FORESTRY & SURFACE WATER ACIDIFICATION

(FORWATER)

2008

MARY KELLY-QUINN[I], ROBERT CRUIKSHANKS[I], JAMES JOHNSON[I] ,

RONAN MATSON[I], JAN-ROBERT BAARS[I] & MICHAEL BRUEN[2],

[1] –School of Biology and Environmental Science

2 - School Of Architecture, Landscape & Civil Engineering

TABLE OF CONTENTS

SUMMARY

This project set-out to investigate the presence and extent of acidification associated with coniferous forestry in Ireland and to assess the risk of impact with respect to different geological settings. In the selection of forested sites it was aimed to represent a combination of the risk factors in terms of catchment forest cover and acid-sensitive geology that were perceived to have the greatest potential for acidification. This study was designed to allow comparisons of the hydrochemical and ecological quality of two groups of sites, forested and non-forested, control or reference sites in two geological settings (igneous/metamorphic and sedimentary) with four dominant soil types (peat, podzolic/lithosolic, poorly drained gleys and well drained mineral soil). The 239 control and forested sites were categorised to reflect a gradient in catchment forest cover (control (<5% forest cover) – 73 sites, and three coniferous forestry bands (5-25, 25-50 and >50% coniferous cover with 27, 41 and 98 sites, respectively). Water samples were collected from all sites on three separate dates and covered a range of flow conditions. Macroinvertebrate samples were collected in spring 2007 and electrofishing was carried out at 19 paired sites in summer 2007.

The pH results analyses suggested that most of the streams were episodically acidic with a small group more likely to be circum-neutral. Overall, the pH results indicated increased acidity at some sites associated with forestry on peat and podzolic/lithosoilic soils on both igneous/metamorphic and sedimentary geology and to some extent on poorly drained gleys. Furthermore, the frequency of low pH readings was substantially higher among some groups of forested sites than the control sites. Certainly the minimum pH for both peat and podzolic/lithosolic sites on igneous/metamorphic geology began to fall below the lower limit of the control sites when forest cover exceeded values in the region of 25-30%. The same applied to peat sites draining sedimentary geology. Sites on podzolic/lithosolic soils on sedimentary geology did not have minimum pH values below the lower limit of the control sites until forest cover exceeded 60%. A similar threshold might be applied to sites on poorly drained gleys but the level of replication was too low for this decision. The presence of forestry tended to depress site pH and alkalinity. Calculations suggested that dilution makes a variable contribution to loss of alkalinity and in many cases the forested sites showed a slightly higher % value. Anion titration was detected in all events examined. The principal contributors were organic acids and excess sulphate, particularly in the east.

Overall, the biological data largely mirrored the trends for the acidity variables. Several macroinvertebrate metrics (taxon richness, ephemeropteran richness, abundance of baetids, EPT richness, diversity indices), which showed a strong relationship with pH, were also shown to vary significantly across the forest cover bands or to correlate with % forest cover. The analyses on the individual metrics highlighted similar % forest thresholds for risk of impact as described for the hydrochemistry. When a selection of non-correlated metrics were combined it was clear that a large proportion of sites in the >50% cover band, and a smaller number of the 25-50% band, had some degree of impairment. However, not all forested sites were impaired and further research must target these sites to better understand the mechanisms governing responses to acid impact. Finally, the length of stream impacted by forest-mediated acidification is likely to vary depending on geological and other catchment characteristics.

The fish analyses was limited to 19 paired sites with similar habitat but did highlight significant differences in fish catch and density between the control and forested groups. This difference was mainly attributed to low numbers of fry (salmon and trout) in the forested streams.

In terms of identified knowledge gaps we need to determine the acidification risk associated with each of the key forestry practices from site preparation to felling. More detailed spatial and temporal analyses of the chemical characteristics of waters draining sedimentary geology is required for more precise mapping of acid sensitivity. The contribution of organic acids to the acid pulses in control and forested sites on both igneous and sedimentary geology and the process contributing to their release needs to be further explored. The influence of flowpath and drainage on buffering potential and its interaction with acid inputs also requires further research. Finally, with respect to the biota it is critical to understand seasonal and longitudinal changes in the community in response to acid inputs together with the functional significance of any impairment.

1 INTRODUCTION

The Water Framework Directive (WFD) (2000/60/EC), which came into force in December 2000, requires EU member states to implement the necessary measures to prevent deterioration in the status of all bodies of surface and groundwater and where necessary, restore all waters to good ecological and chemical status by 2015. As part of the characterization process, the first risk assessment of the anthropogenic pressures on water resources was undertaken to identify the pressures present in each river basin and the threat they pose to the chemical and ecological status of waterbodies. The resulting "National Characterisation Report for Ireland" (Anon, 2005) identified forestry (SD4) as one of the land-use activities posing a potential risk in terms of diffuse pollution. Among the pressures highlighted as arising from forestry were increased acidification from plantations in acid-sensitive catchments (SD4a), sedimentation from clear fell, harvesting, new plantations, road construction and erosion on steep catchments (SD4b) and eutrophication from fertilisation on steep catchments and forest harvesting on peat soils (SD4c).

The Western River Basin District was given, inter alia, the task of further characterisation of the risks from plantation forests and forest related activities on surface waters and to subsequently developing a programme of measures to address any significant risks. The present project was commissioned to address some of the knowledge gaps pertaining to the acidification risk. The keys questions addressed by this research were:

1. Is there any evidence for a forest effect on hydrochemistry and biology?
2. What is the impact on the aquatic biota?
3. What conditions pose the greatest risks

The research sought to identify patterns that indicate risk and inform a 'precautionary approach' through the first programme of measures.

The work was prefaced with a literature review which clarified the state of knowledge on forest-mediated acidification. The literature review (Johnson *et al.* unpublished) followed the Source–Pathway–Receptor model of the risk assessment and the key findings are summarised here.

The Source of the Pressure

The role of forestry in the acidification of surface waters is primarily attributed to the interception of atmospheric pollutants coupled with the inability, in sensitive areas, of the substrate soils and geology to buffer the acidity (Jenkins *et al*., 1990; Ormerod *et al.*, 1991). The magnitude of the pressure exerted by the scavenging effect of forests depends primarily on (a) the pollutant load and (b) the percentage of catchment forest cover. The extent of the pressure is likely to vary with tree species with some species, such as Sitka spruce (*P. sitchensis*), being more effective scavengers of pollutants than others. The pollutant load at a site is further influenced by emission levels, climatic conditions such as the frequency and magnitude of rainfall events, the amount of annual rainfall, prevailing wind direction and air mass circulation patterns as well as site characteristics such as elevation and aspect, tree species, stand age and structure. Few studies have attempted to clarify the relationship between the extent of catchment afforestation and surface water chemistry across a range of catchment characteristics but there is general agreement in the literature that an increase in forest cover has the potential to increase the acidification pressure. However, as yet there is no guidance on the threshold above which adverse impacts are apparent on stream chemistry or biology in acid-sensitive areas.

Sea salt driven acid pulses can occur in coastal catchments. Forests capture marine ions as wet or dry deposition (Farrell, 1995; Harriman, Anderson and Miller, 1995). During storm events, high inputs of Na^+ can displace other cations. The associated chloride ion is largely conservative and most of it is quickly leached. As it passes through acidic soil to associated drainage water it can be accompanied by H^+ and Al^{3+}. Other processes with potential to contribute to acidification include (a) uptake of base cations by trees and subsequent removal by harvesting, (b) oxidation and mineralization of organic matter producing organic acids and (c) alterations to site hydrology resulting in the reduced residence time of water and (d) the short-term release of nitrate following the large-scale felling of forest sites in acid-sensitive catchments. Certainly increased drying of soil and altered drainage increases the oxidation of organic matter and generates carboxylate anions, increases mineralization of organic matter and consequently potentially increases in losses of ammonium and/or nitrate as well as sulphate to drainage waters (Hornung *et al.*, 1995). However, the contribution of these to acidification processes in acid-sensitive areas has not been adequately assessed.

Pathway Susceptibility

The pathway susceptibility is primarily controled by catchment geology. The available literature indicates that water bodies susceptible to acidification are located in catchments dominated by slow weathering bedrock such as granite and quartzite with shallow carbonate free soils as well as areas of sandy, siliceous soils and highly weathered old leached soils (Hornung *et al.,* 1990). In Ireland, granitic areas in the west and northwest (Allott et al., 1990; Bowman, 1991; Allott *et al.,* 1997) as well as the east have been shown to be acid-sensitive (Kelly-Quinn *et al.,* 1996a; Kelly-Quinn, Tierney & Bracken, 1997). The potential for acidification on Old Red Sandstone is likely to be less but this is not fully established and is addressed in the current project.

Catchment size and hydrology/drainage also have a bearing on the susceptibility of running waters to acidification. High drainage rates and steep topography of small upland catchments reduces the contact time for runoff with bedrock and soil and consequently the time for soils to impart buffering capacity to the runoff water (Waters and Jenkins, 1992). As a result, waters draining smaller catchments may be more acidic and have higher concentrations of inorganic monomeric aluminium reflecting a higher proportion of runoff from the acidic mineral soils in the catchment. In larger catchments, the overall residence time of water in soil system is longer and it is therefore likely to be more effectively neutralized.

The Sensitivity of the Receptor

The sensitivity of the receptor shows as an increases in acidity and in many cases elevated aluminium concentrations (Ormerod *et al*., 1991). The response of the biota to forest-mediated acidification has been well documented (examples Clenaghan *et al.,* 1998; Harriman and Morrison, 1982; Stoner, Gee & Wade, 1984; Ormerod *et al.,* 1991, Ormerod & Wade, 1990; Allott *et al.,* 1997; Tierney, Kelly-Quinn & Bracken, 1998). Biological impacts associated with acidification in streams include 1) reductions in or total elimination of fish populations, 2) reductions in taxon richness and elimination of some acid-sensitive macroinvertebrate groups (particularly the Ephemeroptera) and 3) changes in the quality of primary producers (Stoner, Gee & Wade, 1984; Ormerod, Wade and Gee, 1987; Rees and Ribbens, 1995; Tierney, Kelly-Quinn & Bracken, 1998). The studies carried out in Ireland to date have highlighted some impact in areas of Wicklow (Tierney, Kelly-Quinn and Bracken, 1998) and Galway and south Mayo (Allott *et al.,* 1997) but no acidification-related impacts of aquatic fauna were detected for sites on Old Red Sandstone in Cork (Giller *et al.,* 1997).

However, the latter authors noted that some macroinvertebrate communities at medium altitude (200-300m) with medium to high levels of forestry (25 to> 50%) seemed to resemble communities at higher altitudes (>300m) than sites with low levels of forest cover at a medium altitude. The present study set out to undertake more extensive sampling of Old Red Sandstone to further assess the potential for acidification impacts. A further issue arising from the AQUAFOR and indeed more recent WATERAC projects was that the occurrence of detectable impact in rivers (as evidenced by loss of macroinvertebrate taxa and salmonids) appeared to be rather patchy. Therefore, the current study set out to target good numbers of sites with a combination of perceived high risk factors, so that a better measure of the extent of impact could be achieved.

2. SITE SELECTION AND METHODOLOGY

2.1 SITE SELECTION

This study was initially designed to allow comparisons of the hydrochemical and ecological quality of two groups of sites, forested and non-forested, control or reference sites. This approach was adopted following consultation with the Forest and Water National Steering Committee members (EPA, Cóillte, Irish Forest Service, Marine Institute, Central Fisheries Board and National Parks and Wildlife) and is the current approach required by the WFD and widely applied in freshwater studies. The forested sites were to represent a combination of the risk factors perceived to have the greatest potential to facilitate acidification impacts. Percentage forest cover in the catchment and geographical location were considered to influence the magnitude of the acidification pressure. Factors influencing pathway susceptibility were geology and soil type. A total of 239 sites (Appendix A), both control and forested, were therefore selected to provide wide geographical coverage within acid-sensitive geologies (igneous/metamorphic geology and Old Red Sandstone) and to encompass combinations of geology and soil type (peaty and mineral). Four categories of soil were targeted, well drained mineral, poorly drained gleys, podzolic-lithosols and peats [categories followed consultation with Cóillte, Irish Forest Service and WRBD]. The forested sites were selected to have >25% catchment forest cover, the majority of which was closed canopy representing mature forests in the forest cycle. Catchment is here defined and applied throughout this study as the drainage basin to the study site, alternatively referred to as site watershed. Control sites were chosen within each of the regions where the forested sites were located. To ensure comparability, where possible control and forested sites were within the same larger river catchment, although adjacent catchments were selected in some instances. However, the geographic spread of the control and forested catchments were similar.

Control sites were initially chosen to represent catchments with no forestry. However, during the course of the study access to updated forest inventory information revealed variable amounts of forest in some of the control catchments. Consequently, all 239 control and forested sites were re-categorised to reflect the gradient in forest cover (control (<5% forest cover) – 73 sites, and three coniferous forestry bands (5-25, 25-50 and >50% coniferous cover) with 27, 41 and 98 sites, respectively). The numbers of sites in each forestry/soil/forest cover category are given in Table 1 and their location is indicated in Fig. 1. Broadleaf forestry represented a small percentage of the total forest cover and was not considered in the

analyses. Therefore, the forest cover values referred to throughout this report represent coniferous plantation. There was no significant land-use pressure in the control catchments apart from some rough grazing, although historical influences could not be ruled out entirely.

Table 1: Numbers of site sampled in each geology/soil category/forest cover category

Geology/Soils	**Forest Cover**				
Igneous/Metamorphic	**<5%**	**5-25%**	**25-50%**	**>50%**	**Totals**
Peat	22	9	6	17	54
Podzolic/Lithosolic	11	4	9	8	32
Sedimentary					
Peat	20	4	11	30	65
Podzolic/Lithosolic	13	2	2	22	39
Gleys	6	3	7	11	27
Well Drained Mineral	2	5	6	9	22
Totals	**74**	**27**	**42**	**97**	**239**

In each region the sites were selected on first to third order streams. A number of additional sites were located further downstream to examine longitudinal changes in hydrochemistry and aquatic biota. Every effort was made to control for slope, elevation and catchment size assessed using channel length and stream complexity as indicated on OSI maps (1:50000) and by restricting the majority of the sites to sub-catchments. Access to the sites was facilitated by Coillte/Irish Forest Service, many of which were remote with the only access by foot.

Each sites was represented by a 50 (macroinvertebrate sampling) to100 (fish sampling) metre stretch. Physical descriptions for each of the sites were derived from GIS and from onsite measurements of stream width (four measurements), depth (four measurements taken in randomly selected pools in the stretch), substrate composition, mesohabitat cover and flow condition (low, elevated and flood). Substrate was estimated as the percentage cover of bedrock, boulder (26-200cm), cobble (6-26cm), gravel (0.4-6cm), sand (0.06-0.2cm) and silt

(<0.06cm). Mesohabitats were assigned to three categories, including riffles, glides and pools.

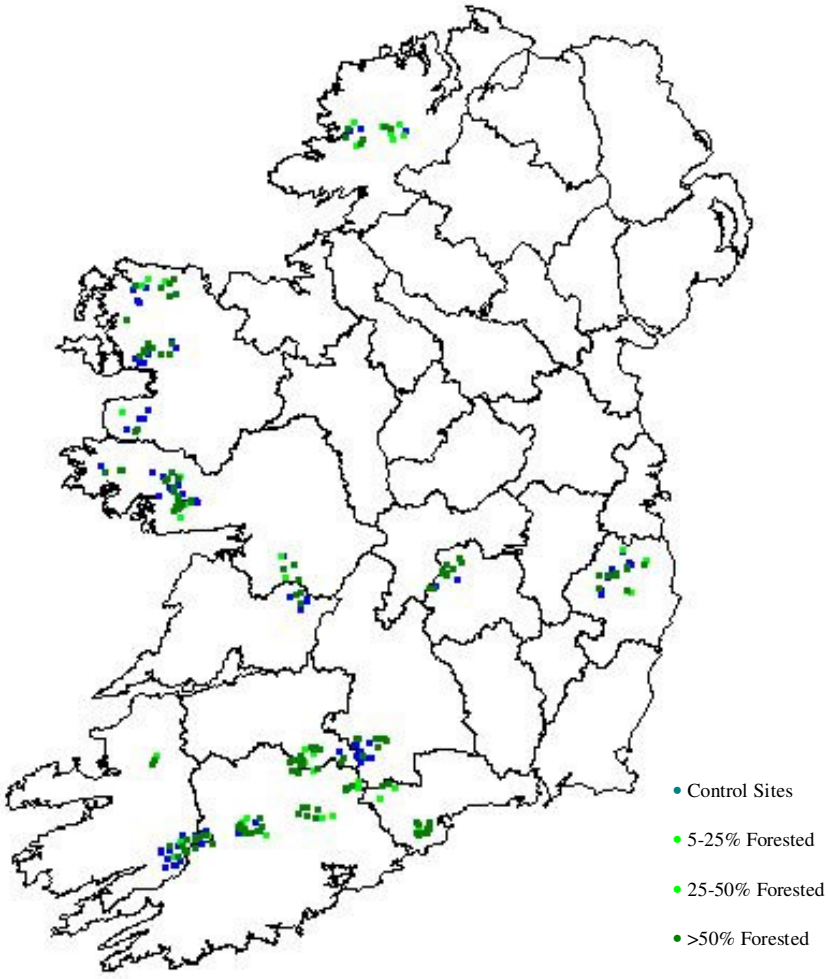


Fig. 1: Distribution of hydrochemical and macroinvertebrate sampling sites (colour coded for forest-cover categories).

2.2 HYDROCHEMISTRY

Water samples were collected from all sites in clean one-litre and 250ml polypropylene bottles (pH). Readings of conductivity (µS/cm), and oxygen (% saturation and mg/l O_2) were taken on site using automatic field meters and probes. All samples were sent to the Aquatic Services Unit at the Environmental Research Institute (ERI, UCC) for analysis within 24-hours of collection. A full suite of hydrochemical analyses were carried out using the methodologies outlined in Table 2. Three rounds of water sampling were undertaken, 2/5/07-6/6/07, 13/11/07-21/12/07 and 18/3/08-16/4/08. The aim was to sample each site at variable flow conditions, from low flow to flood. It was however not possible to obtain flood samples for all sites. Furthermore, it was often difficult to ascertain the stage in the hydrograph represented on any one date and therefore flow condition was simply recorded as low, elevated or flood. Additional samples were taken upstream and downstream of the forestry block on selected source streams. Samples were collected at comparable locations on control streams at similar distances from the source (as indicated on OSI maps).

Sources of acidity and those potentially responsible for any observed differences between forested and control sites were evaluated by examination of the results for sulphate, nitrate, chloride and organic carbon. The potential loss of alkalinity during elevated flow due to dilution by precipitation was assessed using the following formula applied by Kowalik *et al.* (2007):

$$\text{Alkalinity Dilution} = \frac{(((\sum BC_{low} - \sum BC_{high})/\sum BC_{low})\ Alk._{low})}{(Alk._{low} - Alk._{high})}$$

BC=Base Cations, Alk.=Alkalinity

All concentrations are entered in units of µeq/l. High percentage values close to 100% indicate that dilution is strongly affecting buffering. Lower values indicate reduced likelihood of dilution and possible titration by an acid anion.

Titration Ratio

Loss of alkalinity due to titration by an acidic anion is evidenced by changes in the following ratio.

$$\text{Titration Ratio} = \text{Alkalinity} / \sum BC$$

This can be confirmed with the results from the titration ratio equation. The proportional contribution of acid anions to any titration processes was calculated as Anion/∑Acid Anions (Kahl *et al.,* 1992). Losses in ANC due to sea salt effects were evaluated from changes in the concentrations of Cl^- and Na^+ between low and high flow as applied by Evans *et al* (2008).

Table 2: Methods applied in the chemical analyses

Parameter	Method	Unit
pH	WTW pH330i pH meter	-
Conductivity	WTW LF330 Conductivity meter @ 25°C	µS/cm @ 25ºC
Alkalinity	Gran Titration	mg/l $CaCO_3$
Total Hardness	ETDA Titration	mg/l $CaCO_3$
Colour	Colorimetric method using platinum/cobalt solution as colour standard	Hazen Units
Dissolved Total Organic Carbon	SHIMADZU TOC-VCPH TOC analyzer	mg/ lDTOC
Soluble Reactive Phosphorus	Automated Molybdate method using Lachat™ Quikchem FIA	mg/l SRP
Total Phosphorus	Manual molybdate method after sample digestion	mg/l TP
Ammonia	Automated salicylate method using Lachat™ Quikchem FIA	mg/l Ammonia
Total Organic Nitrogen	Automated colourimetric method using Lachat™ Quikchem FIA after cadmium reduction	mg/l TON
Nitrate	Subtraction nitrite from TON	mg/l Nitrate
Nitrite	Manual colourimetric method	mg/l Nitrite
Total Monomeric Aluminium	Graphite furnace AAS	µg/l Aluminium
Inorganic Aluminium	Graphite furnace AAS after Amberlite™ Resin fractionation	µg/l Aluminium
Calcium	Automated IC method using Lachat™ Quikchem IC	mg/l Ca^{2+}
Magnesium	Automated IC method using Lachat™ Quikchem IC	mg/l Mg^{2+}
Potassium	Automated IC method using Lachat™ Quikchem IC	mg/l K^+
Sodium	Automated IC method using Lachat™ Quikchem IC	mg/l Na^+
Chloride	Automated IC method using Lachat™ Quikchem IC	mg/l Cl^-
Sulphate	Automated IC method using Lachat™ Quikchem IC	mg/l SO_4^{2+}
Suspended Solids	Gravimetric method after filtering through GF/C filter paper and dried at 104°C	mg/l SS
Silicate	Manual colourimetric method	mg/l Si
Sodium Dominance Index (SDI)	Calculated	%

2.3 MACROINVERTEBRATES

Benthic macroinvertebrate samples were collected over a six week period in 2007 from the beginning of April until the second week in May at the 239 sites. Additional samples were collected on source streams above and below forest blocks and at similar points on comparable streams. A multi-habitat sampling approach was employed involving kick samples of 1-minute duration taken using a standard pond net (mesh – 1mm). The time spent sampling each mesohabitat (riffle, pool glide) was proportional to its percentage representation in the study site (Wright, 1995). Habitats contributing less than 5% of the stable habitat in the reach were not sampled (Barbour *et al.,* 1997). An additional one minute was spent carrying out hand searches for attached invertebrates. Sampling was initiated downstream of the reach and proceeded upstream. To avoid the confounding effects of shading the forested sites were downstream of the forest within open, un-shaded reaches at least 20m downstream of the forest block. Six samples were collected at each site and preserved using 70% alcohol (IMS). These were sorted in the laboratory and the macroinvertebrates were removed and identified to the lowest taxonomic unit possible using FBA keys (Table 3). Identified samples were stored in 70% alcohol (IMS).

Table 3: Level of identification for macroinvertebrate groups.

Taxon	Level of Taxonomic Identification
Plecoptera	Species
Ephemeroptera	Species
Trichoptera	Genus/species
Coleoptera	Genus/species
Chironomidae	Subfamily
Simuliidae	Genus/
Other Diptera	Family/genus/species
Odonata	Genus/species
Hemiptera	Genus/species
Mollusca	Species
Hirudinea	Species
Oligochaeta	Order

2.4 FISH

Streams were selected on a paired catchment basis (one non-forested catchment, one forested catchment) to represent similar physical characteristics including catchment area, elevation and slope. With few exceptions paired streams were selected on the same main channel, in close proximity so that both streams had the same source fish population within the main channel. Fig. 2 illustrates this approach. In total, 38 sites were fished (19 non-forested and 19 forested sites) (Table 4, Fig. 3). Each site was fished using backpack electrofishing equipment (Safari Research 550D backpack model). Site habitat characteristics including numbers of riffles, glides and pools, stream width (four measurements) and pool depth (four measurements) were recorded on site, along with on-site measurements of oxygen and conductivity. The time taken to fish each site was recorded to compute fishing effort so that comparisons could be made between forested and non-forested sites and corrected if necessary for differences in effort.

A single-pass approach was adopted for the electrofishing sampling. Several investigations have evaluated the 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 the first pass and the total population size estimated from three or more passes (Hayes & Baird, 1994; Jones and Stockwell, 1995; Kruse, Hubert & Rahel, 1998; Mitro and Zale, 2000; Arnason, Antonsson & Einarsson, 2005; Bertrand, Gido & Guy, 2006) and it 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, the 100m stretch fished in the present study was more than adequate to satisfied this condition. To avoid the problem of shading sampling in the forested catchments took place outside of the forest, usually immediately downstream (circa 20m). All species encountered were captured and identified. The salmonids were measured (fork length) and weighed. Scales were removed from a representative sample and retained for age analysis. After capture all fish were held in keep-nets to ensure their complete recovery before being returned to the river.

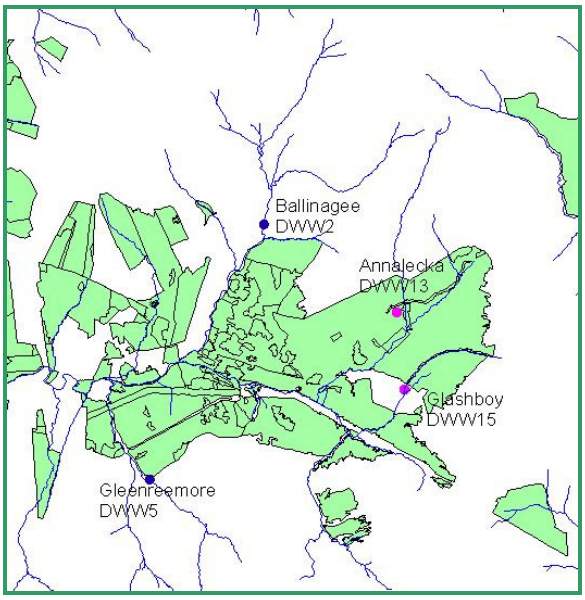


Fig. 2: Illustration of the paired site selection approach in the King's River catchment, Co. Wicklow. Site DWW2 was paired with DWW15 and DWW5 was paired with DWW15.

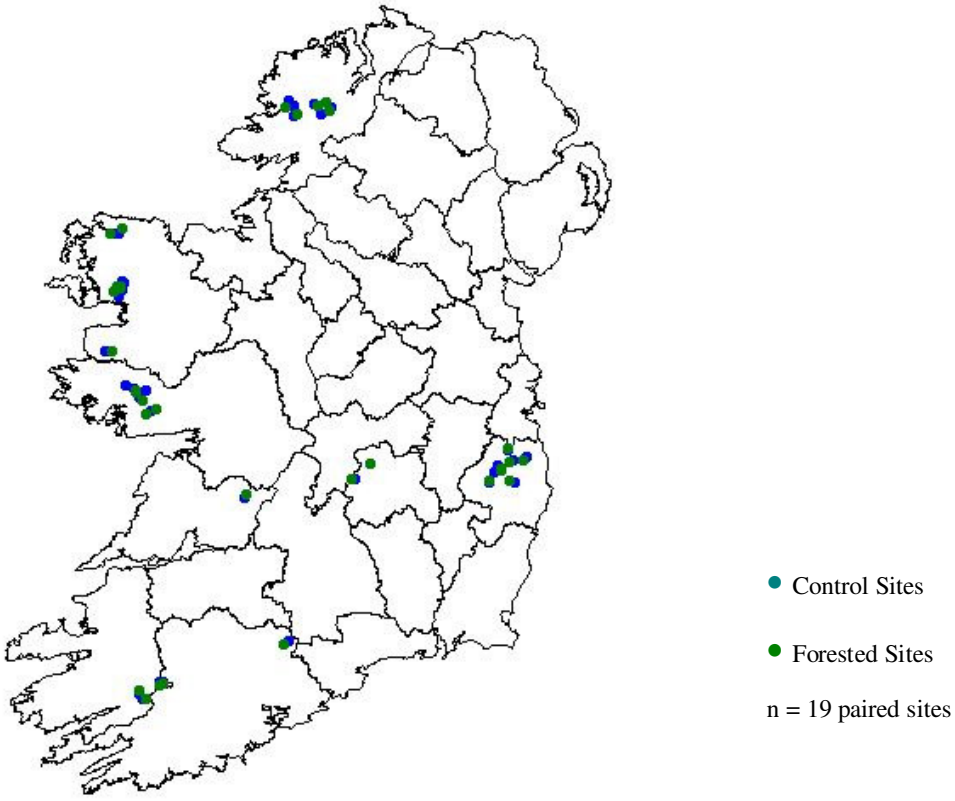


Fig. 3: Location of electrofishing site pairs

Table 4: Location of paired fishing sites

Main System	River		Site Code	Control/Forest	Easting	Northing	Main System	River		Site Code	Control/Forest	Easting	Northing
Kings	Ballinagee	BALLIN1	DWW1	Control	304462	204045	Kings	Annalecka	ANNA1	DWW13	25-50%	306426	202755
Kings	Glencreemore	GLEEN1	DWW5	Control	302788	200283	Kings	Glashaboy	GLASH1	DWW26	25-50%	306535	201611
Kings	Knickeen	KNICK1	DWW6	Control	299726	195072	Oilitigh	Oilitigh	OILI1	DWW17a	25-50%	299286	196067
Cloghoge	Cloghoge	CLOG1	DWW8	Control	312761	207455	Inchavore	Inchavore	INCH1	DWW19	25-50%	311004	206075
Srahmore	Srahmore	SRAH1a	DM11a	Control	096560	305240	Srahmore	Srahmore	SRAH2	DM23	>50%	095227	306980
Srahmore	Glenamong	GAMON4	DM8	Control	093918	304178	Srahmore	Glenamong	GAMON2	DM10	25-50%	092809	303819
Srahmore	Glenamong	GAMON5	DM9	Control	094080	303977	Srahmore	Glenamong	GAMON6	DM9a	25-50%	094019	303524
Owengarve	Callowswallagh	CALLOW1	DM22	Control	093465	298191	Owengarve	Glendahurk	CALLOW1	DM3	25-50%	091103	300931
Glenamoy	Glenamoy	GMOY3	DM19	Control	093801	332628	Glenamoy	Glenamoy	GMOY4	DM20/M8	25-50%	095106	335854
Glenamong	Fiddaunatoreen	FREEN1	DM6	Control	095099	301909	Glenamong	Glenamong	GAMON1	DM7	25-50%	094472	302777
Glenumerra/Glendavock	Glenumerra	GLENU1	DM24	Control	085739	267708	Glenumerra/Glendavock	Glenumerra	GLENU2	DM25	25-50%	089975	267651
Owenree	Owenree	OREE1	DG11	Control	101586	246870	Owenwee	Owenwee	OWEE2	DG22	>50%	103160	245498
Maumwee L. Inflow	Maumwee	MAUM1	DG24	Control	097255	248472	Owenwee	Owenwee	OWEE3	DG23	>50%	103292	245896
Owenriff	Owenriff	ORIFF1a	DG33	Control	105151	242453	Owenriff	Glengawbeg	GBEG1a	DG27	25-50%	106686	240525
Owenboliska	Owenboliska	OLISKA3	DG7	Control	111162	234916	Owenboliska	Owenboliska	OLISKA1	DG13	>50%	114582	235506
Owenboliska	Owenboliska	OLISKA3	DG7	Control	111162	234916	Owenboliska	Owenboliska	OLISKA6	DG15	>50%	108519	232725
Glenleheen	Glenleheen	GHEEN1	DD18	Control	190732	404350	Gweebara	Gweebara	GBAR3	DD11	25-50%	185974	402744
Deele	Deele	DEEL1	DD2	Control	211261	403208	Deele	Deele	DEEL2	DD19	5-25%	208901	405512
Elatagh	Elatagh	ELAT3	DD16	Control	202238	405218	Elatagh	Elatagh	ELATA5	DD14a	>50%	204295	403973

2.5 GEOGRAPHICAL IMAGING AND SITE CHARACTERISATION

Site co-ordinate readings were recorded from a GPS handset at all sites. Arcview™ 3.3 was used to plot site distributions and delineated catchment basins for all sites. The Geoprocessor™ extension program allowed the calculation of various catchment characteristics including geology, soil (and sub-soil) coverage composition, percentage catchment forestry, catchment land-use and catchment area. Catchment delineation was performed by Compass Informatics™. Catchment characteristics were derived using this delineation, the EPA geology database and the most up-to date FIPS database. FIPS 07 was under development during this process and represented the best available data in March 2008. The key information extracted from FIPS 07 included species composition, forest cover, and felling history. The age of the tress was not available. Daniel McInerney, SBES, UCD, undertook in part the geo-processing as outlined.

As mentioned previously two broad geology categories were assigned on the basis of the dominant rock grouping, igneous/metamorphic or sedimentary. The igneous/metamorphic category was predominantly composed of granite, but also included mica schist, quartzite, Diorite, Gabbro and Dolerite while the sedimentary group included mainly Old Red Sandstone, Coal Measures and some Carbonate geology.

Soils were assigned to four groups. This categorisation followed an agreement on interpretation of Irish Forest Soils and Teagasc Soil Categories. The initial catchment soil mapping was based on the Teagasc Soils Map database distributed by the EPA. However, the accuracy of this was unclear. Subsequently, the Forest Service undertook to cross reference the IFS site specific data from 1,732 sites (re-categorised as per the four soil categories) with that held on the Teagasc National Soil cover data. The highest (73%) agreement was within peats, followed by well drained minerals/gleys (62%). Lower percentage agreement was obtained for poorly drained gleys (46%), podzolic/lithosolics (37%) and peaty gleys (21%). Additional soil surveying in a selection of the study catchments was deemed necessary to validate the soil designation. Priority was given to catchments where sites showed variable responses in the aquatic biota to the presence of forestry. A total of 106 catchments were targeted for soil surveying. Using a series of systematic grids (250, 300 and 500m), 1,196 sample points were created and sampled. The attributes data captured was based on the NFI methodology. The results again highlighted

the highest confidence in the designation of peats and well drained mineral soils (report by John Redmond to WRBD). When the catchment soil allocations were finalised the dominant soil type was used in all subsequent analyses as it was considered to have the greatest potential for influencing the stream hydrochemistry.

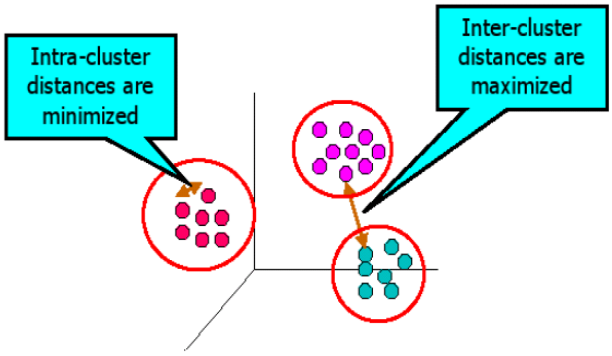
2.6 STATISTICAL ANALYSES

Extensive databases for biological and physico-chemical parameters were generated in Excel™. Univariate and multivariate analyses were performed using SPSS™ v. 12.0.1, STATISITICA™ v. 7.1, Community Analysis Package (CAP™ v. 3.1) and Ecological Community Analysis (ECOM™ v. 2.0). The AQEM Project (ASTERICS 3.10™) program was used to generate over 40 water quality and macroinvertebrate metrics [using Europe version]. Impairment in terms of the various biological metrics and hydrochemical parameters was detected using metric values outside of two standard deviations (or 95% confidence interval) of the control site values as expressed by Resh *et al*. (1988). A similar approach was used by Johnson *et al*. (2005) to develop a clearfelling impact metric. Data from 1st and 2nd order sites were combined following preliminary analyses which indicated no significant relationship between catchment size and taxon richness. These catchment sizes ranged from 21.4 to 661.8ha. Sites with catchment sizes greater than 700ha were excluded but were included in the analyses of longitudinal patterns. Sites with catchments less than 18ha were also excluded.

The hydrochemical data were used to derive means and minimum/maximum values for each parameter. The minimum/maximum values were considered to represent the worst case scenarios and were used to test relationships with forest cover, other catchment descriptors and hydrochemical variables as well as the biological metrics.

Cluster analyses was carried out on the hydrochemical and biological datasets. Clustering is the process of finding groups of objects (or data) such that those in a group are similar (or related) to one another and different from (or unrelated to) the objects in other groups. Some defined distance measure such as the Euclidean distance is often used to determine proximity of the data in a cluster. The k-means clustering algorithm (Hartigan and Wong, 1979) is one of the simplest unsupervised learning algorithms for this partitioning when the number of

clusters (k) is known or specified a priori. A good method will produce high quality clusters with high intra-class similarity and low inter-class similarity (see figure below). The quality of a clustering method is measured by its ability to discover some or all of the hidden patterns. The quality of a clustering result also depends on both the similarity measure (like Simpsons, Bray Curtis of Jaccards) used by the method and its implementation.



In regression analysis or modelling, the clustering helps determine if there are groups of similar data that *might* exhibit a similar response (which might require a specific model or set of parameters) and also if the available data do not cover or span the region of interest. When applied to response variables it clusters those with a similar pattern of responses (which may or may not have a specific physical interpretation).

The k-means algorithm (Hartigan & Wong, 1979) used in the present study is one of the simplest numerical methods used to implement clustering and works as follows:

1. The number of clusters required must be chosen in advance and a significance tolerance for stopping the iterations.
2. An initial position in the data space is chosen for each cluster. These should be as far apart as possible and should cover the range of the data space as well as possible.
3. The Euclidian distance from each point in the data set to all cluster centroids is calculated and each data point is then associated with the nearest centroid. Thus a cluster of data points is associated with each centriod.
4. The actual centroid of the points associated with each cluster is calculated and replaces the previous centriod of that cluster.

5. Steps 3 and 4 are repeated until the change in the centroid positions is less than some specified tolerance.
6. The solution is the set of clusters when the tolerance is satisfied.

Clustering was carried out separately using metrics to describe the chemical signature (hydrogen (max), alkalinity (max) monomeric aluminium (max) cations (min), DOC (max) and organic acids), physical descriptors that may affect the magnitude of the pressure and run-off potential (% coniferous forest, catchment area, slope & area) and selected biological response metrics (taxon richness & ephemeroptera richness/abundance).

2.7 QUALITY CONTROL

Quality control procedures were employed for macroinvertebrate sorting and identification. Previously sorted samples were re-checked for missed specimens to check for % accuracy. At most 10 individuals were recovered representing well below 3% of the total macroinvertebrates initially sorted from the samples. A number of specimens from each identified taxon were checked by an independent taxonomist, Dr Gustavo Becerra Jurado. Quality control of data inputting to the physico-chemical and biological databases was also undertaken. The macroinvertebrate databases from UCD and UCC were reviewed for inconsistencies in taxonomy.

3. RESULTS

3.1 HYDROCHEMISTRY

Conductivity

The river sites examined were typically low conductivity waters. In fact, over 80% of the sites recorded maximum conductivity reading below 150µS/cm (Fig. 4). The highest value recorded was 295 µS/cm in a tributary of the River Loobagh which drains sedimentary geology. Overall, there was no significant differences in the mean readings across geology and soil site groupings (Fig. 5) although the sedimentary sites on well drained mineral soils had marginally higher values. In terms of a forest effect significant differences were detected across the forest cover bands only in sites draining igneous/metamorphic geology and peaty soils (Kruskall Wallis-H(3,65)=13.328, P=0.004).

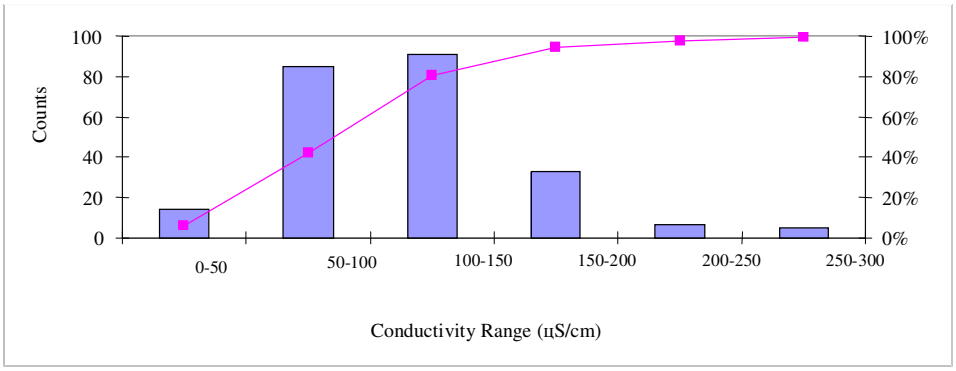


Fig. 4: Frequency distribution and accumulative percentage of conductivity readings from all dates and sites.

The differences between low and high flow readings were highly variable and were typically less than 100µS/cm but one sedimentary site on peat recorded a difference of 205µS/cm, the highest value was associated with low flow. In other sites the highest conductivity readings were associated with high flow.

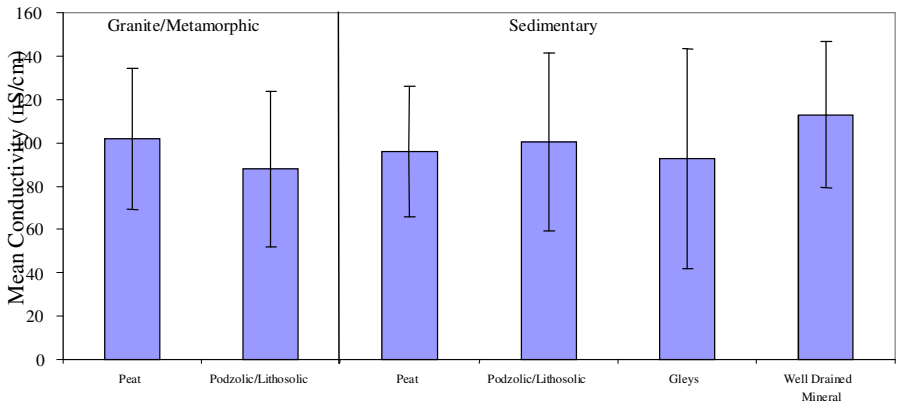


Fig. 5: Variation in mean conductivity reading across geology and soil groups.

pH

The pH readings for the various sampling dates were highly variable (Figs. 6 and 7). Most of the sites appeared to be episodically acidic. Some, especially those draining well drained mineral soils, were more circum-neutral in character. Much of the variation within sites could be related to differences in flow conditions, the low pH values were generally associated with elevated flow. However, as previously mentioned it was difficult to know the stage of the hydrograph represented and full flood conditions were not encountered at many of the sites. It is therefore possible that the highest acidity levels were not captured by the sampling programme.

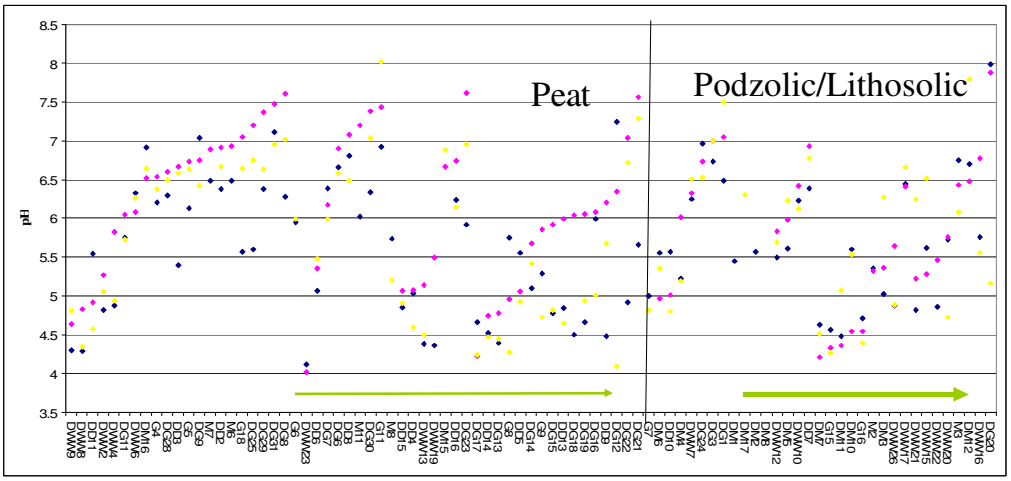


Fig. 6: Distribution of pH readings from sites draining peat and podzolic/lithosolic soils on granite/metamorphic geology. Sites within each soil group are ordered according to increasing forest cover as indicated by the green arrow. The various colours represents the three sampling dates.

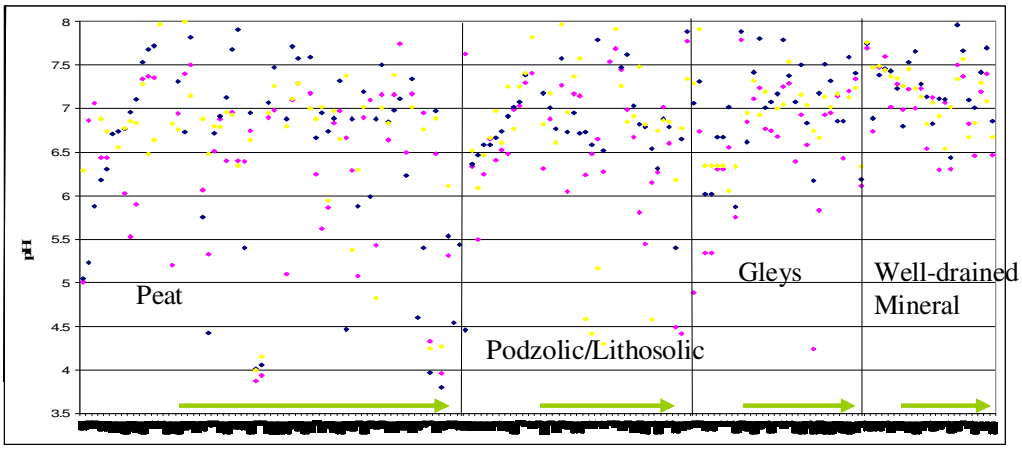


Fig. 7: Distribution of pH readings from sites draining various soils on sedimentary geology. Sites within each soil group are ordered according to increasing forest cover as indicated by the green arrow. The various colours represents the three sampling dates.

Despite the uncertainty relating to flow conditions it should be noted that a good number of control and forested sites were sampled in any one area under the same weather/flow conditions. The randomised sampling should permit assessment of pH changes in relation to forest cover. The data were initially analysed across the forestry bands. Minimum pH was selected for analysis of the worst case condition. On igneous/metamorphic geology minimum pH was significantly different across the forest bands (Soil Type: Peats Minimum pH: Kruskall-Wallis-H(3,55) = 15.8426, p = 0.0012; Soil Type: Podzolic Lithosolic Minimum pH: Kruskall-Wallis -H(3,31) = 9.228, p = 0.0264 – Fig. 7). Some of the lowest values were associated with high forest cover, particularly on peats. The results were similar when maximum hydrogen ion concentrations were analysed.

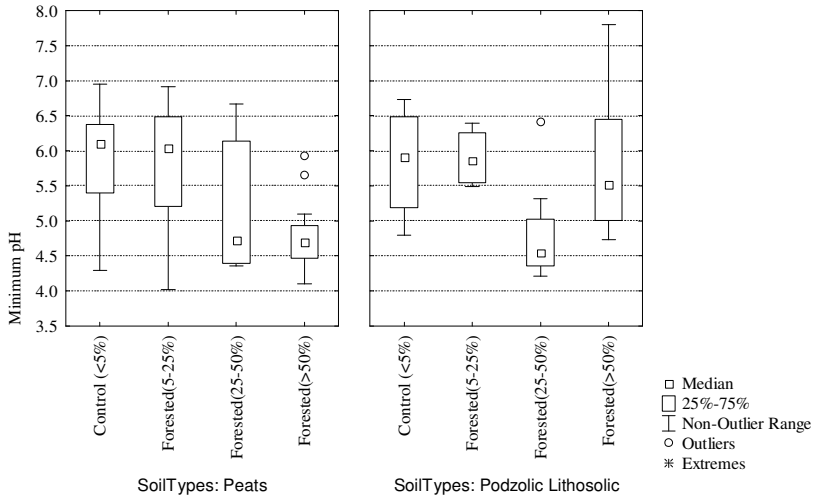


Fig. 8: Box plots of minimum pH values for sites within four forest cover bands draining granite/metamorphic catchments with different dominant soil types.

A similar pattern was recorded on sedimentary geology but none of the differences was statistically significant. Although the pH of sites on well drained mineral soils decreased across the forestry bands the streams remained circum-neutral (Fig. 9).

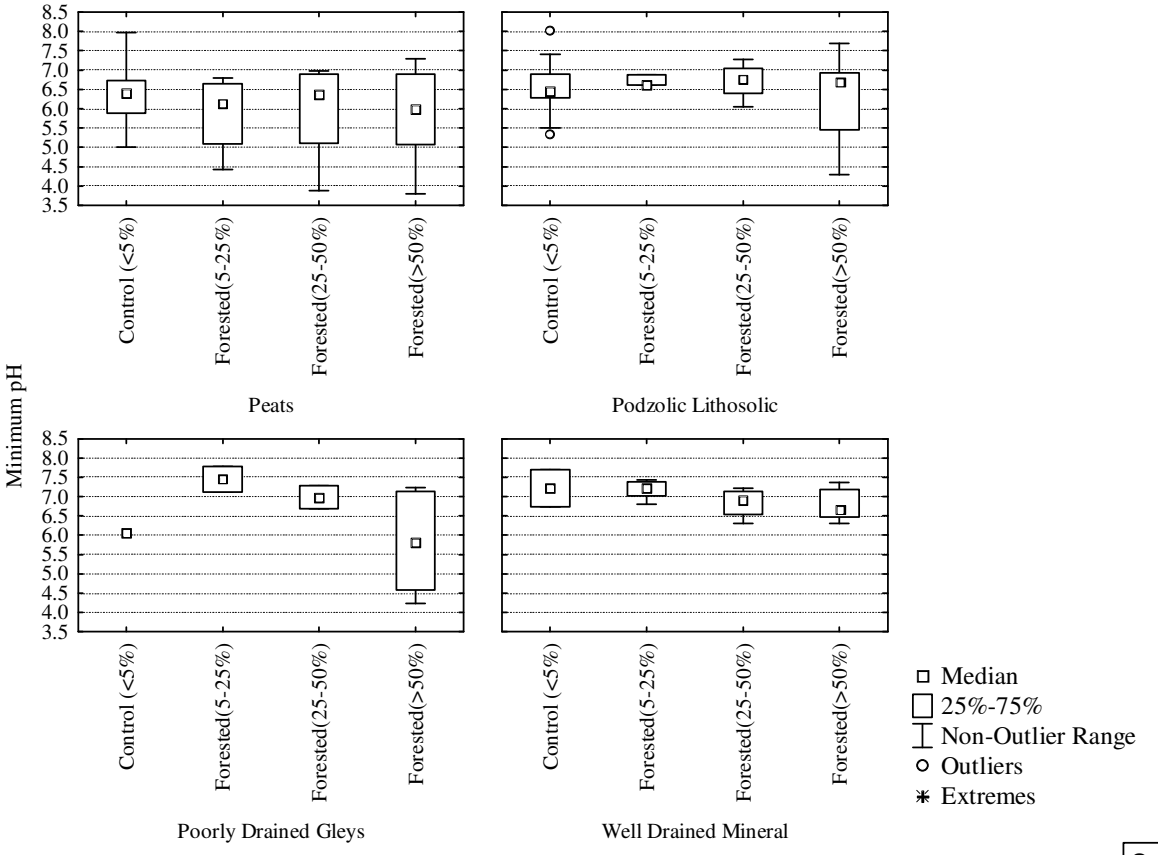


Fig. 9: Box plots of minimum pH values for sites within four forest cover bands draining sedimentary geology with different dominant soil types.

The relationship between minimum pH and % forest cover was also examined for both geological settings. On igneous/metamorphic geology streams draining peat showed a significant decrease ($r = -0.6834$, $p = 0.0000002$) in pH with increasing forest cover (Fig. 10). Although control and sites with low forest cover had some pH reading as low as the more heavily forested sites the latter sites had fewer readings in the circum-neutral range. The minimum pH for both peat and podzolic/lithosolic sites fell below the lower limit of the control sites when forest cover exceeds values in the region of 25-30%. The relationship on peat on sedimentary geology was also significant ($r = -0.2515$, $p = 0.0505$) and largely similar to that on igneous/metamorphic geology except that more readings were in the circum-neutral

range, except when forest cover exceeded 80% (Fig. 10). The podzolic/lithosolic sites did not show a significant correlation between minimum pH and % forest cover. However, it should be noted that the minimum pH values fell below the lower limit for the control sites when forest cover exceeded 60% (Fig. 11).

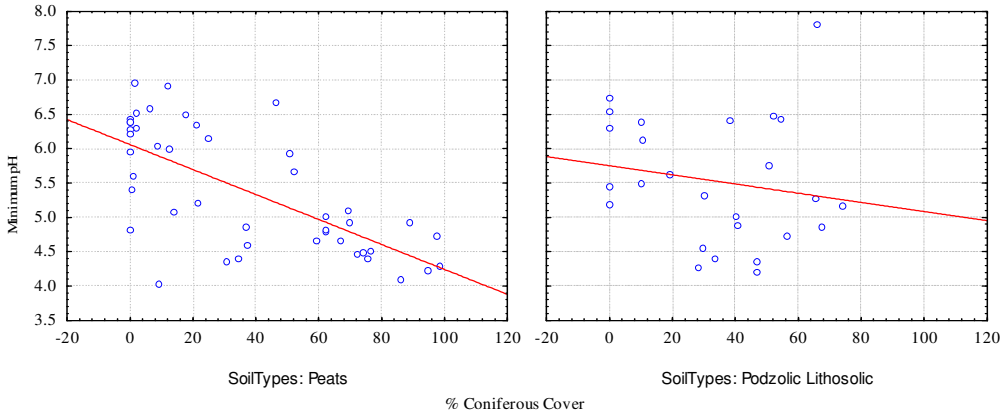


Fig. 10: Relationship between minimum pH and % forest cover for sites draining granite/metamorphic geology with different dominant soil type.

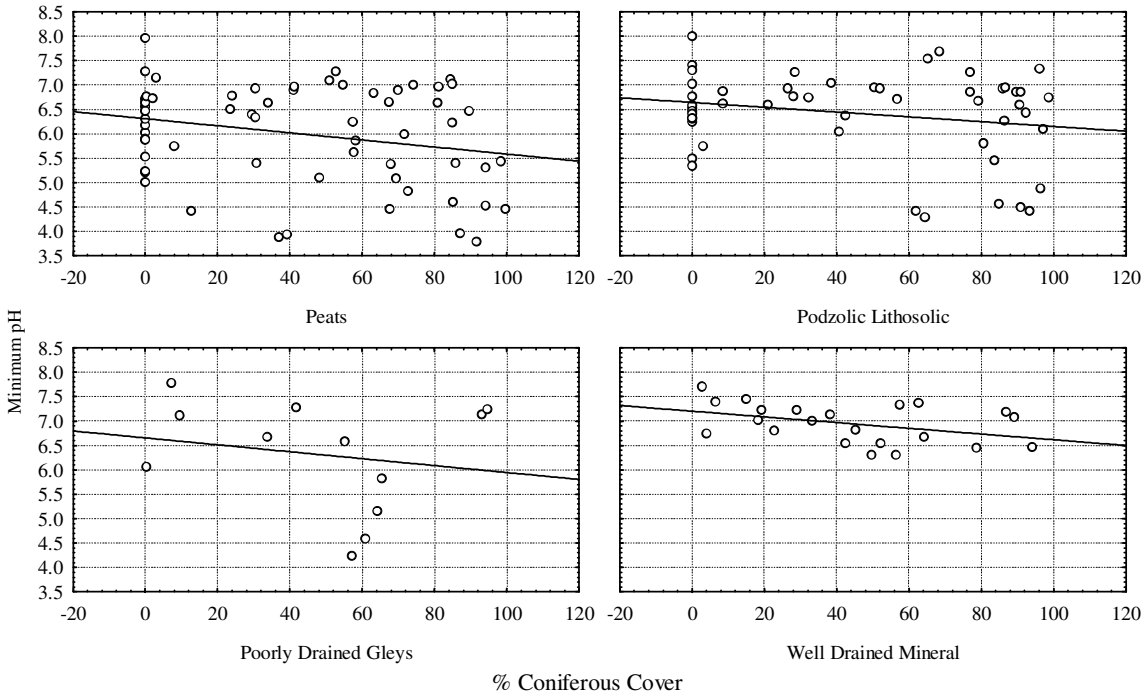


Fig. 11: Relationship between minimum pH and % forest cover for sites draining sedimentary geology with different dominant soil types.

To further analyse these pH data for a possible forest effect it was hypothesised that the number of pH readings below 5.0 would increase across the forest cover bands. This was

based on previous research in upland Wicklow streams which suggested that the duration of low pH values in some forested streams exceeded that in control moorland streams (Kelly-Quinn *et al*., 1996a). Tables 5 and 6 present the results for each geological setting. In igneous/metamorphic catchments the number of pH readings <5.0 was substantially higher than the result for the control sites in the 25-50% and >50% forestry bands for peat sites and the 25-50% band for the podzolic/lithosolic sites.

Table 5: Numbers of pH readings ≤5 in each of the samplings rounds 1-3 and as % percentage of overall samples for sites draining granite/metamorphic geology.

Soil Type	% Forest Cover Band	1	2	3	Total Samples	% total samples with pH≤5
Peat soil						
	<5	4/22	3/22	4/22	66	**17**
	5-25.	1/9	1/8	0/7	24	**8.3**
	25-50	4/6	2/7	3/8	21	**43**
	>50	10/17	4/16	12/16	49	**53**
Podzolic/Lithosolic						
	<5	0/11	2/8	1/8	11	**7**
	5-25.	0/4	0/4	0/4	16	**0**
	25-50	6/9	5/9	4/8	26	**57**
	>50	2/8	0/8	1/8	24	**8**

The result was similar for peat sites on sedimentary geology. On podzolic/lithosolic soils pH values <5.0 were only encountered when forest cover exceeded 50%. The same applied to sites on the poorly drained gleys. No pH readings <5 were recorded in catchments dominated by well drained mineral soils (Table 6).

Table 6: Numbers of pH readings ≤5 in each of the samplings rounds 1-3 and as % percentage of overall samples for sites draining sedimentary geology.

Soil Type	% Forest Cover Bands	1	2	3	Total Samples	% total samples with pH≤5
Peat	**<5**	1/17	1/16	0/15	32	**4**
	5-25.	1/4	1/4	0/4	12	**17**
	25-50	2/10	2/10	2/9	29	**21**
	>50	7/27	3/27	3/27	81	**16**
Podzolic/Lithosolic	**<5**	0/13	0/13	0/13	39	**0**
	5-25.	0/2	0/1	0/2	6	**0**
	25-50	0/2	0/2	0/2	6	**0**
	>50	0/22	3/22	4/22	66	**11**
Poorly drained Gleys	**<5**	0/6	0/6	0/6	18	**0**
	5-25.	0/3	0/3	0/3	9	**0**
	25-50	0/7	0/7	0/5	19	**0**
	>50	0/11	1/11	0/11	33	**3**
Well-drained Mineral	**<5**	0/2	0/4	0/2	6	**0**
	5-25.	0/3	0/3	0/6	13	**0**
	25-50	0/7	0/7	0/6	20	**0**
	>50	0/9	0/9	0/9	27	**0**

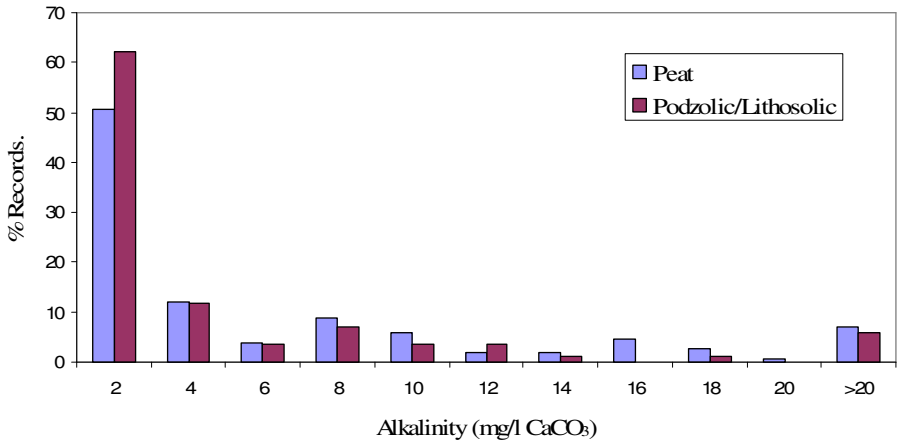


Fig. 12: Distribution of alkalinity readings from sites on predominately peat and podzolic/lithosolic soils in igneous/metamorphic catchments.

Alkalinity, Sodium Dominance Index and Aluminium

The majority of the alkalinity readings from sites draining igneous/metamorphic geology with either peat or podzolic/lithosolic soil cover fell below 2 mg/l $CaCo_3$ (Fig. 12). A few sites had values >20 mg/l $CaCO_3$, these catchments were influenced by variable amounts of carbonate geology in the catchment. On the same soils in catchments dominated by sedimentary geology readings were more evenly distributed across the alkalinity range with over 20% higher than 20 mg/l $CaCO_3$ (Fig. 13).

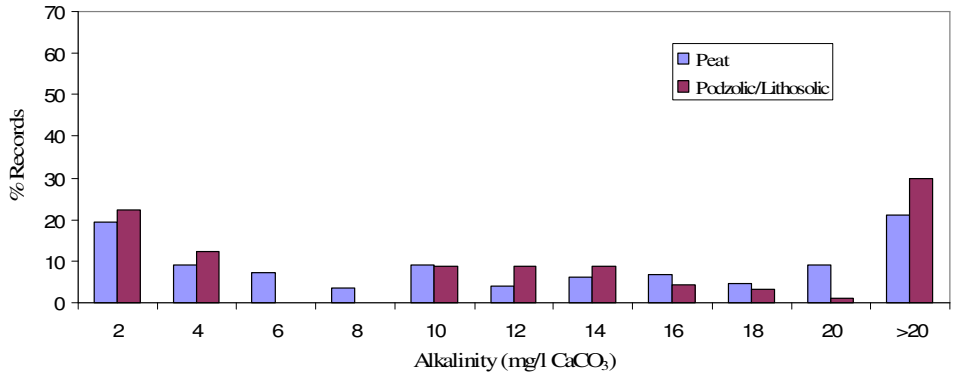


Fig.13: Distribution of alkalinity readings from sites on predominately peat and podzolic/lithosolic soils on sedimentary geology.

As expected, sites draining well drained mineral soil were more buffered and the majority of readings were greater than 20 mg/l $CaCO_3$ (Fig. 14).

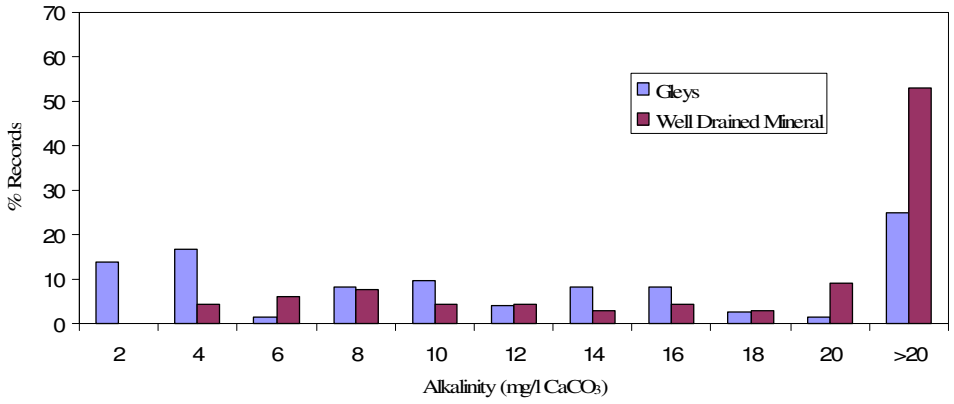


Fig. 14: Distribution of alkalinity readings for sites on predominately peat and podzolic/lithosolic soils in igneous/metamorphic catchments.

In terms of minimum alkalinity the highest number of values <8mg/l $CaCO_3$ were associated with the following geological setting; peat/igneous/metamorphic (96%), podzolic/lithosolic/igneous/metamorphic (95%), peat/sedimentary (72%), podzolic/lithosolic/sedimentary (52%). The presence of forestry tended to depress the site alkalinity as can be seen from Fig.15 which compares the distribution of alkalinity readings from control and heavily forested sites on peat and igneous geology. The effects of forestry were most obvious when minimum alkalinity was considered. Minimum alkalinity decreased significantly across the forest cover bands on peat/igneous/metamorphic (Kruskall-Wallis- $H(3,55) = 14.6122$, $p = 0.0022$), podzolic/lithosolic/ igneous/metamorphic (Kruskall-Wallis - $H(3,31) = 8.0601$, $p = 0.0448$), but this trend was not statistically significant in the other geological settings. However, in the case of peat on granite the occurrence of negative minimum alkalinity increased across the forest cover range. Only the forested sites on podzolic/lithosolic soils recorded negative alkalinity values, some were detected on igneous/metamorphic geology when forest cover exceeded circa 25%. On sedimentary geology negative alkalinity values were recorded at some sites when forest cover exceeded 60%. It should be noted that not all forested sites exhibited this loss of buffering capacity.

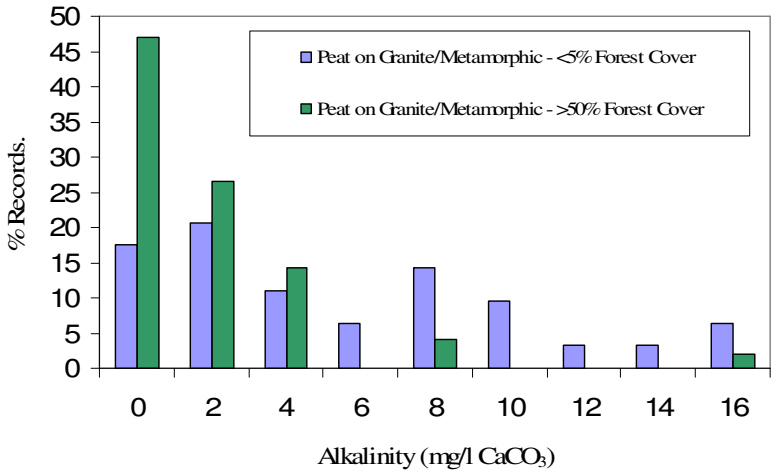
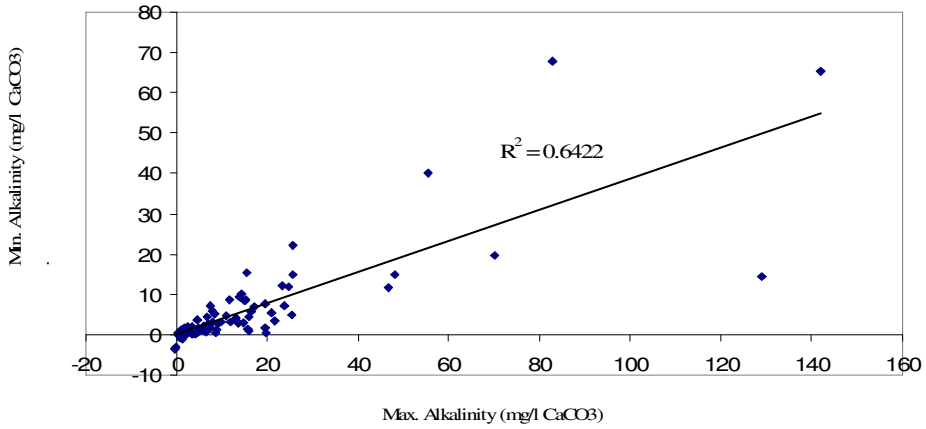


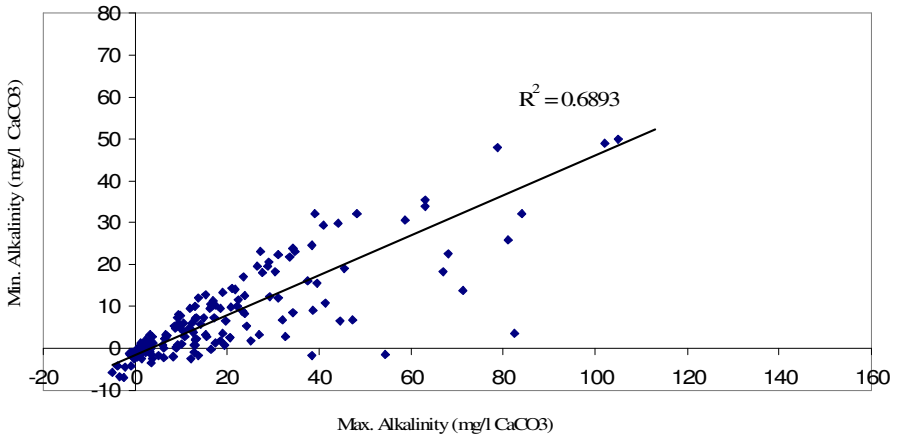
Fig. 15: Distribution of alkalinity readings for sites on predominately peat soils in igneous/metamorphic catchments with and without forest cover.

The relationship between maximum and minimum alkalinity values illustrates the level of change in buffering capacity between the two extremes. The majority of the sites that recorded zero or negative alkalinity had maximum alkalinity below 8 mg/l $CaCO_3$ (Fig. 16a).

In contrast, several forested sites with maximum alkalinity up to 20 mg/l $CaCO_3$ and higher exhibit zero or negative minimum alkalinity values (Fig. 16b).



(a)



(b)

Fig. 16: Relationship between maximum and minimum alkalinity for (a) control sites,(b) sites with >25% forest cover.

Over 90% of the sites dominated by peat or podzolic/lithosolic soil on igneous/metamorphic geology had Sodium Dominance values >60% supporting their acid-sensitive designation. On sedimentary geology the results were more variable, with an increasing proportion of the sites falling below 50% SDI as one moved from peat, through podzolic/lithoslic soils to the gleys and well drained mineral soils. In addition, there were greater differences between maximum and minimum SDI values.

Total aluminium concentrations were highest at sites draining predominantly peat (Figs. 17 & 18). In the two geological settings total aluminium increased significantly with increasing forest cover. The trend for sites influenced by podzolic/lithosolic soils was only significant on sedimentary geology.

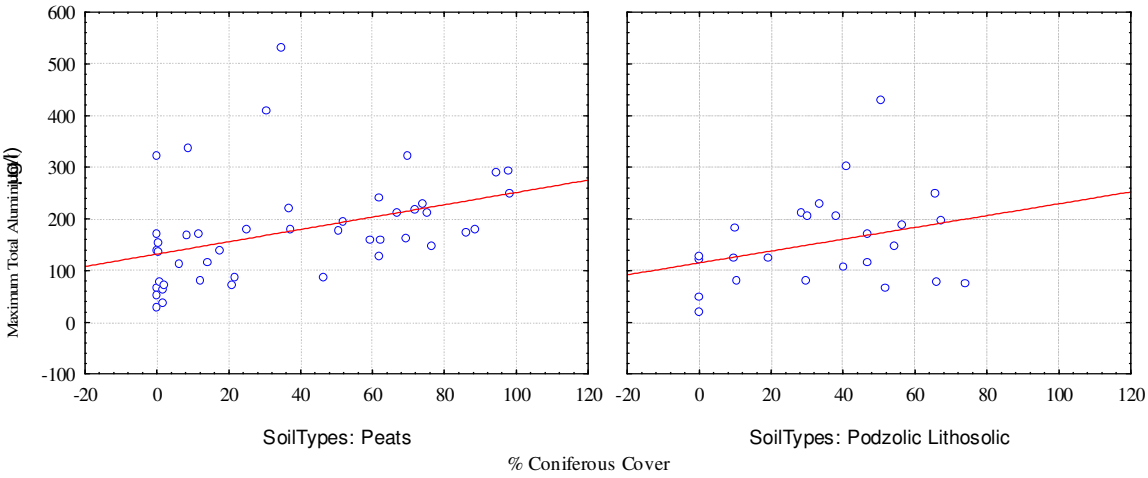


Fig. 17: Relationship between forest cover and maximum total aluminium concentrations on granite/metamorphic geology with peat and podzolic/lithosolic soils.

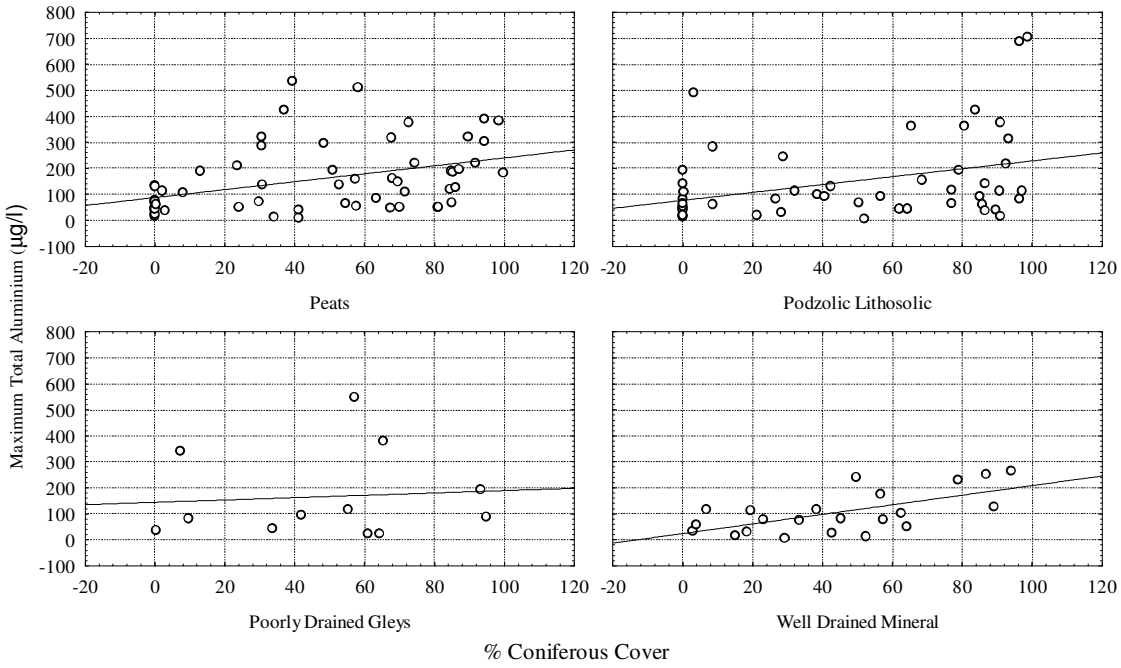


Fig. 18: Relationship between forest cover and maximum total aluminium concentrations on sedimentary geology with various dominant soil types.

Few measurement of labile monomeric aluminium were made and most were for forested sites. Maximum aluminium concentrations ranged from 17.0 – 348 ug/l. No significant correlation with forest cover was detected which may be a factor of the sample size.

Sources of Acidity

Dissolved Total Organic Carbon (DTOC)

As expected the variation in background DTOC concentrations, as illustrated by the control sites, reflected the organic nature of the dominant catchment soils. The highest concentrations were recorded from sites draining peat on igneous/metamorphic geology (Fig. 19) followed by peat on sedimentary geology (Fig. 20). In these two settings DTOC concentrations showed a significant increase across the forest cover bands (Peats on Igneous/metamorphic-Kruskall Wallis-H(3,55) = 19.7422, P = 0.0002; Peats on Sedimentary-Kruskall Wallis-H(3,61) = 12.5833, p = 0.0056).

.

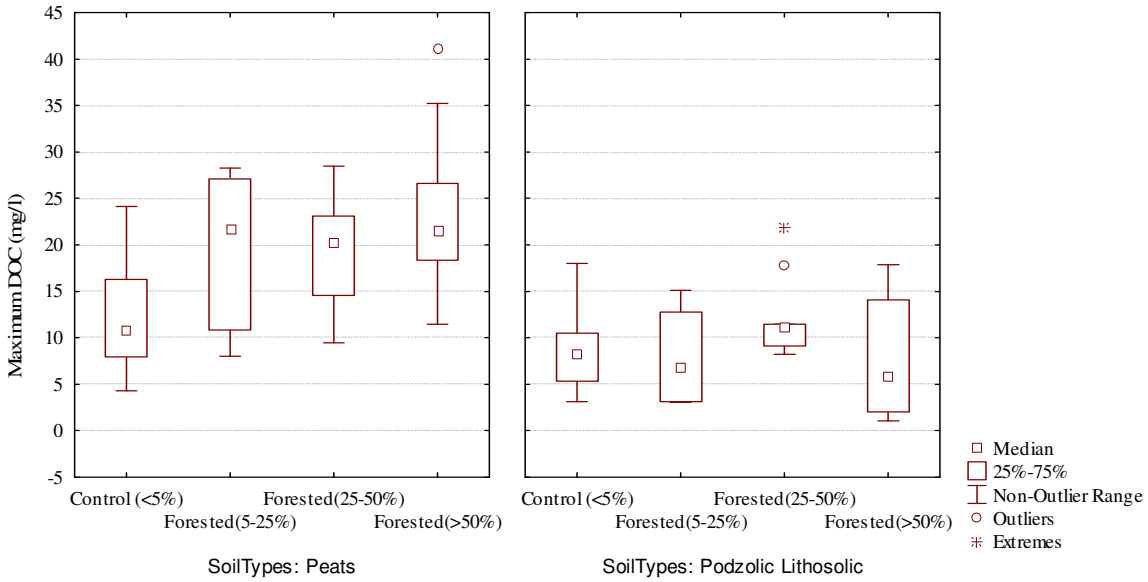


Fig. 19: Box plots of maximum DTOC values from sites within four forest cover bands draining igneous/metamorphic catchments with different dominant soil types.

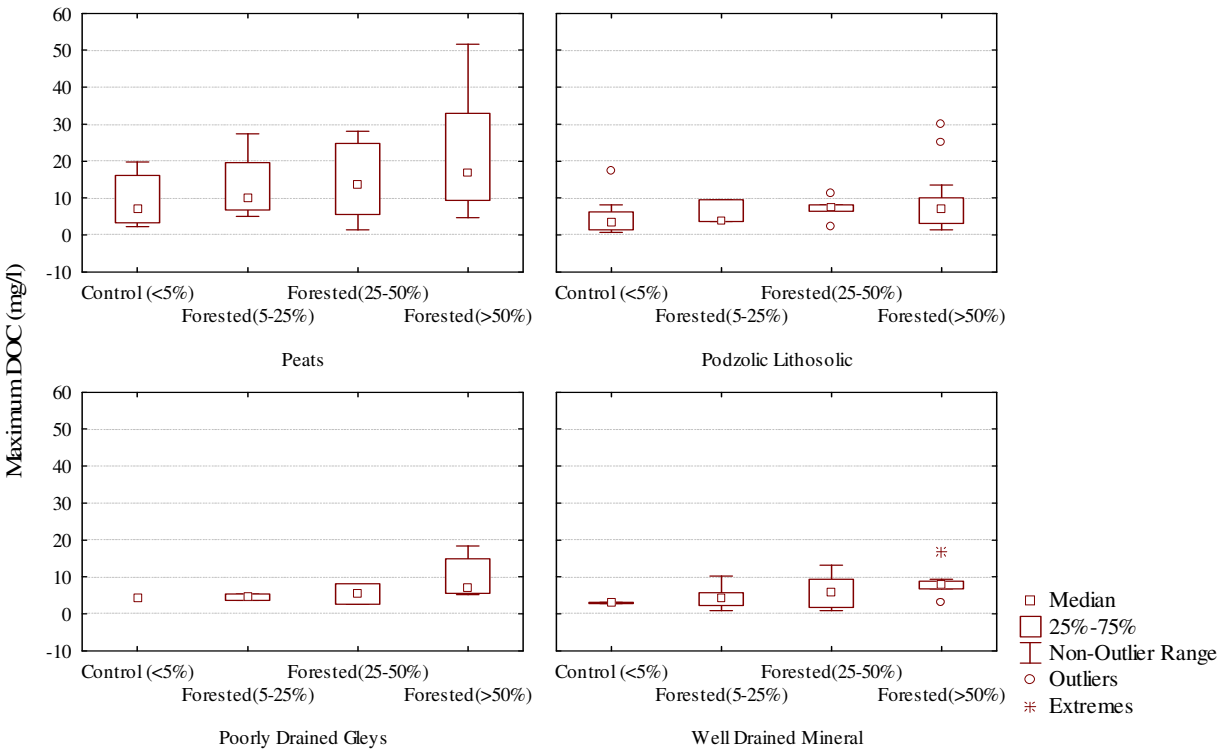


Fig. 20: Box plots of maximum DTOC values from sites within four forest cover bands draining sedimentary geology with different dominant soil types.

Excess or Non-Marine Sulphate

Maximum non-marine sulphate differed significantly across forestry bands for sites draining peat in both geological settings (Igneous/metamorphic – Kruskall-Wallis-H(3,55) = 8.1725, p = 0.0426; Sedimentary Geology – Kruskall-Wallis-H(3,62) = 25.0755, p = 0.00001) The trend was similar for podzolic/lithosolic soils but was only significant on sedimentary geology. Interestingly on mineral soils non-marine sulphate decreased significantly across the forest-cover bands (Kruskall-Wallis-H(3,34) = 9.2423, p = 0.0262).

Nitrate

Sites draining peats and podzolic/lithosolic soils in both geological settings recorded maximum nitrate concentrations largely below 0.5 mg/l NO_3 and there was no significant correlation with forest cover. Concentrations were higher at sites on gleys (0.05-2.75 mg/l NO_3) and well drained mineral soils (0.18-6.22 mg/l NO_3). The latter sites recorded a significant decrease in nitrate concentration across the forest cover gradient.

Chloride

Maximum chloride concentrations ranged from 6.10 to 44.96 mg/l at sites draining peat on igneous/metamorphic geology. Values were up to 10mg/l lower at peat sites on sedimentary geology and marginally lower at sites draining podzolic/lithosolic soils. The relationship with forest cover was only significant for sites on peat in both geological settings.

Calculations suggested that dilution makes a variable contribution to loss of alkalinity and in many cases the forested sites showed a slightly higher % value. Anion titration was detected in all events examined. The principal contributors were organic acids and sulphate. Excess sulphate only made a contribution in the Wicklow sites and at one site in Galway. The contribution of nitrate across all sites was insignificant. The contribution of sea salts to acidification was also low and only one significant sea-salt event was detected at one site in Galway.

Comparison of Source Streams – Upstream and Downstream of Forestry

On igneous geology, no significant differences for any of the chemical variables were found between control sources and the sources sampled above forests (both 5-25% and >25% forest bands), (Mann-Whitney, $P>0.05$). A similar result was noted on sedimentary geology (Mann-Whitney, $P>0.05$). However, it should be noted that on sedimentary geology, only two sites were sampled above the forestry in the 5-25% and >25% categories. Therefore, both of these forestry bands had to be combined into a single forest category and compared to the control sources. This result suggests that all sources (control and above forests) had no significant differences.

Control sources did not differ significantly from the downstream sites on the same stream in terms of pH and alkalinity (Wilcoxon Ranked Sign Test, $P>0.05$). However, chloride, NM sulphate and sodium were significantly higher (Wilcoxon; Chloride: $Z = -2.757$, $P = 0.006$; NM Sulphate: $Z = -2.114$, $P = 0.034$; Sodium: $Z = -3.371$, $P = 0.001$) downstream. Sites downstream of the 5-25% forested band on igneous/metamorphic geology differed significantly from their sources in terms of pH, SDI, chloride, sulphate, NM sulphate, sodium, magnesium, calcium, NM calcium, total hardness and non-marine hardness (Wilcoxon, $P<0.05$). Sites downstream of >25% forest cover had significantly higher total

monomeric aluminium, chloride, sulphate, NM sulphate and sodium (Wilcoxon, P<0.05) than their respective sources.

On sedimentary geology sites, the downstream control sites recorded significant differences from their corresponding sources for chloride, sulphate, NM sulphate, sodium, NM sodium and NM magnesium (Wilcoxon, P<0.05). On sedimentary geology only one site pairing represented the 5-25% forest cover category however, a difference in NM Ca was noted (Wilcoxon, P<0.01). The >25% forested category on sedimentary geology presented significant differences between downstream and source for pH, hydrogen, alkalinity, SDI, NM sodium, magnesium, NM magnesium, calcium, NM calcium, total hardness and NM hardness (Wilcoxon, P<0.05 and P<0.001). Results are presented in Table 7. The higher sodium levels at downstream sites were accompanied by higher magnesium and calcium values. This maintained the SDI values as the overall ratio of cations remained quite similar at the source and downstream sites.

Table 7: Significant results from paired analysis for selected chemical variables at downstream and source sites (Wilcoxon Ranked Sign Test).

Igneous Metamorphic Sites	Parameter	Wilcoxon (Z)	P value	Sedimentary Sites	Parameter	Wilcoxon (Z)	P value
Control <5% Forest	Chloride	2.757	0.006	**Control <5% Forest**	Chloride	-3.516	<0.001
	NM Sulphate	-2.114	0.034		Sulphate	-3.206	0.001
	Sodium	-2.371	0.001		NM Sulphate	-2.999	0.003
					Sodium	-2.999	0.003
5-15% Forest Cover	pH	-1.503	0.028		NM Sodium	-3.154	0.002
	Chloride	-3.11	0.002		NM Magnesium	-2.223	0.026
	SDI	-2.062	0.039				
	Sulphate	-2.97	0.003	**5-15% Forest Cover**	NM Calcium	-2.201	0.028
	NM Sulphate	-3.11	0.001				
	Sodium	-3.18	0.001	**>25% Forest cover**	pH	-2.971	0.003
	Magnesium	-2.551	0.011		Hydrogen	-2.621	0.009
	Calcium	-2.831	0.005		Alkalinity	-2.345	0.019
	NM Calcium	-2.481	0.013		SDI	-2.342	0.019
	Total Hardness	-2.9	0.004		NM Sodium	-2.201	0.028
					Magnesium	-2.622	0.009
					NM Magnesium	-2.271	0.023
>25% Forest cover	Total Aluminium	-3.068	0.002		Calcium	-2.622	0.009
	Chloride	-2.425	0.001		NM Calcium	-2.411	0.016
	Sulphate	-2.516	0.012		Total Hardness	-2.691	0.007
	NM Sulphate	-2.0	0.012		NM Hardness	-2.621	0.009
	Sodium	-3.555	<0.001				

Although the above analyses highlighted only two site grouping that recorded lower downstream pH than at the sources there were several individual sites within other groups

that followed this pattern. Several of the igneous sites in Co. Wicklow were more acidic downstream than their corresponding sources. These sites included those on the Annalecka, Lugduff and Glashaboy rivers. These sites had ~40-70% catchment cover of coniferous forest.

3.2 MACROINVERTEBRATES

Community Composition

In total, over 318,000 individual specimens were sorted and identified to the lowest possible taxonomic level from the 239 study sites. These yielded a total of 204 distinct taxa. The most diverse group was the Trichoptera followed by the Coleoptera (Table 8).

Table 8: Taxon richness in the major taxonomic groups

Taxon	Richness
Trichoptera	61
Coleoptera	53
Diptera	29
Ephemeroptera	18
Plecoptera	17
Gastropoda	8
Odonata	5
Crustacea	3
Hirudinea	4
Hemiptera	2
Neuroptera	1
Lamellibranchia	1

Some of these taxa were highly localised, such as the mayfly species, *Ameletus inopinatus* Eaton, found only in samples collected in Wicklow and Donegal. Other mayfly, such as *Baetis rhodani* (Pictet.) and *Leptophlebia vespertina* (Linn.) were more ubiquitous. Species such as *Caenis rivulorum* Eaton and the caddis-fly, *Sericostoma personatum* (Kirby & Spence) were considered acid-sensitive as they were located in more buffered regions on sedimentary geology. More acid-tolerant species including, *Ameletus inopinatus*, *Siphlonurus lacustris* (Eaton), *Leptopheblia vespertina* and *Plectrocnemia conspersa* (Curtis) were present in higher abundances in areas of weathering tolerant, acid-sensitive, igneous geologies.

In general the mean abundances of macroinvertebrates was significantly higher at the sedimentary sites (One-way ANOVA; $F_{(1,5)} = 59.058$, P = 0.002; Fig. 21).

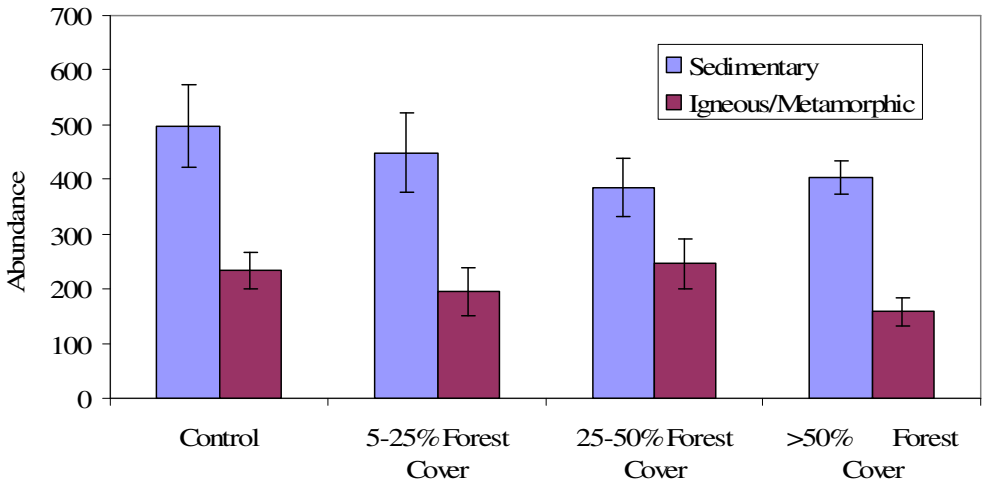
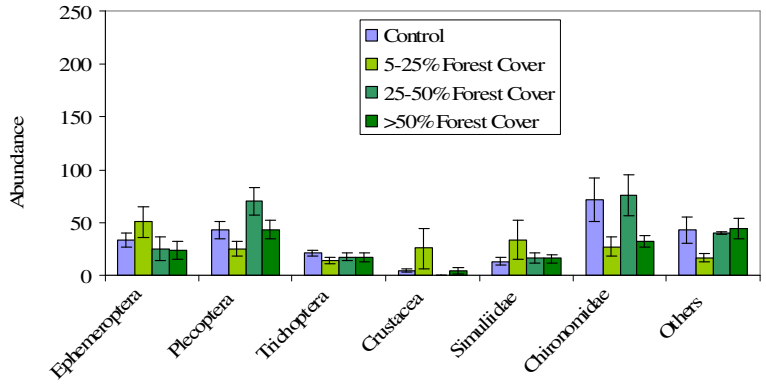


Fig. 21: Mean macroinvertebrate abundances at the sedimentary and igneous/metamorphic sites across the four forest cover bands.

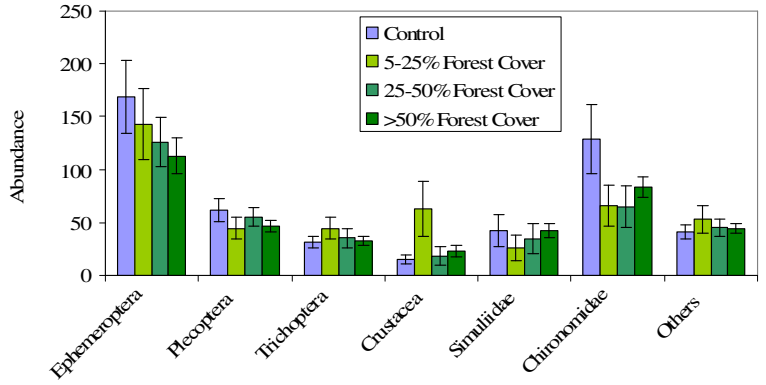
The higher total macroinvertebrate abundances at the sedimentary sites could be largely attributed to the Ephemeroptera and Chironomidae (Fig. 22). In both geological settings the Ephemeroptera was reduced in abundance at sites in the two highest forest cover bands. At the igneous/metamorphic sites the reduction in ephemeropteran abundance was largely balanced by an increase in the numbers of Plecoptera. This did not occur at the sites draining sedimentary geology and consequently overall abundance declined gradually across the forest cover bands.

Selection of Macroinvertebrates Metrics

Approximately 45 different water quality and diversity metrics were generated for the dataset using the AQEM (ASTERICS 3.10™) computer software. Those which were most appropriate for Ireland and which showed a significant correlation with pH were selected to detect impacts due to acidification. These included taxon richness (Fig. 23), ephemeropteran richness, ephemeropteran abundance, trichopteran richness, *Baetis* abundance, %EPT.



(a)



(b)

Fig. 22: Mean abundances of the major taxonomic groups at sites on (a) igneous/metamorphic and (b) sedimentary geology. Standard error bars are included.

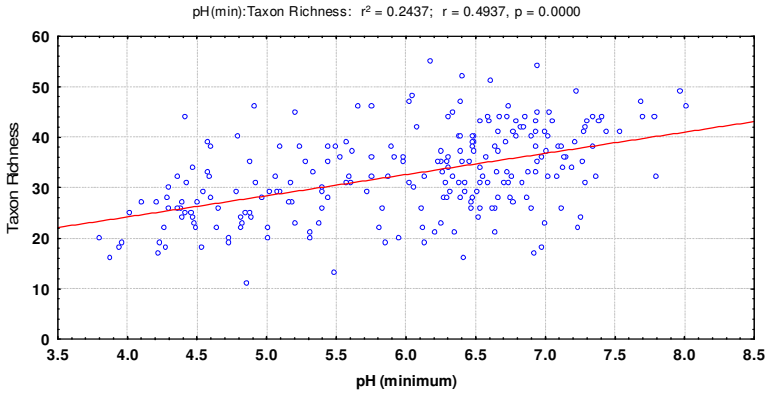


Fig. 23: Relationship between taxon richness and minimum pH across all sites.

Taxon Richness

On igneous/metamorphic geology taxon richness ranged from 11 to 52, the lower value was from a forested catchment (DWW22, 67% forest cover). The range of values was similar on sedimentary geology (16 at L5, 37% forest cover to 55 at one of the control sites). There was a significant decline in taxon richness with increasing coniferous forest cover on peat and well drained mineral sites on sedimentary geology (Kruskall-Wallace; Sedimentary/Peat – $H_{(2,65)} = 10.4252$, $P<0.05$; Sedimentary/Well Drained Mineral – $H_{(2,34)} = 9{,}4919$, $P<0.05$, Fig. 24). The differences were not statistically significant for igneous/metamorphic sites. However, the number of sites with taxon richness less than 30 was highest in the >50% forest cover band compared to the control group (Fig. 25) draining peats, and the pattern was retained when sites on podzolic/lithosolic soils were added to the analysis (Fig. 26).

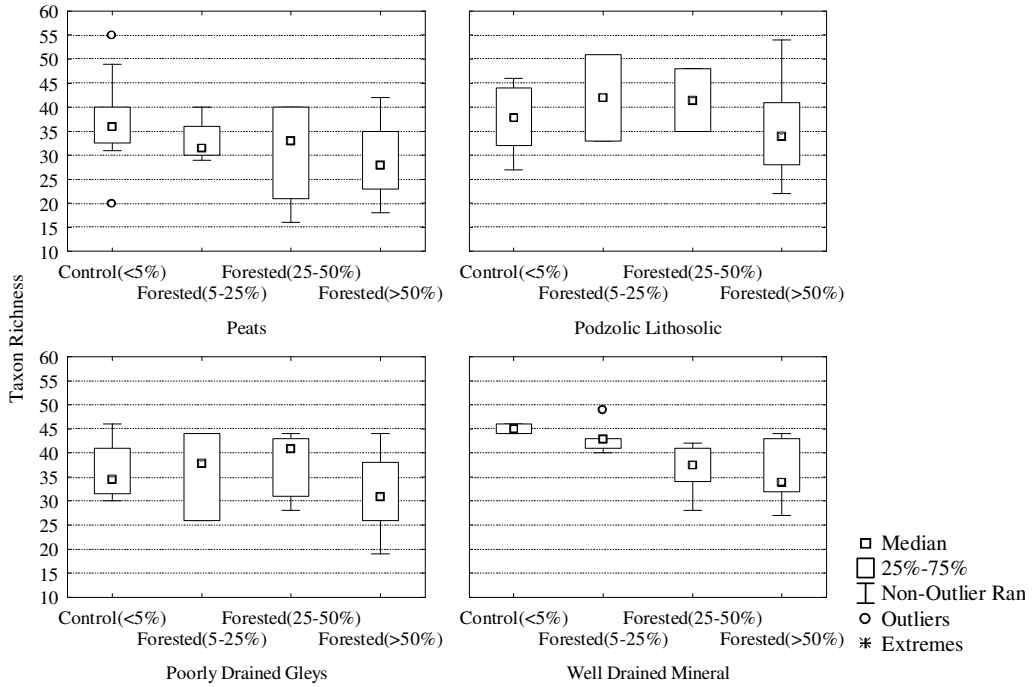


Fig. 24: Box plots of taxon richness for sites on sedimentary geology with different soil categories.

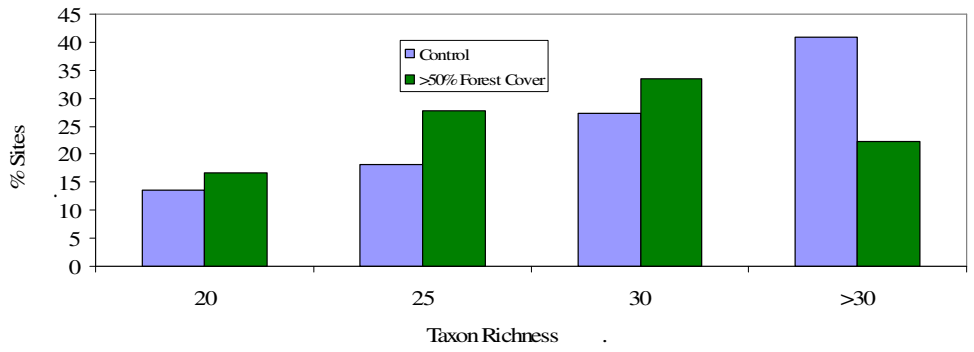


Fig. 25: Comparison of the distribution of taxon richness at sites draining peat on igneous/metamorphic geology.

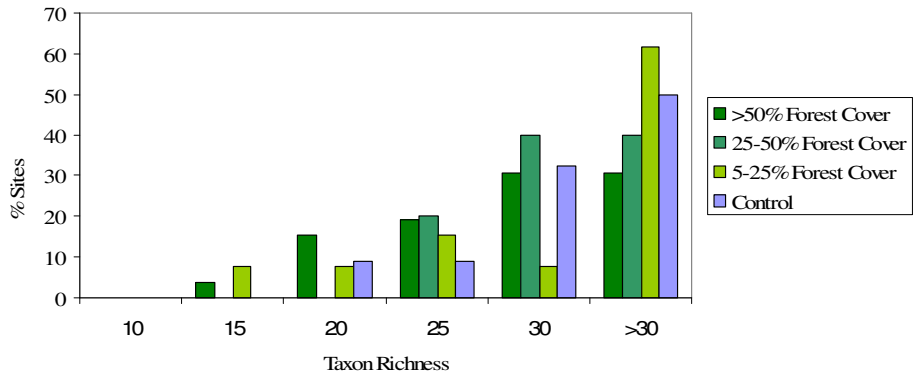


Fig. 26: Distribution of taxon richness counts at sites draining peat and podzolic/lithosolic soils on igneous/metamorphic geology.

Ephemeropteran Richness

Ephemeropteran richness reached a maximum of 8 species on igneous/metamorphic geology with one additional species on the sedimentary geology. In both settings some of the forested sites recorded a low diversity of Ephemeroptera.

Ephemeropteran richness was significantly negatively correlated with % forest cover at sites draining peats ($r=-0.4640$, $P<0.001$) and podzolic lithosols ($r=-0.3884$, $P<0.05$) on igneous/metamorphic geology. As forest cover increased an increasing number of sites

recorded low taxon richness (Fig. 27). On sedimentary geology (Fig. 28) ephemeropteran richness was again significantly correlated to percentage conifer cover on peat ($r = -0.5378$, $P<0.001$) and well drained mineral soils ($r = -0.4855$, $P<0.001$).

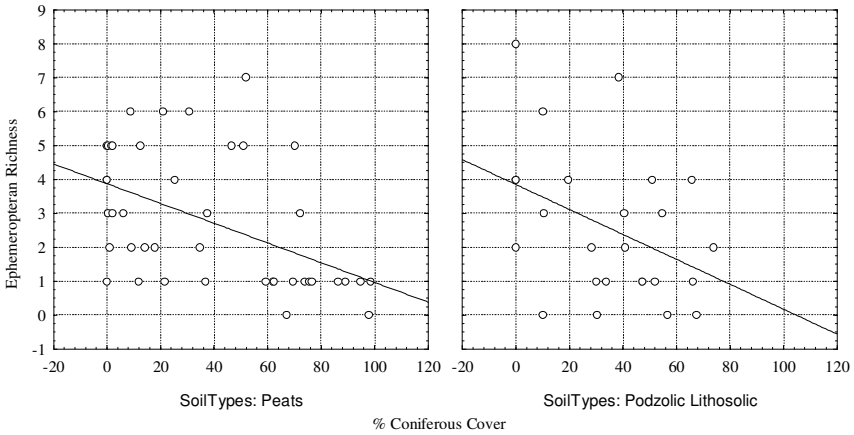


Fig. 27: Relationship between ephemeropteran richness and forest cover at sites draining peat and podzolic/lithosolic soils on igneous/metamorphic geology.

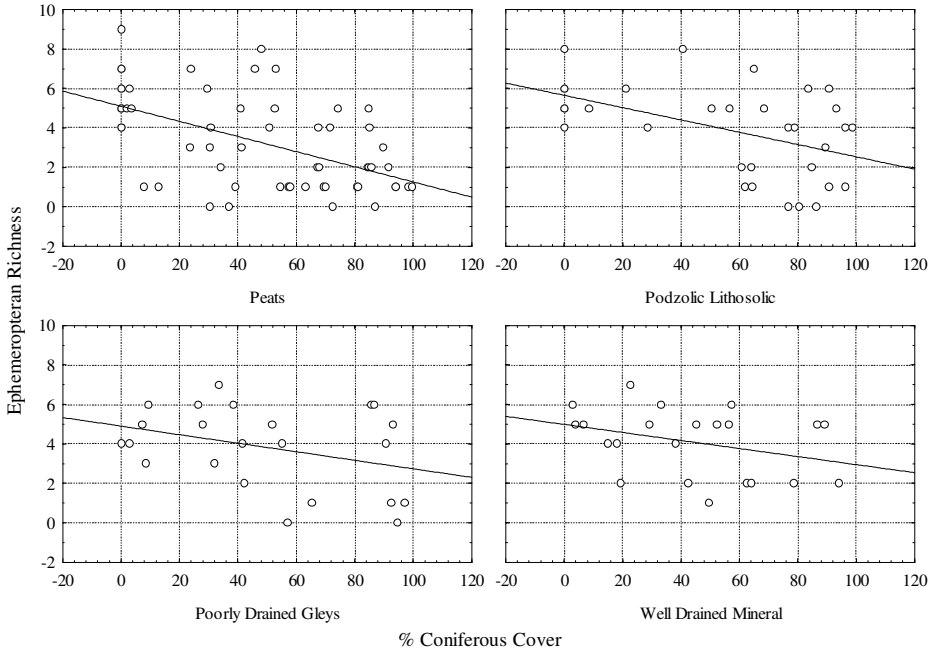


Fig. 28: Relationship between ephemeropteran richness and forest cover at sites draining different soils on sedimentary geology.

As was highlighted for total taxon richness, an increasing number of sites showed a reduction in ephemeropteran richness (Table 9) along the forest cover gradient. Some 13% of sites draining peat on igneous/metamorphic geology in the >50% coniferous cover band were devoid of Ephemeroptera, while a further 75% only had one species present. On podzolic/lithosilic soils there relatively few Ephemeroptera at sites in the three forestry bands. A decline in the occurrence of *Baetis* spp. occurred across the forestry bands. For example 9% of the control sites on peat had low numbers of *Baetis* spp. compared to 71% of sites in the >50% forest cover band. The replacement of *Baetis* by more acid-tolerant species (e.g. *Siphlonurus lacustris*) was a feature of the latter group of sites.

Table 9: Percentage distribution of ephemeropteran taxon richness counts for sites on (a) peats and (b) podzolic/lithosolic soils on igneous/metamorphic geology

(a)

		Forest Cover		
Richness	**Control**	5-25%	25-50%	>50%
0	0	0	0	13
1	5	22	14	75
2	18	33	14.5	0
3	36	11.5	14	6
4	9	0	14.5	0
5	23	11.5	29	6
6	5	22	14	0
7	0	0	0	0
8	4	0	0	0

(b)

		Forest Cover		
Richness	**Control**	5-25%	25-50%	>50%
0	0	25	11	25
1	18	0	44	25
2	9	0	22	12.5
3	9.5	25	11.5	12.5
4	36	25	0	25
5	9.5	0	0	0
6	9	25	0	0
7	0	0	11.5	0
8	9	0	0	0

The pattern was similar for peat sites on sedimentary geology where a low number of Ephemeroptera was recorded in the two top forestry cover bands (Table 10). On podzolic/lithosolic soils only sites in the >50% band recorded no mayfly and a large proportion of the sites in the >50% band had just one or two species present. *Baetis* spp. were only absent from sites (10%) on podzolic/lithosolic soils in the >50% forest cover band. At sites on poorly drained gleys and well drained mineral soils there was little differences in the distribution of ephemeropteran counts across the forest cover bands (Table 11).

Table 10: Percentage distribution of ephemeropteran taxon richness counts for sites draining (a) peats and (b) podzolic/lithosolic soils on sedimentary geology.

(a)

		Forest Cover		
Richness	**Control**	5-25%	25-50%	>50%
0	0	0	18.5	7
1	0	50	9	43
2	0	0	9	20
3	15	25	18.5	3.5
4	10	0	9	13
5	40	0	9	10
6	15	0	9	0
7	10	25	9	3.5
8	5	0	9	0
9	5	0	0	0

(b)

		Forest Cover		
Richness	**Control**	5-25%	25-50%	>50%
0	0	0	0	8
1	0	0	0	8
2	0	0	0	8
3	0	0	0	8
4	23	0	50	17.5
5	23	50	0	25
6	31	50	0	17.5
7	15	0	0	8
8	8	0	50	0
9	0	0	0	0

Table 11: Percentage distribution of ephemeropteran taxon richness counts for sites on (a) poorly drained gleys and (b) well drained mineral soils on sedimentary geology.

(a)

		Forest Cover		
Richness	**Control**	5-25%	25-50%	>50%
0	0	0	0	0
1	33.3	0	0	33
2	0	0	14.5	0
3	33.3	33.3	14	0
4	33.3	0	14.5	22.5
5	0	33.3	14	22
6	0	33.3	29	22.5
7	0	0	14	0
8	0	0	0	0
9	0	0	0	0

(b)

		Forest Cover		
Richness	**Control**	5-25%	25-50%	>50%
0	0	0	0	0
1	0	0	0	0
2	0	20	20	44.5
3	0	0	0	0
4	0	40	40	0
5	50	20	20	44.5
6	50	0	0	11
7	0	20	20	0
8	0	0	0	0
9	0	0	0	0

Ephemeropteran Abundance

A reduction in abundance of indicator taxa can often highlight environmental stress and it is considered to be a useful early warning indicator of impact. No significant correlation was detected between ephemeropteran abundance and forest cover for sites draining either peat or podzolic/lithosolic soil types on igneous/metamorphic geology. The same applied to these soil types on sedimentary geology. However, a significant relationship was detected for sites located on well drained mineral soils on sedimentary geology ($r = -0.6358$, $P>0.001$).

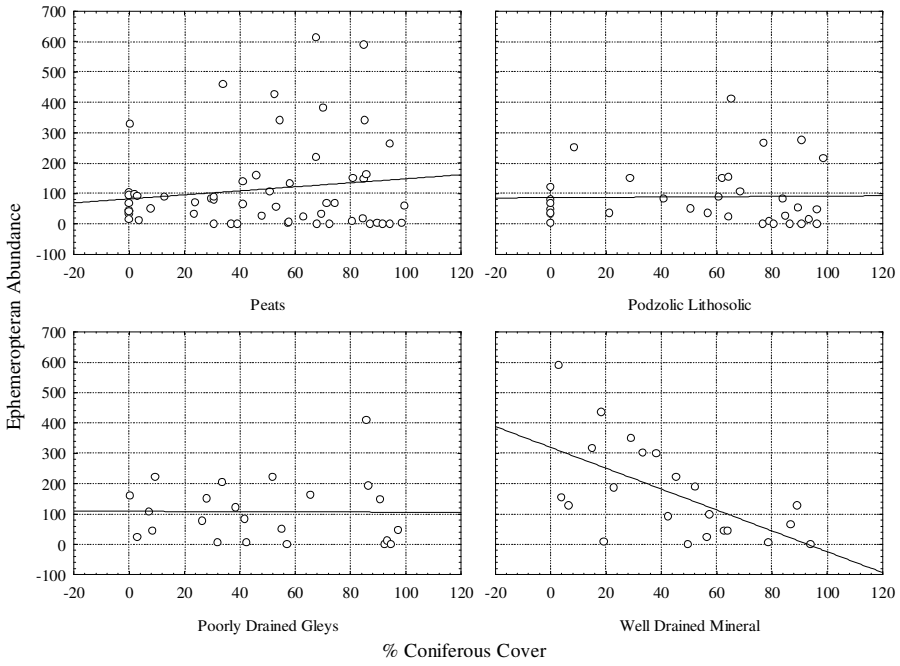
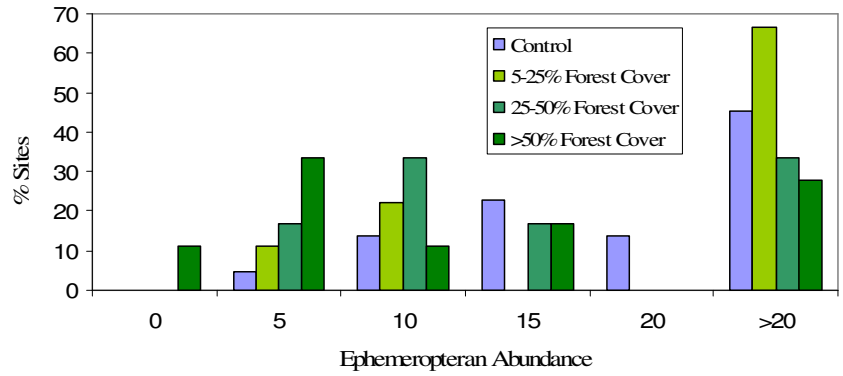


Fig. 29: Relationship between % forest cover and ephemeropteran abundance on sedimentary geology

The cluster of high mayfly abundance noted on sedimentary peats between pH levels 6.5 and 7.5 (Fig. 29) corresponds to a cluster of highly buffered, high pH sites in Co. Cork. Despite high levels of coniferous forest cover at these sites, the occurrence of variable amounts of mineral soils among the peats improved buffering capacity and allowed for higher abundances of mayfly (particularly *Baetis rhodani*). Despite the lack of a strong correlations between abundance and % forest cover it was clear that on peat and podzolic soil in both geological settings the number of sites with low numbers (zero and <5) of ephemeropteran specimens increased across the forest cover bands. The results are illustrated for sites on (a) peat and (b) podzolic/lithosolic soils in Fig. 30.



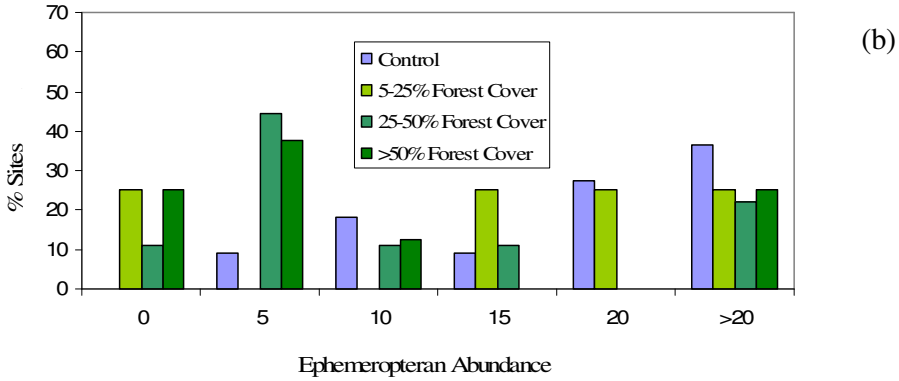


Fig. 30: Frequency distribution of ephemeropteran abundance counts assigned to 6 abundance categories (0; ≤5; ≤10; ≤15; ≤20; >20) at sites draining (a) peat and (b) podzolic/lithosolic soils on igneous/metamorphic geology.

EPT Metric

Ephemeropteran (E), plecopteran (P) and trichopteran (T) richness values are used to calculate EP and EPT metric. In the present study EPT richness correlated significantly (R^2= 0.735, P<0.0001) with EP richness and therefore only one of these was applied in the analyses. The variability plot of EPT richness indicated a shift in distribution towards the lower end of the scale as one moved across the forest cover bands (Fig. 31). While no significant differences in median EPT was detected across forestry bands on igneous/metamorphic geology (P>0.05), significant declines in both metrics were found for peat sites on sedimentary geology (Kruskall-Wallis- EPT – $H_{(3,65)}$ = 10.0914, P = 0.0178, Fig. 32). A similar trend occurred on well drained mineral soil but the relationship was not significant (P>0.05). It is worth noting that the median EPT of sites draining podzolic/lithosolic soils fell well below the control median in the >50% forest cover band.

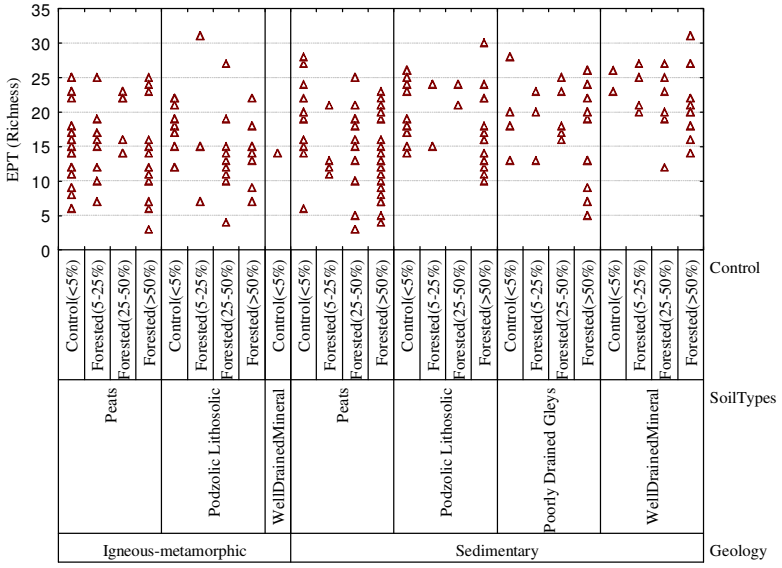


Fig. 31: Variability plot of EPT across coniferous forest bands for each geology and soil category.

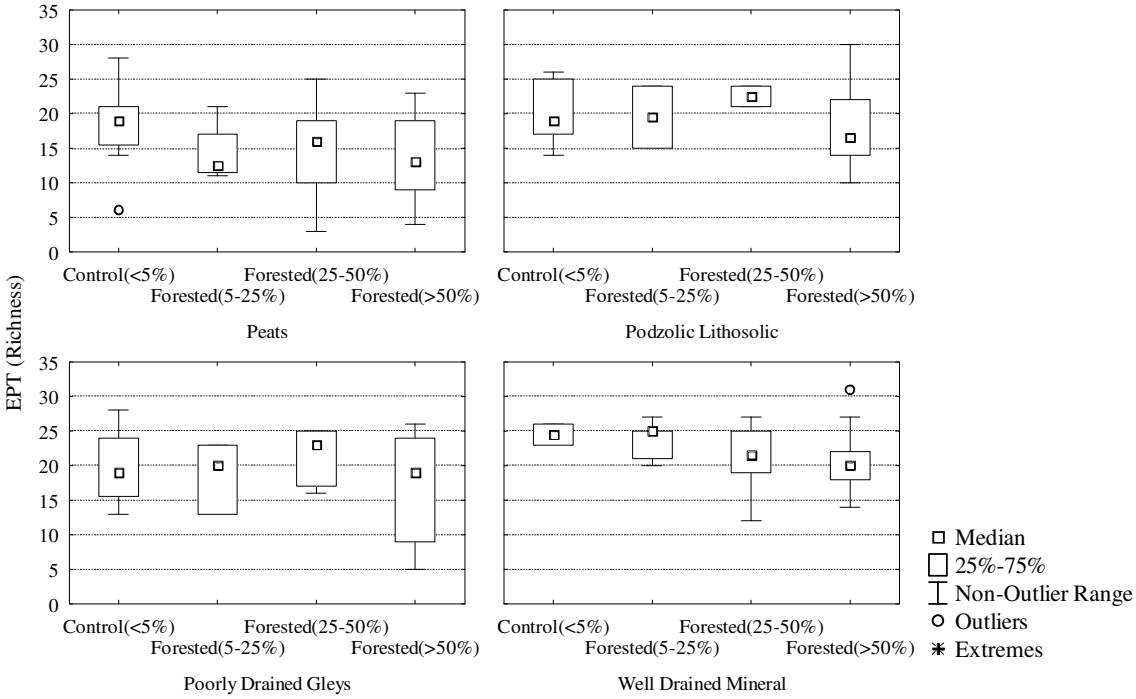


Fig. 32: Box plots for EPT richness on sites draining sedimentary geology.

The relationship between EPT richness and % forest cover is further explored in the correlation plots (Fig. 33). The correlation was significant for peat sites in both geological

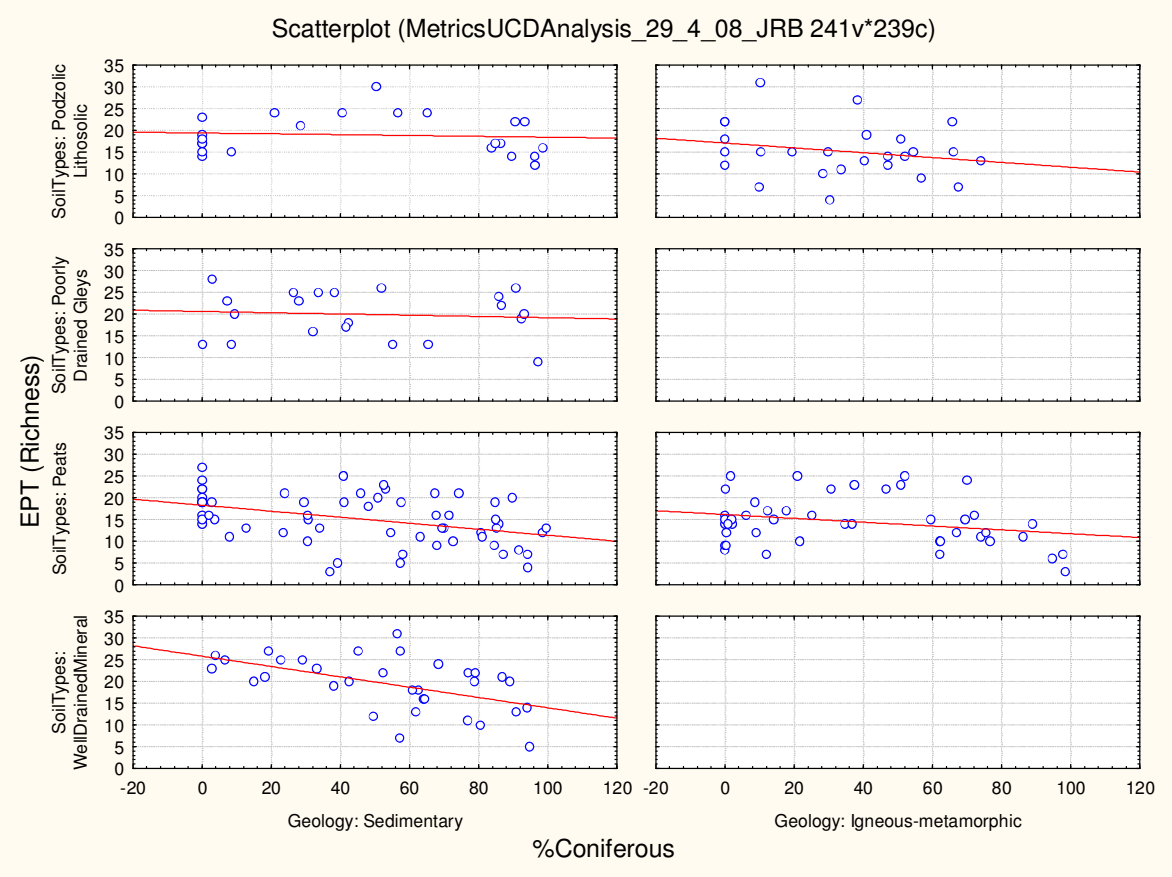


Fig. 33: Relationship between EPT richness and % forest cover in the various geological settings.

settings and site EPT richness began to fall below the lower limit of the control sites when forest cover exceeded circa 25-30% . This also applies to sites on podzolic/lithosolic soils on igneous/metamorphic geology. On sedimentary geology the podzolic sites recorded low EPT above 50% forest cover.

BMWP & ASTP Metrics

While Kruskall-Wallis tests on BMWP did not show a significant difference between forest bands on peats and podzolic/lithpsolic sites on igneous/metamorpgic geology there were nonetheless strong trends of decreasing BMWP across the forest cover bands. Peat sites on sedimentary geology demonstrated a significant decrease in BMWP with increasing forest cover bands ($H_{(3,65)} = 10.3406$, $P = 0.0159$). While the trend was only statistically significant

for peats some sites on podzolic/lithosols and poorly drained gleys showed a distinct decrease in BMWP scores in the >50% forest cover band. The Biological Monitoring Working Party (BMWP) Score was significantly correlated with % coniferous cover for sites draining peat soils on both igneous/metamorphic (r = -0.3209, P = 0.03, Fig. 34) and sedimentary rock (r = -0.4195, P = 0.0009) geology (Fig. 35). The relationship was not significant for podzolic/lithosolic soils but here again it should be noted that scores for sites on igneous/metamorphic and sedimentary geology began to fall below the control BMWP scores when forest cover exceed 25% and 60%, respectively. Average Score Per Taxa (ASTP) significantly correlated with increasing % coniferous cover at sedimentary peats sites only (r = -0.3679, P = 0.0041). A significant difference was also detected between forest cover bands for sedimentary peat sites, with ASTP decreasing as forest bands increased ($H_{(3.65)}$ = 7.9399, P = 0.0473)

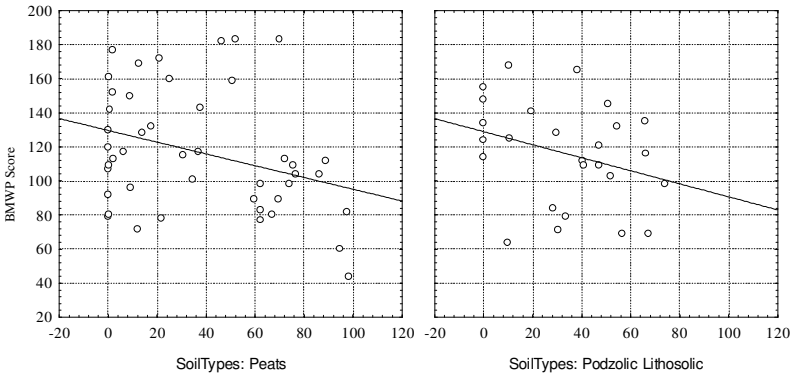


Fig. 34: Relationship between BMWP and % coniferous cover for soils on igneous/metamorphic geology.

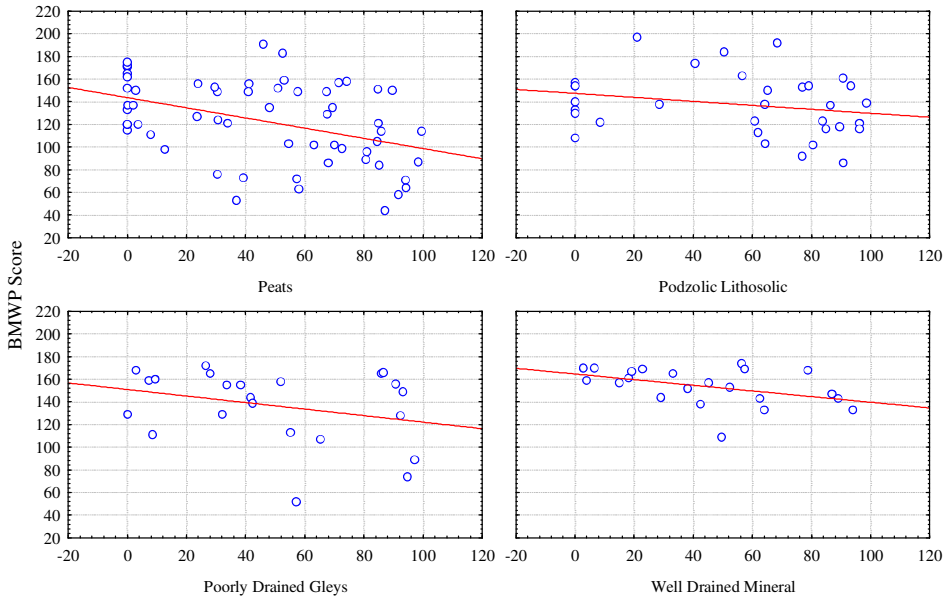


Fig. 35: Relationship between BMWP scores and % forest cover for sites draining sedimentary geology.

Community Diversity and Species Evenness

A trend of decreasing scores for both the Simpson and Margaley diversity indices was noted across the forest cover bands. However, the differences between bands was only significant for peat sites in both geological settings. The correlation with % forest cover was highly significant (P=0.001) for sites draining peat on sedimentary geology (Fig. 36). Species evenness varied greatly across sites and settings and none of the trends was statistically significant at P<0.05. However, the pattern, previously discussed, whereby some forested sites fell below the minimum values of the control sites at forest cover was repeated but the number of sites involved was much fewer.

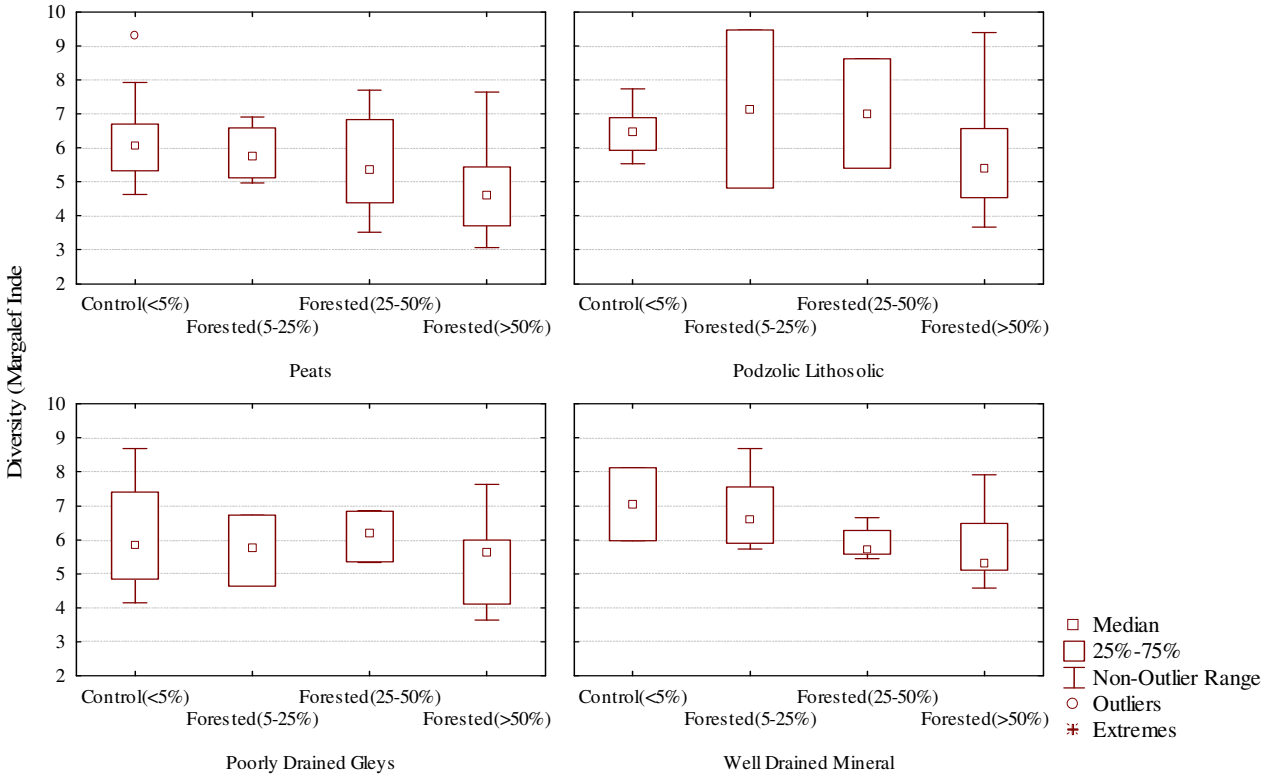


Fig. 36: Box plot of the Margalef diversity Index scores at sites on various soil types on sedimentary geology

Clustering of Biological, Chemical and Physical Metrics

The k-means algorithm was used to implement separate clustering of the 239 sites based on selected chemical, physical and biological (macroinvertebrate response) metrics. In each case the algorithm was asked to select four clusters.

The four biological clusters represented a gradient in the three metrics, group 1 having the highest richness and cluster 4 the lowest. The latter contained a large proportion of the impaired sites (31%, Table 12a). Thirty-three of these sites (44.5%) also grouped into the chemical cluster (cluster 1, Table 13), the centroids of which represented the most acidic conditions (Table 12b). Finally, 52 of the sites in the impoverished biological cluster 4 (66.6%, Table 13) also appeared in the physical cluster with the highest levels of percentage coniferous cover (cluster 3) (Table 12c). The location of the biological cluster is shown in Fig. 37. Sites within cluster 4 occurred in all regions.

Table 12: Centroid values for a) biological, b) chemical and c) physical clusters

(a)

Site	Taxon_Richness	Ephemeropteran_Abundance	Ephemeropteran_Richness
cluster1	42.53	108.13	6.14
cluster2	38.61	465.11	4.61
cluster3	33.23	55.91	3.93
cluster4	**25.59**	**37.75**	**1.14**

(b)

Site	H+(max)	Alk(min)	Al(max)	Na(max)	Ca(min)	Mg(min)	K(min)	Cl(max)	Colour(max)	Est. Organic Matter	DOC(max)	Organic Acid (max)
cluster1	**25.75**	**0.26**	**195.49**	**15.82**	**2.26**	**1.37**	**0.25**	**28.76**	**182.96**	**52.62**	**21.18**	**6.57**
cluster2	0.08	30.56	118.85	8.62	12.78	3	0.55	15.74	23.35	17.52	6.43	6.34
cluster3	3.47	6.26	257.16	11.26	3.89	1.6	0.42	20.48	64.12	34.62	14.23	10.92
cluster4	5.32	5.06	102.68	7.7	2.77	1.3	0.21	13.11	34.23	32.61	5.67	3.56

(c)

Site	%Coniferous	Slope	Elevation
cluster1	8.7	0.06	297.36
cluster2	13.83	0.16	292.02
cluster3	**71.24**	**0.06**	**187.81**
cluster4	11.31	0.04	113.4

Table 13: Numbers of sites loading into both a) biological and chemical clusters and b) biological and physical clusters.

(a)

		Chemical			
		1	**2**	**3**	**4**
Biological	**1**	2	10	11	29
	2	1	9	1	6
	3	19	7	14	20
	4	**33**	5	16	20

(b)

		Physical			
		1	**2**	**3**	**4**
Biological	**1**	11	6	17	20
	2	4	3	6	5
	3	12	5	20	21
	4	8	5	**52**	13

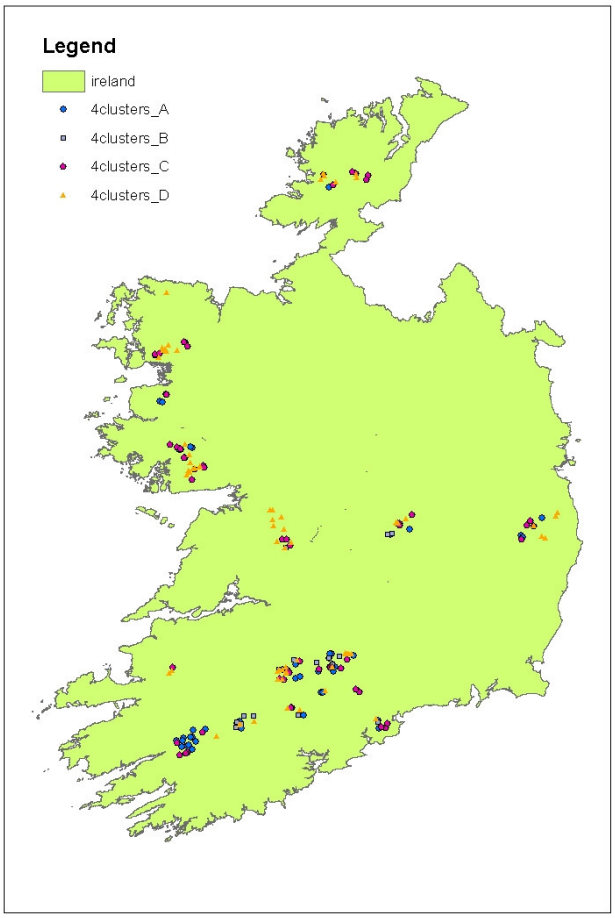


Fig. 37: Location of the biological site clusters.

Multivariate Analyses

Much of the analyses presented thus far deals with trends in individual metrics. The multivariate analyses examined the relationships between the invertebrate communities at each site. To better visualise the similarities between sites in terms of community composition NMDS plots, with Bray Curtis as the similarity measure, were generated using the key indicator groups, Ephemeroptera, Plecoptera and Trichoptera, on site groups within the selected geological settings. Plot were prepared for sites on peat and podzolic/lithosolic soils draining granite and peat sites on sedimentary geology. These were highlighted earlier as showing responses to forest cover.

Fig 38 is a plot of the sites draining peat on igneous/metamorphic geology. The control sites are positioned largely to the left of the plot. The three most acidic control sites (DWW4,8 & 9 –Co. Wicklow; DG9-Co. Galway) with low taxon richness sit on the right side of the group outline. At the other end of the plot sites M6, M7 and DWW6 (Knickeen, Co. Wicklow) represent sites with high total taxon richness and good representation of Ephemeroptera (richness and abundance) and Plecoptera. While there is, as expected, some overlap with the most acidic controls, sites in the highest forest cover bands fall largely on the right side of the plot. Those on the extreme right show impairment in a number of metrics. The only unimpaired sites within this region (DG21, 22 & 23-Owenwee River, Co. Galway) lie well within the main grouping of control sites.

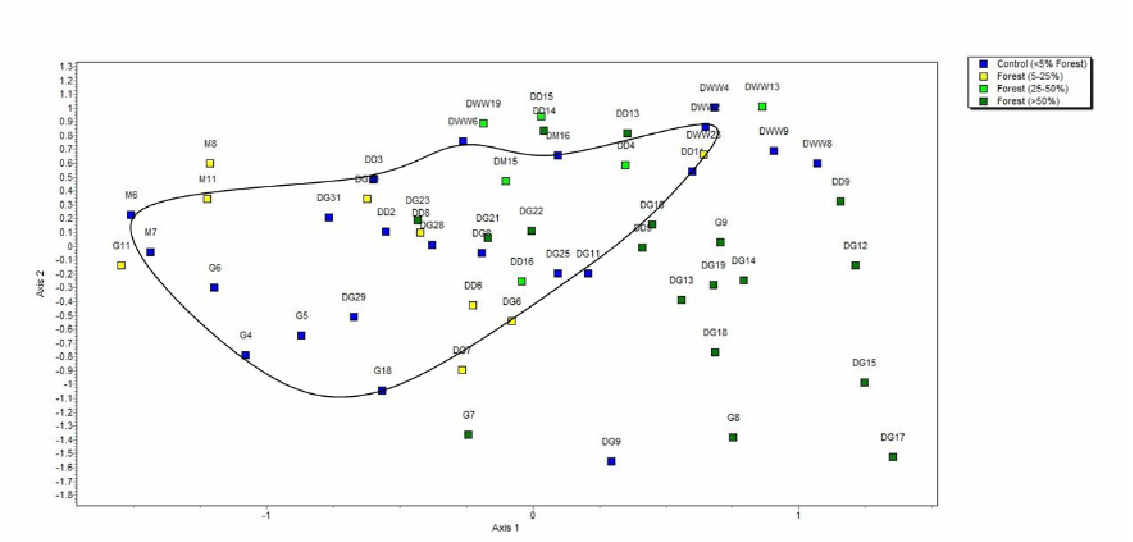


Fig. 38: NMDS plot of Bray Curtis similarity measure of EPT community composition at sites draining peat on igneous/metamorphic geology. Outline of control sites excluded the acidic outliers.

A similar picture emerged with respect to sites draining podzolic/lithosolic on igneous geology. Impaired sites within the three forestry bands plot on the right side of Fig. 39. In contrast, the unimpaired afforested sites (DWW15 , 16 and 17) with high EPT taxon richness and abundances plot to the left.

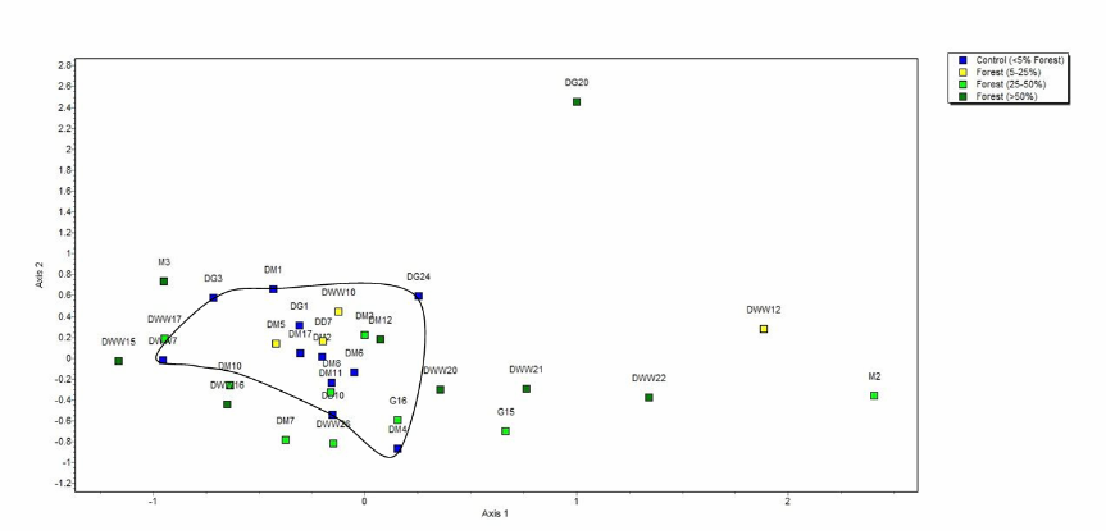


Fig. 39: MDS plot of Bray Curtis similarity measure of EPT community composition at sites draining podzolic/lithosolic soils on igneous/metamorphic geology.

The control peat sites on sedimentary geology form a closer cluster than seen on igneous geology (Fig. 40). Here sites with low EPT richness and abundance largely plot outside the grouping of control sites, most of which show some impairment in the metrics applied earlier. The frequency of impacted sites occurring outside of the control site grouping increases with increasing forest cover.

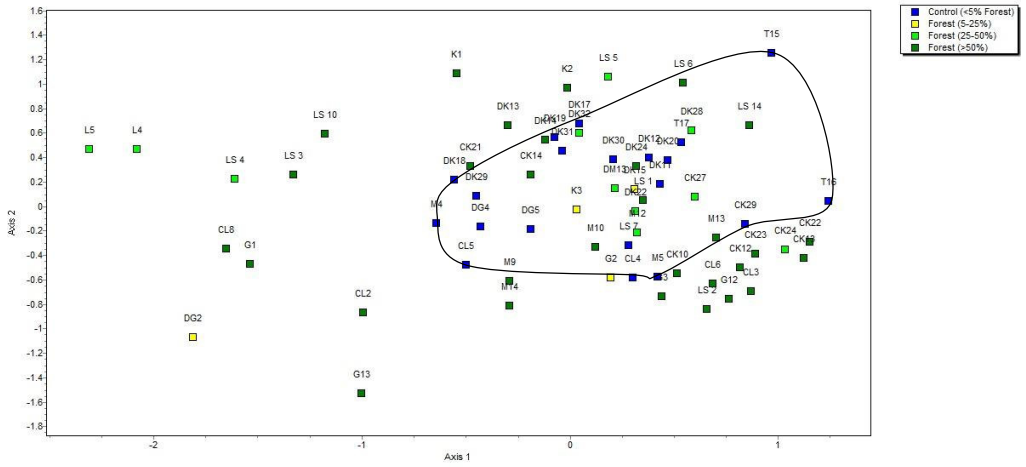


Fig. 40: MDS plot of Bray Curtis similarity measure of EPT community composition at sites draining peat on igneous/metamorphic geology.

Evaluation of the Degree of Biological Impairment

Five metrics (ephemeropteran richness, abundance *Baetis* spp., trichopteran richness, evenness and diversity indices) which were not autocorrelated were selected to evaluate potential impairment. The metrics selected target known indicator taxa as well as abundance, evenness and diversity aspects of the community. As outlined in the methods section, sites with metric values below two standard deviations of the mean of the control site values were considered impaired. Table 14 shows the number of sites in each geological setting that shows impairment for 1 to 5 metrics and gives an overall estimate of the number of sites that fail on two or more metrics.

Table 14: Estimation of the number of impaired sites as indicated by low metric scores for 5 metrics in each geological and forest cover setting.

Geological setting	1	2	3	4	5	Total Sites	% Sites with >2 impacted sites
Igneous/Metamorphic							
Peat							
5-25%	22.2	22.2	0	33.3	0	9	55.6
25-50%	33.3	16.7	16.7	16.7	0	7	50.0
>50%	5.6	22.2	16.7	11.1	27.8	17	77.8
Podzolic/Lithosolic							
5-25%	0	75.0	0	0	0	4	*
25-50%	11.1	11.1	33.3	22.2	0	9	66.7
>50%	25.0	25.0	37.5	0	0	8	62.5
Sedimentary							
Peat							
5-25%	0	25.0	50.0	0	0	4	*
25-50%	27.3	18.2	9.1	27.3	9.1	11	63.6
>50%	6.7	10.0	20.0	33.3	20.0	30	83.3
Podzolic/Lithosolic							
5-25%	0	0	50.0	0	0	2	*
25-50%	0	0	50.0	0	0	2	*
>50%	22.7	22.7	18.2	9.1	18.2	22	68.2
Poorly drained Gleys							
5-25%	0	33.3	0	0	0	3	*
25-50%	14.3	28.6	0	0	0	7	28.6
>50%	9.1	18.2	0	18.2	18.2	11	54.5

- low replication

The % impaired sites increased across the forest cover bands on igneous/metamorphic geology. The same applied for peat sites on sedimentary geology. Podzolic/lithosolic sites on sedimentary geology recorded significant impairment in the >50 forest band. It should however be noted that replication was low in the other two forest bands. Some 55% of sites on poorly drained gleys in the >50 forest band failed on more than two metrics. None of the well drained mineral sites failed on more than two metrics. Four sites recorded low ephemeropteran abundances, however there was low replication of the control sites.

Comparison of Source and Downstream Communities

As outlined in the methods a number of forested streams were sampled above and below the forest. Control sites were sampled at equivalent points. No significant differences were detected between source and downstream sites in pairwise comparison statistics using macroinvertebrate metrics (richness and abundance data for total taxa, Ephemeroptera, Plecoptera, Trichoptera and Coleoptera). However, community differences were revealed by the multivariate analyses.

Fig. 41 shows the NMDS plot based on Sorensen similarity measure. It clearly shows that the control downstream sites (blue) were distinctly different from their sources (red). It also indicates that the macroinvertebrate communities from forested downstream sites (green) were similar to the sources from both forested (yellow) and control sites (red).

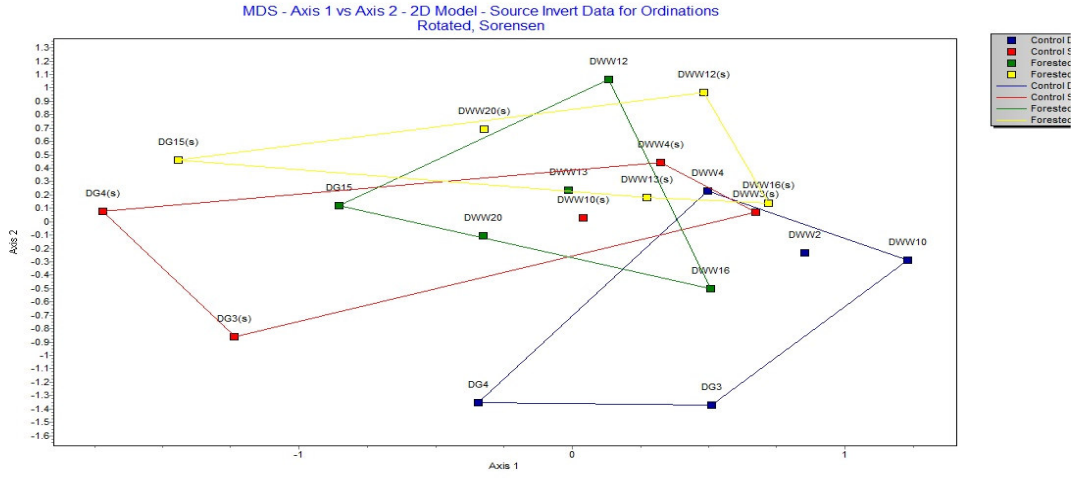


Fig. 41: MDS plot of Sorensen similarity measure of community composition at sites upstream and downstream of forestry and at similar locations on control sites.

Longitudinal Variation in Macroinvetebrate Metrics

Several catchments were sampled at several sites to illustrate longitudinal changes in macroinvertebrate community and to evaluate the potential distance downstream that a forest effect might be detected. The example presented here is the King's catchment, Co. Wicklow. Some of the forested headwater sites had poor total taxon richness and ephemeropteran richness compared to the control sites (Table 15) and this was maintained well down the system to Site Kings1 (Fig. 42).

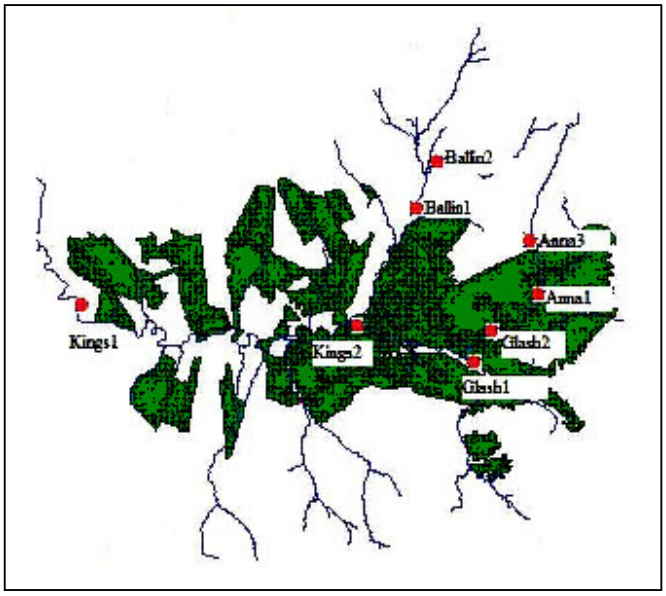


Fig. 42 Sites sampled along the length of the King's River, Co. Wicklow.

Table 15: Locaton of sites sampled in the King's catchment and recorded metric scores for total taxon and ephemeropteran richness

River		% Forest Cover	Taxon Richness	Ephem. Richness
Annalecka Brook	ANNA3	0	25	2
	ANNA1	34.71	26	2
Ballinagee River	BALLIN2	0	24	5
	BALLIN1	0	34	5
Glashaboy River	GLASH2	40.9	35	2
	GLASH1	65.72	35	4
King's River	KINGS2	36.63	22	3
	KINGS1	28.26	25	2

Ephem.=Ephemeroptera

Further Evaluation of Potential Longitudinal Patterns in Macroinvertebrate Recovery from Forest Effects

Forested sites in counties Wicklow (Vartry stream, DWW20 / VART1) and Cork (Foherish river, DK26 / FOHER1) which recorded a paucity of Ephemeroptera during the spring 2007 sampling season were re-visited in spring 2008. In each case a nearby control/reference site was also sampled, the Bohill river (DK23 / BOHIL1) in Co. Cork and the nearby non-forested tributary of the Vartry catchment, Co. Wicklow (DWW10 / VAR1). The controls were selected to be comparable in terms of geology, soil type, elevation, catchment area, slope, aspect and catchment size. The two paired streams were then sampled approximately 800 metres from source and every 500 metres thereafter over a two kilometre stretch of the streams. The Cork sites drained areas of Old Red Sandstone, while all but one of the Wicklow sites were situated on Palaeozoic sediments. Soils types within the catchment of each paired stream were also comparable. The sampling sites where located between 200m-360m asl. Sites 1 in all cases were located at >300m; Sites Nos. 2 between 275-300m; Sites Nos. 3 between 250-275m; Sites Nos. 4 below 250m. The forested sites in Wicklow were coded WKF1, WKF2, WKF3 and WKF4, while open (control) sites were labelled WKO1, WKO2, WKO3 and WKO4. A similar site designation was used for the Cork sites (Figs. 43 & 44).

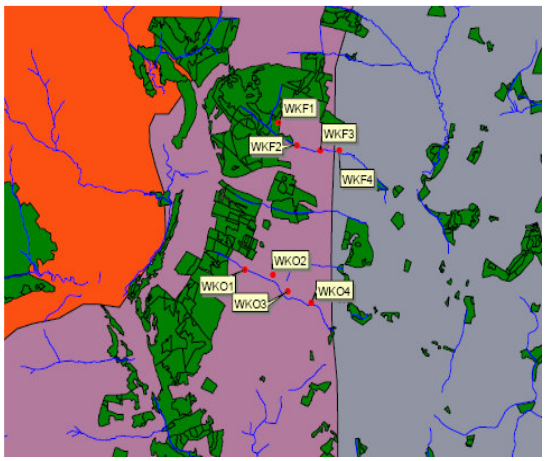


Fig. 43: Location of Wicklow forested and control sites

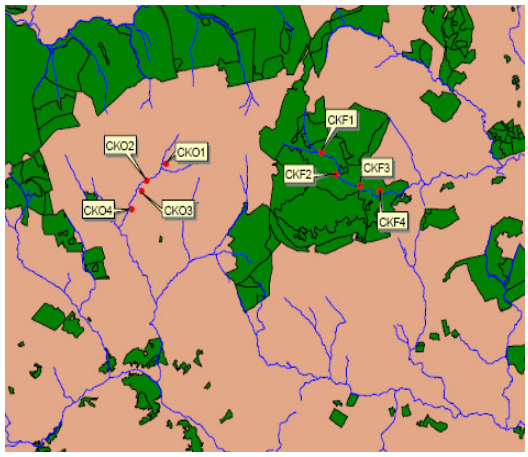


Fig. 44: Location of Cork forested and control sites.

The community composition was dominated by Chironomidae and other dipteran larva in the Wicklow (WKO-64%; WKF-87%) and Cork sites (CKO-57%; CKF-63%). The Ephemeroptera represeneted a lower percentage of the fauna at the Wicklow forested sites (2%) compared to the control sites (10%). The Tricoptera varied little across sites ranging from 4-7%. The Plecoptera accounted for 15% (CKF) to 11% (CKO) of the total abundances in the Cork sites compared to 2% (WKF) and 3% (WKO) sites. Crustaceans were particularly abundant in non-forested sites; WKO and CKO accounting for 9% and 8% of the total abundances. However, they accounted for only 4% of the fauna in forested sites in Cork and where absent from forested sites in WKO. In Wicklow, non-forested sites supported between 7 and 25 taxa, while forested sites had between 4 and 14. In Cork, non-forested sites recorded between 14 and 24 taxa, while between 8 and 21 taxa were found at forested sites. The differences between the paired control and forested sites was significant (Wicklow – Wilcoxon test: $Z=-2.598$, $P<0.01$; Cork – Wilcoxon test: $Z=-2.096$, $P<0.05$) (Fig. 45).

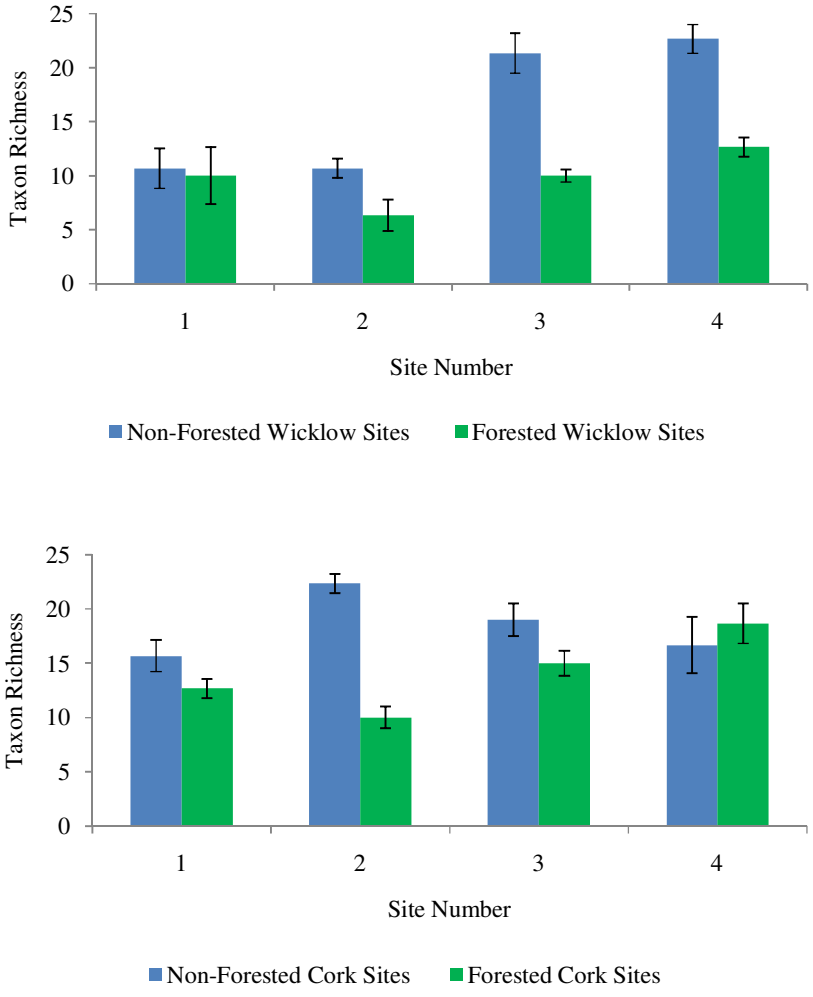


Fig. 45: Mean taxon richness at forested and non-forested sites in Wicklow and Cork.

Low ephemeropteran richness was a common feature of the first 2 sites in each forested catchment (Figs. 46 & 47). The only species present was *Baetis rhodani*. While this indicated a slight recovery in terms of species richness in comparison to the original sampling period of the study (April-May 2007 - in which no mayfly were found), the abundances were far lower than those for the corresponding controls in both Cork and Wicklow. Wilcoxon paired Test, showed there was a significant differences in the mean ephemeropteran richness between forested and non-forested sites in Cork ($Z=-0.2366$; $P<0.05$) and Wicklow ($Z= -3.781$; $P<0.001$). However, in Cork ephemeropteran richness differed significantly only between the first three forested and control site pairs. The fourth, located almost 2.5km from the source, and still within close proximity to the forestry, was not showing impairment (Fig. 45).

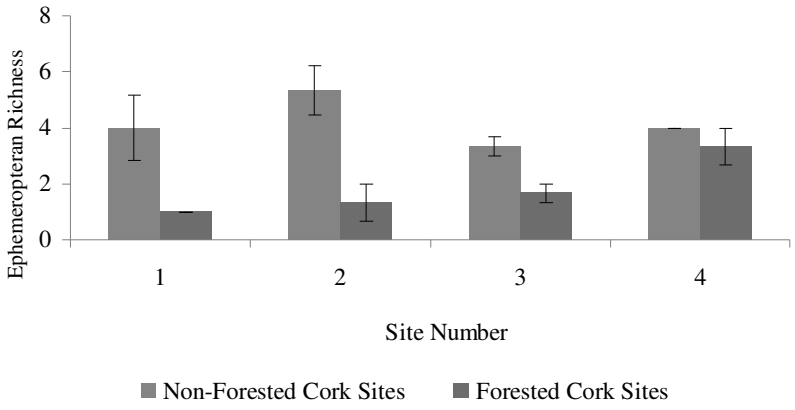


Fig. 46: Mean ephemeropteran richness at forested and non-forested sites in Cork

In Wicklow the significantly lower ephemeropteran richness at the forested sites persisted down to Site 4. (Kruskall-Wallace ANOVA, $P<0.05$) (Fig. 47). Trichopteran richness was also significantly higher at the control sites in Wicklow ($Z= -3.776$' $P<0.001$) and Cork ($Z= -3.530$, $P <0.001$) but here again the differences between the site pairs was eliminated by Site 4 in Cork.

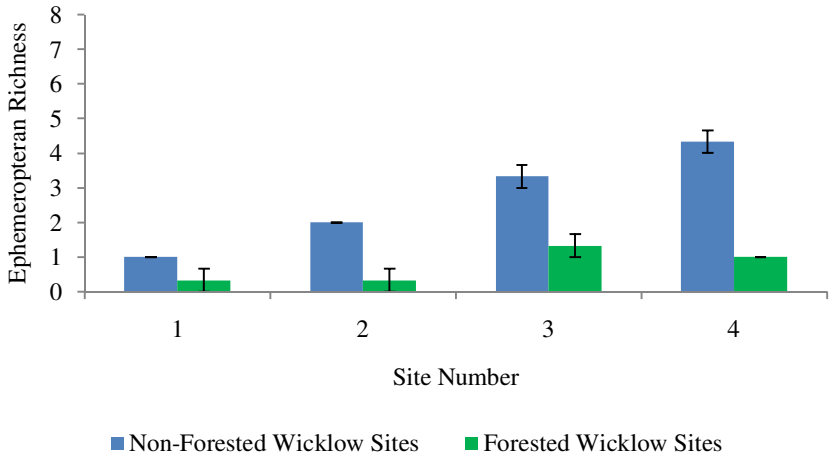


Fig. 47: Mean ephemeropteran richness at forested and non-forested sites in Wicklow.

The longitudinal pH profile at the time of sampling is illustrated (Fig. 48). Although all sites were circum-neutral there was at least a 0.5 unit of differences between the control and forested site pairs and the differences was greatest for the first two sites.

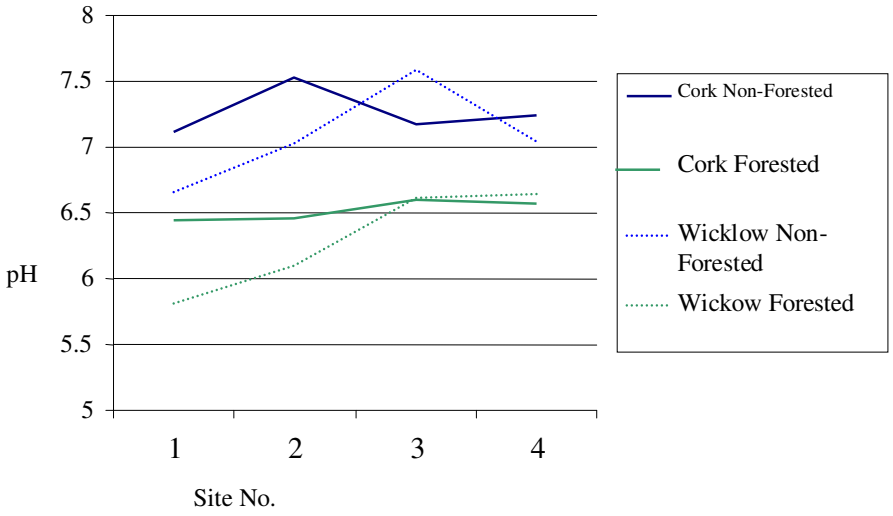


Fig. 48: Longitudinal pH profile for sites sampled in Wicklow and Cork.

3.3 FISH

As outlined in the methods sites were selected on a paired-catchment basis. To further ensure that all site pairings were comparable, the coverage of each habitat type (riffle (P=0.38), glide (P=0.402), pool (P=0.175)), depth (P=0.822), width (P=0.705), wetted area (P=0.812), conductivity (P=0.492) and time fishing (effort – m^2/min, P=0.12) were examined between pairs. In each case there were no significant differences (Wilcoxon Ranked Sign Test, P>0.05) detected, making all pairings comparable for further analysis.

The results of the Wilcoxon Signed Ranked Test on abundance of salmonids at the two site grouping are given (Table 15). Overall the catch of trout was higher than salmon (Figs. 49 & 50), few salmon were caught in the Wicklow sites. The Galway sites recorded the highest catches of salmonids. Total salmonid catch differed significantly between the paired control and forested sites with the lowest numbers at the forested sites. Significant differences were also detected for total salmon, trout and salmon fry (Table 16). In all cases, there were fewer individuals at forested sites (Figs. 49 & 50). The differences were not significant for adult trout or adult salmon.

Table 16: Wilcoxon Ranked Sign Test results for comparison of salmonid abundances between non-forested and forested sites.

	Z-Value	P-Value
Total Salmonid Abundance	-2.939	0.003*
Total Trout Abundance	-1.731	0.083
Trout Adult Abundance	-1.168	0.243
Trout Fry Abundance	-2.049	0.041*
Total Salmon Abundance	-2.194	0.028*
Salmon Adult Abundance	-1.55	0.121
Salmon Fry Abundance	-2.366	0.018*

* Significant to P = 0.05
Wilcoxon Sign Ranked Tests

Fig. 49: Mean abundances of salmon captured at forest and non-forested sites. Standard error bars included.

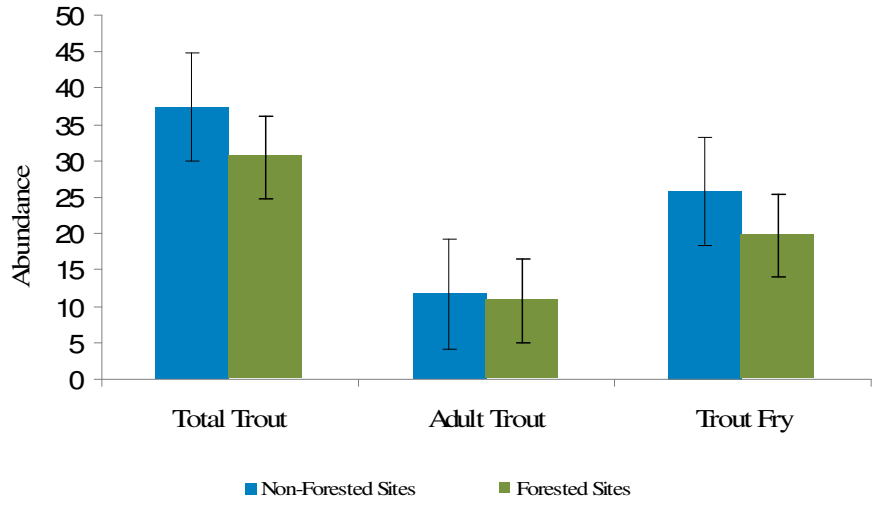


Fig. 50: Mean abundances of trout captured at forest and non-forested sites. Standard error bars included.

Salmonid densities (fish/m^2) were also compared between the paired control and forested sites. The results were similar to those described for fish catch. Total trout density ($Z = 2.45$, $P = 0.014$) and trout fry ($Z = 2.50$, $P = 0.0122$) density differed between the two site groups.

The differences were significant for adult trout. Total salmon ($Z = 2.73$, $P = 0.006$) and fry ($Z = 3.54$, $P = 0.0003$) density was significantly lower in the forested sites.

The length frequency distribution of salmonids across all sites is shown in Fig. 51. The populations were generally dominated by 1+ fish ranging in length from 9 to 15 cm. The numbers of larger fish were highly variable across sites. Fry numbers were generally lower at the forested sites. An evaluation of growth differences between control and forested sites was undertaken for fry. The data were examined on a regional basis to incorporate any natural variation in growth. No significant differences were detected in any of the regions or across the total dataset (Mann-Whitney, $P>0.05$, Table 17).

Table 17: Mann-Whitney test statistic results ($P>0.05$) for a comparison of the length of trout fry in control and forested sites in four regions.

Region	Mann-Whitney Test Statistic
Wicklow	0.260
Mayo	0.279
Galway	0.314
Donegal	0.082

Several sites were noted for their low abundances or paucity of salmonids (Table 18). These sites were all highly forested with the exception of one control site on the Cloghoge river in Co. Wicklow (CLOG1). Kelly-Quinn *et al.* (1996b) suggest that the combination of site elevations >400 m.a.s.l along with steep slopes can exclude salmonid fish from streams. However, while several of the sites in Table 18 were approaching the elevation cut-off, the slopes were not steep. None of the sites had any known barriers to fish movement.

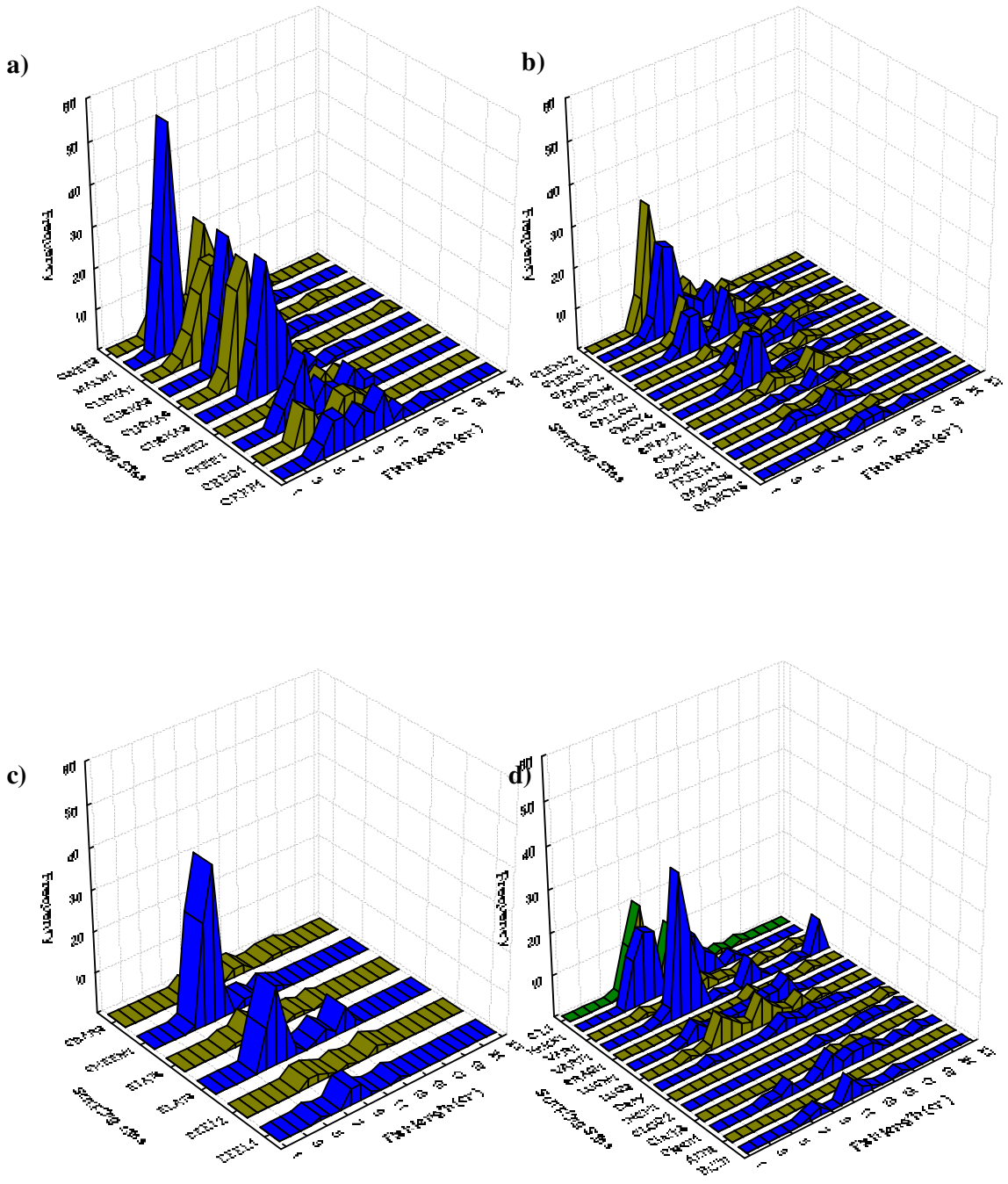


Fig. 51: Fish population structure of the salmonids caught in the control (blue) and forested (green) sites in the Galway (a), Mayo (b), Donegal (c) and Wicklow (d) regions. The stream pairs are maintained alongside each other for comparative purposes.

Table 18: Sites with absences of salmonid fish.

All Fish Absent	Fry Absent	Adult Fish Absent
ANNA1 (25-50% Forest)	ANNA1 (25-50% Forest)	ANNA1 (25-50% Forest)
	GAMON5 (>50% Forest)	CROE1 (5-25% Forest)
	INCH1 (25-50% Forest)	
	GMOY1 (>50% Forest)	
	GLASH2 (25-50% Forest)	
	CORRIB1 (>50% Forest)	
	CORRIB2 (25-50% Forest)	
	SRAG1 (5-25% Forest)	
	CLOG1 (Control <5% Forest)	

4. DISCUSSION

This project set-out to investigate the presence and extent of any acidification associated with coniferous forestry in Ireland and to assess the risk of impact with respect to different geological settings. In the selection of forested sites it was aimed to represent a combination of the risk factors in terms of catchment cover and acid-sensitive geology that were perceived to have the greatest potential for acidification. The large number of sites selected allowed for good spatial coverage but it did limit the amount of water sampling that could be undertaken. The aim was to sample each site at variable flow conditions, from low flow to flood. It was however not possible to obtain flood samples for all sites as a result of their geographic spread and remote locations. Furthermore, it was often difficult to ascertain the stage in the hydrograph represented on any one date. Nevertheless, within any one region a good representation of control and forested sites were sampled within the same timeframe and usually under the same flow/weather conditions.

The pH results analyses suggested that most of the streams were episodically acidic with a small group more likely to be circum-neutral. Overall, the pH results indicated increased acidity at some sites associated with forestry on peat and podzolic/lithosoilic soils on both igneous/metamorphic and sedimentary geology and to some extent on poorly drained gleys. Two components of these results require clarification. Firstly, while a small number of the control sites, especially on peat/granite, recorded minimum pH values as low as some of the forested sites the frequency of low pH readings was substantially higher among some groups of forested sites. So the critical issue may be that the frequency and duration of acid pulses can be higher in some forested catchments. Previous intensive monitoring of acid pulses in a heavily afforested stream in the Wicklow mountains alluded to this (Kelly-Quinn, Tierney & Bracken, 1997). The current dataset, unfortunately, does not have sufficient data to further test this hypothesis. This association emerged when forest cover in the site catchment exceeded 25-30%. Factors controlling the severity and duration of acid pulses require further research to better target measures. The second issue to consider is whether the current results suggest a forest-cover threshold above which the risk of acidification increases. Certainly the minimum pH for both peat and podzolic/lithosolic sites on igneous/metamorphic geology began to fall below the lower limit of the control sites when forest cover exceeded values in the region of 25%. The same applied to peat sites draining sedimentary geology. Sites on podzolic/lithosolic soils on sedimentary geology did not have

minimum pH values below the lower limit of the control sites until forest cover exceeded 60%. A similar threshold might be applied to sites on poorly drained gleys but the level of replication is too low for this decision.

It is important to point out that not all sites within the high forest-cover bands had low minimum pH. It may be that the sampling did not capture the peak of the acidity or alternatively the sites are buffered against pH change. Indeed, when alkalinity was examined, many sites, particularly those on peat/podzolic/lithosolic on sedimentary geology, had alkalinity values well above 20 mg/l $CaCO_3$. Further analyses of 57 of these sites confirmed that 78% have some alkaline sub-soils or carbonate geology within the catchment. Interestingly, a number of sites with maximum alkalinity >20 mg/l $CaCO_3$ recoded high flow alkalinity values of close to zero. This was also mirrored in the SDI results for these sites. Evaluation of the differences in the flow pathway between base and flood conditions in forested catchments is clearly required to better understand factors controlling buffering potential. Overall, the greatest variation in alkalinity was recorded on sedimentary geology which may relate to more complex geology with, as already mentioned, occurrences of some carbonate soils or rocks among the largely acid-sensitive geology. More detailed spatial and temporal analyses of the chemical characteristics of waters draining sedimentary geology is required for more precise mapping of acid sensitivity and this should be an element of future research projects.

The presence of forestry tended to depress site pH and alkalinity. Calculations suggested that dilution makes a variable contribution to loss of alkalinity and in many cases the forested sites showed a slightly higher % value. Anion titration was detected in all events examined. The principal contributors were organic acids and sulphate. Excess sulphate only made a contribution in the Wicklow sites and at one site in Galway. The contribution of nitrate across all sites was insignificant. The contribution of sea salts to acidification was low and only one significant sea-salt event was detected at one site in Galway. Similar variability in contributing variables has been reported by Kowalik *et al.* (2007). Overall, it is likely that a combination of dilution and higher organic acidity concentrations, and occasionally excess sulphate, contributed mostly to the differences in acidity between control and heavily afforested catchments. Reasons for the differences in organic acidity are unclear and may relate to the effects of patterns of drying and wetting and other climatic factors associated with forest soils (Raveh and Avnimelech, 1978; Worrall, Burt and Adamson, 2004). This

represents another key knowledge gap and if addressed may help to develop focused forestry practices that minimize the risk of acidification. Indeed, given that the focus of the current study was on mature forestry, we need to determine the acidification risk associated with each of the key forestry practices from site preparation to felling.

In terms of the macroinvertebrtae analyses the control sites draining sedimentary sites were more productive in terms of biomass than the igneous/metamorphic sites but the ranges for taxon richness were similar. The higher total macroinvertebrate abundances at the sedimentary sites could be largely attributed to the Ephemeroptera and Chironomidae. This may relate to slightly higher pH and cation concentrations in the former. In both geological settings the Ephemeroptera were reduced in abundance at sites in the two highest forest cover bands, a factor of increasing pH. The sensitivity of the Ephemeroptera to acidification is well established through field observations as well as stream microcosm experiments (e.g. Courtney &Clements, 1998) Interestingly, at the igneous/metamorphic sites the reduction in ephemeropteran abundance was largely balanced by an increase in the numbers of Plecoptera. This did not occur at the sites draining sedimentary geology and consequently overall abundance declined gradually across the forest cover bands. Most the analyses in relation to forestry effects was performed separately for the two geological settings to avoid any confounding effects of differences in taxon abundances.

Overall, the biological data largely mirrored the trends for the acidity variables. In fact pH was the key variable structuring the community, a feature that is commonly reported in the literature (Ormerod and Edwards, 2006; Sutcliffe and Carrick, 1973). The study on the source communities illustrated two key findings. Firstly, the macroinvertebrate communities found in control source streams differ substantially from those at sites located further downstream and therefore one must be cautious about making comparisons between sites upstream and downstream of forestry if source sites are included. Secondly, the study showed that the macroinvertebrates communities downstream of the forest were more similar to the source sites than the downstream counterpart on the non-forested stream. This implies some impact of forestry inputs, most likely relating to acidity.

Several metrics (taxon richness, ephemeropteran richness, abundance of baetids, EPT richness, diversity indices), which showed a strong relationship with pH, were also shown to vary significantly across the forest cover bands or to correlate with % forest cover bands.

Ephemeroptera were absent from several sites in the >50 forest bands on peat and podzolic/lithosolic soils. A striking finding was that the number of sites with low ephemeropteran richness and abundance increased across the forest cover bands. The analyses on the individual metrics highlighted similar % forest thresholds for risk of impact as described for the hydrochemistry. When a selection of non-correlated metrics were combined it was clear that a large proportion of sites in the >50% cover band, and a smaller number of the 25-50% band, had some degree of impairment. These same sites were shown to have a different invertebrate community to the control sites as indicated on the NMDS plots. The implications of the detected impairment for overall ecological health and functioning requires further research.

It should be highlighted that, as for pH, not all sites within the high forest cover bands showed impact. Most of those that were impacted recorded minimum pH values well below 5.0 and alkalinity values below or close to zero but this was not consistent across the soil/geological categories. In fact, some of the non-impacted sites also became substantially acidic. Further research must target these sites to better understand the mechanisms governing responses to acid impact under naturally acidic conditions. It is likely that both the degree and duration of acidity are important factors.

Some limited analyses of the season and longitudinal extent of forest impact was undertaken. The results suggest that impact may be seasonal and that recovery in some of the metrics (e.g. baetid numbers) takes places. The potential for seasonal recovery may be dependent on climatic factors *viz.* severity, duration and frequency of precipitation leading to acid pulses; but also life history patterns of the biota (e.g. baetids). The limited data also suggest that recovery may occur over a shorter distance on sedimentary geology than on igneous/metamorphic geology but this is likely to be controlled by many interacting factors that change with distance from the source, such as catchment size, forest cover, inputs from other sub-catchments as well as geology/soils. The longitudinal responses to forestry as a land-use activity needs to be addressed by further research.

The fish analyses was limited to 19 paired sites with similar habitat but did highlight significant differences in fish catch and density between the control and forested groups. This difference was mainly attributed to low numbers of fry (salmon and trout) in the forested streams. Low recruitment in forested streams is most likely to be related to pH as previously

highlighted by Kelly-Quinn, Tierney and Bracken (1993) in upland streams in Co. Wicklow. However, there may be other contributing factors such as discharge and availability of food.

In conclusion this study has addressed the objectives as set out in the introduction. It has detected increases in acidity and biological impairment associated with forest plantations. The risk appears to be a factor of soil geology/soil and increasing catchment cover with the greatest impact occurring above a coniferous forest cover of 50%. The sources and pathways of acid inputs needs to be better clarified and related to forest activities to allow further refinement of the programme of measures.

5. REFERENCES

Allott, N., Mills, W.R.P., Dick, J.R.W., Eacrett, A.M., Brennan, M.T., Clandillon, S., Phillips, W.E.A., Critchley, M., and Mullins, T.E. (1990) *Acidification of Surface Waters in Connemara and South Mayo. Current Status and Causes*, Dublin: duQuesne Limited, Economic and Environmental Consultants, pp 61.

Allott, N., Brennan, M., Cooke, D., Reynolds, J., and Simon, N. (1997) Stream chemistry, hydrology and biota, Galway-Mayo region. In: *A study of the effects of stream hydrology and water chemistry in forested catchments on fish and macroinvertebrates*, AQUAFOR Report 4, COFORD, Dublin, Ireland.

Anon. (2005) *The characterisation and analysis of Ireland's river basin districts in accordance with Section 7(2 & 3) of the European Communities (Water Policy) Regulations 2003 (SI 722 of 2003).* National Summary Report (Ireland).

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.

Barbour, M.T., Gerritsen, J. and Snyder, B.D. (1997) *Revision to Rapid Bioassessment Protocols for Use in Streams and Rivers.* EPA 841-D-97-002, Office of Water, United States Environmental Protection Agency, Washington, D.C., USA.

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.

Bowman, J. (1991) Acid sensitive surface waters in Ireland. Environmental Research Unit, Dublin. pp 321.

Clenaghan, C., Giller, P.S., O'Halloran, J., and Hernan, R. (1998) Stream macroinvertebrate communities in a conifer-afforested catchment in Ireland: relationships to physico-chemical and biotic factors. *Freshwater Biology* 40: 175-193.

Courtney, L.A. and Clements, W.H. (1998) Effects of acidic pH on benthic macroinvertebrate communities in stream microcosms. *Hydrobiologia* 379: 135-145.

Economou, A.N., Schmutz, S., Melchior, A. and D. Heidvogel (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.

Evans, C.D., Reynolds, B., Hinton, C., Hughes, S., Norris, D., Grant, S. and Williams, B. (2008) Effects of decreasing acid deposition and climate change on acid extremes in an upland stream. *Hydrology and Earth System Sciences* 12: 337-351.

Farrell, E.P. (1995) Atmospheric deposition in maritime environments and its impact on terrestrial ecosystems. *Water, Air, & Soil Pollution*, 85: 123-130.

Giller, P., O'Halloran, J., Kiely, G., Evans, J., Clenaghan, C., Hernan, R., Roche, N., and Morris, P. (1997) An Evaluation of the Effects of Forestry on Surface Water Quality and Ecology in Munster. In *A study of the effects of stream hydrology and water quality in forested catchments on fish and invertebrates* AQUAFOR Report 2, COFORD, Dublin, Ireland: COFORD, pp 77.

Harriman, R., and Morrison, B.R.S. (1982) Ecology of streams draining forested and non-forested catchments in an area of central Scotland subject to acid precipitation. *Hydrobiologia*, 88: 251-263.

Harriman, R., Anderson, H., and Miller, J.D. (1995) The role of sea-salts in enhancing and mitigating surface water acidity. *Water, Air, & Soil Pollution*, 85: 553-558.

Hartigan, J.A., Wong, M.A., (1979) Algorithm AS 139: A k-means clustering algorithm. *Applied Statistics* 28: 100-108.

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.

Hornung, M., Le-Grice, S., Brown, N., and Hughes, S. (1990) The role of geology and soils in controlling surface water acidity in Wales. In *Acid Waters in Wales* (R.W.

Edwards, J.H. Stoner, and A.S. Gee, eds.), Dordrecht: Kluwer Academic Publishers, pp 55-66.

Hornung, M., Bull, K.R., Cresser, M., Ullyett, J., Hall, J.R., Langan, S., Loveland, P.J., and Wilson, M.J. (1995) The sensitivity of surface waters of Great Britain to acidification predicted from catchment characteristics. *Environmental Pollution*, 87: 207-214.

Jenkins, A., Cosby, B.J., Ferrier, R., Walker, T.A.B., and Miller, J.D. (1990) Modelling stream acidification in afforested catchments. An assessment of the relative effects of acid deposition and afforestation. *Journal of Hydrology*, 120: 163-181.

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

Johnson, M.J., Giller, P.S., O'Halloran, J., O'Gorman, K. and 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.

Kahl, J.S., Norton, S.A., Haines, T.A., Rochette, E.A., Heath, R.H. and Nodvin, S.C. (1992) Mechanisms of episodic acidification in low-order streams in Maine, USA. *Environmental Pollution* 78: 37-44.

Kelly-Quinn, Tierney, D. and Bracken, J.J. (1993) Survival of salmon eggs planted in upland streams. *Aquaculture and Fisheries Management* 24: 791-796.

Kelly-Quinn, M., Tierney, D., Coyle, C. and Bracken, J.J. (1996a) Factors affecting the susceptibility of Irish soft-water streams to forest-mediated acidification. *Fisheries Management and Ecology* 3, 287-301.

Kelly-Quinn, M., Tierney, D., Roche, W. and Bracken, J.J. (1996b) Distribution and abundance of trout populations in moorland and afforested upland nursery streams in Co. Wicklow. *Biology and Environment*, *Proceeding of the Royal Irish Academy.* 96B, No. 3 127-140.

Kelly-Quinn, M., Tierney, D. and Bracken J.J. (1997) Stream Chemistry, Hydrology and Biota, Wicklow Region. In: *A study of the effects of stream hydrology and water quality in forested catchments on fish and invertebrates*. AQUAFOR Report 3, COFORD, Dublin. 92pp. ISBN:0 9523938 4 0

Kowalik, R.A., Cooper, D.M., Evans, C.D. and Ormerod, S.J. (2007) Acidic episodes retard the biological recovery of upland British streams from chronic acidification. *Global Change Biology* 13: 2439-2452.

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.

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.

Ormerod, S.J., Wade, K.R., and Gee, A.S. (1987) Macrofloral assemblages in upland Welsh streams in relation to acidity, and their importance to invertebrates. *Freshwater Biology*, 18: 545–557.

Ormerod, S.J., and Wade, K.R. (1990) The role of acidity in the ecology of Welsh lakes and streams. In *Acid Waters in Wales* (R.W. Edwards, J.H. Stoner, and A.S. Gee, eds.), Dordrecht: Kluwer Academic Publishers, pp 95-109.

Ormerod, S.J., Rutt, G., Weatherley, N.S., and Wade, K. (1991) Detecting and managing the influence of forestry on river systems in Wales: Results from surveys, experiments and models. In: *Irish Rivers: Biology and Management* (M.W. Steer, ed.) Royal Irish Academy, Dublin. pp 163-184.

Ormerod, S.J. and Edwards, R.W. (2006) The ordination and classification of macroinvertebrate assemblages in the catchment of the River Wye in relation to environmental factors. *Freshwater Biology*, 17: 533-546.

Raveh, A. and Avnimelech, Y. (1978) The effect of drying on the colloidal properties and stability of humic compounds. *Plant Soil*, 50: 545-552.

Rees, R.M., and Ribbens, J.C.H. (1995) Relationships between afforestation, water chemistry and fish stocks in an upland catchment in south west Scotland. *Water, Air, & Soil Pollution*, 85: 303-308.

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*, 7: 433-455.

Stoner, J., Gee, A., and Wade, K. (1984) The effects of acidification on the ecology of streams in the upper Twyi catchment in West Wales. *Environmental Pollution, A*, 36: 125-157.

Sutcliffe, D.W. and Carrick, T.R. (1973) Studies on mountain streams in the English Lake District I. pH, calcium and the distribution of invertebrates in the River Duddon, *Freshwater Biology*, 3:, 437-462.

Tierney, D., Kelly-Quinn, M., and Bracken, J.J. (1998) The faunal communities of upland streams in the eastern region of Ireland with reference to afforestation impacts. *Hydrobiologia*, 389: 115-130.

Waters, D., and Jenkins, A. (1992) Impacts of afforestation on water quality trends in two catchments in mid-Wales. *Environmental Pollution*, 77: 167-172.

Worrall, F., Burt, T. and Adamson, J. (2004) Can climate change explain increases in DOC flux from upland peat catchments? *Science of the Total Environment*, 326: 95-112.

Wright, J.F. (1995) Development and use of a system for predicting the macroinvertebrate fauna in flowing waters. *Australian Journal of Ecology*, 20: 181-197.

APPENDIX A

Location of the 239 sampling sites

Main system	Invertebrate Site Code	Water Chemistry Site Code	County	Grid Reference	Easting	Northing
Trib of River Blackwater	CK1	CK1	Cork	W 717 958	171785	95834
Caher River	CK10	CK10	Cork	W 452 863	145243	86306
Dripsey River	CK12	CK12	Cork	W 416 857	141683	85758
Dripsey River	CK13	CK13	Cork	W 415 858	141503	85862
Trib of Caher River	CK14	CK14	Cork	W 444 869	144463	86909
Glennaharee River	CK15	CK15	Cork	W 459 889	145995	88996
Glengarriff Stream	CK17	CK17	Cork	W 454 922	145465	92232
Fermoyle River	CK19	CK19	Cork	W 394 919	139459	91961
Trib of River Blackwater	CK2	CK2	Cork	W 671 973	167196	97330
Ownagluggin River	CK21	CK21	Cork	W 370 872	137010	87248
Ownagluggin River	CK22	CK22	Cork	W 377 875	137788	87521
Ownagluggin River	CK23	CK23	Cork	W 377 875	137749	87501
Ownagluggin River	CK24	CK24	Cork	W 384 877	138438	87701
Carrigduff River	CK25	CK25	Cork	W 357 884	135710	88412
Carrigduff River	CK26	CK26	Cork	W 354 883	135486	88331
Crinnaloo River	CK27	CK27	Cork	W 370 892	137044	89243
Aghalode River	CK28	CK28	Cork	W 384 851	138427	85148
Trib of River Laney	CK29	CK29	Cork	W 353 855	135399	85576
Trib of River Blackwater	CK3	CK3	Cork	W 666 970	166688	97055
Bregoge River	CK30	CK30	Cork	R 595 134	159550	113454
Castlepook River	CK31	CK31	Cork	R 606 137	160600	113785
Trib of Bregoge River	CK32	CK32	Cork	R 620 133	162013	113334
Fluckane Stream	CK33	CK33	Cork	R 631 131	163110	113186
Trib of River Funshion	CK34	CK34	Cork	R 693 140	169333	114043
Trib of Sheep River	CK35	CK35	Cork	R 720 149	172068	114983
Garrane River	CK36	CK36	Cork	R 594 179	159451	117961
Trib of Ross River	CK4	CK4	Cork	W 647 968	164725	96815
Trib of River Bride	CK6	CK6	Cork	W 740 926	174075	92612
Bunnaglanna River	CK8	CK8	Cork	W 709 928	170926	92860
Ballycorban River	CL2	CL2	Clare	R 635 891	163510	189122
Trib of Scarriff River	CL3	CL3	Clare	R 639 891	163978	189115
Bow River	CL4	CL4	Clare	R 663 904	166353	190427
Bow River	CL5	CL5	Clare	R 669 917	166934	191795
Corlea River	CL6	CL6	Clare	R 616 938	161652	193837
Trib into Lough Atorick	CL8	CL8	Clare	R 641 940	164190	194005
Muchnagh	MUCH1	DC1	Cork/Tipperary	R 882 071	188278	107108
Douglas	DOUG3	DC10	Cork/Tipperary	R838 057	183827	105741
Muchnagh	MUCH2	DC12	Cork/Tipperary	R 868 069	186850	106913
Araglin	ARAG1	DC13	Cork/Tipperary	S 007 066	200701	106648
Araglin	ARAG2	DC14	Cork/Tipperary	S 006 067	200602	106743
Araglin	ARAG3	DC15	Cork/Tipperary	S 006 068	200662	106800
Geeragh River	GEER1	DC16	Cork/Tipperary	R 824 178	182482	117850
Geeragh River	GEER2	DC17	Cork/Tipperary	R 829 193	182988	119382
Burncourt River	BURN1	DC18	Cork/Tipperary	R 937 192	193779	119290
Trib Araglin	ARA1	DC2	Cork/Tipperary	R 904 062	190438	106297
Glenakeefe	GKEEF1	DC4	Cork/Tipperary	S 062 060	206269	106014
Glennandaree	GLENN1	DC5	Cork/Tipperary	S 040 077	204056	107794
Sheep	SHEEP1	DC6	Cork/Tipperary	R 910 178	191095	117840

Main system	Invertebrate Site Code	Water Chemistry Site Code	County	Grid Reference	Easting	Northing
Sheep	SHEEP3	DC7	Cork/Tipperary	R 906 201	190631	120168
Sheep	SHEEP2	DC8	Cork/Tipperary	R 893 203	189384	120327
Douglas	DOUG2	DC9	Cork/Tipperary	R 850 056	185087	105620
Gweebarra River	GBAR1	DD10	Donegal	B 857 044	185721	404452
Gweebarra River	GBAR3	DD11	Donegal	B 861 026	186163	402664
Elatagh River	ELATA2	DD13	Donegal	C 043 039	204371	403953
Elatagh River	ELATA4	DD14	Donegal	C 045 041	204500	404109
Elatagh River	ELATA1	DD15	Donegal	C 043 039	204307	403947
Elatagh River	ELAT3	DD16	Donegal	C 022 052	202224	405222
Trib Deele River	DEEL1	DD2	Donegal	C 112 031	211241	403177
Stranagoppoge	STRAN2	DD3	Donegal	G 912 976	191262	397661
Gweebarra River	GBAR2	DD4	Donegal	B 854 038	185497	403876
Gweebarra River	GBAR4	DD5	Donegal	B 840 005	184082	400536
Sruhanboy River	SRUHA1	DD6	Donegal	C 048 017	204803	401716
Trib of Strachashell River	STRAC1	DD7	Donegal	G887 965	188777	396441
Cloghroe River	CROE1	DD8	Donegal	C102 009	210272	400935
Stranagoppoge	STRAN1	DD9	Donegal	G 924 992	192448	399263
Trib Owenmore	OWEN2	DG1	Galway	L 913 731	91317	273196
Trib. Owenree	OREE1	DG11	Galway	M 015 468	101571	246870
Owenboliska	OLISKA2	DG12	Galway	M 142 360	114238	236010
Owenboliska	Oliska1	DG13	Galway	M 145 355	114582	235506
Owenboliska	OLISKA7	DG14	Galway	M 084 322	108464	232252
Owenboliska	OLISKA6	DG15/G7	Galway	M 085 327	108519	232725
Owenboliska	OLISKA5	DG16	Galway	M 080 332	108003	233271
Lough More	MORE1	DG17	Galway	M 068 310	106835	231025
Owenboliska	OLISKA4	DG18	Galway	M 104 345	110492	234516
Sruffaunanulra River	SRUFF1	DG19	Galway	M 090 379	109007	237978
Trib Bunowen River	BOWEN1	DG2	Galway	L 837 757	83779	275767
Trib Lough Corrib	CORRIB1	DG20	Galway	M 057 485	105770	248510
Owenwee River	OWEE1	DG21	Galway	M 025 452	102550	245210
Owenwee River	OWEE2	DG22	Galway	M 032 455	103259	245508
Owenwee River	OWEE3	DG23	Galway	M 033 458	103301	245877
Trib to Maumwee Lough	MAUM1	DG24	Galway	L 973 484	97370	248412
Trib to Owenriff	ORIFF2	DG25	Galway	M 083 423	108363	242381
Glengawbeg River	GBEG2	DG28	Galway	M 056 409	105672	240995
Glengawbeg River	GBEG3	DG29	Galway	M 053 410	105371	241083
Trib Owenmore	OWEN3	DG3	Galway	L 929 728	92970	272821
Owenakilla River	OWENK1	DG30	Galway	M 097 465	109748	246534
Gowlaun River	GLAUN1	DG31	Galway	M 089 471	108947	247150
Trib Owenwee	OWENN2	DG4	Galway	L 951 771	95168	277166
Trib Owenwee	OWENN1	DG5	Galway	L 945 771	94573	277143
Loughanillaunmore	LOUGH1	DG6	Galway	M 098 283	109823	228341
Owenboliska	OLISKA3	DG7/G6	Galway	M 112 345	111290	234524
Owendunnakilla	OKILLA1	DG8	Galway	M 165 364	116552	236440
Knockbane river	KBANE1	DG9	Galway	M 171 353	117126	235361
River Loo	LOO1	DK1	Cork/Kerry	W 048 796	104813	79671
Clydagh	CLYDA9	DK11	Cork/Kerry	W 176 843	117667	84397
Clydagh	CLYDA10	DK12	Cork/Kerry	W165 842	116554	84248
Clydagh	CLYDA6	DK13	Cork/Kerry	W 159 829	115954	82944
Clydagh	CLYDA1	DK14	Cork/Kerry	W 183 833	118389	83368

Main system	Invertebrate Site Code	Water Chemistry Site Code	County	Grid Reference	Easting	Northing
Clydagh	CLYDA2	DK15	Cork/Kerry	W 222 845	122212	84523
Roughty River	ROUGH3	DK16	Cork/Kerry	W 068 751	106817	75194
Roughty River	ROUGH1	DK17	Cork/Kerry	W 072 712	107266	71289
Roughty River	ROUGH2	DK18	Cork/Kerry	W 065 710	106572	71000
Clydagh	CLYDA3	DK19	Cork/Kerry	W 210 865	121066	86565
River Loo	LOO2	DK2	Cork/Kerry	W 045 782	104554	78231
Clydagh	CLYDA8	DK20	Cork/Kerry	W 201 844	120194	84468
Owgarriv River	OWGAR1	DK22	Cork/Kerry	W 100 800	110094	80028
Bohill River	BOHIL1	DK23	Cork/Kerry	W 195 806	119594	80696
Clydagh	CLYDA7	DK24	Cork/Kerry	W 206 845	120620	84561
Foherish River	FOHER1	DK26	Cork/Kerry	W 241 807	124186	80767
Inchamore River	INCHMR1	DK27	Cork/Kerry	W117 775	111751	77576
Aughboy River	AUGHB1	DK28	Cork/Kerry	W 125 783	115953	78398
Slievenaneav River	SLIEVE1	DK29	Cork/Kerry	W 089 800	108969	80033
Roughty River	ROUGH5	DK3	Cork/Kerry	W 038 743	103836	74388
Trib to Flesk River	FLESK1	DK30	Cork/Kerry	W 102 842	110254	84282
Roughty River	ROUGH4	DK31	Cork/Kerry	W 097 730	109706	73061
Inchamore River	INCHMR2	DK32	Cork/Kerry	W 120 773	112095	77334
Kealgorm	KEAL1	DK4	Cork/Kerry	W 012 771	101263	77185
Kealgorm	KGORM1	DK5	Cork/Kerry	W 007 782	100724	78269
Trib to Slaheny River	SLAH1	DK6	Cork/Kerry	W 029 700	102958	70085
Garrrow River	GARW1	DK8	Cork/Kerry	W 081 753	108180	75351
Glenthomas River	GTHOM1	DM1	Mayo	L 889 999	88932	299925
Glennamong River	GAMON2	DM10	Mayo	F 928 038	92869	303830
Trib Srahmore River	SRAH1	DM11	Mayo	F 965 054	96560	305240
Trib Skerdagh River	SKERD1	DM12	Mayo	G 012 023	101267	302346
Trib Crumpaun River	CRUM2	DM13	Mayo	G 046 021	104638	302162
Trib Crumpaun River	CRUM1	DM14	Mayo	G 073 047	107325	304771
Fiddaungrave	FIDD2	DM15	Mayo	G 061 069	106136	306948
Fiddaungal	FIDD1	DM16	Mayo	G 055 076	105540	307619
Glennamong River	GAMON3	DM17	Mayo	F 941 037	94103	303724
Glenthomas River	GTHOM2	DM2	Mayo	F 887 003	88783	300376
Glendahurk River	GHURK3	DM3	Mayo	F 910 009	91083	300953
Glendahurk River	GHURK1	DM4	Mayo	L 909 985	90982	298562
Glendahurk River	GHURK2	DM5	Mayo	F 912 007	91275	300780
Fiddaunatoreen	FREEN1	DM6	Mayo	F 950 019	95086	301904
Glennamong River	GAMON1	DM7	Mayo	F 944 027	94493	302799
Glennamong River	GAMON4	DM8	Mayo	F938 304	93899	304237
Vartry	VAR1	DWW10	Wicklow	O 204 092	320426	209227
Derrybaun River	DERRY1	DWW12	Wicklow	T 133 946	313316	194649
Annalecka Brook	ANNA1	DWW13	Wicklow	O 067 026	306721	202647
Glashaboy	GLASH1	DWW15	Wicklow	O 053 013	306599	201610
Garryknock	GARRY1	DWW16	Wicklow	O 026 022	302636	202250
Oiltiagh Brook	OILI1	DWW17	Wicklow	S 991 958	299141	195835
Inchavore River	INCH1	DWW19	Wicklow	O 110 060	311100	206200
Ballinagee River	BALLIN2	DWW2	Wicklow	O 047 047	304702	204744
Vartry	VART1	DWW20	Wicklow	O 190 069	319044	206912
Lugduff Brook	LUG3	DWW21	Wicklow	T 110 955	311096	195572
Lugduff Brook	LUG1	DWW22	Wicklow	T 111 957	311123	195749

Main system	Invertebrate Site Code	Water Chemistry Site Code	County	Grid Reference	Easting	Northing
Sraghoe Brook	SRAG1	DWW23(f)	Wicklow	O 097 135	309741	213564
Glashaboy	GLASH2	DWW26	Wicklow	O 065 016	306500	201700
Annalecka Brook	ANNA3	DWW4	Wicklow	O 065 033	306512	203325
Knickeen River	KNICK1	DWW6	Wicklow	S 998 952	299854	195214
Slaney River	SLAN1	DWW7	Wicklow	S 995 937	299593	193771
Cloghoge River	CLOG1	DWW8	Wicklow	O 130 074	313029	207418
Cloghoge River	CLOG2	DWW9	Wicklow	O 126 076	312627	207642
Trib of Owendalulleegh River	G1	G1	Galway	R 627 996	162760	199671
*	G11	G11	Galway	M 548 106	154844	210646
Trib of Boleyneendorrish River	G12	G12	Galway	M 565 052	156586	205222
Trib of Boleyneendorrish River	G13	G13	Galway	M 565 052	156556	205208
Trib into Derryclare Lough	G15	G15	Galway	L 830 498	83047	249800
Trib into Derryclare Lough	G16	G16	Galway	L 827 493	82718	249356
Trib of Owenglin River	G18	G18	Galway	L 740 513	74043	251332
Trib of Owendalulleegh River	G2	G2	Galway	M 566 015	156641	201559
Owenaglanna River	G3	G3	Galway	M 612 065	161215	206594
*	G4	G4	Galway	M 563 105	156391	210551
Owendunnakilla	G5	G5	Galway	M 160 363	116027	236399
Trib of Owenboliska River	G6	G6	Galway	M 112 350	111216	235073
*	G7	G7	Galway	M 085 327	108534	232725
Trib of Owenboliska River	G8	G8	Galway	M 099 328	109902	232848
Trib into Seecon Lough	G9	G9	Galway	M 086 359	108638	235904
Trib of Smearlagh River	K1	K1	Kerry	Q 969 170	96915	117058
Dromaddamore River	K2	K2	Kerry	Q 982 182	98243	118241
Trib of Smearlagh River	K3	K3	Kerry	Q 990 204	99014	120483
Barranahown River	L10	L10	Longford	R 702 244	170264	124474
Trib of Ahaphuca River	L12	L12	Longford	R 721 237	172168	123782
Trib of Awbeg River	L13	L13	Longford	R 618 188	161896	118844
Trib of Awbeg River	L14	L14	Longford	R 616 187	161672	118760
Trib of Awbeg River	L15	L15	Longford	R 597 190	159776	119073
Trib of Awbeg River	L16	L16	Longford	R 597 190	159757	119065
Trib of Assaroola River	L17	L17	Longford	R 817 227	181764	122760
Trib of River Loobagh	L2	L2	Longford	R 638 204	163838	120431
Trib of Keale River	L3	L3	Longford	R 650 186	165078	118651
Trib of River Ogeen	L4	L4	Longford	R 638 168	163859	116884
Trib of River Ogeen	L5	L5	Longford	R 646 171	164626	117113
Trib of Keale River	L6	L6	Longford	R 660 175	166024	117505
Trib of River Loobagh	L8	L8	Longford	R 694 217	169474	121787
Barranahown River	L9	L9	Longford	R 684 242	168480	124258
Delour River	LS1	LS1	Laois	N 285 029	228597	202935
Trib of River Barrow	LS10	LS10	Laois	N 331 054	233181	205441
Trib of Delour River	LS12	LS12	Laois	S 246 970	224699	197009
Trib of Delour River	LS13	LS13	Laois	S 237 966	223754	196687
Trib of Delour River	LS14	LS14	Laois	S 225 967	222549	196740
Delour River	LS2	LS2	Laois	N 281 031	228188	203180
Delour River	LS3	LS3	Laois	N 285 032	228549	203287

Main system	Invertebrate Site Code	Water Chemistry Site Code	County	Grid Reference	Easting	Northing
Delour River	LS4	LS4	Laois	N 276 038	227675	203807
Delour River	LS5	LS5	Laois	N 297 033	229712	203334
Delour River	LS6	LS6	Laois	N 295 019	229594	201996
Trib of Mountrath River	LS7	LS7	Laois	S 348 997	234839	199782
Trib of River Barrow	LS8	LS8	Laois	N 366 079	236680	207933
Trib of Glenummera River	M1	M1	Mayo	L 905 676	90547	267687
Trib of Glenora River	M10	M10	Mayo	G 046 339	104608	333981
Glennafrankagh	M11	M11	Mayo	G 028 347	102811	334720
Trib of Altderg River	M12	M12	Mayo	G 010 322	101088	332253
Coolin River	M13	M13	Mayo	G 047 283	104744	328324
Sruffaunmuinganierin	M14	M14	Mayo	G 072 288	107280	328874
Trib of Glenummera River	M2	M2	Mayo	L 896 674	089611	267419
*	M3	M3	Mayo	F 855 178	085545	317826
Trib of Glencullin River	M4	M4	Mayo	F 911 255	091155	325560
Trib of Glencullin River	M5	M5	Mayo	F 898 261	089800	326190
*	M6	M6	Mayo	F 878 309	087822	330950
Trib of Glenamoy River	M7	M7	Mayo	F 909 331	090976	333125
Trib of Glenamoy River	M8	M8	Mayo	F 951 358	95114	335885
Bellanaminnaun River	M9	M9	Mayo	G 066 365	106695	336507
Trib of Sheep River	T1	T1	Tipperary	R 893 204	189359	120401
Trib of River Aherlow	T10	T10	Tipperary	S 016 279	201647	127960
Trib of River Aherlow	T11	T11	Tipperary	S 002 280	200240	128048
Trib of River Aherlow	T12	T12	Tipperary	R 993 279	199374	127905
Trib of River Aherlow	T13	T13	Tipperary	R 988 280	198875	128086
Trib of River Aherlow	T14	T14	Tipperary	R 979 284	197987	128490
Trib of River Aherlow	T15	T15	Tipperary	R 948 263	194857	126398
Trib of River Aherlow	T16	T16	Tipperary	R 945 263	194527	126387
Clydagh River	T17	T17	Tipperary	R 887 255	188710	125536
Clydagh River	T18	T18	Tipperary	R 884 253	188495	125348
Trib of River Aherlow	T19	T19	Tipperary	R 901 279	190180	127915
Trib of Sheep River	T2	T2	Tipperary	R 906 201	190631	120157
Trib of River Aherlow	T20	T20	Tipperary	R 891 279	189140	127993
Trib of Burncourt River	T3	T3	Tipperary	R 907 220	190702	122059
Trib of Burncourt River	T4	T4	Tipperary	R 908 220	190815	122056
Trib of Burncourt River	T5	T5	Tipperary	R 922 205	192276	120535
Trib of Burncourt River	T6	T6	Tipperary	R 955 200	195582	120028
*	T7	T7	Tipperary	R 991 247	199159	124754
Trib of River Suir	T8	T8	Tipperary	S 029 270	202978	127038
Trib of River Aherlow	T9	T9	Tipperary	S 017 279	201719	127950
Trib of Licky River	W1	W1	Wexford	X 212 855	221260	85579
Trib of Licky River	W2	W2	Wexford	X 224 878	222457	87880
Trib of Licky River	W4	W4	Wexford	X 174 848	217494	84858
Trib of Licky River	W6	W6	Wexford	X 190 859	219066	85973
Trib of Goish River	W7	W7	Wexford	X 170 885	217047	88586
Trib of Goish River	W8	W8	Wexford	X 159 904	215946	90467
Goish River	W9	W9	Wexford	X 171 894	217105	89485