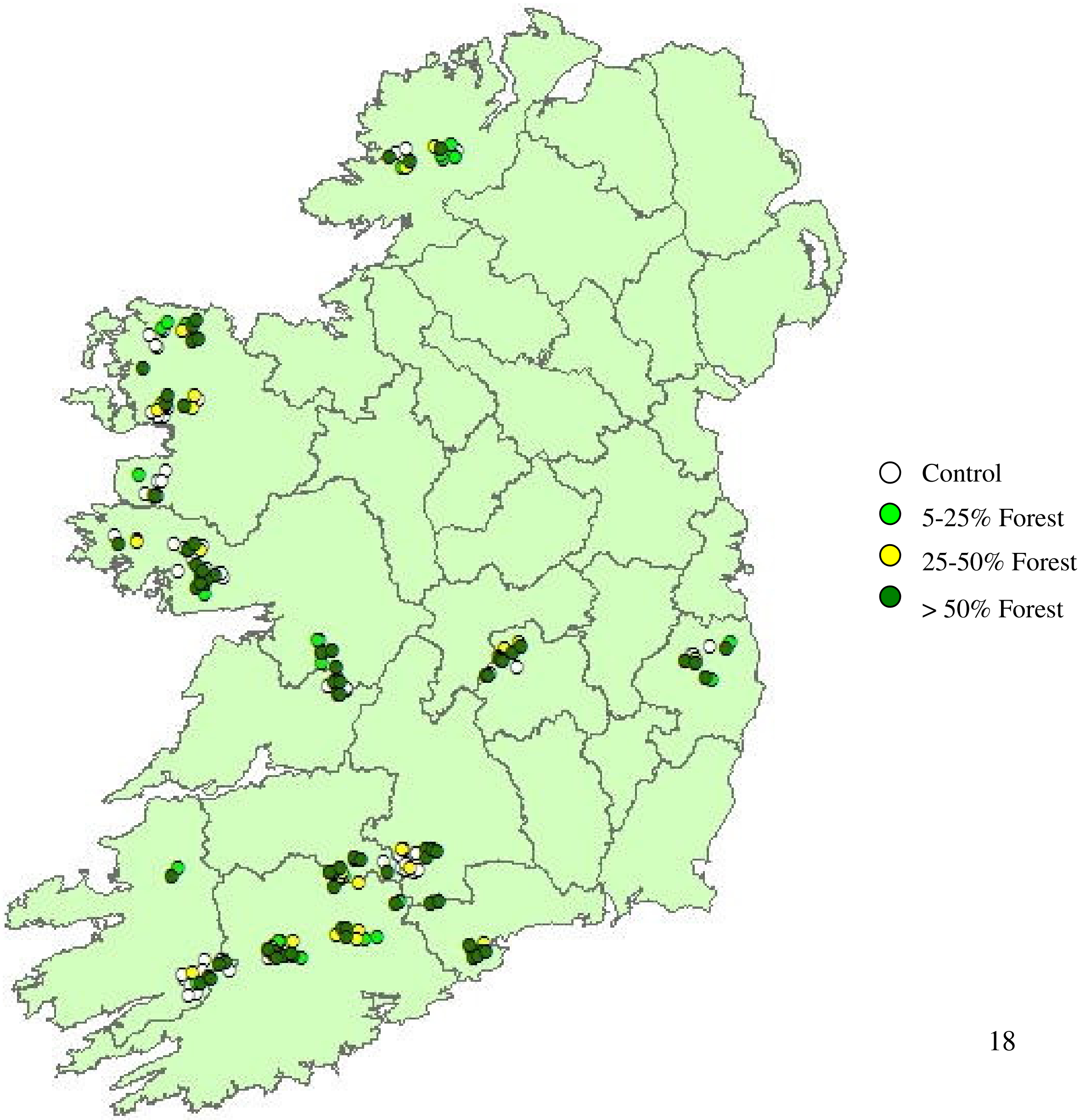


Table 2. Water chemistry parameters measured and methods used.

Parameter	Method	Limits of detection
pH	WTW pH meter 320 following 2-point calibration on fresh pH buffer solutions	N/A
Conductivity	WTW LF330 Conductivity meter calibrated with standardised KCl after Standard Methods (APHA 1989) Method 2510-B	1 µS cm^{-1}
Alkalinity	Standardised HCl (0.02N) titration with BDH 4.5 indicator for samples above Conductivity of 150µS/cm and Gran titration for samples with less than 150µS/cm conductivity	0.1 mg $CaCO_3$ l^{-1}
Total Hardness	ETDA Titration	5 mg $CaCO_3$ l^{-1}
Colour	Colorimetric method	1 Hazen
Dissolved Total Organic Carbon	SHIMADZU TOV-V_{CPH}	1 mg DTOC l^{-1}
Soluble Reactive Phosphorus	Lachat Quik-Chem 8000 FIA Method following 0.45µm filtration	0.001 mg SRP l^{-1}
Total Phosphorus	Murphy & Reily Method (Molybdate – Ascorbic Acid) following digestion of the unfiltered sample with persulphate and sulphuric acid (autoclave)	0.005 mg TP l^{-1}
Ammonia	Lachat Quik-Chem 8000 FIA method following 0.45µm filtration	0.002 mg N l^{-1}
Total Organic Nitrogen	Lachat Quik-Chem 8000 FIA method following 0.45µm filtration	0.05 mg TON l^{-1}
Nitrate	Subtraction nitrite from TON	0.01 mg Nitrate l^{-1}

Nitrite	Manual colourimetric method	0.001mg Nitrite l^{-1}
Total Monomeric Aluminium	Graphite furnace AAS	0.5 µg Al l^{-1}
Inorganic Aluminium	Graphite furnace AAS after Amberlite™ resin fractionation	0.5 µg Al l^{-1}
Calcium	Automated IC method using Lachat™ Quik-Chem 8000	0.5 mg Ca l^{-1}
Magnesium	Automated IC method using Lachat™ Quik-Chem 8000	0.2 mg Mg l^{-1}
Potassium	Automated IC method using Lachat™ Quik-Chem 8000	0.1 mg K l^{-1}
Sodium	Automated IC method using Lachat™ Quik-Chem 8000	0.2 mg Na l^{-1}
Chloride	Automated IC method using Lachat™ Quik-Chem 8000	1 mg Cl l^{-1}
Sulphate	Automated IC method using Lachat™ Quik-Chem 8000	0.5 mg SO_4 l^{-1}
Suspended Solids	Gravimetric using Whatman GF/C filters and drying at 103-105 °C as per Standard Methods (APHA 1989) Method 2540-B	0.1 mg SS l^{-1}
Silicate	Manual colourimetric method	0.01 mg Si l^{-1}

2.3 MACROINVERTEBRATES

Macroinvertebrate sampling was carried out in order to assess possible differences in communities due to coniferous plantation forests and forest operations. In particular, this section of the field sampling aimed to identify potential areas of eutrophication and sedimentation impact in forested catchments in Ireland.

Methodology

Benthic macroinvertebrate samples were collected using standard 1mm pond nets during April/May 2007. A multi-habitat sampling approach was employed using kick samples of 1-minute duration. Four samples were collected at each site and preserved using 70%

ethanol (IMS). During macroinvertebrate sampling, data was collected on both habitat and substrate coverage in streams.

On return to the laboratory, samples were sorted in illuminated white plastic trays into taxonomic groups. All specimens were identified to the lowest possible taxonomic level (Table 3) using the appropriate keys. Identified samples were preserved and stored in 70% alcohol (IMS). For high abundances of taxa sub-sampling was employed during sorting. Quality control procedures were employed for macroinvertebrate sorting and identification. Previously sorted samples were re-checked for missed specimens. Identified specimens were also re-examined with each identified species being confirmed by an independent taxonomist. Quality control was also used in checks of data inputting in physico-chemical and biological databases.

Table 3. Level of identification for each taxonomic group

Taxonomic Group	Level Of Identification
Ephemeroptera	Species
Plecoptera	Species
Trichoptera	Species
Coleoptera	Species
Crustacea	Species
Hirudinea	Species
Odonata	Species
Diptera	Family/Genus
Mollusca	Family/Genus
Hemiptera	Family/Genus
Other taxa	Family/Order

2.4 PHYTOBENTHOS (DIATOMS)

Diatom sampling was carried out in order to assess potential shifts in communitiy structure in relation to coniferous plantation forests and forest operations.

Methodology

Diatoms were sampled in accordance with the DARES Project collection protocol (Kelly *et al*., 2005) during April/May 2007. Five cobbles were removed from the main flow of the stream in riffle/glide habitats and the upper surface scrubbed using a toothbrush until all algal material and biofilm was collected in a small white plastic sorting tray along with a small volume (50ml) of stream water. The composite sample, from all five cobbles, was added to a 250ml labelled wide-rimmed bottle and preserved using 70% ethanol (IMS).

Diatom samples collected in the field were sent to Dr Martyn Kelly (Bowburn Consultancy Ltd., Durham) for microscopy identification and analysis.

2.5 Bedload Sediment

Sediment sampling was undertaken to assess the levels of fine bedload sediment in relation to coniferous forests and forest operations and its potential impact on stream biology.

Methodology

Fine bedload sediment was collected using a modified Surber sampler (delimiting an area of 0.09m^2) in April/May 2007. The sampler had a front mesh of 1000µm diameter to stop larger sediment particles from being collected. The surber net mesh was 50µm, so sediment ranging from 50-1000µm was collected in the sampling bottle attached to the surber.

The sampler was placed on the streambed in an area that was typical of the reach being sampled, usually within a riffle area. The substrate was then disturbed with a hand trowel to dislodge sediment for a duration of 5 minutes or until there was no more visible sediment being dislodged. The sample was then transferred into a polypropylene container and taken to the laboratory for analysis.

Samples were oven dried over a period of 48 hours at 65°C and then sieved at two fractions (250-1000μm and 50-250μm) and then weighed.

2.6 Salmonid Electrofishing

This survey assessed the potential impact on salmonids due to coniferous forests and forest operations.

Methodology

Streams were selected on a paired basis (1 forested catchment, 1 non-forested catchment) with similar physical characteristics including catchment area, elevation, slope etc. and were generally 2nd order headwater streams. In total, 34 paired sites (68 sites) were fished and are shown in Fig. 2.

Habitat parameters of forested and control streams, such as the number of riffles, glides and pools, conductivity, stream width and depth were recorded on site, along with hydrochemical parameters.

Fish populations were sampled by single-pass backpack electrofishing (Safari Research 550D backpack model). Several investigations have evaluated accuracy and usefulness of single-pass electrofishing to estimate abundance or relative abundance of salmonids in streams. These studies have indicated that there is a significant relationship between number of fish caught in first pass and the total population size estimated from three or more passes (Hayes & Baird, 1994; Jones and Stockwell, 1995; Kruse *et al.*, 1998; Mitro and Zale, 2000; Arnason *et al.*, 2005; Bertrand *et al.*, 2006) and is therefore a sensitive method for detecting differences in relative abundance. The FAME protocol recommends at least 10-20 times the wetted width be fished (Economou *et al.,* 2002). As the majority of the selected sites were approximately 2m wide, 100m was considered sufficient to satisfy this condition. Forested stream sites used for analysis had at least 20% mature closed-canopy cover. Data collected at each site included species identity, length, weight and age of all individual fish. From the data collated salmonid population density and biomass was calculated.

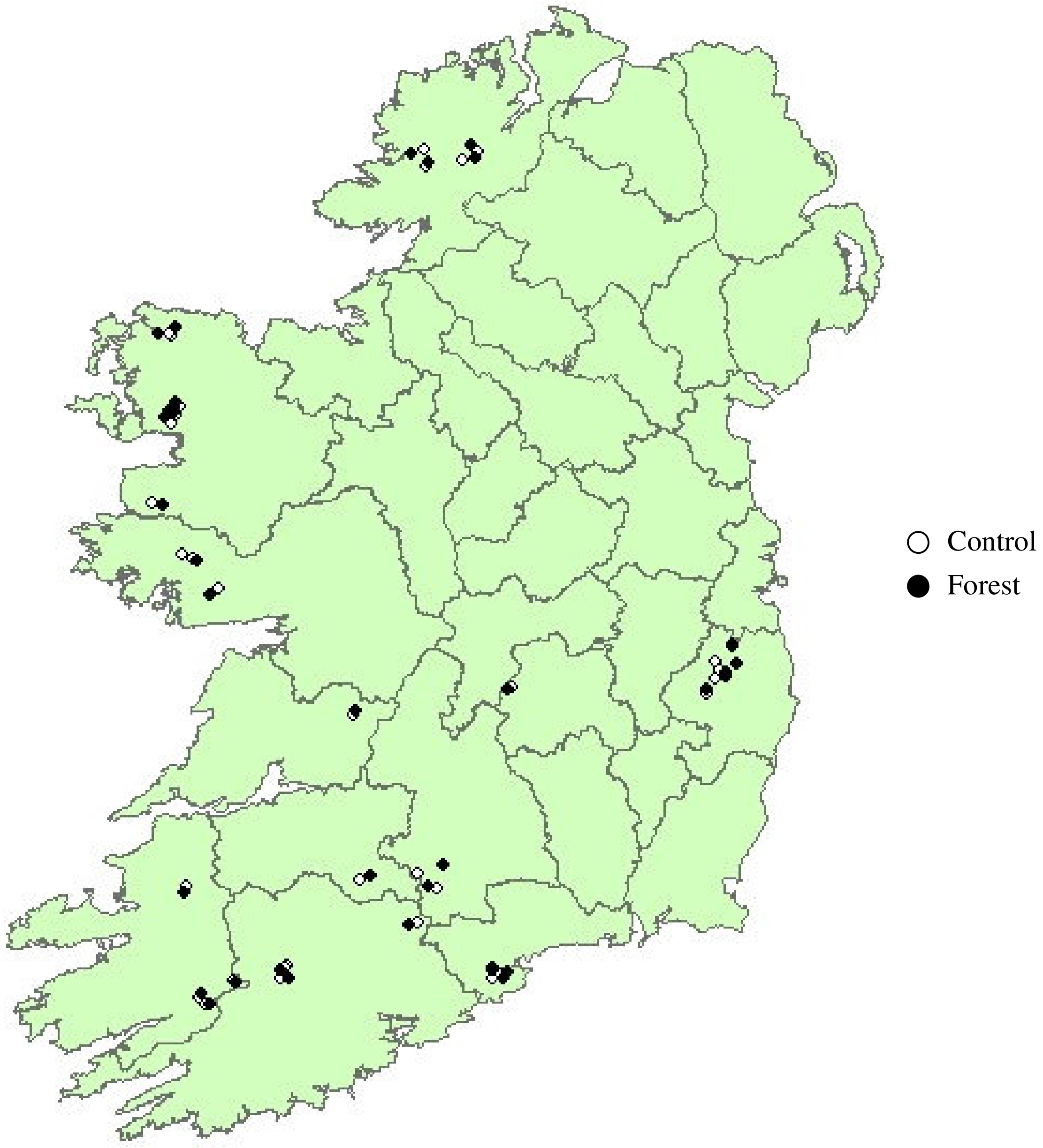
2.6 Geographical Imaging

Readings were recorded from a GPS handset at all sites. Arcview™ 3.3 was utilised in order to plot site distributions and delineate catchment basins for all sites. The Geoprocessor™ extension program allowed the calculation of various catchment characteristics including; geology, soil (and sub-soil) coverage, percentage catchment forestry, catchment land-use, catchment area. Catchment delineation was undertaken by Compass Informatics™, while land-use, (sub)-soil and geology coverage was carried out by UCD.

2.7 Statistical Analyses

Extensive databases for biological and physico-chemical parameters were generated in Excel™. Analyses of various parameters were carried out using univariate and multivariate using SPSS™ v. 12.0.1, Primer-E v. 6.0, and PC-Ord. Many analyses utilised included analysis of variance (ANOVA), *t*-tests, correlations and regressions, and PCA. The multitude of macroinvertebrate metrics were generated using AQEM (ASTERICS 3.10) software. Impact at sites in terms of various biological metrics and hydrochemical parameters was detected using a significant shift outside of two standard deviations (or 95% confidence interval) as expressed by Resh *et al.* (1988). The same metric was used to develop a clearfelling impact metric by Johnson *et al.* (2005).

Fig 2. Distribution of paired fishing sites.



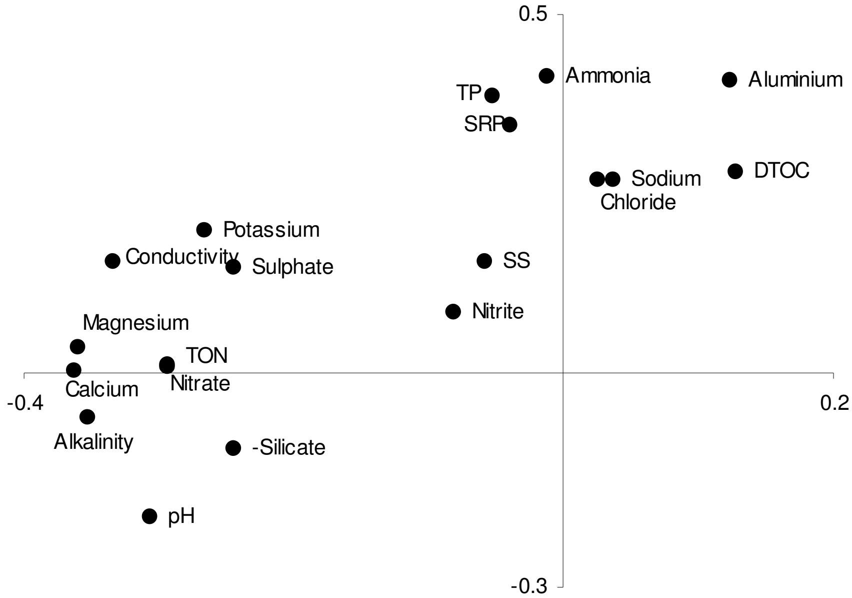
3. RESULTS

3.1 WATER CHEMISTRY

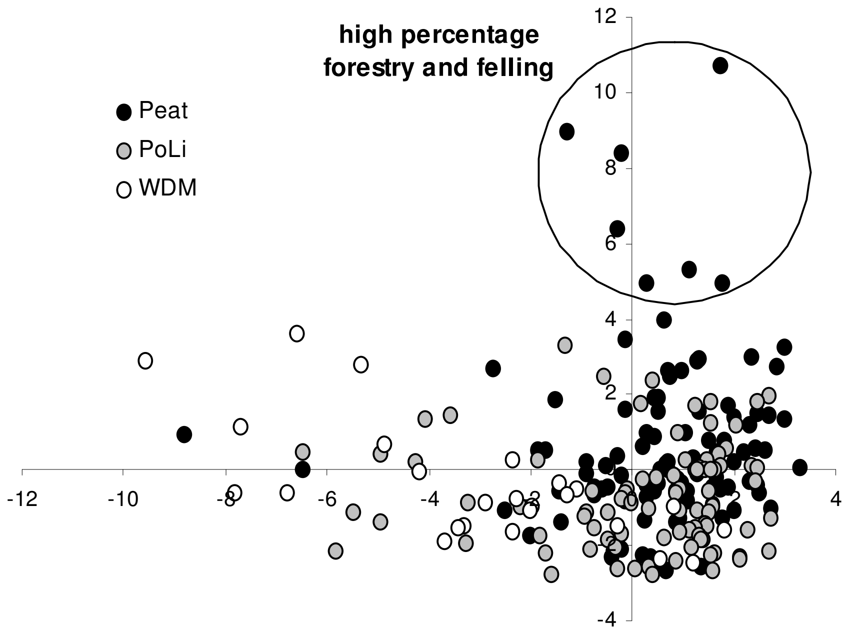
Fig.3 shows the PCA results of the water chemistry and the individual sites in relation to soil type.

Fig. 3 Principal Components Analysis (PCA) of (a) the water chemistry parameters measured and (b) the individual sites in relation to soil type (Peat = Peats, PoLi = Podzolic Lithosolic, WDM = Well Drained Mineral) for all sites.

(a)



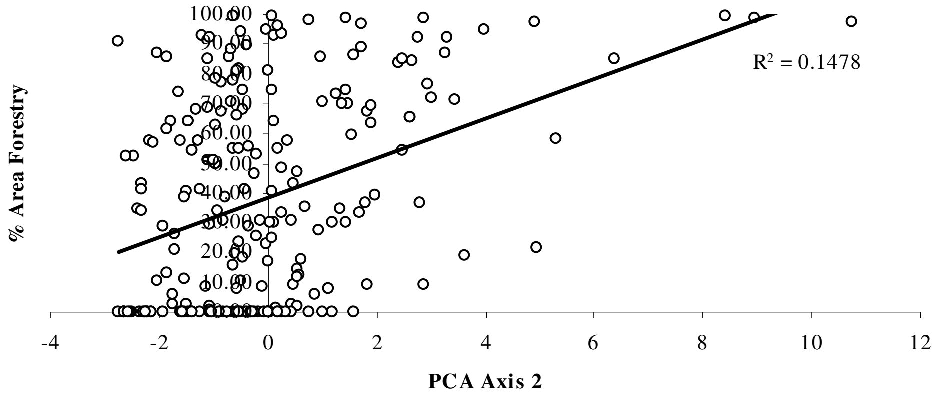
(b)



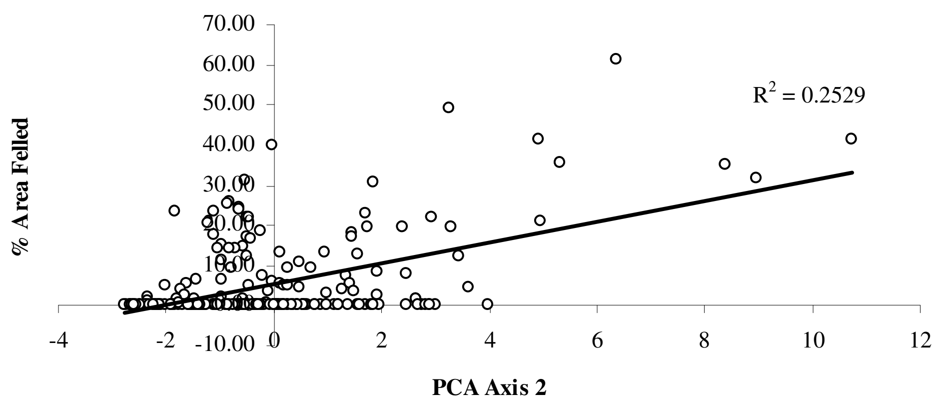
Sites with high percentage forestry and high percentage felling have been circled. These sites correspond to high mean levels of Soluble Reactive Phosphorus (SRP), Total Phosphorus (TP), Ammonia and Dissolved Organic Carbon (DTOC). The high percentage forest and percentage felled sites were pulled from the data by the PCA axis 2 score. Indeed, correlations of percentage area forestry (r = 0.384; $p < 0.001$) and percentage area felled (r = 0.503; $p < 0.001$) against PCA axis 2 score show highly significant relationships (Fig. 4).

Fig.4 Correlations of (a) % area forest and (b) % area felled (past 5 years) against PCA axis 2 score for all sites.

(a)



(b)

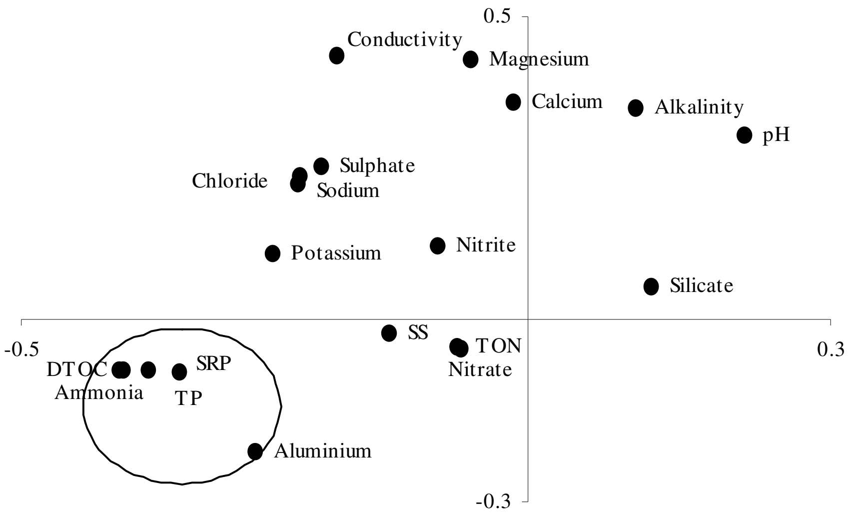


The majority of sites that have a high percentage of forestry and/or high percentage of felling tend to be located on peat soils. As peat soils are more likely to be vulnerable to environmentally significant phosphorus losses, peat soils were investigated further. The

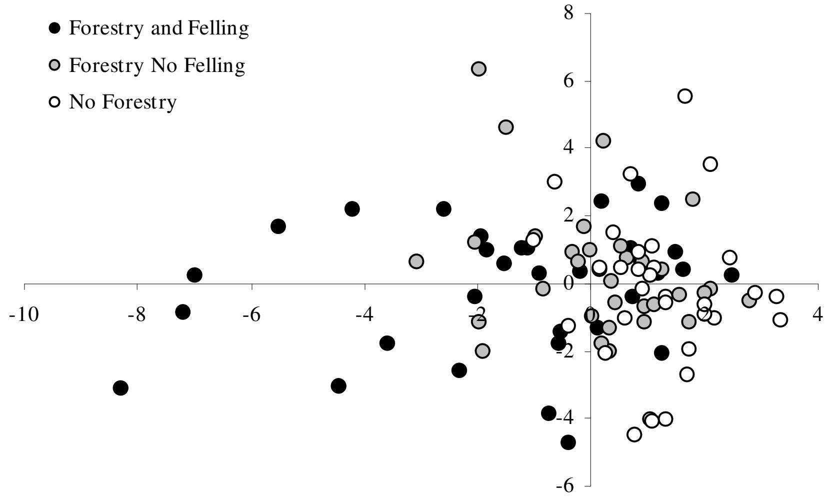
PCA results of water chemistry and the individual sites on peat soils in relation to percentage forestry and percentage felling are shown in Fig. 5. The sites which have high percentage forestry and/or high percentage felling are pulled on PCA axis 1 and correspond to high mean levels of Soluble Reactive Phosphorus (SRP), Total Phosphorus (TP), Ammonia, Dissolved Organic Carbon (DTOC) and Aluminium. Therefore there is a forestry/felling effect evident on peat soils where the greater the percentage of forestry within a catchment combined with a high percentage of the catchment felled will increase the risk of elevated phosphorus, ammonia and DTOC levels entering the watercourse.

Fig.5 Principal Components Analysis (PCA) of (a) the water chemistry parameters measured and (b) the individual sites on peat soils in relation to percentage forestry and percentage felling.

(a)



(b)

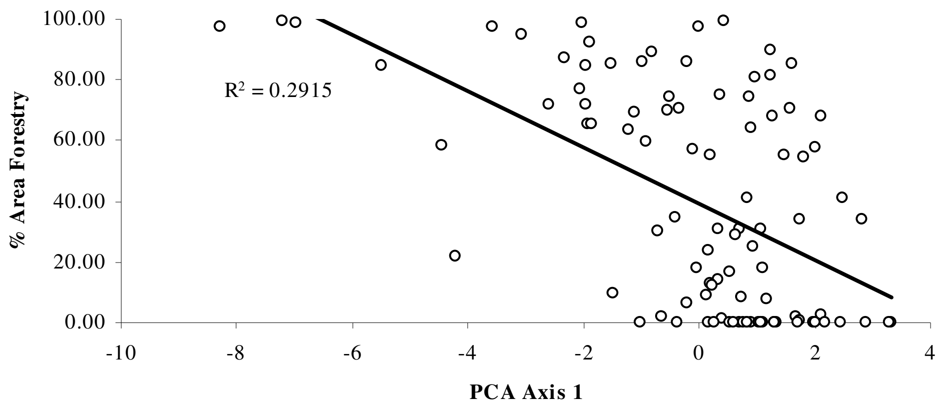


Indeed, there was a significant correlation between > 50% forestry combined with felling and DTOC on peat soils (r = 0.607; $P < 0.01$) but no significant correlation was observed between > 50% forest with no felling and DTOC on peat soils (r = 0.377; $P > 0.05$). There was no significant relationship for sites with < 50% forestry combined with either felling (r = 0.041; $P > 0.05$) or no felling (r = 0.121; $P > 0.05$) and DTOC on peat soils.

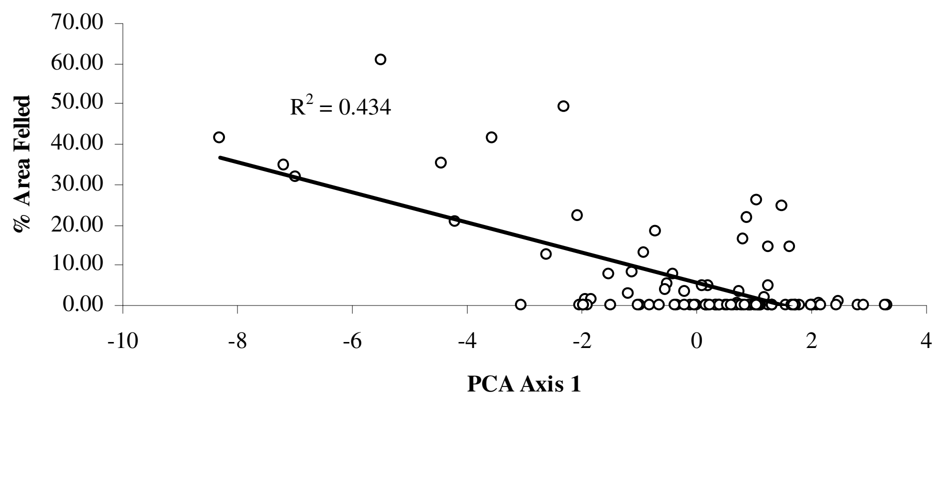
For peat sites the high percentage forest and percentage felled sites were pulled from the data by the PCA axis 1 score. Correlations of percentage area forestry (r = 0.540; $p < 0.001$) and percentage area felled (r = 0.659; $p < 0.001$) against PCA axis 1 show highly significant relationships (Fig. 6).

Fig.6 Correlations of (a) % area forest and (b) % area felled (past 5 years) against PCA axis 1 score for peat soils.

(a)



(b)



3.1.1 Soluble Reactive Phosphorus (SRP)

The mean Soluble Reactive Phosphorus (SRP) values for each soil type in relation to percentage of catchment forested are shown in Fig.7. On the peat soils, a significant difference in mean SRP was shown in relation to the percentage of forestry within the watershed. Sites that had greater than 50% forestry within the watershed had significantly greater mean SRP values than those found in watersheds with less than 50% forestry (χ^2 (3) = 26.545; $P < 0.001$). This was also reflected in the podzol-lithosol soils, although there was no significant difference between sites that had greater than 50% forestry and sites with 25-50% forestry within the watershed (z = 1.792; $P > 0.05$). The overall mean SRP values for the podzol-lithosol soils were generally low. In well drained mineral soils mean SRP increased with decreasing forest cover with the exception of sites with less than 5% forestry. However, it should be noted that the number of sites in this category (n = 2) was extremely low.

Fig.7 Mean SRP (mg P/l) values for each soil type in relation to % of catchment forested. Treatments with different letters above standard error bars are significantly different ($P < 0.05$).

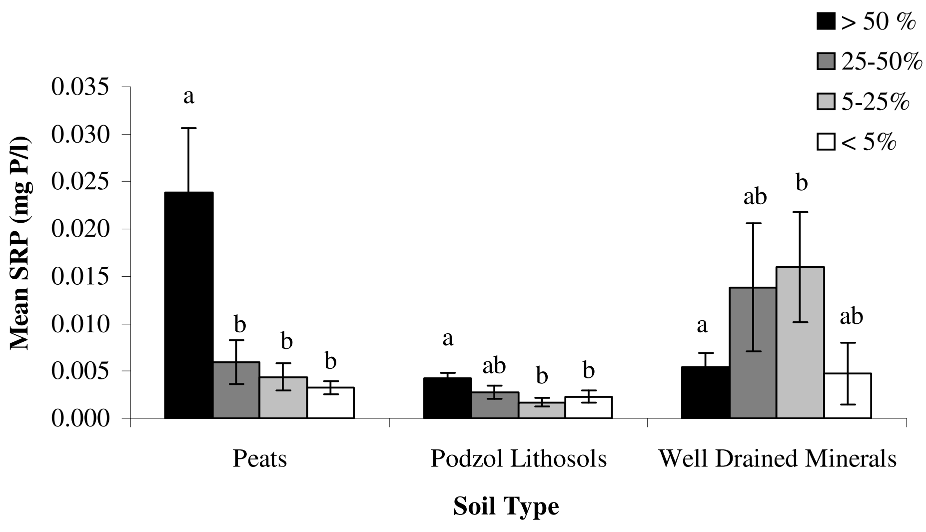


Fig.8 Mean SRP (mg P/l) values for peat and podzol lithosol soils in relation to % of catchment felled. Data presented are for sites with > 50% forestry. Treatments with different letters above standard error bars are significantly different ($P < 0.05$).

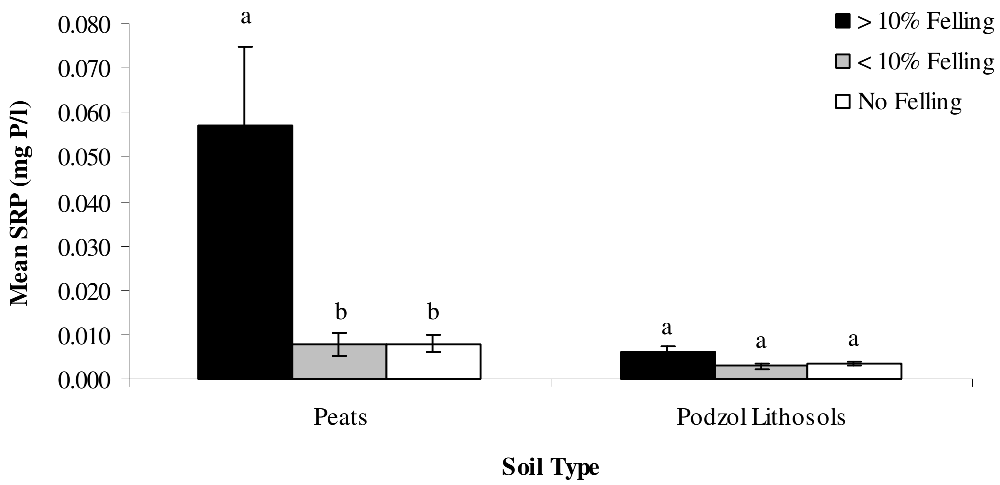


Fig. 8 shows the percentage felling for catchments that have greater than 50% forest cover. The mean SRP was significantly greater on peat soils with > 10% felling compared to catchments with < 10% or with no felling (χ^2 (2) = 13.875; P = 0.001). There was no significant difference between the three felling levels on the podzol lithosol soils (χ^2 (2) = 3.883; $P > 0.05$). There was not enough data available for the well drained mineral soils to perform a viable statistical test.

The Draft European Communities Environmental Objectives (Surface Waters) Regulations 2008 has been introduced to transpose into Irish law the measures needed to give effect to the environmental objectives of the Water Framework Directive relating to surface waters. These define ‘high’ and ‘good’ water quality status according to a suite of chemical and biological parameters. The ‘high’ and ‘good’ water quality status for mean SRP in surface waters (MRP given in the Regulations, but it is known that these two fractions have similar values and are highly correlated) are as follows:

Mean MRP (mg P/l)

High status = ≤ 0.025 mg P/l

Good status = $\geq 0.025 \leq$ 0.035 mg P/l

Fig.9 Percentage forestry in the catchment against mean SRP (mg P/l) for all sites. Dashed line represents ‘good’ status and solid line represents ‘high’ status.

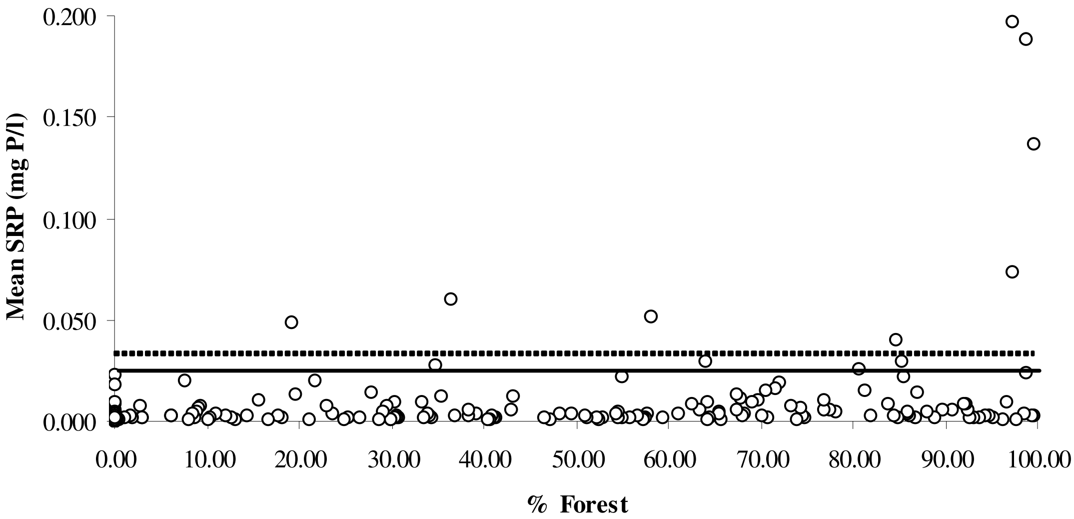


Fig. 9 shows the mean SRP (mg P/l) recorded for all sites in relation to the percentage forestry within the sites catchment. The two levels of water quality status (‘high’ and ‘good’) are represented by a solid and dashed line respectively. The majority of sites (94.4%) satisfied the ‘high’ status quality with only 8 sites (3.7%), from a total of 214 sites, failing to reach ‘good’ status. A summary of the SRP status of the streams in the dataset is given in Table 4.

Table 4. Phosphorus water quality status of streams

Mean SRP concentration	Status	Percentage of sites
> 0.035 mg P/l	Failed Status	3.7
0.025 – 0.035 mg P/l	Good Status	1.9
< 0.025 mg P/l	High Status	94.4

Fig.10 Percentage forestry in the catchment against mean SRP (mg P/l) for all sites. Sites are separated into forest sites with either felling or no felling activities, and control sites. Dashed line represents 'good' status and solid line represents 'high' status.

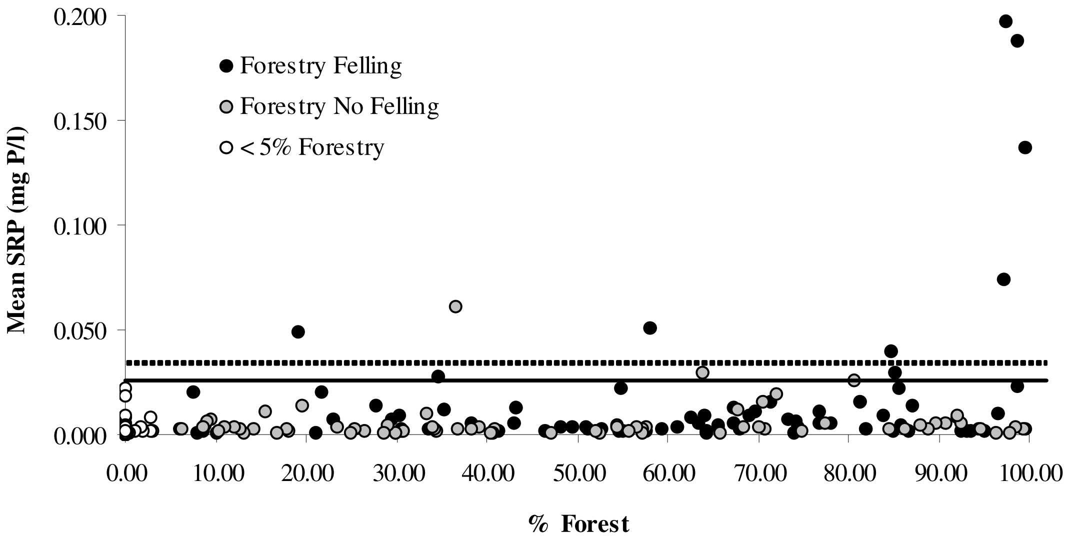


Fig. 10 shows the mean SRP (mg P/l) recorded for all sites in relation to the percentage forestry and presence/absence of felling activity within the catchment. The two levels of water quality status ('high' and 'good') are represented by a solid and dashed line respectively. All sites with less than 5% forestry (i.e. control sites) had 'high' status water quality in relation to SRP levels. The water quality status of the streams changed slightly on sites that had more than 5% forestry and no felling activity, with 1.4% of streams failing 'good' status (Table 5). There was more of a shift in water quality status with sites that had more than 5% forestry combined with felling activities, although the general trend was still to have a high percentage (88.9%) of 'high' status sites (Table 5). Within this category, 8.6% of streams failed 'good' status. Although this is a small percentage of the overall number of sites the risk remains that sites with high levels of forestry combined with a high percentage of the catchment felled have the potential to release high levels of SRP (mg P/l) into watercourses and as a result fail the Regulations.

Table 5. SRP (mg P/l) water quality status of the study streams in relation to % forestry and presence/absence of any felling activity. The percentage of the sites for each 'Status' category are given.

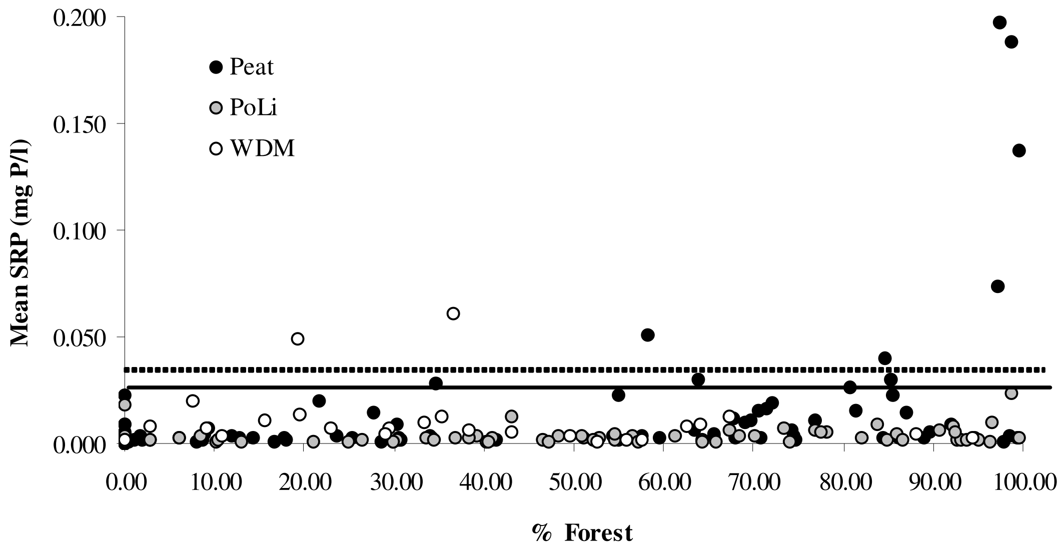
Mean SRP concentration	Status	< 5% Forestry (No Felling)	> 5% Forestry (No Felling)	> 5% Forestry (Felling)
> 0.035 mg P/l	Failed Status	0.0 %	1.4 %	8.6 %
0.025 – 0.035 mg P/l	Good Status	0.0 %	2.9 %	2.5 %
< 0.025 mg P/l	High Status	100.0 %	95.7 %	88.9 %

Fig. 11 shows the mean SRP (mg P/l) recorded for all sites in relation to the percentage forestry and soil type within the catchment. The two levels of water quality status ('high' and 'good') are represented by a solid and dashed line respectively. The sites that had the highest levels of mean SRP (mg P/l) were found on peat soils, with 5.9% of peat sites failing 'good' status (Table 6). Although Table 6 shows a greater percentage of well drained mineral sites failing 'good' status than peat sites, these were only represented by two sites. As the number of sites for both soil types within this category are small (Well drained mineral, n = 2; Peats, n = 6) no significance could be detected between the respective means (Well drained mineral mean for failed status sites = 0.055 mg P/l; Peats mean for failed status sites = 0.115 mg P/l). The well drained mineral sites that failed 'good' status had a percentage forest cover below 40%, so there could potentially be influences of other landuses on the SRP values recorded. All podzolic lithosolic sites sampled were of 'high' status with a mean SRP value of 0.003 mg P/l.

Table 6. Mean SRP (mg P/l) water quality status of the study streams in relation to soil type. The percentage of the sites for each 'Status' category are given for each soil type.

Mean SRP concentration	Status	Peats	Podzolic Lithosolic	Well Drained Mineral
> 0.035 mg P/l	Failed Status	5.9 %	0.0 %	8.0 %
0.025 – 0.035 mg P/l	Good Status	4.0 %	0.0 %	0.0 %
< 0.025 mg P/l	High Status	90.1 %	100.0 %	92.0 %

Fig.11 Percentage forestry in the catchment against mean SRP (mg P/l) for all sites. Sites are separated further into soil type (Peat = Peats, PoLi = Podzolic Lithosolic, WDM = Well Drained Mineral). Dashed line represents 'good' status and solid line represents 'high' status.



The mean SRP (mg P/l) recorded for sites with felling activities (either > or < 10% felling) for each soil type is shown in Fig. 12 in relation to the percentage forestry within the catchment. Of the 7 sites that failed ‘good’ status, 6 of these were located on peat soils with > 10% felling and approximately ≥ 60% forest within the study catchment. The other remaining site was located on well drained mineral soil with < 10% felling and < 20% forestry within the catchment. Table 7 shows clearly the potential risk of the combined effect of forestry and felling on peat soils, with 16.7% of sites failing ‘good’ status. This is in contrast to peat sites with no felling, where no sites failed ‘good’ status (Table 7). From the present dataset there was no potential risk of elevated SRP levels from forestry and/or felling on podzolic lithosolic soils, with all sites (n = 88) having ‘high’ status. The results of the well drained mineral soils in relation to failed sites should be read with caution as the total number of sites were reduced (n = 25) compared to those of the peat and podzolic lithosolic soils. Indeed, there was only one site that failed for each well drained mineral category.

It was noted that mean SRP levels were significantly greater in smaller streams (< 2m wide) than larger streams (> 2m wide) (F = 7.709; P < 0.05) and this relationship will be explored later in the dilution studies.

Fig.12 Percentage forestry in the catchment against mean SRP (mg P/l) for all sites that have felling activity. Sites are separated further into soil type (Peat = Peats, PoLi = Podzolic Lithosolic, WDM = Well Drained Mineral) with > 10% or < 10% felling within the catchment. Dashed line represents ‘good’ status and solid line represents ‘high’ status.

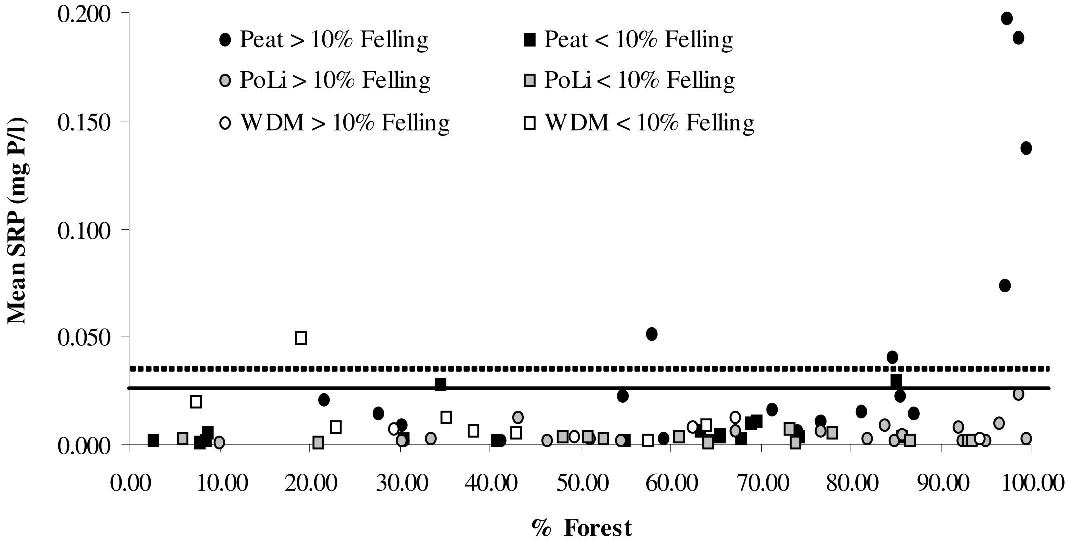


Table 7. Mean SRP (mg P/l) water quality status of the study streams in relation to soil type (Peat = Peats, PoLi = Podzolic Lithosolic, WDM = Well Drained Mineral) and presence/absence of any felling activity. The percentage of the sites for each 'Status' category are given.

Mean SRP concentration	Status	WDM (Felling)	WDM (No Felling)	PoLi (Felling)	PoLi (No Felling)	Peat (Felling)	Peat (No Felling)
> 0.035 mg P/l	Failed Status	7.7 %	8.3 %	0.0 %	0.0 %	16.7 %	0.0 %
0.025 – 0.035 mg P/l	Good Status	0.0 %	0.0 %	0.0 %	0.0 %	5.6 %	3.1 %
< 0.025 mg P/l	High Status	92.3 %	91.7 %	100.0 %	100.0 %	77.8 %	96.9 %

3.1.2 Ammonia

The mean Ammonia values (mg N/l) for each soil type in relation to percentage of catchment forested are shown in Fig.13. The highest mean Ammonia value obtained (0.048 mg N/l) was located on peat soils with > 50% forestry. On the peat soils, a significant difference in mean Ammonia was shown in relation to the percentage of forestry within the watershed. Sites that had greater than 50% forestry within the watershed had significantly greater mean Ammonia values than those found in watersheds with 25-50% forestry (z = 2.824; $P < 0.01$) and < 5% forestry (z = 5.077; $P < 0.001$). There was no significant difference between sites with > 50% forestry and sites with 5-25% forestry (z = 1.546; $P > 0.05$). The only significant difference of sites sampled on the podzol-lithosol soils was between sites with 25-50% forestry and < 5% forestry (z = 2.935; $P < 0.01$). Like SRP, the overall mean Ammonia values for the podzol-lithosol soils were generally low. There was no significant difference between sites on well drained mineral soils (χ^2 (3) = 1.641; $P > 0.05$). A high mean Ammonia of 0.046 mg N/l was recorded for the 5-25% forest band, due to the presence of one high value, which is represented by the large standard error bar in Fig.13.

Fig.13 Mean Ammonia (mg N/l) values for each soil type in relation to % of catchment forested. Treatments with different letters above standard error bars are significantly different ($P < 0.05$).

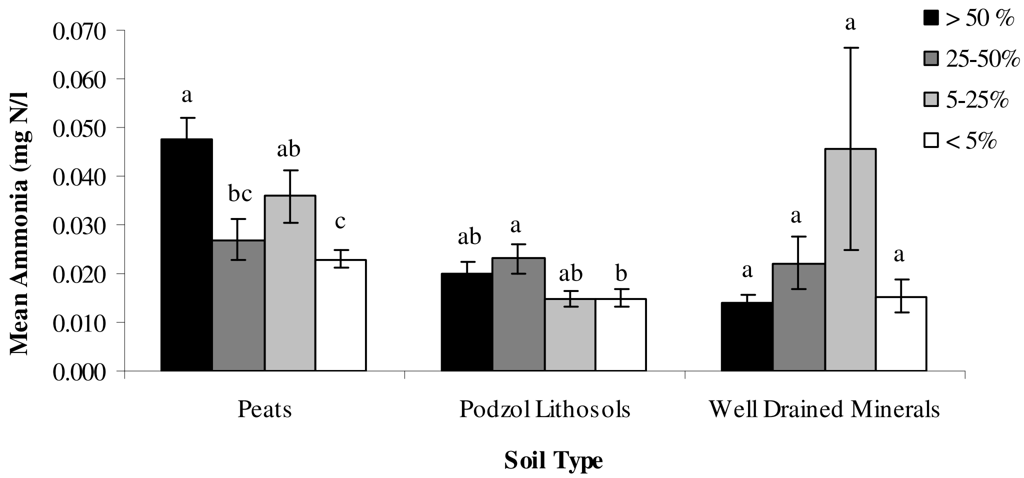


Fig.14 Mean Ammonia (mg N/l) values for peat and podzol lithosol soils in relation to % of catchment felled. Data presented are for sites with > 50% forestry. Treatments with different letters above standard error bars are significantly different ($P < 0.05$).

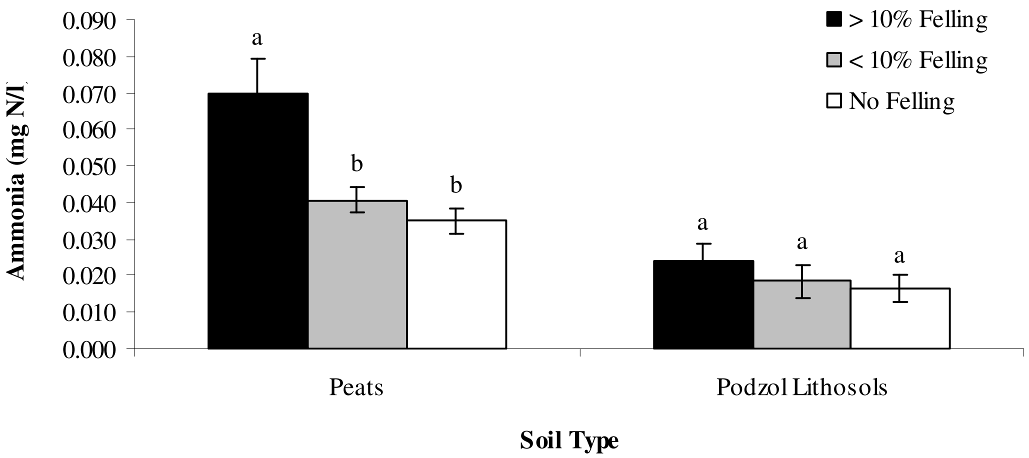


Fig. 14 shows the percentage felling for catchments that have greater than 50% forest cover. The mean Ammonia values were significantly greater on peat soils with > 10% felling compared to catchments with < 10% or with no felling (χ^2 (2) = 11.011; $P < 0.05$). There was no significant difference between the three felling levels on the podzol lithosol soils (χ^2 (2) = 2.400; $P > 0.05$). There was not enough data available for the well drained mineral soils to perform a viable statistical test.

The 'high' and 'good' water quality status for mean Ammonia (mg N/l), stated in the Draft European Communities Environmental Objectives Regulations 2008 in surface waters are as follows:

Mean Ammonia (mg N/l)

High status = ≤ 0.040 mg N/l

Good status = ≥ 0.065 ≤ 0.040 mg N/l

Fig. 15 shows the mean Ammonia (mg N/l) recorded for all sites in relation to the percentage forestry within the sites catchment. Again, like SRP, it is evident that most sites satisfy the Regulations with 81.3% of sites reaching 'high' status, but there are slightly more sites (12 sites; 5.6%) than SRP failing to reach ‘good’ status. A summary of the Ammonia status of the streams in the dataset is given in Table 8.

Fig.15 Percentage forestry in the catchment against mean Ammonia (mg N/l) for all sites. Dashed line represents ‘good’ status and solid line represents ‘high’ status.

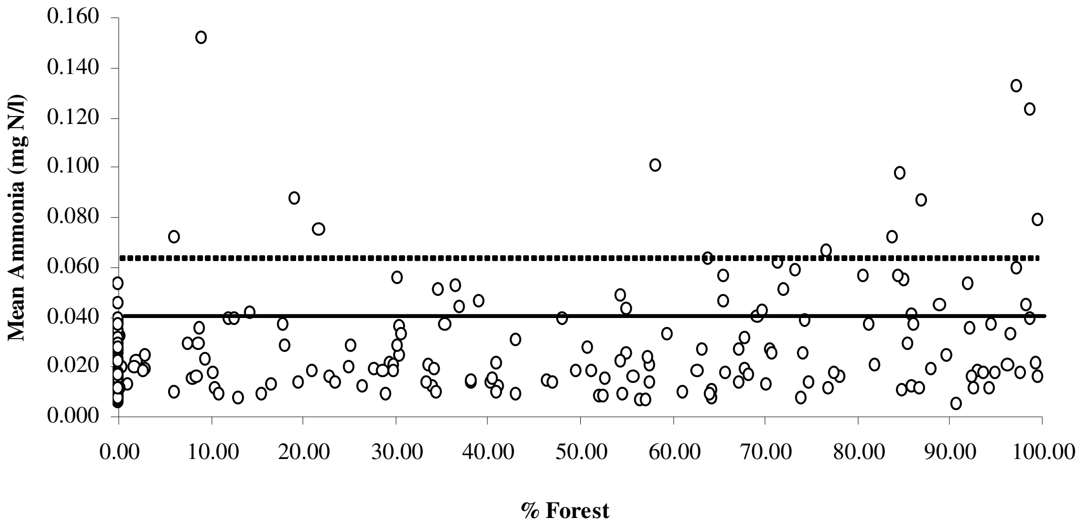


Table 8. Ammonia water quality status of streams

Mean Ammonia concentration	Status	Percentage of sites
> 0.065 mg N/l	Failed Status	5.6
0.040 – 0.065 mg N/l	Good Status	13.1
< 0.040 mg N/l	High Status	81.3

Fig. 16 shows the mean Ammonia (mg N/l) recorded for all sites in relation to the percentage forestry and presence/absence of felling activity within the catchment. The majority of sites (96.9%) with less than 5% forestry (i.e. control sites) had ‘high’ status water quality in relation to Ammonia levels. The water quality status of the streams changed slightly on sites that had more than 5% forestry and no felling activity, with 2.9% of streams failing ‘good’ status (Table 9). There was more of a shift in water quality status with sites that had more than 5% forestry combined with felling activities, although there was still a high percentage (71.6%) of ‘high’ status sites (Table 9). Within this category, 12.3% of streams failed ‘good’ status. As with SRP there is a potential risk of sites with high levels of forestry combined with a high percentage of the catchment felled to release high levels of Ammonia (mg N/l) into watercourses and as a result fail the Regulations.

Fig.16 Percentage forestry in the catchment against mean Ammonia (mg N/l) for all sites. Sites are separated into forest sites with either felling or no felling activities, and control sites. Dashed line represents ‘good’ status and solid line represents ‘high’ status.

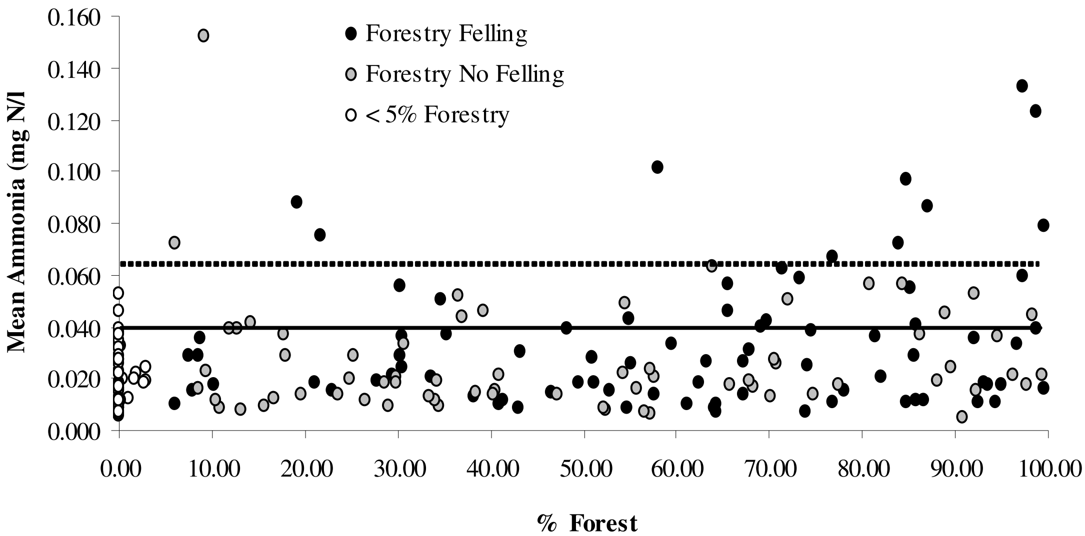


Table 9. Mean Ammonia (mg N/l) water quality status of the study streams in relation to % forestry and presence/absence of any felling activity. The percentage of the sites for each ‘Status’ category are given.

Mean Ammonia concentration	Status	< 5% Forestry (No Felling)	> 5% Forestry (No Felling)	> 5% Forestry (Felling)
> 0.065 mg N/l	Failed Status	0.0 %	2.9 %	12.3 %
0.040 – 0.065 mg N/l	Good Status	3.1 %	18.8 %	16.1 %
< 0.040 mg N/l	High Status	96.9 %	78.3 %	71.6 %

The mean Ammonia (mg N/l) recorded for all sites in relation to the percentage forestry and soil type within the catchment is shown in Fig. 17. The sites that had the highest levels of mean Ammonia (mg N/l) were found on peat soils, with 8.9% of peat sites failing ‘good’ status (Table 10). The well drained mineral soils had a similar percentage of sites that failed ‘good’ status (8.0%) but, as mentioned previously, the number of sites within this category were small (*n* = 2). The well drained mineral sites that failed ‘good’ status had a percentage forest cover below 20%, so there could potentially be influences of other landuses on the Ammonia values recorded. Only 1.1% (1 site) of the podzolic lithosolic sites sampled failed ‘good’ status with 92.1% reaching ‘high’ status (Table 10).

Fig.17 Percentage forestry in the catchment against mean Ammonia (mg N/l) for all sites. Sites are separated further into soil type (Peat = Peats, PoLi = Podzolic Lithosolic, WDM = Well Drained Mineral). Dashed line represents ‘good’ status and solid line represents ‘high’ status.

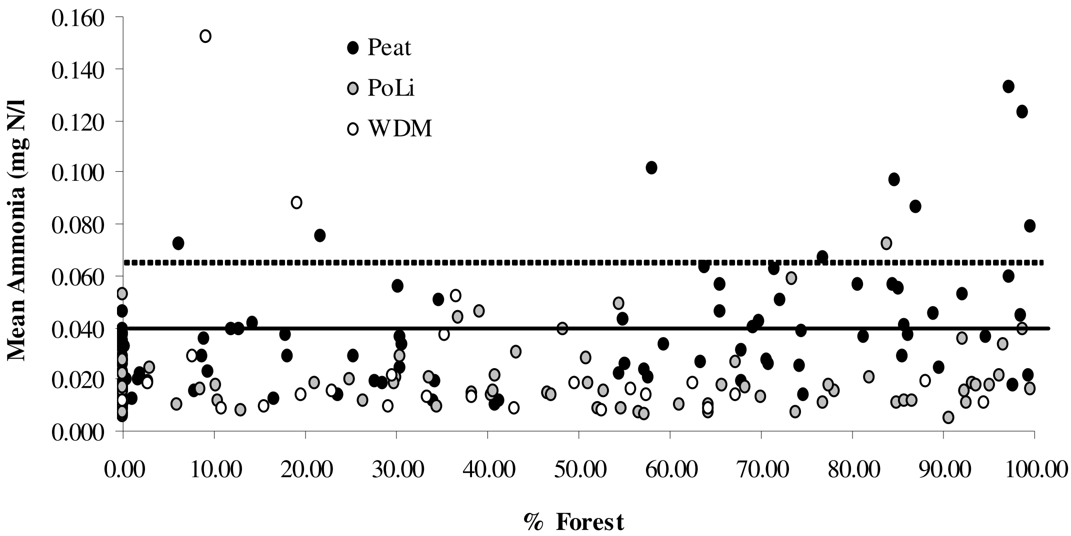


Table 10. Mean Ammonia (mg N/l) water quality status of the study streams. The percentage of the sites for each ‘Status’ category are given for each soil type.

Mean Ammonia	Status	Peats	Podzolic	Well Drained

concentration			Lithosolic	Mineral
> 0.065 mg N/l	Failed Status	8.9 %	1.1 %	8.0 %
0.040 – 0.065 mg N/l	Good Status	20.8 %	6.8 %	4.0 %
< 0.040 mg N/l	High Status	70.3 %	92.1 %	88.0 %

The mean Ammonia (mg N/l) recorded for sites with felling activities (either > or < 10% felling) for each soil type is shown in Fig. 18 in relation to the percentage forestry within the catchment. Of the 10 sites that failed ‘good’ status, 8 of these were located on peat soils with > 10% felling and the majority having approximately ≥ 60% forest within the study catchment. Of the two remaining sites, one was located on podzolic lithosolic soil with > 10% felling and > 80% forestry and the other on well drained mineral soil with < 10% felling and < 20% forestry within the catchment. Table 11 shows clearly the potential risk of the combined effect of forestry and felling on peat soils, with 22.2% of sites failing ‘good’ status. This is in contrast to peat sites with no felling, where only 1.5% failed ‘good’ status (Table 11). From the present dataset there was a small potential risk of elevated Ammonia levels from combined forestry and felling on podzolic lithosolic soils (2.9%), but the majority of sites (> 90%) achieved ‘high’ status with or without felling activity. The results of the well drained mineral soils in relation to failed sites should be read with caution as the total number of sites were reduced (n = 25) compared to those of the peat and podzolic lithosolic soils. Indeed, there was only one site that failed for each well drained mineral category.

Fig.18 Percentage forestry in the catchment against mean Ammonia (mg N/l) for all sites that have felling activity. Sites are separated further into soil type (Peat = Peats, PoLi = Podzolic Lithosolic, WDM = Well Drained Mineral) with > 10% or < 10% felling within the catchment. Dashed line represents ‘good’ status and solid line represents ‘high’ status.

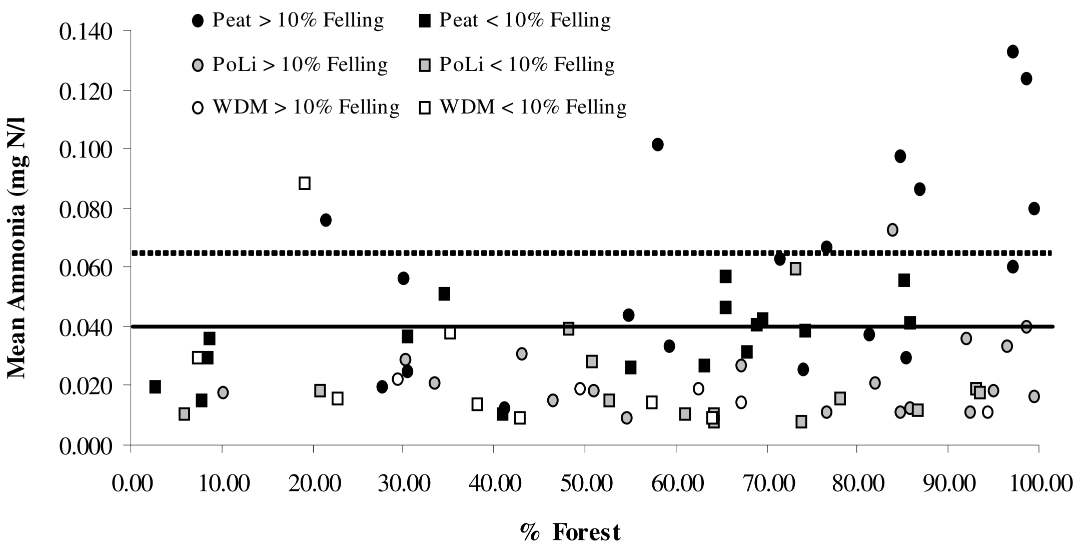


Table 11. Mean Ammonia (mg N/l) water quality status of the study streams in relation to soil type (Peat = Peats, PoLi = Podzolic Lithosolic, WDM = Well Drained Mineral) and presence/absence of any felling activity. The percentage of the sites for each 'Status' category are given.

Mean Ammonia concentration	Status	WDM (Felling)	WDM (No Felling)	PoLi (Felling)	PoLi (No Felling)	Peat (Felling)	Peat (No Felling)
> 0.065 mg N/l	Failed Status	7.7 %	8.3 %	2.9 %	0.0 %	22.2 %	1.5 %
0.040 – 0.065 mg N/l	Good Status	0.0 %	8.3 %	5.9 %	7.3 %	30.6 %	15.4 %
< 0.040 mg N/l	High Status	92.3 %	83.4 %	91.2 %	92.7 %	47.2 %	83.1 %

3.1.3 Total Organic Nitrogen (TON)

The mean TON values (mg N/l) for each soil type in relation to percentage of catchment forested are shown in Fig.19. On the peat soils, there was a significant difference in mean TON between the sites with 25-50% forestry and all other forest categories (χ^2 (3) = 7.422; $P < 0.05$), although the overall levels of TON recorded were low. On the podzolic lithosolic soils, sites with > 50% forestry had significantly higher values of TON compared to sites with 25-50% and < 5% (χ^2 (3) = 13.337; $P < 0.01$), but not with sites with 5-25% forestry. As with peat soils, the mean values recorded on podzolic lithosolic soils were low. In well drained mineral soils the mean TON increased with decreasing forest cover with the exception of sites with less than 5% forestry. However, it should be noted that the number of sites in this category ($n = 2$) was extremely low. The mean TON of sites with 5-25% forestry was significantly greater than the other forest categories (χ^2 (3) = 14.128; $P < 0.01$). Indeed, well drained mineral soils had significantly greater mean values of TON compared to peat and podzolic lithosolic soils at each forest cover category (> 50% - χ^2 (2) = 43.878; $P < 0.001$; 25-50% - χ^2 (2) = 14.269; $P = 0.01$; 5-25%

- χ^2 (2) = 16.199; $P < 0.001$) with the exception of < 5% where well drained mineral soils were only represented by 2 sites.

Fig. 20 shows the percentage felling for catchments that have greater than 50% forest cover. The mean TON was significantly greater on peat soils with > 10% felling compared to catchments with < 10% felling (z = 2.255; $P < 0.05$), but not significant with catchments with no felling (z = 1.858; $P > 0.05$). There was no significant difference between the three felling levels on the podzolic lithosolic soils (χ^2 (2) = 2.252; $P > 0.05$). Although there were significantly greater values of TON in streams in catchments with felling, the values of TON were markedly low. There was not enough data available for the well drained mineral soils to perform a viable statistical test.

Fig.19 Mean TON (mg N/l) values for each soil type in relation to % of catchment forested. Treatments with different letters above standard error bars are significantly different ($P < 0.05$).

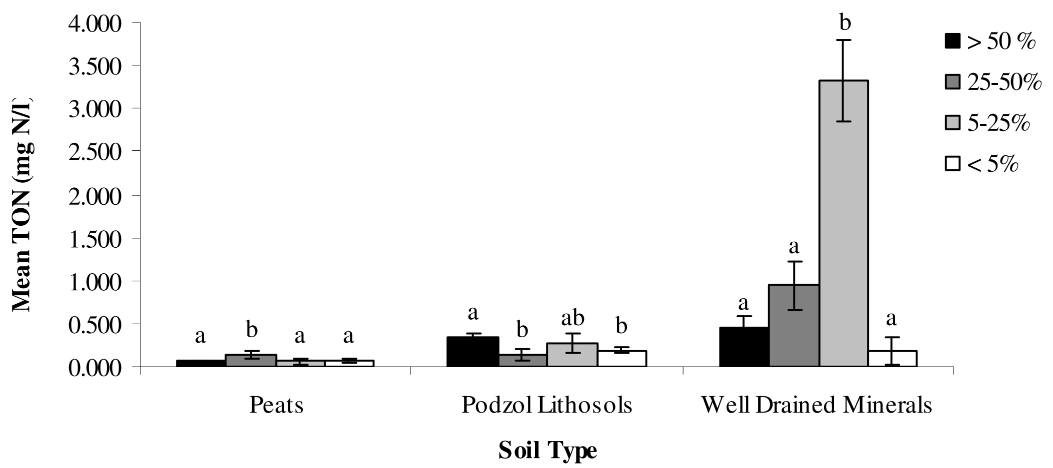
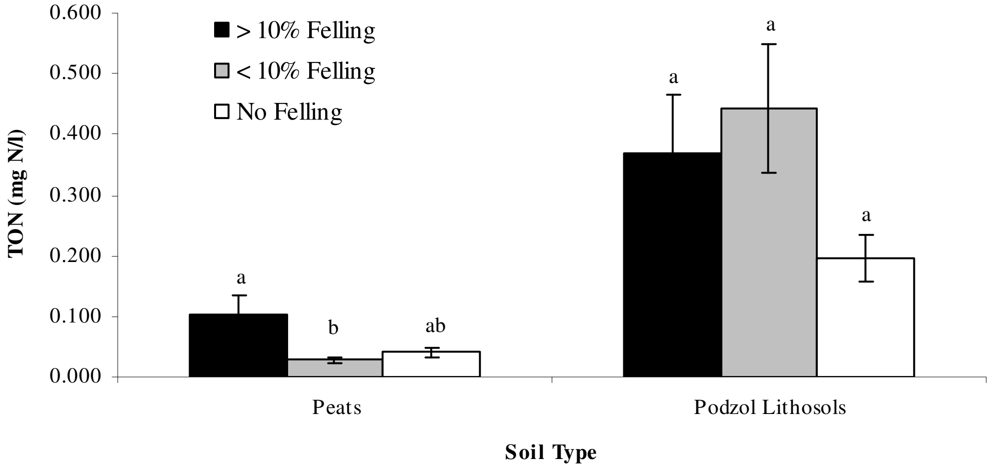


Fig.20 Mean TON (mg N/l) values for peat and podzolic lithosolic soils in relation to % of catchment felled. Data presented are for sites with > 50% forestry. Treatments with different letters above standard error bars are significantly different ($P < 0.05$).



3.2 DILUTION STUDY

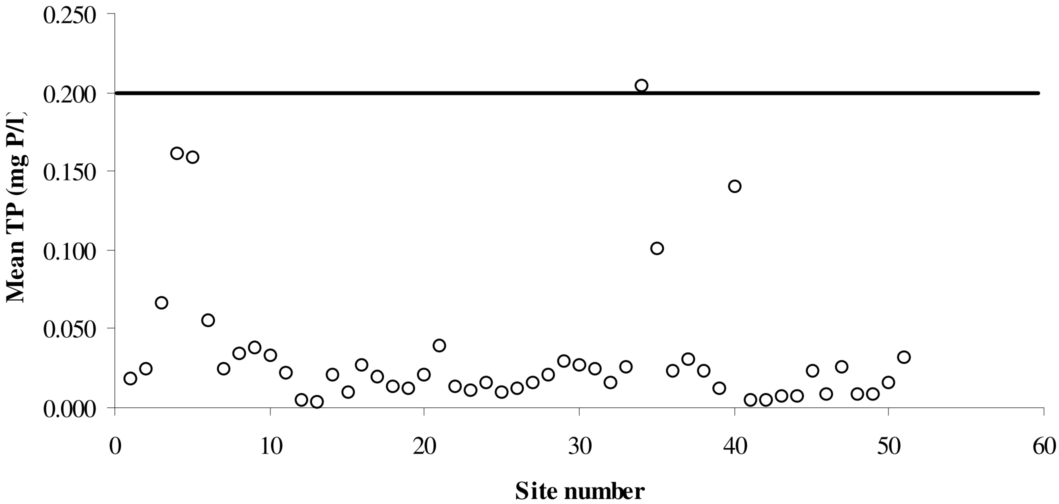
The mean SRP, TP and Ammonia values obtained for the 51 additional sites sampled in the southern Slieve Aughty mountains region are shown in Fig. 21. Six of the 51 sites (11.8%) sampled failed 'good' status for SRP levels, four (7.8%) of the sites failed 'good' status for Ammonia levels, and 1 (2%) of the sites failed the EPA limit for TP levels (Fig. 21).

Fig.21 Mean (a) SRP (mg P/l), (b) TP (mg P/l) and (c) Ammonia (mg N/l) values recorded for all additional sites in the southern Slieve Aughty mountains, Co. Clare. Dashed line represents 'good' status and solid line represents 'high' status for SRP and

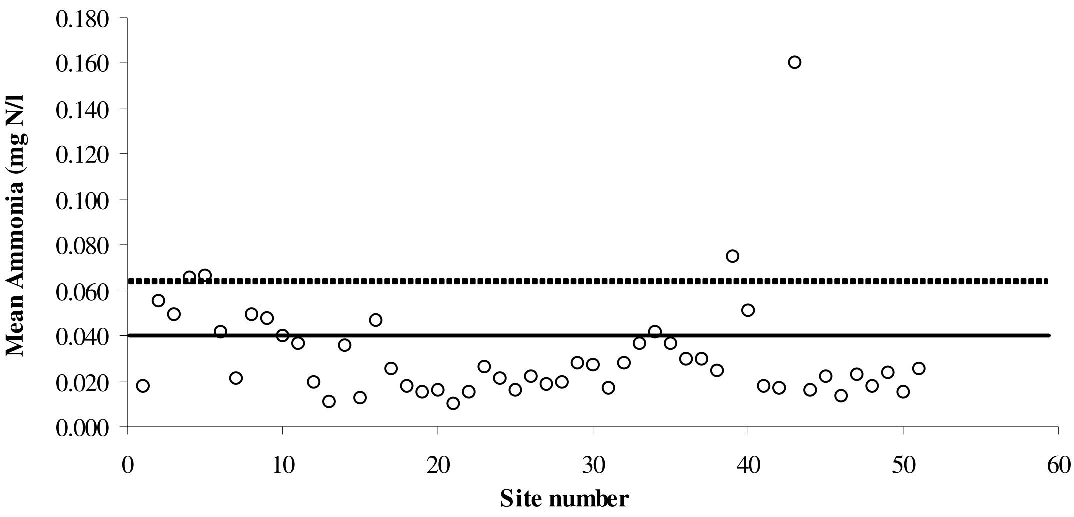
Ammonia. Solid line for TP represents the EPA 'limit value indicative in order to reduce eutrophication' of 0.2 mg P/l (salmonid waters).

(a)

(b)



(c)



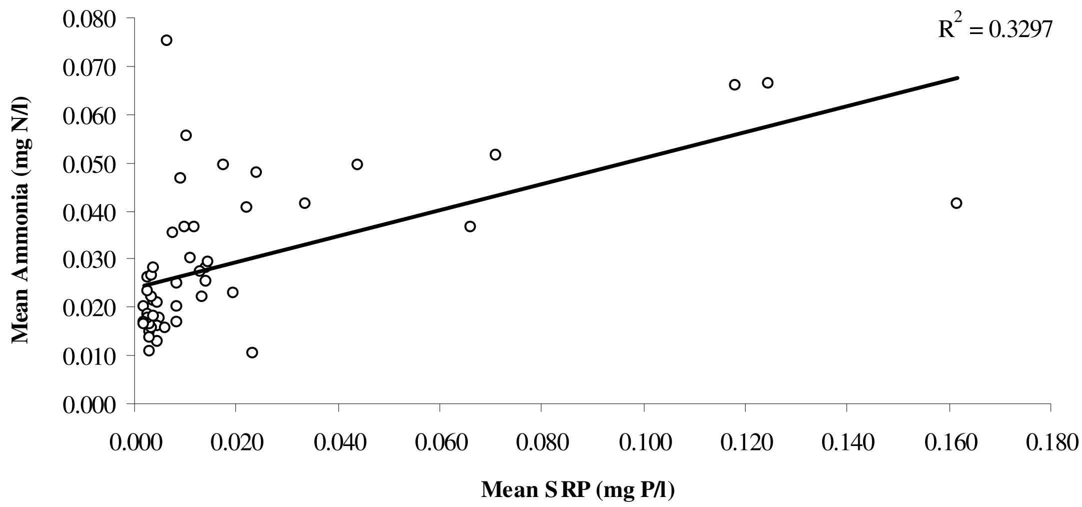
There was a significant relationship between mean SRP and mean Ammonia ($r = 0.315$; $P < 0.05$). After the removal of the one detected outlier from the dataset, this correlation became stronger ($r = 0.575$; $P < 0.001$) with Ammonia levels increasing with observed increases in SRP levels (Fig. 22). As SRP and TP are highly correlated ($r = 0.984$; $P < 0.001$) a similar significant relationship was observed between TP and Ammonia ($r = 0.594$; $P < 0.001$). When the data was separated by soil type, the correlation between SRP and Ammonia was $r = 0.504$; $P < 0.05$ for peat soils and $r = 0.521$; $P < 0.01$ for podzolic lithosolic soils (Fig. 22).

The majority of the sites sampled in this region were of 'good' or 'high' water quality status, but we wanted to investigate the sites that were failing in more detail. Two rivers where high levels of SRP and/or Ammonia were recorded were selected for further sampling; the Corra and the Ballycorban Rivers.

The sampling areas for the Corra River are given in Fig. 23 (numbered 1-13, with sample 1 being the furthest upstream site and sample 13 the furthest downstream site). Water samples were taken from tributaries as well as from the main system of the Corra River and covered a distance of approximately 7km. The levels of SRP and Ammonia are shown in Fig. 24.

Fig.22 Correlation of mean SRP (mg P/l) against mean Ammonia (mg N/l) for (a) all the sampled sites and for (b) all the sampled sites divided by soil type (PoLi = Podzolic Lithosolics, Peats = Peat).

(a)


R^2 = 0.3297
Mean Ammonia (mg N/l)
Mean SRP (mg P/l)
0.000
0.010
0.020
0.030
0.040
0.050
0.060
0.070
0.080
0.000
0.020
0.040
0.060
0.080
0.100
0.120
0.140
0.160
0.180

(b)

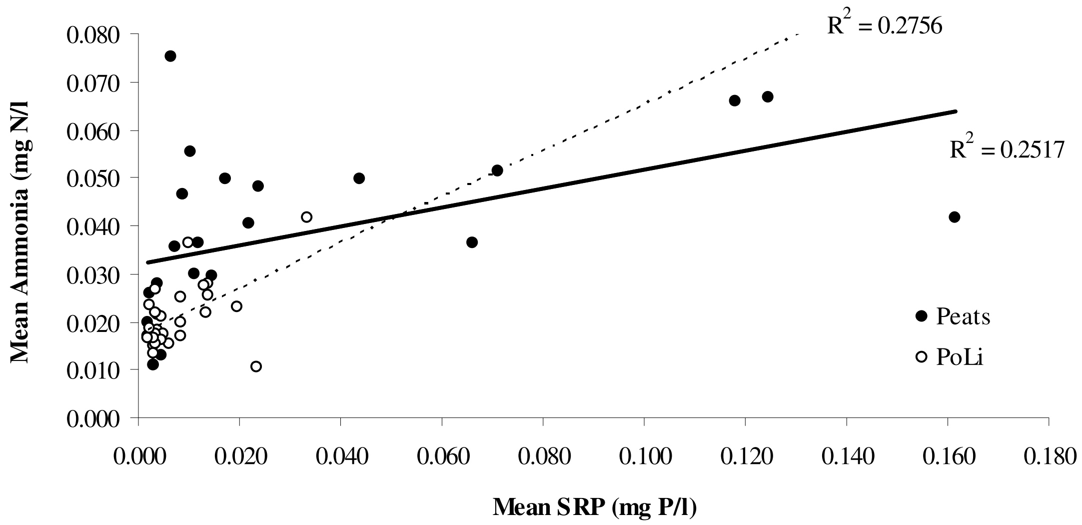

R^2 = 0.2756
R^2 = 0.2517
Peats
PoLi
Mean Ammonia (mg N/l
Mean SRP (mg P/l)
0.000
0.010
0.020
0.030
0.040
0.050
0.060
0.070
0.080
0.000
0.020
0.040
0.060
0.080
0.100
0.120
0.140
0.160
0.180

Fig. 23 Map of sampling points on Corra River, Co. Clare.

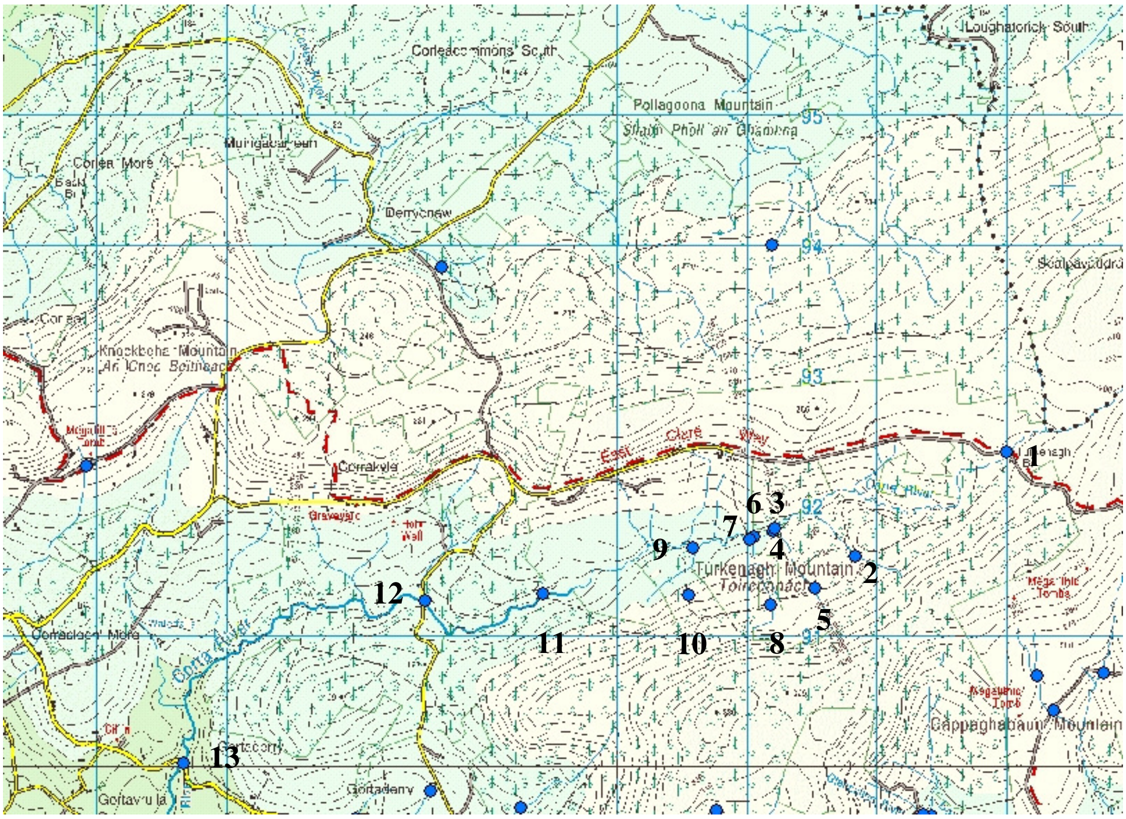
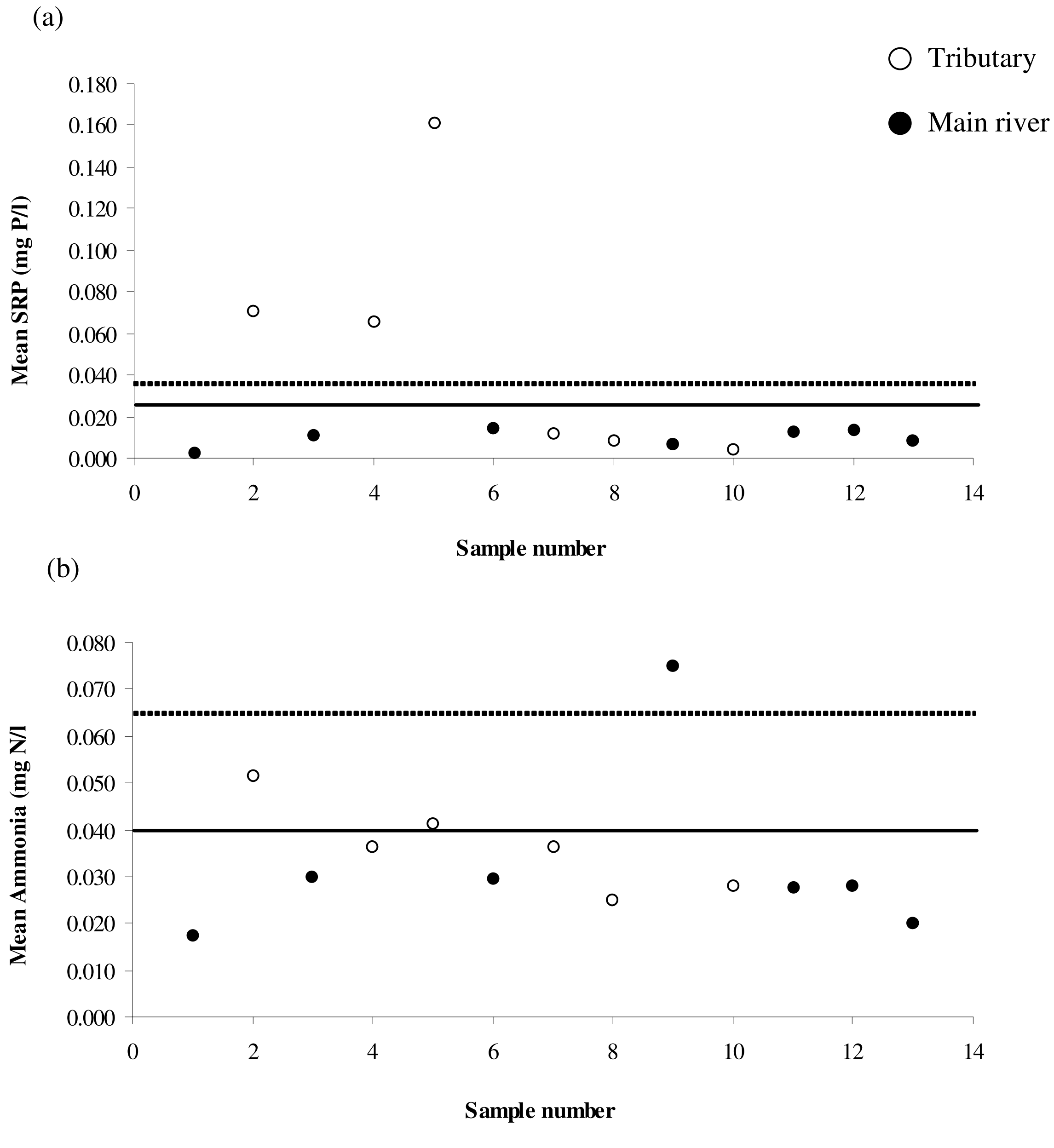


Fig.24 Corra River tributary (clear circles) and main river (black circles) values for (a) mean SRP (mg P/l) and (b) mean Ammonia (mg N/l). Dashed line represents 'good' water quality status and solid line represents 'high' water quality status for both parameters.

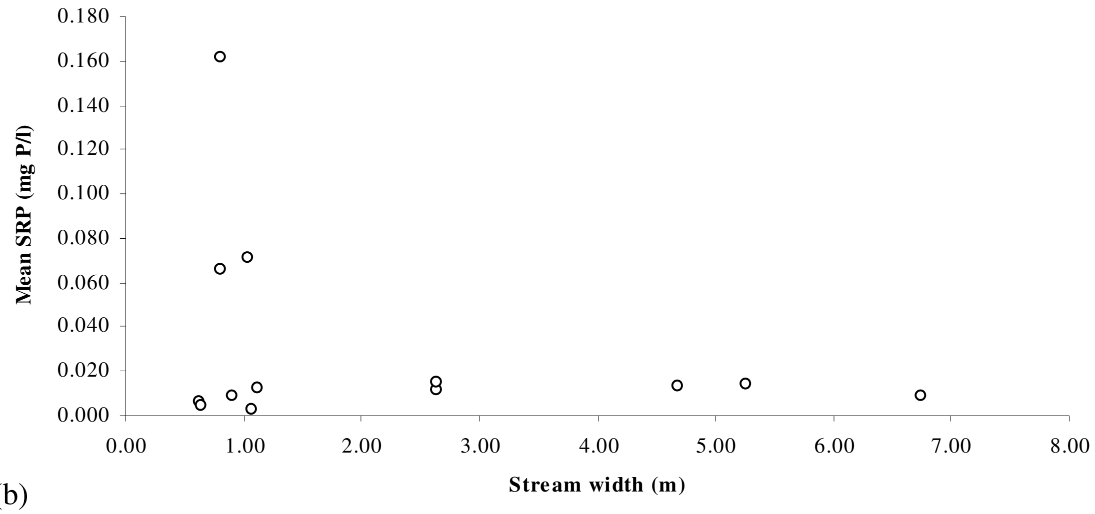


Upstream Downstream

Three samples taken from the tributaries of the Corra River showed high levels of mean SRP, which all failed ‘good’ status. Sampling in the main river immediately after where the tributary enters the main stretch, the SRP values were reduced/diluted to such an extent that the water quality was of ‘high’ status. The water quality remained at this status for the subsequent samples downstream from both tributaries and the main river. Only 1 site had elevated Ammonia levels that failed ‘good’ water quality status and this was taken from the main river at sampling point 9. Apart from this sampling point the tributaries tended to have higher Ammonia values than the main river, but these satisfied the Regulations at at least the ‘good’ status level. The data shows the importance at sampling at a multitude of sample points to obtain a better overall picture of the hydrochemistry of a particular site.

Fig.25 Relationship between stream width (m) and (a) mean SRP (mg P/l) and (b) mean Ammonia (mg N/l) for the Corra River.

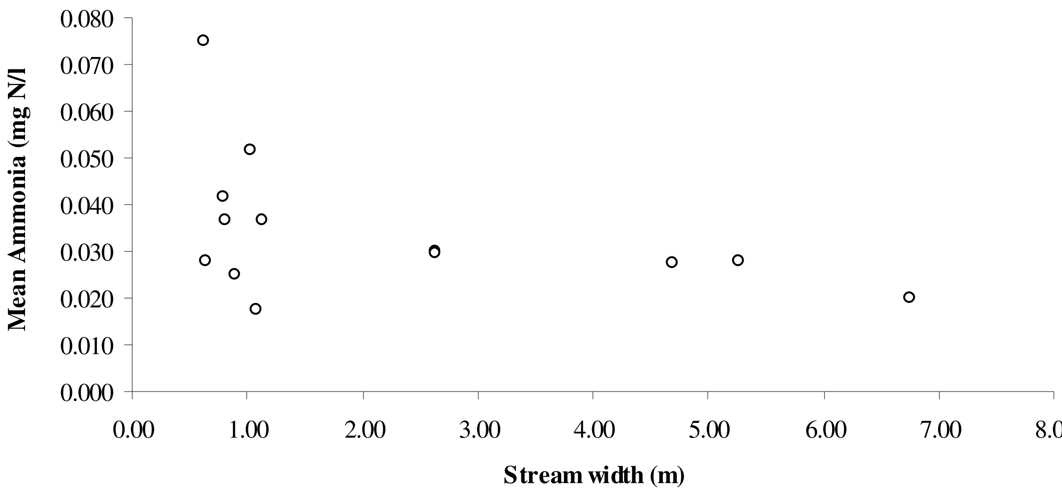
(a)



(b)

Although the correlations were not significant (SRP, r = 0.295; $P > 0.05$; Ammonia, r = 0.442; $P > 0.05$) it is clear from Fig. 25 that with an increase in the stream width there was a decrease in the levels of both SRP and Ammonia in the river. Streams of 1m width or less had the highest levels of both SRP and Ammonia so impacts can be detected in these small streams, whereas further downstream the impacts will be diluted with increasing stream width.

The sampling areas for the Ballycorban River are given in Fig. 26 (numbered 1-8, with sample 1 being the furthest upstream site and sample 8 the furthest downstream site). Water samples were taken from the main system of the Ballycorban River and covered a



distance of approximately 4km. The levels of SRP and Ammonia in relation to proximity to felling activity are shown in Fig. 27.

Fig.26 Map of sampling points on Ballycorban River, Co. Clare.

5

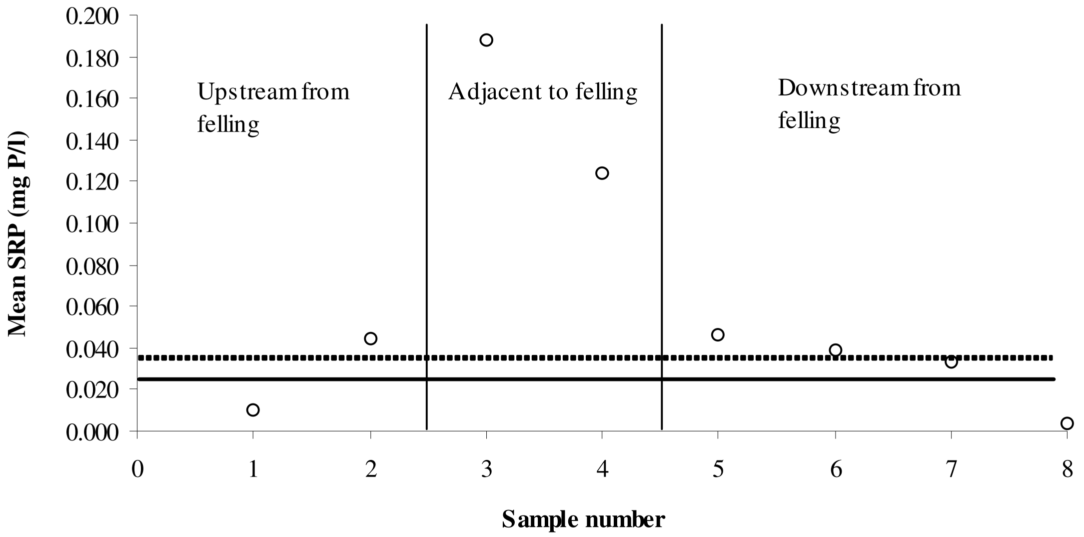
6

7

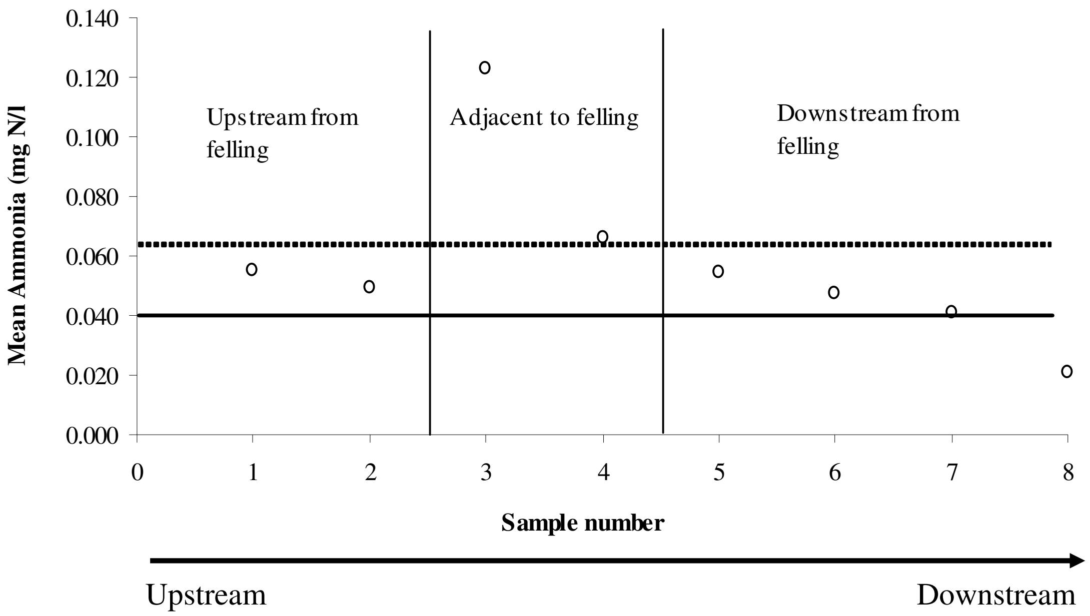
8

Fig.27 Ballycorban River values for (a) mean SRP (mg P/l) and (b) mean Ammonia (mg N/l). Dashed line represents 'good' water quality status and solid line represents 'high' water quality status for both parameters. Sample sites spilt into 3 sections in relation to proximity to felling activity (Upstream from felling, Adjacent to felling, Downstream from felling).

(a)



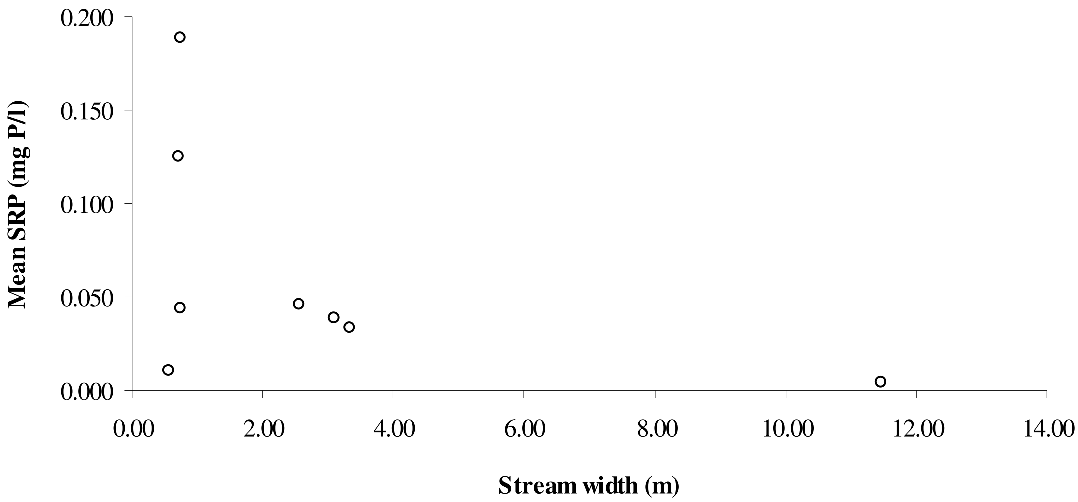
(b)



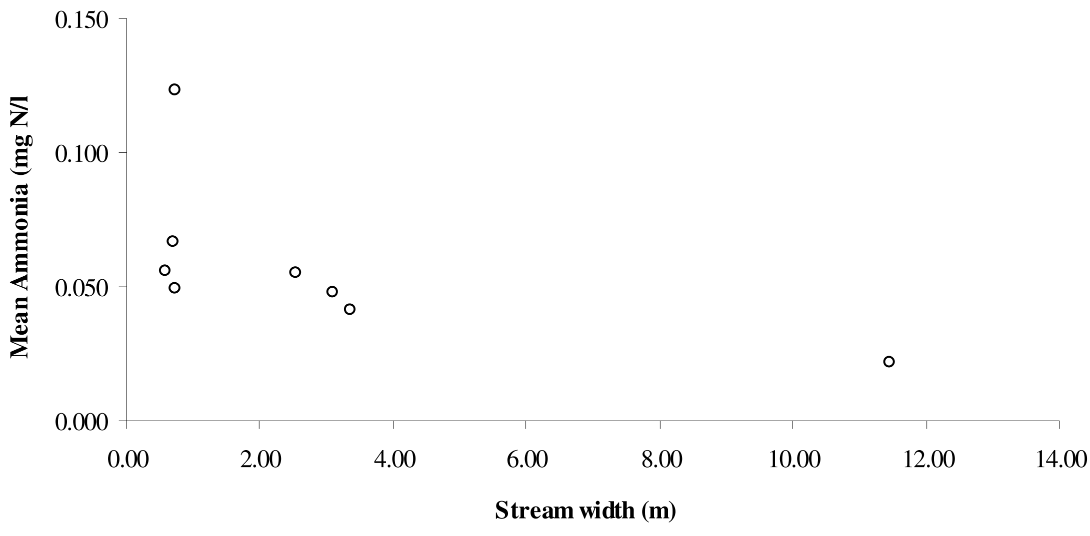
In the Ballycorban River, two sites were sampled upstream from the felling activity, two sites adjacent to the felling and four sites downstream from the felling. Of the 8 samples collected, five had mean SRP levels that failed 'good' status, with the two sites adjacent to felling giving the highest recorded values (Fig. 27). Both sampling point 1 and point 8 had levels of SRP that were of 'high' status, with sampling point 7 having 'good' status.

Fig.28 Relationship between stream width (m) and (a) mean SRP (mg P/l) and (b) mean Ammonia (mg N/l) for the Ballycorban River.

(a)



(b)



Two of the 8 sites had elevated Ammonia levels that failed 'good' water quality status, with these sites again being adjacent to the felling. The two upstream of felling sites were of ‘good’ status as were the subsequent three sampling points downstream from the felling, with the furthest downstream site exhibiting ‘high’ status. There was a very strong positive correlation between the mean SRP and Total Ammonia for this particular stream (r = 0.914; $P = 0.001$). The data shows the importance at sampling at a multitude of sample points to obtain a better overall picture of the hydrochemistry of a particular site.

Although the correlations were not significant (SRP, r = 0.474; $P > 0.05$; Ammonia, r = 0.606; $P > 0.05$) it is evident again that with an increase in the stream width there was a decrease in the levels of both SRP and Ammonia in the river. Streams of ≤ 2m width had the highest levels of both SRP and Ammonia so the potential risk of detecting an impact increases when sampling streams within this size range.

3.3 Sediment

3.3.1 Suspended Solids (SS)

The mean SS values (mg/l) for each soil type in relation to percentage of catchment forested are shown in Fig.29. There was no significant difference between forest cover categories for peat soils (χ^2 (3) = 1.418; $P > 0.05$) or well drained mineral soils (χ^2 (3) = 0.769; $P > 0.05$). There was significantly greater SS levels in streams on podzolic lithosolic soils with > 50% forestry and 5-25% forestry compared to sites with < 5% forestry (χ^2 (3) = 11.513; $P < 0.01$). There was no significant difference between soil types at each forest cover category with the exception of peat and podzolic lithosolic soils at the < 5% forestry category, where peat soils had a significantly greater mean SS level (z = 3.281; $P = 0.001$). Even though statistically significant differences were detected between and within soil types in relation to percentage forestry within the catchment, the SS levels recorded in this study were well below the recommendations of the Salmonid Waters Regulations 1988 (≤ 25 mg/l) and Surface Water Regulations 1989 (50 mg/l) with the maximum mean SS recorded being 23.22 mg/l.

Fig.29 Mean Suspended Solid (mg/l) values for each soil type in relation to % of catchment forested. Treatments with different letters above standard error bars are significantly different ($P < 0.05$).

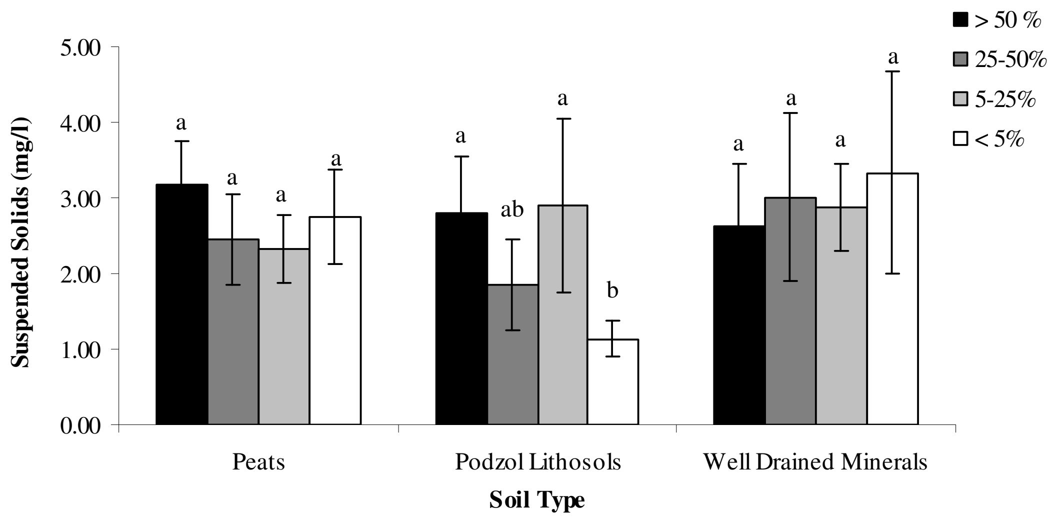
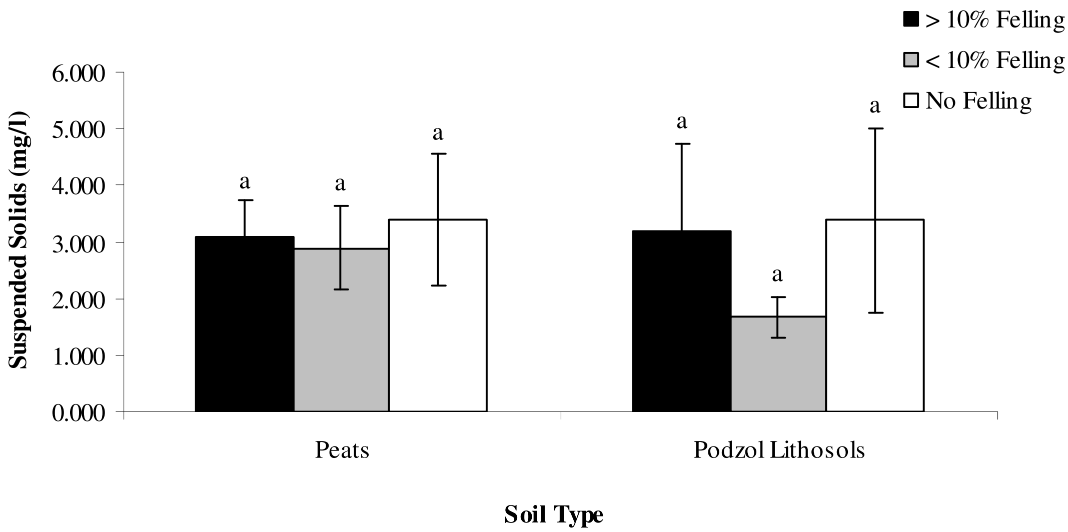


Fig. 30 shows the percentage felling for catchments that have greater than 50% forest cover. There was no significant difference between felling levels for either peat (χ^2 (2) = 0.962; $P > 0.05$) or podzolic lithosolic soils (χ^2 (2) = 0.536; $P > 0.05$) in relation to SS levels. Again it should be noted that the values of SS recorded were markedly low compared to Regulation values. There was not enough data available for the well drained mineral soils to perform a viable statistical test.

Fig.30 Mean Suspended Solid (mg/l) values for peat and podzol lithosol soils in relation to % of catchment felled. Data presented are for sites with > 50% forestry. Treatments with different letters above standard error bars are significantly different ($P < 0.05$).



3.3.2 Bedload Sediment

The mean bedload sediment values for each soil type in relation to percentage of catchment forested are shown in Fig.31. There were no significant differences between forest cover categories for peat soils (χ^2 (3) = 4.292; $P > 0.05$), podzolic lithosolic soils (χ^2 (3) = 7.095; $P > 0.05$) or well drained mineral soils (χ^2 (3) = 2.120; $P > 0.05$). Although no significant differences were detected there was a tendency for the higher percentage forestry categories (> 50% and 25-50%) streams to have higher bedload sediment than the lower forestry categories (5-25% and < 5%), in particular on the peat and podzolic lithosolic soils. There was no significant difference between soil types at each forest cover category, > 50% (χ^2 (2) = 1.367; $P > 0.05$); 25-50% (χ^2 (2) = 1.250; $P > 0.05$); and < 5% (χ^2 (2) = 0.117; $P > 0.05$), with the exception of the 5-25% category (χ^2 (2) = 6.989; $P < 0.05$) where there was significantly more bedload sediment in the well drained mineral sites compared to the peat and podzolic lithosolic soils.

Fig.31 Mean fine bedload sediment (g) for each soil type in relation to % of catchment forested. Treatments with different letters above standard error bars are significantly different ($P < 0.05$).

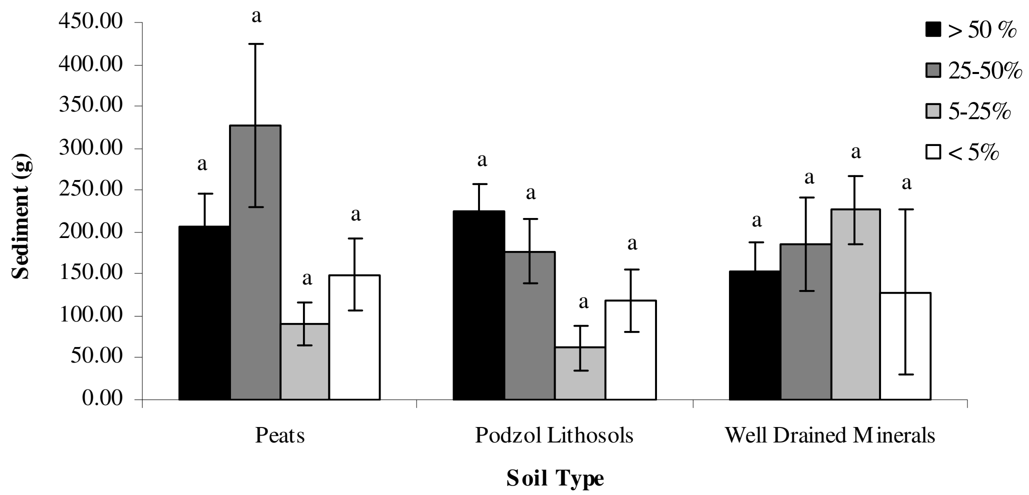
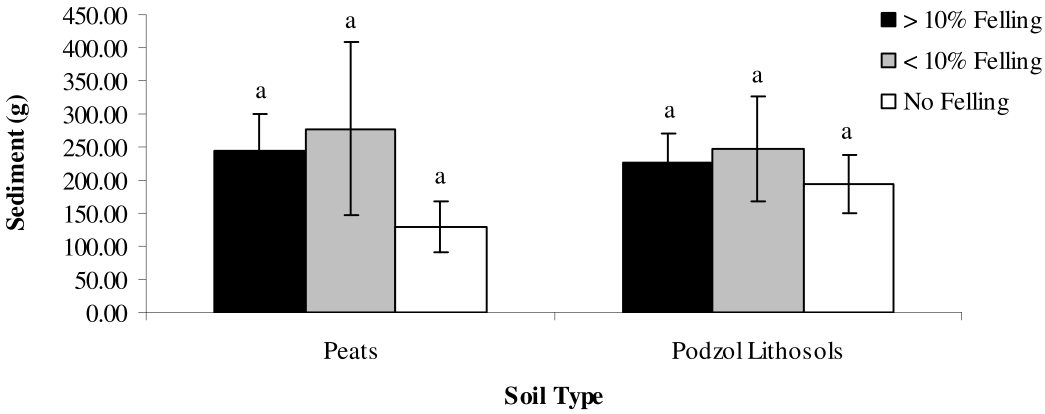


Fig. 32 shows the percentage felling for catchments that have greater than 50% forest cover in relation to bedload sediment. There was no significant difference between felling levels for either peat (χ^2 (2) = 2.148; $P > 0.05$) or podzolic lithosolic soils (χ^2 (2) = 0.216; $P > 0.05$). Again, although there was no significant difference detected between sites, catchments that had some felling activity tended to have higher bedload sediment levels than catchments that had no felling activities. Although further work is needed, a potential risk exists in highly forested catchments combined with felling activities. There was not enough data available for the well drained mineral soils to perform a viable statistical test.

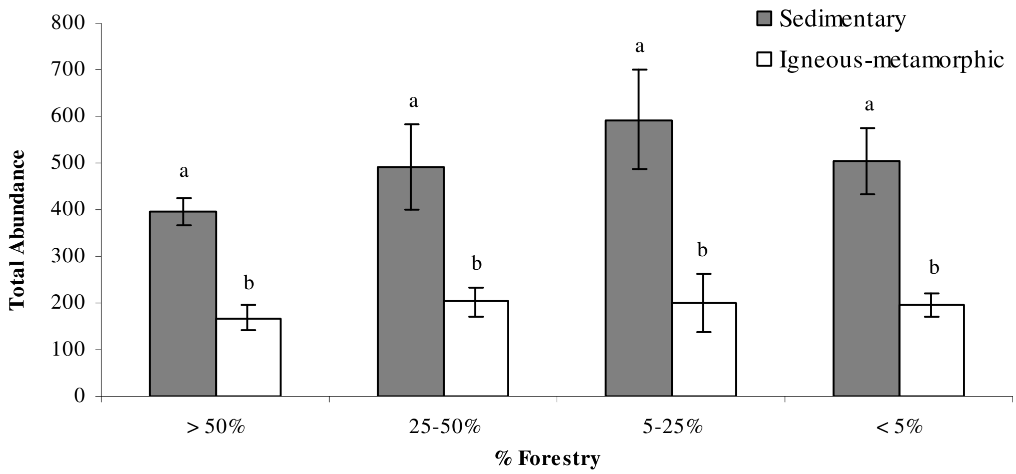
Fig.32 Levels of mean fine bedload sediment (g) for each soil type in relation to % area felled in catchments with > 50% forestry. Treatments with different letters above standard error bars are significantly different ($P < 0.05$).


450.00
400.00
350.00
300.00
250.00
200.00
150.00
100.00
50.00
0.00
Sediment (g)
a
a
a
a
a
a
■ > 10% Felling
■ < 10% Felling
□ No Felling
Peats
Podzol Lithosols
Soil Type

3.4 MACROINVERTEBRATES

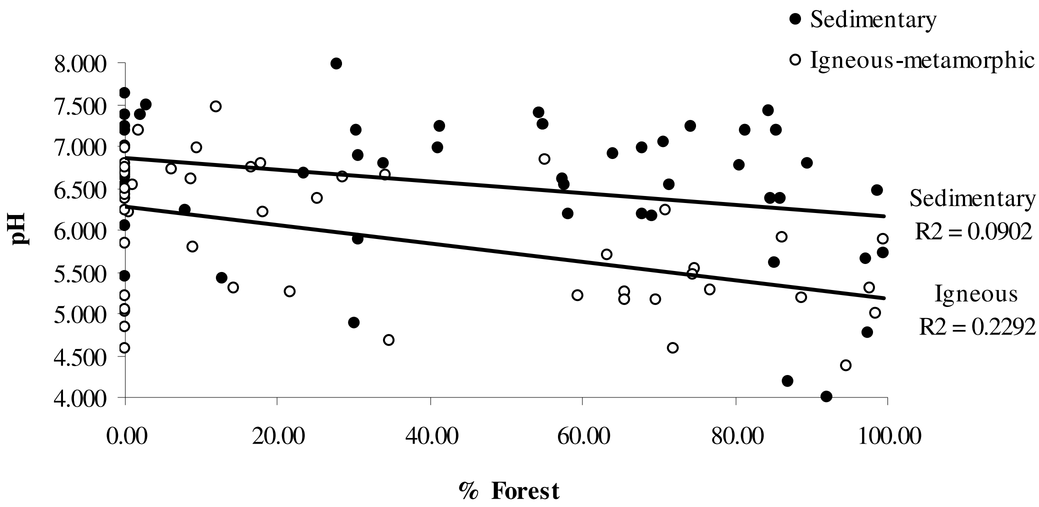
In total, 318,000 individuals were collected and identified during the period of this study, spanning 206 taxa and covering all the major invertebrate groups. The total abundance of macroinvertebrates was found to be greater on sedimentary geology than igneous-metamorphic at all four forestry categories (Fig. 33), > 50% (z = 5.102; $P < 0.001$); 25-50% (z = 2.190; $P < 0.05$); 5-25% (z = 2.713; $P < 0.01$); and < 5% (z = 3.221; $P = 0.001$). There was a trend for the macroinvertebrate abundance to increase with decreasing forestry on sedimentary geology, with a significantly greater abundance detected between the 5-25% forestry sites compared to the > 50% forestry sites (z = 1.975; $P < 0.05$). No such trend was observed in the igneous-metamorphic geology sites.

Fig.33 Macroinvertebrate mean total abundance for each forest cover category on sedimentary and igneous-metamorphic geology. Treatments with different letters above standard error bars are significantly different ($P < 0.05$).



As the peat soils have shown significant impacts of nutrients such as SRP and Ammonia with high percentage forestry within the catchment combined with felling, the peat soils will be explored in further detail. Fig. 34 shows the relationship between the percentage forest within a catchment and pH on the two types of geology on peat soils.

Fig.34 Relationship between % forest in catchment and pH on the two types of geology on peat soils.



Igneous-metamorphic geology showed a significant correlation between % forest within the catchment and pH ($r = -0.479$; $n = 50$; $P < 0.001$). The sedimentary geology also showed a significant correlation between % forest within the catchment and pH but to a lesser degree ($r = -0.300$; $n = 51$; $P < 0.05$).

pH is known to be a main driver in invertebrate assemblage structure. For this reason, calculations investigated all sites on peat soils irrespective of pH value, and sites on peat soils that had greater than pH 6 (Fig. 35 and Fig. 36). From the PCA results the same suite of hydrochemical parameters are observed in both cases with high SRP, TP, Total Ammonia and DTOC being associated with high percentage forestry and presence of felling activity.

Fig.35 Principal Components Analysis (PCA) of (a) the water chemistry parameters measured and (b) the individual sites in relation to % forestry and (c) the individual sites in relation to % felling on peat soils.

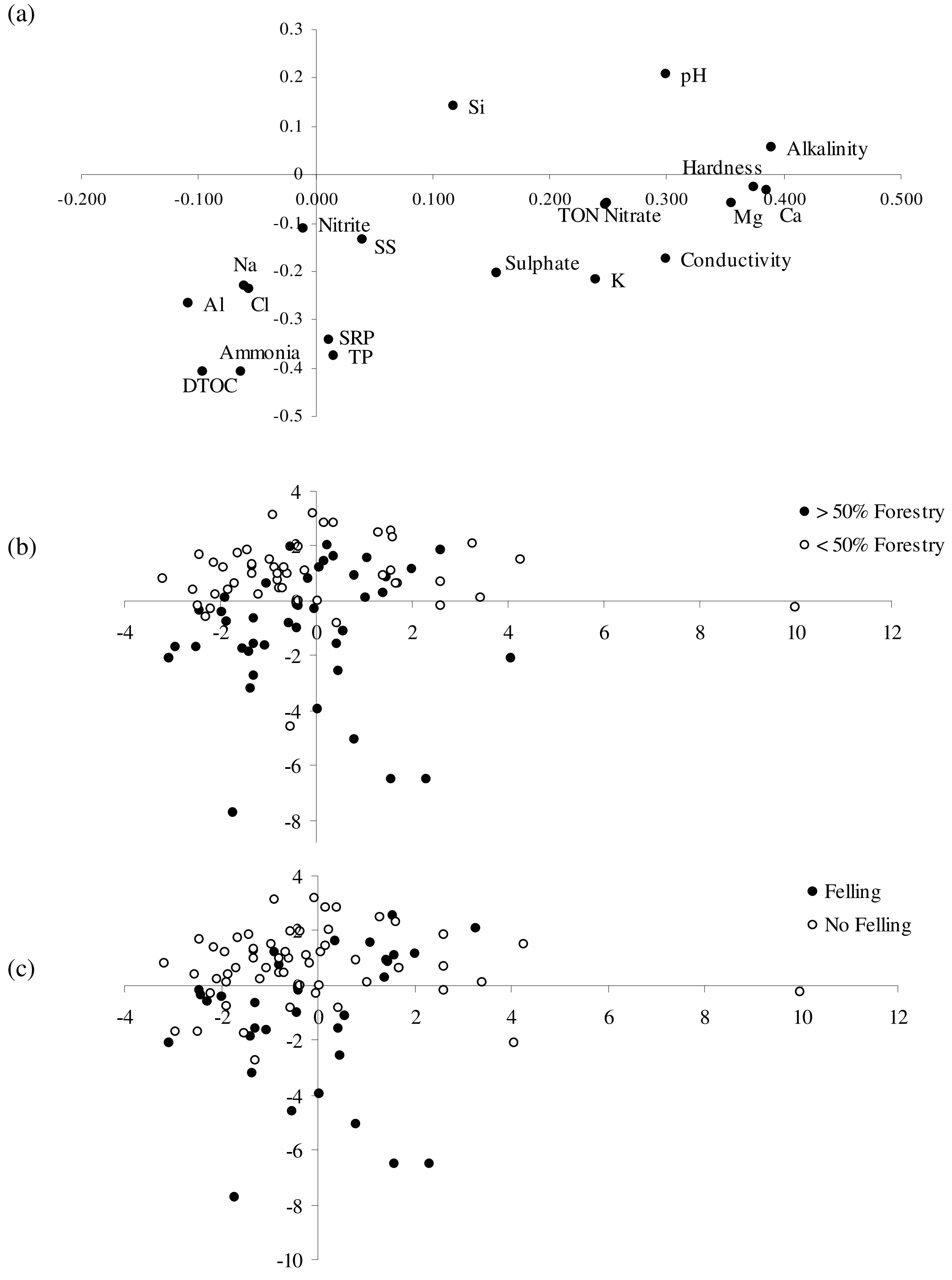
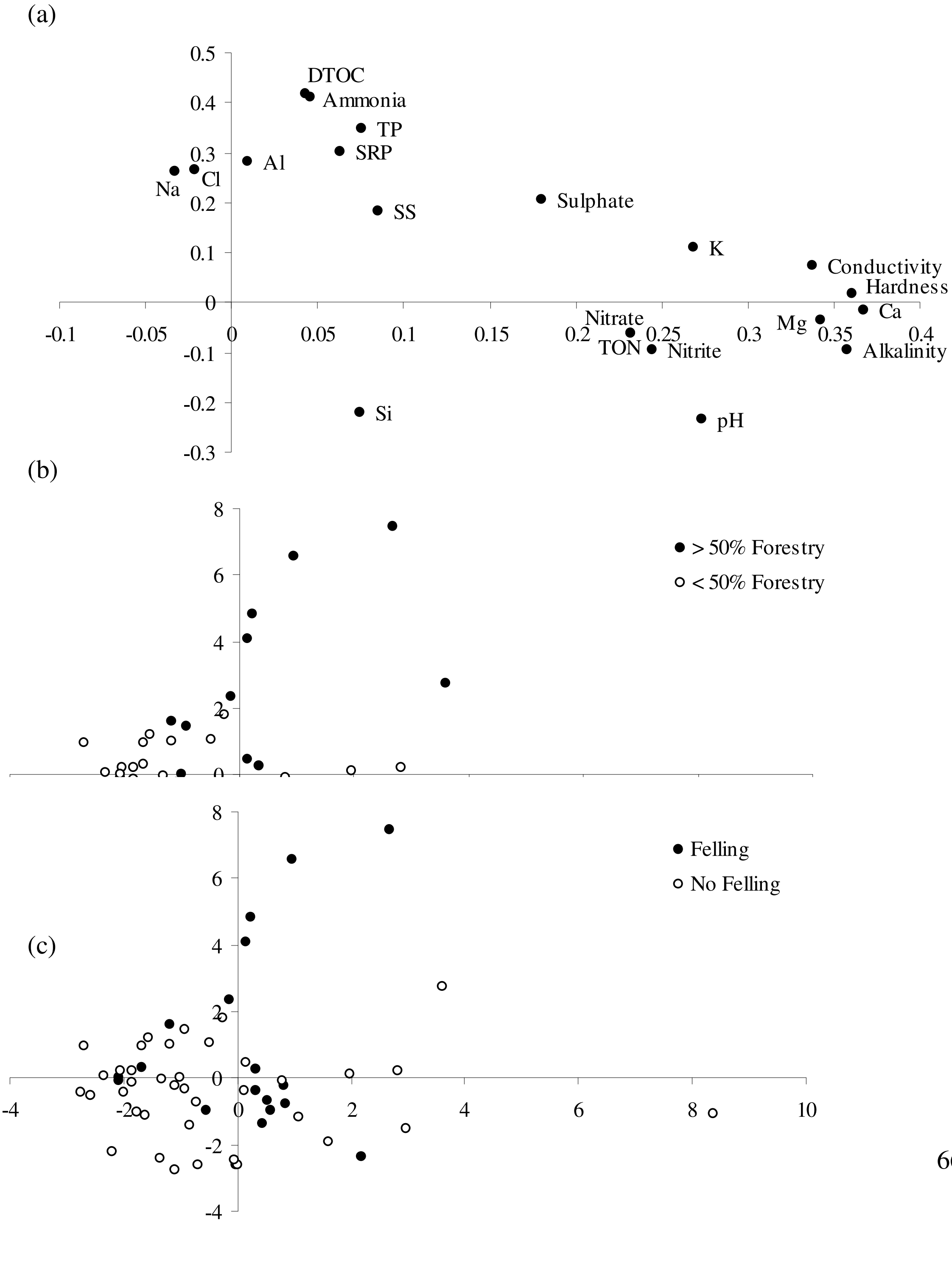


Fig.36 Principal Components Analysis (PCA) of (a) the water chemistry parameters measured and (b) the individual sites in relation to % forestry and (c) the individual sites in relation to % felling on peat soils with pH values > 6.



The PCA axis 2 is pulling these sites out for both datasets and therefore axis 2 was correlated against a suite of invertebrate metrics to identify the presence of significant relationships "over and above" the impact derived from pH. The results of the correlations are shown in Table 12. Of the 14 metrics presented, 11 are significantly correlated with PCA axis 2 when all sites are examined. By removing the potential impact of pH (examing only sites with pH > 6), the correlations were run again with 6 of the metrics showing a significant relationship with PCA axis 2. The most significant relationship was with Plecopteran richness, where the richness decreased with increased levels of SRP, TP, Ammonia and DTOC. Plecopterans are known to be sensitive to eutrophication and this impact would appear to be the case in this study on peat sites with high forestry and presence of felling activity. There were significant correlations with taxon richness, Ephemeropteran richness, Trichopteran richness, EPT richness, and the BMWP score observed in the complete peat dataset that were not present in the peat sites with pH values > 6 (Table 12).

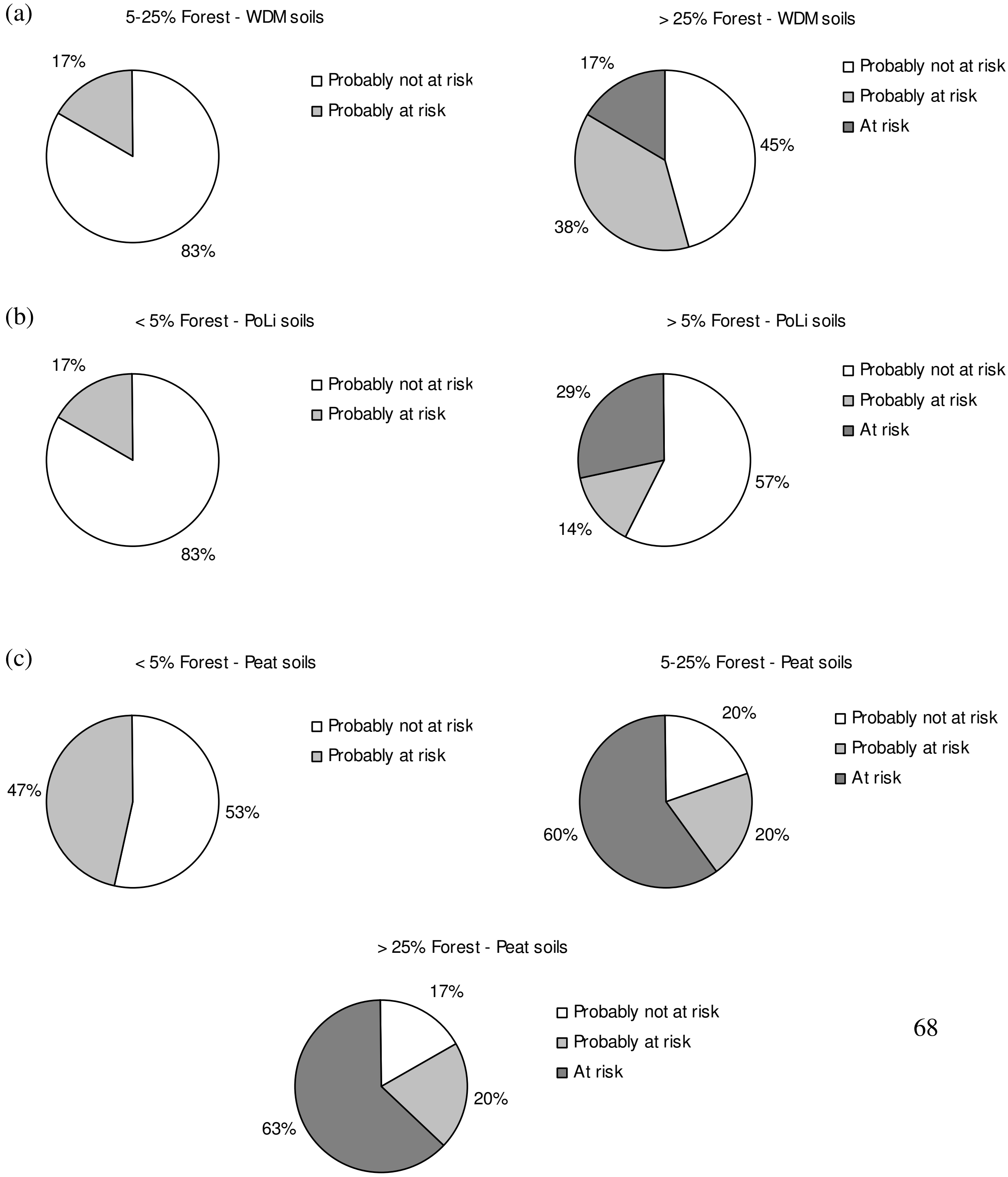
Table 12. Macroinvertebrate metrics for all peat sites and peat sites with pH values > 6. * denotes significant correlation at 0.05 level; ** denotes significant correlation at 0.01 level.

	All peat sites	Peat sites with pH > 6
Total Abundance	0.234*	0.270*
Taxon Richness	0.316**	0.134
Ephemeropteran Abundance	0.253*	0.312*
Ephemeropteran Richness	0.330**	0.157
Plecopteran Abundance	0.218*	0.312*
Plecopteran Richness	0.320**	0.387**
Trichopteran Abundance	0.062	0.111
Trichopteran Richness	0.271**	0.102
EPT Abundance	0.274**	0.330*
EPT Richness	0.382**	0.241
BMWP Score	0.382**	0.208
ASPT	0.401**	0.307*

Simpson-Index	0.013	0.204
Shannon Weiner-Index	0.074	0.147

Another tool used to assess the impacts of eutrophication on stream macroinvertebrates is the Small Stream Risk Score (SSRS). The SSRS values for each soil type are given in Fig. 37. It should be noted that not every forest category had enough sites for each soil type and are thus not presented here.

Fig.37 The SSRS values for (a) Well Drained Mineral soils, (b) Podzolic Lithosolic soils and (c) Peat soils.



With increased percentage forestry within a catchment there is a greater risk of failing the SSRS, particularly on peat soils where 60-63% of sites were classified ‘At risk’. There were no sites, for any soil type, that were classified ‘At risk’ where there was < 5% forestry (i.e. control site) in the catchment. For well drained mineral soils, levels of forestry up to 25% illustrated no streams that were ‘At risk’.

3.5 FISH

A total of 68 sites were surveyed during this study. Catchments were grouped in non-forested and forested pairs (n = 34). Pairs were chosen to have similar soils and geological types and analysis was conducted on the physical characteristics (riffle, glide, pool, depth, width, wetted area, conductivity and effort) to ensure that all site pairings were comparable. In each case there were no significant differences (Wilcoxon Ranked Sign Test; $P > 0.05$) in physical features found between the control and forested sites making all pairings comparable for further analysis (Fig. 38 and Table 13).

Fig.38 Within stream habitat differences between control and forest sites

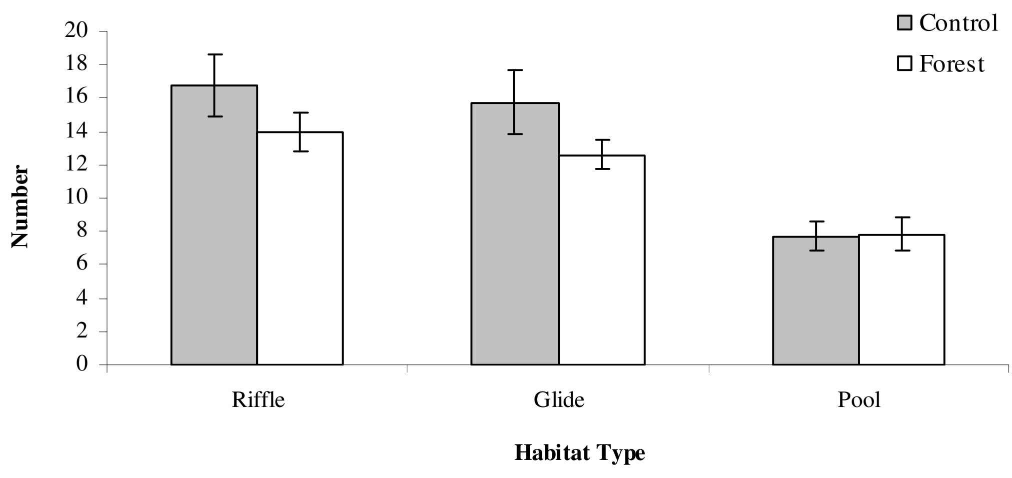


Table 13. Results of the physical characteristics of the paired fishing sites

Physical Parameter	*P*-value
Riffle	0.172
Glide	0.12
Pool	0.946
Depth	0.106
Width	0.945
Wetted Area	0.958
Conductivity	0.135
Effort (m^2 per min)	0.096

The overall total salmonid abundance was significantly greater in the control sites compared to the forest sites (z = 2.052; $P < 0.05$) as shown in Table 14. This relationship was also seen for total salmonid young-of-year abundance, but there was no significant difference between paired sites in relation to adult salmonid abundance (Table 14). There was less of a significant trend between forested and control sites in relation to the trout parameters, with only the trout young-of-year showing a significant increase in abundance in the control sites compared to the paired forest sites (z = 2.003; $P < 0.05$). It was the salmon that expressed the greatest differences between control and forest sites. For each parameter related to salmon (biomass, total abundance, adult abundance and young-of-year abundance) there were significantly greater numbers found in the control sites compared to forest sites, with salmon biomass, salmon adult abundance and young-of-year abundance showing particularly significant declines in the forest sites (Table 14). An observation of high importance, and one that needs further research, is the absence of salmon at many of the forest sites, in particular where salmon are found in the paired control site. In all cases, whether significance was detected or not, there were fewer individuals of trout and salmon at forested sites.

Table 14. Comparison of fish abundance and biomass between paired forest and control fishing sites. * denotes significance at < 0.05 level; ** denotes significance at < 0.01 level.

	z-value
Total salmonid abundance	2.052*
Total adult abundance	1.514
Total young-of-year abundance	2.300*
Trout biomass	1.522
Total trout abundance	1.539
Trout adult abundance	1.317
Trout young-of-year abundance	2.003*
Salmon biomass	2.688**
Total salmon abundance	2.689**
Salmon adult abundance	2.051*
Salmon young-of-year abundance	2.629**

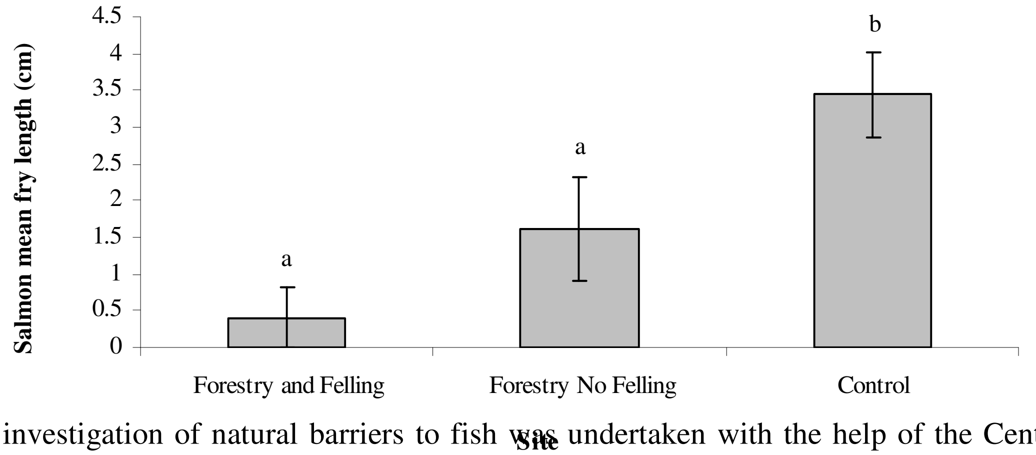
When investigating the sites divided by soil type (peat and non-peat catchments) the percentage occurrence of salmonids is quite striking (Table 15). For instance, salmon fry were found in 39% of control streams on peat catchments, but only in 6% of forest sites. On non-peat catchments the percentage occurrence of salmon fry increased in the forest sites (25% of sites), but relative to the control sites (50%) the percentage was still low. While the percentage occurrence of trout tended to be lower in forest sites than in control sites, the differences were markedly reduced compared to that of the salmon.

Table 15. Percentage occurrence of salmonids in streams with forested and non-forested (control) land use, for peat and non-peat catchments.

Catchment type	Land uses	Trout 1+	Trout 0+	Salmon 1+	Salmon Fry
Peat catchments	Control (n = 18)	83	83	39	39
	Forestry (n = 18)	83	67	11	6
Non-peat catchment	Control (n = 16)	100	94	38	50
	Forestry (n = 16)	75	75	19	25

An examination of trout fry length yielded no significant differences between non-forested and forested sites across the total dataset ($z = 1.823$, $P > 0.05$). However, salmon fry length did show a significant relationship ($z = 3.340$; $P = 0.001$), with salmon fry length greater in control sites than in forest sites. When the sites were further divided into forestry, with or without felling, there was a significant difference in salmon fry length between the control sites and both forestry with felling and forestry without felling (Fig. 39). Even though there was no significant difference between the two forestry categories, the forestry and felling category had reduced mean fry length compared to the forestry and no felling category. As stated earlier, the impact of combined high percentage forestry and percentage felling would appear to be detrimental to salmon growth.

Fig.39 Salmon mean fry length (cm) for forestry and felling, forestry and no felling, and the control sites. Treatments with different letters above standard error bars are significantly different ($P < 0.05$).



An investigation of natural barriers to fish was undertaken with the help of the Central Fisheries Board to ascertain if these were causing the absence of fish from certain sites. From the results no barriers were evident on these sites. Absences and low abundances could be attributed to stress from acid-sensitive conditions or to possible changes in the trophic status.

These results suggest a detrimental effect of coniferous forests on salmonid populations. The characteristic small, upland, gravel streams are potential nursery areas for salmon and trout and the low abundances and absence of fry (mostly of salmon) are particularly indicative of impact on these nursery streams.

4. Discussion and Conclusions

This study has aimed to assess the risks of forestry-derived eutrophication and sedimentation to running waters and to identify and quantify the factors that affect these risks. Several studies have stated that to extrapolate data from one or two catchment studies would have to be done with extreme caution (Cummins & Farrell, 2003; Machava *et al.*, 2007). This present study has adopted a wider approach in that it has covered a large number of catchments encompassing the important parameters of study (e.g. soil type, percentage catchment felling, percentage forest cover, geographical location, slope, etc.) to help identify potential risks of forestry and associated forest activities on the eutrophication and sedimentation of streams. By targeting sites that had a high percentage of forestry within the catchment meant that the potential impacts sampled could be more confidently attributed to forestry and associated forest activities rather than to other landuse practices.

Forests and associated forest activities have been identified as potentially important diffuse sources of Phosphorus (Nisbet, 2001). Fertilisation and harvesting have been found to be the two most important forest operations that could significantly elevate the P concentration in receiving waters if appropriate management is not implemented (Cummins and Farrell, 2003a).

It is clear from our data that in reference to SRP, out of 214 sites, only 8 sites (equivalent to 3.7%) failed 'good' water quality status in relation to The Draft European Communities Environmental Objectives (Surface Waters) Regulations 2008. Of these 8 sites, four in particular had extremely high SRP values. These values were calculated means from three water sampling occasions almost spanning a year from April/May 2007 to Mar/April 2008. Therefore the SRP at these sites was elevated on all three sampling occasions and was not a "once-off" phenomenon. These sites were associated with high percentage forestry (> 90%) on peat soils combined with high percentage felling (30-40%) of the catchment within the last 5 years. Indeed, almost one fifth of all streams on peat soils with felling activities failed 'good' status levels for SRP. The majority of these sites (87.5%) also had a stream width of < 2m.

Results from single catchment studies, in relation to P, within Ireland have produced contrasting results. Cummins and Farrell (2003) found significant losses of P from blanket peatlands in Connemara, Co. Galway attributed to the decomposition of the forest residues and the application of rock phosphate, at rates greater than recommended, on ground that lacked vegetative cover immediately after forest re-establishment. Under current Forest Service guidelines foliar analysis should detect if a reforested site needs fertilisation or not, and if the guidelines had been adhered to the levels of P recorded could have potentially been reduced. In contrast, the Penrich project, whose main aim was to "study the influence of forestry and forest operations on water quality, specifically in the context of eutrophication of surface waters", found no evidence of any negative influence of forest, or forest operations, at two study catchments, one in Co. Wicklow and one in Co. Mayo (Machava *et al.*, 2007). At the Ballinagee site, a blanket peatland catchment, little or no medium/long-term impact on water quality was detected in relation to forestry and associated operations. However, compared to the present study where significant levels of P were detected in sites with greater than 90% forest, the Ballinagee site had approximately 50% of its catchment forested and that the forest operations were confined, at any one time, to a fraction of that area (Machava *et al.*, 2007).

Rodgers *et al.* (2008) found significantly elevated P concentrations after clearfelling and harvesting operations in the Burrishoole catchment, Co. Mayo, a blanket peat catchment. This study utilised automatic water samplers to ascertain P levels during flood events and also during base-flow conditions.

Of the 214 sites sampled, 12 sites (equivalent to 5.6%) failed 'good' water quality status in relation to The Draft European Communities Environmental Objectives (Surface Waters) Regulations 2008 in relation to Ammonia. Again, these values were calculated means from three sampling occasions and therefore there was a continuous high level of Ammonia present within these streams. Sites were generally associated with high percentage forestry (> 75%) on peat soils combined with high percentage felling (30-40%) of the catchment within the last 5 years. Almost ¼ of streams on peat soils with felling activities failed 'good' status levels for Total Ammonia. The majority of sites (90.5%) also had a stream width of < 2m.

Cummins and Farrell (2003b) recorded elevated DTOC levels after felling at a blanket peat catchment in western Ireland. Our data would support this where a significant correlation was identified between sites with > 50% forestry, combined with felling, and DTOC on peat soils. It was the combination of these two factors, high % forestry and % felling, that was important as there was no correlation between sites that had > 50% forestry combined with no felling.

As this study has identified, the location of the actual sampling point is of critical importance. Impacts can be detected in small streams (≤ 1m wide), but when sampled further downstream, where the stream widens and the volume of water is greater, these impacts can be diluted to such an extent that impacts are not detected. This was evident in two areas, Ballycorban and Corra Rivers in Co. Clare, which had both very high values of mean SRP and Ammonia in upstream sites of the main river (≤ 1m stream width) or small tributaries, but which had reached 'good' or 'high' status at downstream sampled sites (≥ 2m stream width). In the Ballycorban River, where the highest levels of SRP were recorded in this study, SRP values could reach 'good' water quality status in a relatively short distance from where 'failed' values were registered (< 500m). What needs to be determined is what occurs within this short distance in relation to SRP uptake/dilution. It would appear imperitive therefore to sample at various locations within a single catchment to achieve a better understanding of the hydrochemistry and further studies are needed to determine the mechanisms of the uptake of SRP and Ammonia in forest streams.

Higher levels of bedload sediment were recorded from peat catchments that were subject to felling activities and had at least ≥ 25% forestry within the catchment. Further studies are needed to determine the distribution of fine sediment downstream of felling activities. This would involve sampling at various locations downstream of the felling over a large distance to attempt to assess the settlement patterns of the fine sediment. The present data found no significant difference in suspended solids between any parameter, and the actual values recorded were low. This was as a result of the sampling method (grab samples) as elevated suspended solids occur on the rise of a flood for a short period of time. To have sampled this exact moment by grab samples would be impossible. Therefore continuous monitoring stations, triggered by rising water levels, would be required to obtain the impacts of suspended solids within these catchments. In the most recent study in Ireland that recorded suspended solids, Rodgers *et al.* (2008) indicated little soil loss from the clearfelled blanket peat area of their study (attributed to no harvesting during wet weather), and the suspended solid concentrations returned to their pre-clearfelling values within a year of the completion of the harvesting activities. Rodgers *et al.* (2008) summised that if blanket peat forest areas are clearfelled in accordance with the Forest Service guidelines, there would be little loss of soil from the felled catchment. As stated above, further studies are needed to assess the bedload sediment levels in these streams because at present there is no official method of collection/sampling or limit/critical value for fine bedload sediment which is of critical importance to many freshwater species, in particular salmonids and the freshwater pearl mussel (*Margaritifera margaritifera*).

Total Organic Nitrogen was higher on mineral soils and negatively correlated with percentage forestry. As expected, as little N is applied to Irish forestry, the values for TON were very low and way below critical values.

There was clear impacts of forestry on benthic macroinvertebrate communities. The SSRS shows greater risk of failing with increasing percentage of forestry cover. When sites were investigated that had pH values greater than 6 there was still significant correlations present between other chemical parameters (such as SRP, Ammonia, and DTOC) and the macroinvertebrate metrics. The most significant relationship was with Plecopteran richness, where the richness decreased with increased levels of SRP, TP, Ammonia and DTOC. Plecopterans are known to be sensitive to eutrophication and this impact would appear to be the case in this study on peat sites with high forestry and presence of felling activity. It has been evident from this study that the chemical parameters of SRP, TP, Ammonia and DTOC are always elevated and closely linked on peat catchments in areas of high percentage forestry combined with felling activity. Further work is needed to clarify what the impacts of these individual elements are and how closely linked they are to one another. For example, further work is needed to detect if the high levels of DTOC are driving the pH values down in felled catchments. Rodgers *et al.* (2008) study indicated that there was no significant change in the macroinvertebrate assemblages following clearfelling. However, it was noted that that the baseline assemblages were fairly depauperate to begin with, comprising only small abundances of acid-tolerant species. The plecopteran species (very sensitive to eutrophication) were unaffected by the clearfelling operations. The depauperate assemblages were concluded to be due to two factors – the acidification effects of the forestry over the last three decades, or the temporal nature of water flow given the size of the stream (Rodgers *et al.*, 2008).

From the fish study, salmonid abundance was significantly greater in the control sites compared to the forest sites. The salmon data gave the most prominent results between control and forest sites. For each parameter related to salmon (biomass, total abundance, adult abundance and young-of-year abundance) there were significantly greater numbers found in the control sites compared to forest sites. An observation of high importance, and one that needs further research, is the absence of salmon at many of the forest sites, in particular where salmon are found in the paired control site.

Again, peat sites on forestry showed the most striking results. Salmon fry were found in 39% of control streams on peat catchments, but only in 6% of forest sites. On non-peat catchments the percentage occurrence of salmon fry was 25%, with 50% found in control sites. While the percentage occurrence of trout tended to be lower in forest sites than in control sites, the differences were markedly reduced compared to that of the salmon. Salmon fry length was significantly greater in control sites than in forest sites. When the sites were further divided into forestry, with or without felling, there was a significant difference in salmon fry length between the control sites and both forestry with felling and forestry without felling. The impact of combined high percentage forestry and percentage felling would appear to be detrimental to salmon growth. After no natural barriers were observed for each site, absence and low abundances could potentially be attributed to stress from acid-sensitive conditions or to possible changes in the trophic status. The latter could be true in paired sites that both have salmon present, with the fry in the forest sites having reduced length, and hence growth. Another possible explanation of fish impairment could be due to Manganese (Mn) levels in the streams. Elevated concentrations of Mn are toxic to fish (Nyberg *et al.*, 1995) and can impair drinking water quality. In Scotland, Heal (2001) showed that conifer afforestation was associated with enhanced Mn in runoff. Mn was leached from conifer foliage and litter, and mature conifers enhanced the loss of Mn from acidified catchment soils. Elevated Mn concentrations in runoff were also observed following harvesting (Heal, 2001). Unfortunately, Mn was not sampled as part of this study but should be included in future studies.

These results suggest a detrimental effect of coniferous forests on salmonid populations. The characteristic small, upland, gravel streams are potential nursery areas for salmon and trout and the low abundances and absence of fry are particularly indicative of impact on these nursery streams.

The results of this project emphasise the complexity of forest-site interactions. This extensive catchment scale network of monitoring sites was designed to capture the range of variation in hydrochemistry and biological loss and to refine the identification of vulnerable site types. The indications from these results are that there is a potential risk of nutrient and bedload sediment in catchments with high percentage forestry within the catchment combined with high percentage of felling on peat soils. Further studies should include more intensive studies in order to elucidate ecosystem processes controlling the retention and release of nutrients and sediment in forested catchments.

5. REFERENCES

Arnason, F., Antonsson, T. and Einarsson, S.M. 2005. Evaluation of single-pass electric fishing to detect changes in population size of Atlantic salmon (*Salmo salar* L.) juveniles. *Icelandic Agricultural Sciences*, **18**: 67–73.

Bertrand, K.N., Gido, K.B. and Guy, C.S. 2006. An evaluation of single-pass versus multiple-pass backpack electrofishing to estimate trends in species abundance and richness in prairie streams. *Transactions of the Kansas Academy of Science*, **109**: 131–138.

Cummins, T. and Farrell, E.P. 2003a. Biogeochemical impacts of clearfelling and reforestation on blanket peatland streams I. phosphorus. *Forest Ecology and Management*, **180**: 545–555.

Cummins, T. and Farrell, E.P. 2003b. Biogeochemical impacts of clearfelling and reforestation on blanket peatland streams II. major ions and dissolved organic carbon. *Forest Ecology and Management*, **180**: 557–570.

Duggan, S.N., O'Halloran J., Giller, P.S. and Johnson, M.J. 2000. The effects of felling and restocking on stream ecosystems: a single catchment study. Unpublished report to Enterprise Ireland, Dublin, Ireland.

Economou, A.N., Schmutz, S., Melchior, A. and Heidvogel, D. 2002. Development, evaluation & implementation of a standardised fish-based assessment method for the ecological status of European rivers: A contribution to the Water Framework Directive (FAME): Defining reference conditions (D3), Final Report. FAME, Austria.

Edwards, M.R., Combs, D.L., Cook, S.B. and Allen, M. 2003. Comparison of single-pass electrofishing to depletion sampling for surveying fish assemblages in small warmwater streams. *Journal of Freshwater Ecology*, **18**: 625–634.

European Union. 2003. WFD Pressures and Impacts Assessment Methodology - Guidance on pressures and impacts methodology. Paper published by the Working Group on Groundwater and Working Group on Characterisation and Reporting, pp 45.

Gallagher, M.B., Johnson, M., O'Gorman, K., O'Halloran, J., Giller, P.S. and Clenaghan, C. 2000. The impact of clearfelling operations on physiochemical parameters of aquatic ecosystems in south west Ireland. *Journal Verhandlungen Internationale Vereinigung für Limnologie*, **27**: 1108–1115.

Giller, P.S., O'Halloran, J., Kiely, G., Evans, J., Clenaghan, C., Hernan, R., Roche, N. and Morris, P. 1997. *A study of the effects of stream hydrology and water chemistry in forested catchments on fish and macroinvertebrates*. AQUAFOR Report 2. An evaluation of the effects of forestry on surface water quality and ecology in Munster. COFORD, Dublin, Ireland.

Hayes, J.W. and Baird, D.B. 1994. Estimating relative abundance of juvenile brown trout in rivers by underwater census and electrofishing. *New Zealand Journal of Marine and Freshwater Research*, **28**: 243–253.

Heal, K.V. 2001. Manganese and land-use in upland catchments in Scotland. *The Science of the Total Environment*, **265**: 169–179.

ITGA (Irish Timber Growers Association). 2006. Forestry and Timber Yearbook, Dublin.

ITGA (Irish Timber Growers Association). 2007. Forestry and Timber Yearbook, Dublin.

Johnson, M.J., O'Gorman, K., Gallagher, M., Clenaghan, C., Giller, P. S. and O'Halloran, J. 2000. The identification and amelioration of the impacts of clearfelling operations on macroinvertebrates and salmonids in Ireland. *Journal Verhandlungen Internationale Vereinigung für Limnologie*, **27**: 1565–1570.

Johnson, M.J., Giller, P.S., O'Halloran, J., O'Gorman, K., Gallagher, M.B. 2005. A novel approach to assess the impact of landuse activity on chemical and biological parameters in river catchments. *Freshwater Biology*, **50**: 1273–1289.

Jones, M.L. and Stockwell, J.D. 1995. A rapid assessment procedure for the enumeration of salmonine populations in streams. *North American Journal of Fisheries Management*, **15**: 551–562.

Kelly, M.G., Yallop, M.L., Hirst, H. and Bennion, H. 2005. Sample collection. Version 2.1. 11pp. Unpublished DARES/DALES protocol. http://craticula.ncl.ac.uk/dares/methods.htm

Kruse, C.G., Hubert, W.A. and Rahel, F.J. 1998. Single-pass electrofishing predicts trout abundance in mountain streams with sparse habitat. *North American Journal of Fisheries Management*, **18**: 940–946.

Machava, J., McCabe, O., O'Dea, P., Cabral, R. and Farrell, E.P. 2007. Forestry operations and eutrophication - PEnrich. Environmental Protection Agency and COFORD, 82pp.

Mitro, M.G. and Zale, A.V. 2000. Predicting fish abundance using single-pass removal sampling. *Canadian Journal of Fisheries and Aquatic Sciences*, **57**: 951–961.

Nisbet, T.R. 2001. The role of forest management in controlling diffuse pollution in UK forestry. *Forest Ecology and Management*, **143**: 215–226.

Nyberg, P., Andersson, P., Degerman, E., Borg, H. and Olofsson, E. 1995. Labile inorganic manganese - an overlooked reason for fish mortality in acidified streams? *Water, Air, & Soil Pollution*, **85**: 333–340.

Resh, V.H., Brown, A.V., Covich, A.P., Gurtz, M.E., Li, H.W., Minshall, G.W., Reice, S.R., Sheldon, A.L., Wallace, J.B. and Wissmar, R.C. 1988. The role of disturbance in stream ecology. *Journal of the North American Benthological Society*, **7**: 433–455.

Rodgers, M., Xiao, L., Müller, M., O'Connor, M., de Eyto, E., Poole, R., Robinson, M. and Healy, M. 2008. Quantification of erosion and phosphorus release from a peat soil forest catchment. STRIVE Report, Environmental Protection Agency and COFORD, 57pp.