Journal: Biology and Environment Proceeding of the Royal Irish Academy 111B, 40-60.

Riparian zone creation in established coniferous forests in Irish upland peat catchments: Physical, Chemical and Biological Implications.

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Abstract

Plantation forests were established on western Irish peatlands before it became apparent that riparian buffer zones were essential for the health of important salmonid habitats and aquatic ecosystems. The option to retrofit a riparian buffer zone several years prior to the clearfelling of the main plantation may lessen the possible effect of the clearfelling on receiving waters and provide some protection against sediment and nutrient runoff. The option to create a RBZ can only be considered if it can be shown that clearfelling this zone of coniferous forestry along the stream does not pose a significant risk to the water bodies in the short term. To assess this risk, the hydrology, water chemistry and biota at three locations in western peatland catchments within mature, harvestable-age forestry plantations were studied prior to, during and immediately after riparian buffer zones were created. Results indicate that water discharge and suspended sediment increased significantly at two experimental sites post-felling. Maximum and minimum daily temperature and pH also increased significantly at two of the sites. The biological results from macroinvertebrate analysis indicated some significant changes in richness and abundance of species post-felling. The juvenile trout (*Salmo trutta* L.) densities remained stable over the sampling period and appeared unaffected by the clearfelling operations.

Key-words: Riparian buffer zone (RBZ), coniferous forestry, upland peat catchment, hydrology, macroinvertebrate and salmonids.

Introduction

Between the 1950s and 1980s, peatlands in the west of Ireland were targeted for afforestation for social and economic purposes. As a result, the state owned forestry company, Coillte, currently owns 200,000ha of forestry in Ireland's western district of which 63% of the total forest area is found on peat soil (58% on blanket bog, 3% on raised bog and 2% on cutaway bogs (Dermot Tiernan Coillte,per comm.). The dominant species planted were Sitka spruce (*Pinus sitchensis)* and Lodgepole pine (*Pinus contorta).* Many of these plantations have low economic potential with a yield class of less than fourteen (the productive capacity of a forest measured in cubic meters per hectare per year) and are classified by Coillte as "Red Areas" (RA). These plantations often did not develop into closed canopy forest stands due to site exposure, infertility, thin soils and poor drainage. Today a good deal of forestry planted on upland peats is classified as "Red Area", and many of the watercourses that rise in, or receive drainage from these forests are salmonid waters. A combination of peat soils, upland catchments and minimal riparian buffer zone (RBZ) protection means that these receiving waters are particularly sensitive to forestry operations, particularly clearfelling. High levels of precipitation in the west of Ireland (Rohan, 1986), over relatively steep sloped catchments, exert a major influence on the hydrology of these upland stream systems (Helliwell et al., 2001; Soulsby et al., 2002) and flash flooding is a common occurrence (Müller, 2000).

Afforestation and reforestation activities can result in both soil exposure and compaction in upland plantations and is now widely accepted to have significant impacts on nearby watercourses (Riipinen et al., 2009a), by changing temperature regime, flow regime (Robinson et al., 2003; Robinson and Dupeyrat, 2004), primary production, organic matter dynamics (Dineen et al., 2007), water chemistry, aquatic ecosystem community structure (Binkley and Brown, 1993; Campbell and Doeg, 1989; Stone et al., 1998; Giller and O'Halloran, 2004; Herlihy et al., 2005) and sediment and nutrients release (Binkley and Brown, 1993; Johnson, 2000). The presence or absence of a RBZ may mitigate or amplify respectively many of these impacts, and RBZ management and implementation are recognised as a means of mitigating the impact of clearfelling (Forest Service, 2000; Nisbet, 2001; COFORD, 2002; Giller et al. 2002; Nisbet et al., 2002; Broadmeadow and Nisbet, 2004; Kreutzweiser et al. 2005 and Parkyn et al. 2005).

Current forest guidelines in Ireland now require a riparian strip of 10-25m wide (the width depending on slope and soil erodability) to be left unplanted when planning commercial afforestation adjacent to water courses (Forest Service 2000). Kiffney et al. (2003) conclude that an uncut riparian buffer of 30m or more on either side of the stream is needed to limit biotic and abiotic changes associated with clearfelling in watersheds where the headwaters are forested. In Canada the concept of varying the width of RBZ (from 20m-300m) is accepted as a means of optimising the effectiveness of the RBZ (Luke et al., 2007), and the width can be determined by the stream morphology, stream order, local topography, aspect and soil classification. Recommendation from research carried out in northwestern America indicates that the retention of 10m RBZ will retain stream bank stability and reduce sedimentation. A RBZ of 15-30m will help to maintain the instream habitat through such outcomes as regulation temperature, litter and wood input (Olson et al. 2007).

However, the majority of coniferous plantations established in Ireland in the 1950s and 1960s were planted right to the edge of any watercourses flowing through the catchments, and a planned, naturally vegetated established RBZ is usually absent.

This problem is not a specifically Irish one, as much of the forestry planted in upland areas of the UK have similar characteristics (Broadmeadow and Nisbet, 2004), having no RBZ in place along watercourses on first rotation plantations. These forests are now reaching felling age (primarily as a single age class), and the impending clearfelling brings with it two main challenges. The first is how to protect the watercourses from sediment and nutrient release from the imminent clearfelling operations. The second is how to manage the second and successive rotations of coniferous forestry without repeating the mistakes of the first, and ensuring that functioning RBZs are established to mitigate against all the impacts mentioned above that may occur in the future as a result of forest operations. Riipinen et al. 2009a emphasise that coniferous planting in Europe is occurring at a faster rate than felling since the 1960s, therefore the potential impacts on water courses are likely to increase in the near future.

In relation to the first challenge, it is widely accepted that a vegetated RBZ is likely to attenuate sediment and nutrient release from clearfelling operations (Carling et al. 2001). It has been previously shown that the greatest risk to receiving water bodies in Ireland as a result of clearfelling is increased suspended sediment (Johnson et al. 2000). Sedimentation of watercourses increases turbidity (Parkyn et al. 2005) thereby reducing dissolved oxygen (DO), restricting sunlight from reaching photosynthetic plants, (Campbell and Doeg, 1989; Nisbet, 2001; Hutton et al. 2007) covering food sources and smothering benthic macroinvertebrate communities. The recommended critical limit of sedimentation in salmonid waters is $25 mgl^{-1}$ (European Communities, 1988). Sediment may carry a higher concentration of nutrients (phosphorus and nitrogen) to streams post-clearfelling (Decker, 2003; Rodgers et al. 2008). The increased nutrient concentrations post-felling may be caused by the loss of nutrient uptake by trees, an increase in decomposition on the forest floor, excess surface runoff and soil leaching. In the case of mature, first rotation coniferous forests in the west of Ireland, there are two options to ensure that a vegetated RBZ is in place before clearfelling to attenuate the impacts of sediment and nutrient runoff: i) leaving a strip of coniferous forestry uncut or ii) retrofitting a more naturally vegetated RBZ several years prior to clearfelling. Leaving a strip of forestry uncut adjacent to the streams has been shown to be effective in minimising suspended sediment and nutrient inputs (Kiffney et al. 2003; Aubertin and Patric, 1974; Borg et al. 1988 and Lynch and Corbett, 1990). However, the combination of exposed sites, strong winds and thin soils in western Irish peatlands means that the risk of wind-throw is high if a relatively thin strip of conifers was left uncut adjacent to streams. In addition, this option does nothing to address the fact that the presence of a monoculture of conifers adjacent to streams causes over shading and reduces the primary production within the stream due to the lack of leaf litter and sunlight (Dineen et al. 2007). In plantation forests the only food source comes from tough conifer needles, which are poor in nutrients (Riipinen et al. 2009b), and this food source is difficult to retain as they are flushed rapidly from the system (Riipinen et al. 2009b). This effect makes plantation forests an even poorer quality food resource for macroinvertebrates and microbial decomposers.

A RBZ of natural vegetation provides dapple shade, woody debris and leaf litter. Dappled shade can reduce in stream plant growth and reduce water temperature (Parkyn et al. 2005). Leaf litter has been recognised as providing the most important trophic basis of the stream food web in forests in the west of Ireland (Dineen et al. 2007), and higher macroinvertebrate densities are linked with leaf litter entering the stream (Giller and Twomey, 1993). The accumulation of woody debris helps retain the leaf litter in the stream. The retrofitting of a more naturally vegetated RBZ may therefore be a more ecologically sound option than leaving a strip of conifers as an RBZ, and one which addresses both the issue of stream protection during clearfelling, and also the long-term ecological functioning of streams flowing through second and successive rotations of commercial coniferous forestry.

This establishment of RBZ in mature coniferous forests is, however, time consuming and awkward. Forestry owners and mangers are unlikely to adopt this practice unless it is proven to be effective. An assessment of the efficacy of this practise is not possible due to the lack of data, as has been hightlighted by work in the UK and Canada (Ormerod et al. 1993; Kiffney, 2003). The first step in this process is to assess the immediate impact of felling and establishing a naturally vegetated RBZ on the adjacent waterbodies. The work described in this paper addresses this first step. Three RBZs were created in coniferous forests by felling the mature coniferous trees within the RBZ, and either allowing the vegetation in the zone to naturally regenerate, or planting native woodland species in a mosaic pattern within the zone (Parrott and MacKenzie, 2000). Macroinvertebrate indices and salmonid composition are used as the prime indicators of water habitat quality and ecological functioning (Parkyn et al., 2005) in this study. These indicator species are also used to aid in the assessment of the Irish river water quality (Tierney et al. 1998; Mc Garrigle et al. 2002), and to measure the recovery of surface water after clearfelling (Cruikshanks et al. 2006). The results presented here describe the immediate short term physical, chemical and biological impacts resulting from the felling operations adjacent to the river.

Methods and Study Location

Three study sites were located on first and second order streams (RA1: Glendahurk, RA2: Altahoney and RA3: Sheskin) in the west of Ireland (Fig. 1). We used two control catchments for comparison with the RA study streams, the Glenamong and Goulaun Rivers (Fig. 1) which form part of the Marine Institute network of monitoring stations within the Burrishoole catchment. High resolution data is available from these control rivers, and of particular data of interest to this work is the water discharge data from the Glenamong river and temperature data from the Goulaun river. The main land use in the Glenamong and Goulaun subcatchments is forestry and commonage. All sites (RA and control) experience a similar moderate climate due to their close proximity to the Atlantic Ocean. The air temperature rarely reaches 22^{o}C in the summer or goes below -5^{o}C in the winter. Snowfall is occasional and is generally restricted to upland areas. Annual rainfall for the region is between 1500mm-2000mm (Rohan, 1986) and annual average precipitation recorded at the Met Éireann weather station in Burrishoole during the monitoring period was 1500mm (Marine Institute, 2006). Periodic or permanent water-logging is characteristic at all sites.

The physical, chemical water parameters and electrofishing survey results presented here were collected over a three-year period, one year prior to felling a RBZ and two years post-felling. The macroinvertebrate results presented were collected for one year pre- felling and for three years post-felling.

Coillte is the landowner of the three sites and commercial forestry within each catchment where the experimental sites are situated varies from 19% at RA1, 42% at RA2 to 100% at RA3. The experimental streams all drain commercial coniferous plantation established without RBZs. However, in some cases owing to the topography of the sites, there were patches of unplanted areas adjacent to the streams, with the width between the streams and the edge of the planted area varying between 0 and 15 metres. Outside the afforested areas, sheep farming is the main agricultural activity. The catchments surrounding RA1 and RA2 were afforested in 1970 and RA3 was planted in 1984. Thus, for this study, RA1 and RA2 were 36 years old prior to harvesting and RA3, 22 years old. The sites contain mixed stands, mainly made up of *Picea sitchensis* (Sitka spruce) and *Pinus contorta* (Lodgepole pine) with *Larix* spp. (Larch) trees found along the riparian strip in RA3 only. Details of sites are in Table 1.

The study sites were felled according to standard operating procedures for harvesting and using a Valmet harvester (10m reach capability) and forwarder for felling and extraction. Brash mats were used by the machine to traverse the sites, thereby reducing compaction of the peat (COFORD, 2002). The streams were crossed in each RA at one location using a timber bridge made from material on-site and this bridge was removed after felling operations were completed. The cut trees were laid adjacent to the brash mat for collection by the forwarder. All work was carried out during the summer months of 2006, when timing, weather and ground conditions were suitable. The plan was to clearfell 300m x 100m along the stream (Fig. 2) (50m either side of the stream and 300m length of the forested edge) which is a larger RBZ than required under current Forest Service guidelines 2000. The area of the RBZ was chosen to give a clear effect from the clearfelling and allow sufficient area to develop a RBZ. Although every effort was made at the early stage of the project to communicate the proposed area to be felled, a larger area was felled than originally proposed at RA1. This was probably owing to the fact that the downstream side of the stream was due to be clearfelled anyway for commercial purposes, and so this operation was carried through right up to the stream edge. Any native shrubs species present on-site were avoided by the harvester.

Figure 1 here:

Table 1 here:

Water parameters analysed for this study included water discharge, suspended sediment, conductivity, pH, dissolved oxygen (DO), colour and temperature. Orphimedes data loggers (OTT Hydrometry Ltd, UK) were installed at each site to measure the water level every fifteen minutes. The salt dilution method was used to generate a rating curve for each study stream. This enables the water level (mm) recorded by the OTTs to be converted into water discharge (l/s) (Rodgers et al., 2008). It is important to note that in high floods, once the water breached the banks, an accurate flow can not be calculated and this is particularly true at RA2 as water breached the bank at 600mm. Suspended Sediments (SS) were measured during flood events using automatic water samplers the Sigma 900 Max portable samplers (www.americansigma.com). These samplers were triggered to start sampling once the water level reached a certain height above zero on the staff gauge (RA1-450mm, RA2-450mm and RA3-400mm). Once activated, the auto samplers took one sample every 10 minutes at RA1/RA3 and 15 minutes at RA2, with a capacity for 24 samples. All samples were taken back to the laboratory, where 200ml was filtered through Whatman GF/C filters (pore size 1.2 µm) to calculate the suspended sediment content (mg/l) using standard methods (APHA 1998). Conductivity, pH and DO were measured using a field probe (WTW 340i) during every site visit. Colour was measured in units of Platinum Cobalt (PtCo) using a HACH (DR/2000)

Spectrophotometer. Temperature data were collected using automatic recorders (Tidbits from Onset) at RA1 and RA2. The temperature recorder in RA3 was not functioning properly until November 2006. Pre- and post-felling analysis was therefore not possible.

Although water quality data were collected for a longer period post-felling, only data from comparable time periods were used to eliminate seasonal factors. Data were examined for normality and transformed where necessary. Comparisons of water discharge, precipitation, pH, conductivity, colour and temperature were made using 2 sample *t*-tests or Mann-Whitney *U*- tests (where data was non-normal, and transformation was not appropriate).

Biological monitoring of these study sites occurred in mid July over four years. Each study site was divided into three macroinvertebrate sampling points (Fig. 2). Point 1 was 150m downstream of the felling, point 2 was in the middle of the felled block and point 3 was 150m upstream of the felling. Juvenile fish stock assessment survey was carried out in late August of each year (one year pre-felling (2005) and two years post-felling (2006-2007)) within the felling block of RA1, RA2 and RA3 streams (Fig. 2) at a location which included several habitat types and a suitable area for catch quantification.

Figure 2 here:

Macroinvertebrates were sampled using a Surber sampler consisting of a quadrat (25cm x 25cm) attached to a 1.0mm mesh net and collection bottle. Six replicate samples were taken at each sampling point randomly in an upstream direction. Samples were stored in 70% Industrial Methylated Spirits (IMS) and identified to the lowest practical taxa level (usually species). Univariate analysis (abundance, taxon richness, number of EPT taxa (Ephemeroptera, Plecoptera, Trichoptera) and Shannon diversity) was conducted in Datadesk, and pre-felling values (2005) were compared with the values from three years post-felling (2006, 2007 and 2008). The analysis was carried out using ANOVA (Analysis of variance) and LSD posthoc tests with sampling point and year as explanatory variables. Data were examined for normality and transformed where necessary. Pre- and post-felling differences in macroinvertebrate communities were analysed using the multivariate analyses ANOSIM (analysis of similarity) and SIMPER (similarity percentage) in CAP (Community Analysis Package version 3.1, Pisces Conservation). ANOSIM looks at the similarity between defined groups and within defined groups, whereas SIMPER breaks down the contribution of each species to the observed similarity or dissimilarity between samples, allowing the identification of the species that are most important in creating the observed pattern of similarity. The data were not transformed and Bray-Curtis Similarity Matrices were used in the analyses. All analysis was carried out using the data from 6 replicates taken on each sampling occasion.

Electrofishing operations were conducted with a 12-volt Safari Research 550D backpack machine adopting a zigzag pattern across the stream (Lehane et al., 2004; Summers et al., 2005). All sites were fished three times in an upstream direction (from bottom to top, for a length of 30m at RA1, 30m at RA2 and 50m at RA3) under similar weather conditions using the catch depletion method (Zippin, 1958). Fish from each pass were held in buckets of water, identified and measured to fork-length to the nearest millimetre. This survey was conducted to determine the species of fish present and the relative population densities of each species. Age determination was carried out on the basis of length frequency analysis and all fish were returned to the stream alive after sampling. The area for all sites was calculated by multiplying the length by the average of 4 or 5 width measurements and this was used to calculate fish densities per m^2.

Results

Water Discharge

Water level analysis was carried out at RA1 from July 2005 to May 2006 (pre-felling) and July 2006 to May 2007 (post-felling), at RA2 from July 2005 to May 2006 (pre-felling) and May 2006 to April 2007 (post-felling) and at RA3 from July 2005 to March 2006 (pre-felling) and Aug 2006 to March 2007 (post-felling). At RA1 the daily sum, daily average, daily maximum and daily minimum were generally lower post-felling ($p<0.05$) although there were several occurrence of very high discharge (Fig. 3) post-felling. Water discharge in the pre-felling period at RA2 and RA3 sites was significantly different from that in the post-felling period with the daily sum, daily average, daily maximum and daily minimum being significantly higher post-felling ($p<0.05$). The median daily maximum at RA1 pre-felling was 48 l s^{-1} while post-felling it was 38 l s^{-1}. At RA2, the median daily maximum pre-felling was 160 l s^{-1} rising to 325 l s^{-1} post-felling while at RA3, the median daily maximum pre-felling was 36 l s^{-1} rising to 70 l s^{-1} post-felling (Fig. 3). The most noteworthy increase in discharge was observed at RA2. Owing to the fact that the rating curve at this site may not have been accurate over levels of 600mm (i.e. flow breaching the river banks) data was also analysed without water levels above 600mm. However, even without the higher flow data, there was still a significant increase in water flow post-felling. Water discharge from the control site on the Glenamong River was analysed over a similar time period as the RA sites (July 2005 to May 2006 pre-felling and July 2006 to May 2007) and there was no significant increase in daily sum, daily average and daily maximum. There was a significant increase in daily minimum water discharge in the Glenamong ($p= 0.04$).

Figure 3 here:

Weather Conditions

Total precipitation from the Marine Institute, weather station was 1451mm pre-felling (June 2005 to May 2006) and 1611mm post-felling (June 2006 to May 2007). Analysis showed that precipitation was not significantly higher in the post-felling period (Mann-Whitney *U*-test, p=0.10). However, it was apparent that there were more outliers in the rainfall data in the post-felling period, so further investigation was carried out on daily and monthly variation in precipitation in the pre- and post-felling periods (Fig. 4). A dry day can be defined as a day with less than 1mm of rainfall recorded, and a wet day is defined as a day with more than 1mm of rainfall recorded (Hundecha and Bárdossy, 2005). The number of dry days recorded pre-felling was 156 and post-felling was 150. The number of wet days recorded pre-felling was 194 and post-felling was 200. The average daily rainfall pre-felling was 7.3mm and post felling was 7.9mm (Table 2).

Heavy or extreme precipitation events can be defined as days with precipitation greater than a given threshold (McElwain and Sweeney, 2007). Long term precipitation from the Marine Institute was analysed from 1959 to 2008 and showed that precipitation of 20mm and greater occurs only 6% of the time. Therefore, we defined heavy precipitation occurrence in the Burrishoole catchment as any precipitation event greater than 20mm of rainfall in a day. The precipitation data from pre- and post-felling periods indicates that there were 3 such occurrences pre-felling and 13 such occurrences post-felling (Fig. 4). Generally, daily precipitation pre- and post-felling periods were similar but the numbers of heavy precipitation occurrences were greater post-felling (Fig. 4).

Figure 4 here:

Table 2 here:

Suspended Sediment.

Suspended sediment was measured at RA1 from July 2005 to July 2006 pre-felling and June 2006 to March 2007 post-felling. At RA2 SS was sampled from Sept '05 to June 2006 pre-felling and from May 2006 to Feb 2007 post-felling and at RA3 from Aug 2005 to April 2006 pre-felling and from Aug 2006 to Feb 2007 post-felling. At RA1 a total of 20 floods were sampled for SS pre-felling and 32 post-felling. At RA2 a total of 32 floods were measured for SS pre-felling and 48 floods post-felling. At RA3 16 floods were measured for SS pre-felling and 21 post-felling. The concentrations of SS at base flow conditions at all sites were generally low (1-8 mg l^{-1}) and the maximum concentrations of SS discharged pre-felling were lower at each site than in the first flood events post-felling. Generally all the maximum SS events occurred within two months of felling (i.e. September 2006). There were some high levels of SS measured post-felling at RA1, however the difference in SS pre- and post-felling was not significant (*t*-test, p=0.98). SS levels were significantly higher in the post-felling period at RA2 (Mann-Whitney *u*-test, p=0.018) when a maximum of

380mg/l was recorded There was also a significant increase in SS post-felling at RA3 (Mann-Whitney *U*-test, $p=0.013$) (Fig. 3).

Conductivity, Colour and pH

Conductivity, colour and pH parameters were measured weekly or biweekly. The conductivity at RA1 ranged from 41µS/cm^{-1} to 158µS/cm^{-1} and colour ranged from 18 to 145 mg l^{-1} PtCo. The pH at RA1 ranged from 4.6 to 6.57 (Fig. 5), which reflects the geology and thin peatland soils in the surrounding catchment. Comparisons of conductivity, colour and pH in the pre-felling (September 2005 to May 2006) and post-felling (September 2006 to May 2007) periods show that there were no significant differences in these variables as a result of the felling (Mann-Whitney *u*-tests, $p>0.05$ in all cases). The pH measurements are described in Fig. 3. At RA2, conductivity ranged from 39 µS/cm to 126µS/cm. Colour ranged from 36 to 260 mg/l PtCo and pH ranged from 3.8 to 7 (Fig. 5). Comparison of conductivity, colour and pH in the pre-felling (May 2005 to May 2006) and post-felling (May 2006 to May 2007) periods show that there was a significant increase in pH (Fig. 3) and a significant decrease in colour post-felling (Mann Whitney u-tests, $p<0.05$), but not in conductivity. The conductivity measurements for RA3 ranged from 60 µS/cm to a maximum of 342 µS/cm and colour ranged from 120 to 500 mg/l PtCo. The pH in RA3 reflects the sandstone geology in the area, with a range from 4.6 to 8.1 (Fig. 5). The median value for pH at RA3 was 6.5, which is much higher than the other two RA sites. Comparisons of conductivity, colour and pH in the pre-felling and post-felling periods show that pH increased significantly post-felling (Mann Whitney *U*-Test, $p=0.03$) (Fig. 3) but that colour and conductivity remained the same at RA3.

Figure 5 here

Temperature

The maximum and minimum daily stream water temperature over the duration of the project at RA1 and RA2 was measured using temperature tidbits (RA1 July 2005 to May 2006 (pre-felling) and July 2006 to May 2007 (post-felling), RA2 September 2005 to May ‘06 (pre-felling) and September 2006 to May 2007 (post-felling)). At RA3, the tidbit battery failed, therefore data are only available from November 2006, with none available pre-felling. Temperature ranged from 2.1°C to 18.2°C at RA1 and from 1.4°C to 15.9°C at RA2. The maximum daily temperatures were significantly higher post-felling ($p<0.001$) at RA1 but there was no significant difference in the daily minimum temperature ($p=0.13$). Both maximum and minimum water temperature were significantly higher in the post-felling period (t-test, $p<0.05$) at RA2 (Fig. 6). Water temperature from the Goulaun River control site was analysed over a similar time period as RA1 and RA2 (September 2005 to April 2006 pre-felling and September 2006 to April 2007). The maximum temperature was significantly higher between in the second time period, but there was no significant increase in minimum temperatures.

Figure 6 here:

Macroinvertebrates

Generally the taxa identified at each RA were typical of streams located in areas with thin peatland soils, low acid neutralising capacity and poor buffering geology (Kelly-Quinn et al. 1997; Clenaghan et al. 1998), with Plecoptera, Trichoptera, Diptera, Ephemeroptera, Coleoptera and Oligochaete dominating the samples. Smaller numbers of Collembola, Amphipoda, Mollusca and Acari were also present in some of the samples. Crustaceans were only present in abundance at RA3. None of the sites were particularly diverse with the number of taxa ranging between 37 at RA1, 41 at RA2 and 58 at RA3.

RA1:

The clearfelling of the RBZ was more extensive than originally planned at RA1. The extra clearfelling was however, carried out down stream and down slope from the study site, therefore it is unlikely that it had an effect on the study stream, except for an increase in light penetration. A total number of 1658 individuals, representing 37 taxa and 9 orders, were sorted and identified at RA1 over the four sampling years.

Analysis of variance (ANOVA) indicated that year and point were both significant sources of variation in taxon richness (Table 3). The interaction and point and year were not significant overall but LSD post-hoc tests highlighted significant differences at each point. At point 1, taxon richness was significantly lower pre-felling in 2005 than post-felling in 2006, 2007 and 2008 ($p< 0.05$). At point 2, there was an increase in taxon richness in each year post-felling in comparison to pre-felling values, although the difference was only significant in 2007 ($p<0.05$). At point 3, the taxon richness did not change significantly over the four years (Fig. 7). Macroinvertebrate abundance also varied between years (Table 3). The macroinvertebrate abundance significantly increased ($p<0.05$) at point 1 from pre- (2005) to post-felling (2006 and 2007). At point 2 there was a significant increase in macroinvertebrate abundance post-felling ($p<0.05$) (2006, 2007 and 2008). At point 3 there was a decrease in abundances from pre- (2005) to post-felling (2006, 2007 and

2008) although the decrease was not significant. Point 3 was located just outside the forestry plantation and should not have been directly affected by felling. Overall, the third year post-felling (2008) showed a decrease in abundance at each sampling point (Fig. 7). There was a significant difference in Shannon Diversity at point 1 pre- (2005) and post-felling (2006) (p<0.05), with no significance difference in diversity at the other points over all sampling years (Fig. 7) (Table 3). EPT taxon richness was significantly higher from pre- (2005) to post-felling (2006) (p<0.05) at point 1, with no significant differences evident at the point 2 and point 3 (Fig. 7) (Table 3).

Figure 7 here

Table 3 here

When comparing the dominant orders at RA1 over the four years, it is obvious that there was a large increase in abundance at point 1 and point 2 from pre- (2005) to post-felling (2006, 2007 and 2008) (Fig. 8). At point 1 (which was 150m downstream from the clearfelling) there was an increase in Plecoptera, Ephemeroptera and Diptera in 2006 and 2007, with a slight decrease in species abundance occurring in 2008 when compared with 2006 and 2007. At point 2 there was a large increase in abundance of Oligochaetes and Diptera between 2005 and 2006. In 2007, there was a shift in the dominant species from Diptera to Coleoptera and Plecoptera and this was also evident in the 2008 sampling. At point 3, the control sampling point at RA1, there was a slight decrease in abundance of species, in particular Diptera and Coleoptera over the four years (Fig. 8).

Figure 8 here.

Multivariate analysis of the macroinvertebrate samples over the four years i.e. pre-felling (2005) and post-felling (2006, 2007 and 2008) was conducted using ANOSIM (analysis of similarity). The analysis showed that macroinvertebrate assemblages sampled pre-felling (2005) were significantly different from those sampled post-felling (2006, 2007 and 2008) at points 1 and 2 ($p<0.05$) when analysed at both taxon and order level. At point 3, significantly different assemblages were also found in 2008 in comparison to 2005 ($p<0.05$). The SIMPER output gives a general pattern of how the order and taxa differs from pre- to post-felling and gives the average abundances for each. The results indicate that Diptera and Plecoptera were the two orders which showed the most change post-felling at RA1 at all sampling points.

Nemoura cinerea Retzius, *Leuctra hippopus* Kemp, *Amphinemura sulciollis* (Steph) and *Chloroperla torrentium* (Pict.), were the dominant Plecoptera and increased in abundance from pre- (2005) to post-felling (2006, 2007 and 2008) at point 1 and point 2. Plecoptera were present at point 3 but did not dominate the samples. The dipteran taxa *Orthocladinae*, *Simuliidae*, *Chironominae*, *Dicranota,* and *Tanypodinae* were particularly abundant at all the three sampling points. At sampling points 1 and 2, Diptera increased in average abundance from pre- (2005) to post-felling (2006, 2007 and 2008). In contrast, at point 3 the average abundance of dipteran decreased from pre-felling (2005) to post-felling (2006, 2007, and 2008). *Plectrocnemia conspersa* (Curtis, 1834), *Glossosoma conformis* (Neboiss, 1963) and *Rhyacophila dorsalis* (Curtis, 1834) were the most abundant Trichoptera at RA1. Trichoptera species were present at point 1 and 2 but the average abundance did not dramatically change, while at point 3 there was an increase in average abundance from pre- to post-felling. The SIMPER analysis found the average abundance of Coleoptera to be significantly different at all points between pre- and post-felling, showing an increase in average abundance at point 1 and 2, with a decrease at point 3. *Limnius volckmari* (Panzer) and *Oulimnius tuberculatus* (Muller, 1806) and *Esolus parallelepipedus* (Muller, 1806) were the most common species. The abundance of Ephemeroptera at RA1 was generally low with *Baetis rhodani* (Pictet, 1844) being the most common taxa sampled. The average abundance increased pre-felling (2005) to post-felling (2006, 2007 and 2008) at point 1.

RA2:

A total of 1574 individuals representing 41 taxa and 10 orders were sorted and identified at RA2 over the 4 sampling years. Ephemeroptera, Plecoptera, Trichoptera, Coleoptera and Diptera dominated the samples from RA2 over the sampling period although Collembola, Oligochaete, Acari and Amphipoda were also present in very low numbers.

Analysis of variance (ANOVA) indicated that taxon richness was significantly lower at point 1 ($p<0.05$) in 2007 when compared with pre-felling values. There was no other significant difference in taxon richness during the study period (Fig. 7) (Table 3). There were no significant chances in abundance over the study period (Fig. 7) (Table 3). The Shannon Diversity Index at point 1 showed a significant decrease in diversity in 2007 and 2008 post-felling ($p<0.05$). The increase in diversity at point 2 in 2008 was significantly different from pre-felling values ($p<0.05$) and diversity did not significantly change at point 3 (Fig. 7) (Table 3). EPT taxa at point 3 decreased significantly between 2005 and 2007 ($p<0.05$) (Fig. 7) (Table 3), but there were no other significant changes at points 1 and 2.

The number of Ephemeroptera decreased at point 1 and 2 from pre- to post-felling. There was an increase in Plecoptera and Coleoptera at point 1 and point 2 post-felling (2006, 2007 and 2008) (Fig 8). ANOSIM analysis confirms that macroinvertebrate assemblages sampled pre-felling (2005) were significantly different from those sampled post-felling (2006, 2007 and 2008) at point 1 and point 2 ($p<0.05$) at both taxon and order level, with no significant difference in taxa and order from pre- to post-felling at point 3. There was a decrease in the average abundance of the main ephemeropteran species, *B. rhodani* and *Serratella ignite* (Poda, 1761) immediately post-felling in 2006 at all points. At point 1 and 2 there was an increase in the average abundance of *L. hippopus, N. cinerea*, and *C. torrentium* from pre- (2005) to post-felling (2006, 2007 and 2008). *Orthocladinae, Simuliidae, Chironmimea*, *Dicranota* and *Tanypodinae* were the most abundant Diptera and there was a decrease in dipteran average abundance from pre- to post-felling at all points. There was a decrease in the average abundance of the trichoptera, *Silo pallipes* (Fabricius, 1775), *R. dorsalis* and *P. conspersa* at point 1 post-felling. Trichoptera were present at point 2 and 3 in low numbers but their average abundance did not dramatically change from pre- to post-felling. *L. volckmari*, *Elmis aenea* (Muller, 1806), *E. parallelepipedus* and *Noterus clavicornis* (DeGeer, 1774) were the abundant coleopteran taxa. Coleoptera were present at point 1 in low abundances, and the average abundance of Coleoptera at point 2 decreased post-felling (2006) but showed an increase in average abundance in 2007 and 2008 post-felling.

RA3:

A total of 1976 individuals, representing 58 taxa and 10 orders, were sorted and identified at RA3 over four years. Ephemeroptera, Plecoptera, Amphipoda, Trichoptera, Diptera, Coleoptera and Oligochaete dominated the samples from RA3 over the sampling programme although Acari, Collembola and Molluscs were also present.

Analysis of variance (ANOVA) indicated that changes in taxon richness at RA3 were not significant at point 1 and point 2 but were significant (p<0.05) at point 3 when comparing 2005 with 2007 and 2008 (Fig. 7) (Table 3). Abundances remained relatively unchanged at point 1 over the four sampling years. Significant changes in abundances occurred at point 2 between 2005 and 2006 and at point 3 between 2005 and 2008 (ANOVA, p<0.05) (Fig. 7) (Table 3). The Shannon diversity index was generally higher pre-felling compared with post-felling diversity values although the decrease was only significant at point 1 and 3 in 2007 and at point 2 in 2008 (p<0.05) (Fig. 7) (Table 3). The number of EPT taxa displayed a similar pattern in that they decreased post-felling at points 1, 2 and 3 but the decrease was only significant at point 1 in 2007 and at point 2 and 3 in 2008 when compared to values pre-felling 2005 (Fig. 7) (Table 3).

The dominant orders at RA3 were Trichoptera, Plecoptera, Ephemeroptera, Diptera, Coleoptera and Amphipoda. There was an increase in Diptera numbers at point 1 and 2 from pre- to post-clearfelling (Fig. 8). A more detailed analysis was carried out using ANOSIM (analysis of similarity) on untransformed data. Macroinvertebrate taxa assemblages sampled pre-felling were significantly different from those sampled post-felling at point 1 in 2006, 2007 and 2008, point 2 in 2006 and point 3 in 2006 and 2008 (p<0.05). When data was analysed at the order level significant differences between pre-felling and post-felling samples were recorded at point 1 in 2007 and 2008, point 2 in 2006 and at point 3 in 2008 (p<0.05).

The SIMPER output highlights the taxon and orders which differed at each point pre- and post-felling and gives the average abundances of each. The average abundance of Diptera, *Chironominae, Simuliidae, Orthocladinae* and *Dicranota* increased at point 1 each year post-felling. At point 2 there was an increase in abundance of Diptera in 2006 and 2007 post-felling and a decrease in abundance in 2008. Point 3 showed an initial increase in average abundance of Diptera in 2006 and a decrease in average abundance in 2007 and 2008 post-felling. Ephemeropteran taxa comprised *B. rhodani, Rhithrogena semicolorata* (Curtis, 1834), *S. ignite, Ecdyonurus spp* (Ecoton, 1868), and *Heptagenia spp* Walsh, 1863. The average abundance of ephemeropteran increased immediately post-felling 2006, and subsequently decreased in 2007 and 2008 at point 1 and point 2. At point 3, there was a decrease in the average abundance of Ephemeroptera post-felling in 2006 and 2008, but an increase in 2007. *Gammarus duebenii* (Lilljeborg) was the only amphipod recorded at this site and there was a small increase in its average abundance post-felling in 2006 at point 1 but a decrease in 2007 and 2008. There was an increase in average abundance at point 2 post-felling (2006, 2007 and 2008). At point 3 there was a decrease in average abundance of Amphipoda post-felling (2006, 2007 and 2008). The plecopteran taxa included *L. hippopus, A. sulciollis*, *C. torrentium,* and *N. cinerea,* but they were not recorded in high abundance.

Juvenile Fish Stock Assessment.

Fishing was carried out during the same time period and under similar weather conditions over three years at each RA site (one year pre-felling (2005) and two years post-felling 2006 and 2007). Each fishing site was within the clearfell area. Brown Trout (*Salmo trutta* L.), Atlantic Salmon (*Salmo salar* L.) and European Eel (*Anguilla*

anguilla L.) were the three species recorded in the survey (Table 4). Length frequency distribution of salmon and trout were determined for the three sites fished. 0+ salmon were less than 8cm, while 1+ salmon were greater than 8cm. Trout less than 9cm were assigned to the 0+ age class, and between 9 and 13 cm to the 1+ age class. Any trout greater than 13cm was assigned to the 2+ age class. Trout were the main fish species recorded at RA1 during the period 2005-2007, with about twice as many 1+ trout as 0+ trout being recorded, with one eel caught in 2006. Numbers were stable over the experimental period (0.22 trout of all age classes per m^2). At RA2 one 1+ salmon was recorded in 2006, resulting from stocking in the stream as part of another experiment. Normally no salmon would be present in this area of the Altahoney River due to an impassable falls lower down the stream. Trout were the main fish species recorded in the survey, and similar to RA1, numbers were fairly stable over the experimental period 2005-2006. 0+ fish (indicative of successful spawning over the winter preceding the fishing period) were recorded in all years, as were 1+ fish. 4 eels were caught in 2005, but none in 2006 and 2007. Trout, salmon and eels were recorded at RA3 in the experimental period 2005-2007, although it would appear that no salmon spawning occurred in the vicinity of the site over the winter of 2005/2006, as no 0+ salmon were recorded in September 2006.

Table 4 here

Discussion

Water discharge into the receiving water bodies showed a significant decrease at RA1 from pre-felling to post-felling periods, whereas there was significantly higher discharge at RA2 and RA3 sites in the post-felling period when compared to the pre-felling period. The increase in discharge was probably as a result of site compaction and disturbance of thin peat soils during the felling and extraction operations. It could also be that the trees in the riparian zone pre-felling regulated the flood events in the stream to some extent. Indeed, our results would support the hypothesis that any established vegetation close to streams (including established conifers) is likely to regulate and moderate water discharge into the stream. It must be noted however, that forest drainage pre-planting and post-planting is likely to induce significant increases in stream flows which may then decrease again as drains block up with vegetation and debris (Robinson et al. 2003). Increases in rainfall in the post-felling period might also be responsible for increased flood events, but rainfall measured at the Marine Institute weather station (6km from RA1, 11km from RA2 and 29km from RA3) was not significantly higher in the post-felling period (1451 mm pre-felling and 1611mm post-felling) although heavy precipitation occurrences were more frequent in the post-felling period (3 vs. 13 occurrences >20mm daily). In order to assess the influence of these high precipitation events, we analysed water discharge at a control site (the Glenamong River) within the same geographical area as the study sites (Fig. 1). There was no significant increase in daily sum, daily average or daily maximum discharge in the same post-felling period although there was a slight increase in daily minimum (unpublished data, Marine Institute), even though the Glenamong river would have been influenced by the same high precipitation events mentioned in the results. Thus the likely cause of increased flood water at RA2 and RA3 sites was the felling operations in the riparian zone, which was the only significant land use change that occurred in the vicinity at that time. Increased flood events will increase the erosion of the river bed and cause bank deterioration, but it is likely that once vegetation becomes established in these riparian zones over the next 5 – 10 years, they will provide some protection against water discharge when it comes to felling the area behind the riparian zones. The lack of change in discharge at RA1 may be owing to the fact that the experimental stream is a t the top of the watershed, and volumes are therefore relatively low.

A total of 169 flood events were analysed for SS for the three RA sites over the study before, during and after clearfelling. SS loads post-felling were significantly higher at RA2 and RA3 ($p<0.05$). Given that the recommended level of SS in salmonid waters is 25mg l^{-1} (European Communities, 1988), it is worth noting that pre-felling SS levels exceeded this on numerous occasions and deteriorated significantly post-felling.

There appeared to be no seasonal pattern in relation to the concentration of suspended sediment measured throughout the project. This is partly because rainfall does not follow a distinct seasonal pattern. Allott et al. (2005) studying sediment transport into Lough Feeagh in the Burrishoole catchment and Rodgers et al. (2008) monitoring clearfelling on blanket peat within the same catchment also noted no seasonal pattern in sediment movement. However, all the maximum sediment events occurred within two months of felling (i.e. September 2006). It is also worth noting the extremely large peak in SS at RA2 (380.0 mg/l) compared to the other sites. RA2 was the only site that had little or no riparian zone in place before this project commenced. The recommended average widths (ranging from 10m-25m) of a riparian zone is based upon the slope of the catchment and the sensitivity of the catchment to soil erosion (Forest Service, 2000; COFORD, 2002). However the capacity of riparian zones to reduce sediment entering the stream environment is also linked with many factors including slope, soil type, rainfall events, land use, width of buffer zone and structure of riparian vegetation (Carling et al. 2001; Broadmeadow and Nisbet, 2002).

Thus riparian zone widths may be linked with the SS concentrations released into the stream post-felling at each RA. The higher levels of SS recorded post-felling are almost certainly related to felling operations in the immediate vicinity of the stream, and the resulting increased water discharge and hence erosion. Results showing an increase in SS post-felling were also reported in another study in the Burrishoole catchment, with maximum SS concentrations of 97.5mg l^{-1} post-felling, compared to a maximum of 64mg l^{-1} pre-felling (Rodgers et al. 2008).

It is clear from work carried out by Kelly-Quinn et al. (1997); Allott et al. (1997); Rodgers et al. (2008) that acid episodes are experienced during high flows and flood events in poorly buffered Irish streams flowing over upland peatland. The streams flowing though RA1 and RA2 are situated on quartzite/schist and were prone to acid episodes during flood events (dropping to pH 4.6 and 3.8 respectively on numerous occasions), whereas at RA3 (on sandstone), the pH generally did not decrease below pH 5. RA1 and RA2 generally would have a lower macroinvertebrate taxa richness and lower fish numbers in comparison to RA3. The significant increase in pH at RA2 and RA3 after clearfelling is intriguing, and hard to explain (median at RA2 increased by 0.6 pH units from 4.9 to 5.5, and median at RA3 increased by 0.35 pH units from 6.42 to 6.75). It is unlikely that the increase is an artefact of the data collection as the number of samples was high, and the samples were well spaced throughout flood events and low flows. In fact, given the increasing water flow through the sites post-felling and the higher number of significant rainfall events, we actually would have expected a lowering of pH at the sites. Cummins and Farrell (2003) attributed an increase in pH after clearfelling to road usage and traffic adjacent to the site. This cannot be a valid explanation in this case as neither site was adjacent to roads with even light traffic, and there were no significant road works in the catchments. There is some alluvium mineral subsoil in the vicinity of RA2 (Altahoney), which if disturbed during the felling operation may have caused the increase in pH to some extent, although blanket peat is the dominant soil type in the area. At RA3, the soil and subsoil type are both blanket peat (Teagasc, 2006).

At RA1 and RA2, the daily maximum stream temperature post-felling was significantly higher than pre-felling daily maximum temperature, and at RA2 the minimum daily temperature was also higher post-felling. Based on these two sites (RA3 data not complete), it is likely that removal of closed forest canopies in the vicinity of streams may cause an increase in water temperature, above what might be expected owing to climatic conditions. This is particularly true for RA1 where the maximum temperature median increased by 1.5ºC after felling. However at RA2, the increase was less marked (0.5ºC), and was very similar to the increase noted in a control site in the Goulaun river (Fig. 1), and can probably be attributed to warmer air temperatures in the post-felling period as recorded at the weather station. In addition RA1 was situated at the top of the watershed with only about 100 meters of river up stream of the site, which means water flowing through the experimental site would not be affected by a large discharge above the site. In contrast, RA2 has approximately 6.5km of stream upstream of the site, which would mean that any temperature increase within the site would be considerably dampened by the amount of water flowing from the top of the water shed. This probably explains the lesser degree of heating at RA2 in comparison to RA1. Temperature increase following clearing of riparian vegetation is to be expected (Malcolm et al. 2004) and it is likely that any increases would lead to increased periphyton biomass and abundance of primary consumers (Kiffney et al. 2003). In the case of streams flowing through forestry on western peatlands in Ireland, this must be considered advantageous, as it is likely that primary production has been suppressed for the last number of decades as a result of significant shading by commercial conifer plantations.

The biological results presented from RA1 here show some changes in the macroinvertebrate community in the stream between the pre-felling and post-felling periods, particularly at the sampling point immediately below the felling zone (point 1) and the sampling point in the middle of the felling zone (point 2). The changes in the macroinvertebrate community at these points were positive in terms of ecological quality, with increases noted in taxon richness, abundance, diversity and number of sensitive (EPT) taxa, particularly at point 1. As Plecoptera are particularly sensitive to decreases in oxygen and sedimentation (McClelland and Brusven, 1980), the increase in the average abundance of Plecoptera at point 1 and 2 in the three years post-felling is particularly noteworthy as this would indicate that prolonged sedimentation was not a factor at this site. Juvenile fish populations seemed to be unaffected by the felling operations. The absence of salmon at RA1 is not surprising, given the small size of the stream and its position in the very top of the headwaters of the catchment.

Similarly to RA1, there were some changes in the macroinvertebrate communities at point 1 and 2 in RA2, notably increases in the average abundances of Plecoptera and Coleoptera, a concurrent decrease in the Ephemeropteran *B. rhodani* and decreases in the Shannon diversity. At point 3 (upstream of felling activities), the post-felling macroinvertebrate communities were similar to those sampled pre-felling. These results lead to the conclusion that while some impact was noted in the macroinvertebrate communities, it was not catastrophic and there were no local extinctions of sensitive species. However, it should be noted that the Altahoney River is generally considered to have quite poor macroinvertebrate communities (Marine Institure, 2006), probably as a result of forestry mediated acidification in the whole catchment which is known to impact on macroinvertebrate communities (Kelly-Quinn et al. 1997; Ormerod et al. 2004). RA2 would therefore already have had quite a limited macroinvertebrate community before this clearfelling even took place. It is likely that any positive impact of this clearfelling on the ecological quality of this river will be delayed, as there is a considerable area of forest (and hence acidification pressure) still standing in the catchment above the experimental site and much of the river is totally enclosed by a canopy of conifers. As with RA1, the juvenile fish at RA2 appeared to be unaffected by felling operations, with similar (if low) numbers of trout recorded in each year of the study.

The geology of the RA3 catchment means that the study stream is not as acidic as RA1 and RA2 and taxon richness is naturally higher. However, there were some significant changes in the macroinvertebrate communities in the post-felling period. At points 1 and 2, an increase in the abundance of chironomids and Diptera may well be the result of increased sediment and detritus blocking the water flow in the 2nd sampling years immediately after felling. There was also a significant decrease in the amount of EPT taxa at each point. Species such as *N. cinerea, S. pallipes, S. ignita,* and *R. dorsalis* were not recorded in 2008, which may indicate some decrease in ecological quality. It must be noted however that none of these species were ever recorded in very high numbers pre-felling and longer term monitoring of this site may be needed to assess whether these species are locally extinct at this site. Fish numbers were lowest in 2006, probably as a result of felling operations, so it was reassuring therefore to find 0+ salmon in 2007, and also a general improvement in fish numbers back to those recorded pre-felling.

Many studies have shown significant effects of sediment deposition or forestry felling on benthic invertebrates in streams (Johnson et al. 2000; Fossati et al. 2001) while others report minimal disturbance (Williams et al. 2002; Kreutzweiser et al. 2005). Owing to the mobility and short life cycles of most macroinvertebrates, the spatial and temporal scale of the study is of crucial importance in determining whether certain forestry practices have long term detrimental impacts on stream biota. Given enough time, short term loss of species may be mitigated by recolonisation from other parts of the water body. This is presuming that the impacts of the forestry activity were such as to affect only a small portion of the habitat. This is probably the case in this study, as the area of riparian zone felled was small in comparison to both the catchments as a whole and the forest area. Generally, it has now been accepted that phased felling (rather than clear felling) of coniferous forestry offers the best hope of minimising long term detrimental impacts on aquatic biota (Neal et al. 2004). This study indicates that felling an area in the order of 2-6 hectares immediately adjacent to streams is unlikely to have severe lasting impacts on the aquatic biota.

There is little literature describing the long term efficacy or the practicality of the forestry management practice described in this paper, the retrospective clearing and establishing of a RBZ in commercial coniferous forests. However, it is a forestry practice that may become increasingly necessary as first rotation forest crops planted in the UK and Ireland reach harvestable age (Broadmeadow and Nisbet, 2004; Robinson and Dupeyrat, 2004). The results from this study indicate that if this practice of clearing the RBZ of conifers several years prior to main crop clearfelling is undertaken, the short term impacts on the aquatic biota are likely to be minimal. This is based on the assumption that a relatively small area of RBZ is cleared at a time, during good weather conditions, with adequate brash mats and with minimal stream crossing. Further research will be required at these experimental sites in 5-10 years to allow assessment of the mitigation effects of these enhanced riparian zones once the main conifer crop behind them is clearfelled. However, it must be noted that given current forestry guidelines, these riparian zones will continue to provide protection to the neighbouring water bodies in the second and successive rotations of commercial forestry, and their early establishment must be considered as a timely intervention.

Acknowledgement

This study was funded by COFORD the National Council for Forest Research and Development in Ireland, and coordinated by Coillte Teoranta and Marine Institute. Thanks to David Sweeney, Joseph Cooney, Michael Murphy and the Marine Institute bursary students in 2005, 2006 and 2007 for assistance in the field and laboratory. Thanks to the Coillte staff involved in all site operations, particularly Kieran McLoughlin. We are grateful to Alistair Pfeiffer for his suppose and advice throughout the project and to two anonymous reviewers of this paper.

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Figure Captions:

Fig 1: Map of locations of the three study and two control sites in the west of Ireland.

Fig 2: Description of the experimental design at each RA study site.

Fig 3: Maximum daily water discharge (l/s) (log_{10} scale), suspended sediment discharge (mg/l) (log_{10} scale) and pH at three experimental sites, RA1, RA2 and RA3, measured before and after felling of coniferous trees in the riparian zone. Boxes represent the 25% and 75% quartiles, whiskers represent the maximum and minimum values and dots indicate the median value.

Fig 4: Daily precipitation recorded at the Marine Institute weather station for a) the pre-felling b) post-felling and c) monthly periods. Gray line indicates threshold for storm occurrence.

Fig 5: pH measurements taken pre- and post-felling at each RA site, the grey line represents time of clearfelling.

Fig 6: Maximum and minimum daily water temperatures at RA1, RA2 and RA3, measured before and after felling of coniferous trees in the riparian zone. Boxes represent the 25% and 75% quartiles, whiskers represent the maximum and minimum values and dots indicate the median value.

Fig 7: A comparison of four years (one year pre-felling 2005 and three years post-felling 2006, 2007 and 2008) of taxon richness, species abundance, Shannon Diversity index and no. of EPT taxon richness at RA1, RA2 and RA3. Values represent the average of six replicate samples and error bars indicate standard deviation.

Fig 8: A comparison of abundance (per surber) of dominant orders over a four year period at RA1, RA2 and RA3. Values represent an average of six replicate from each sampling point over each sampling year.

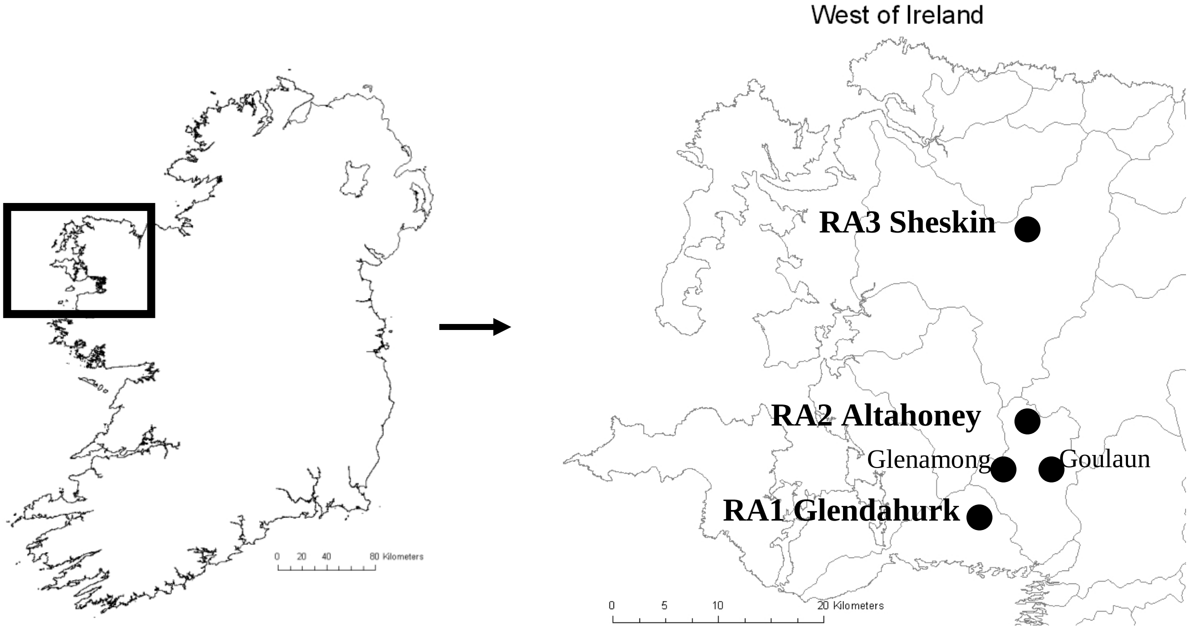

0 20 40 80 Kilometers
West of Ireland
RA3 Sheskin
RA2 Altahoney
Glenamong
Goulaun
RA1 Glendahurk
0 5 10 20 Kilometers

Figure 1:

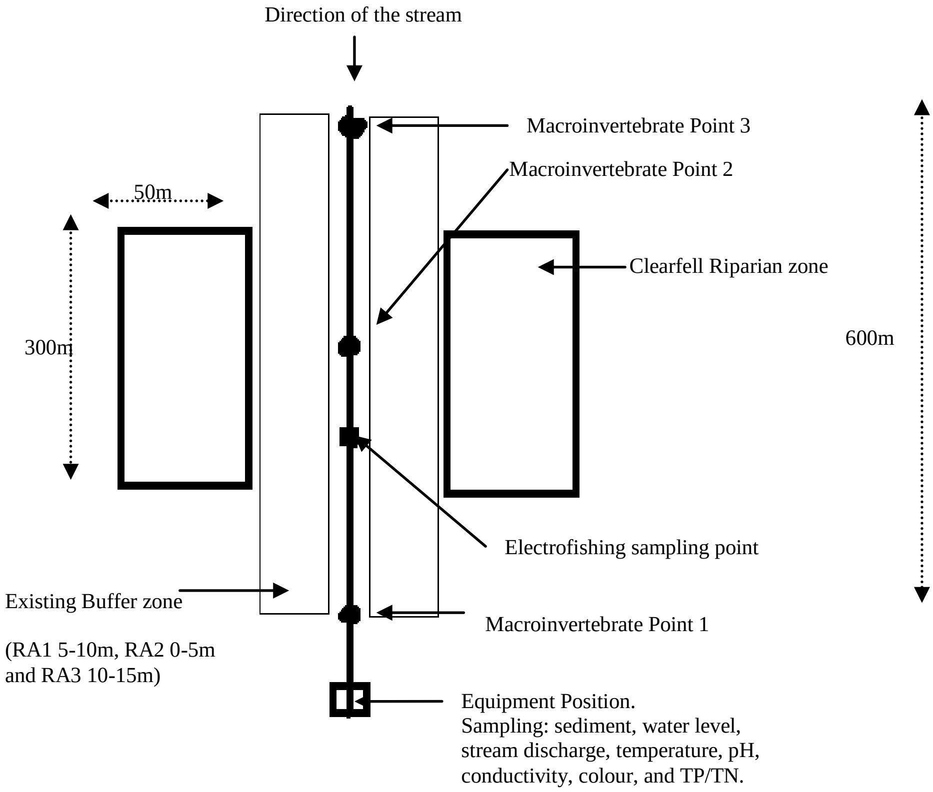

Direction of the stream
Macroinvertebrate Point 3
Macroinvertebrate Point 2
50m
Clearfell Riparian zone
300m
600m
Electrofishing sampling point
Existing Buffer zone
(RA1 5-10m, RA2 0-5m
and RA3 10-15m)
Macroinvertebrate Point 1
Equipment Position.
Sampling: sediment, water level,
stream discharge, temperature, pH,
conductivity, colour, and TP/TN.

Figure 2:

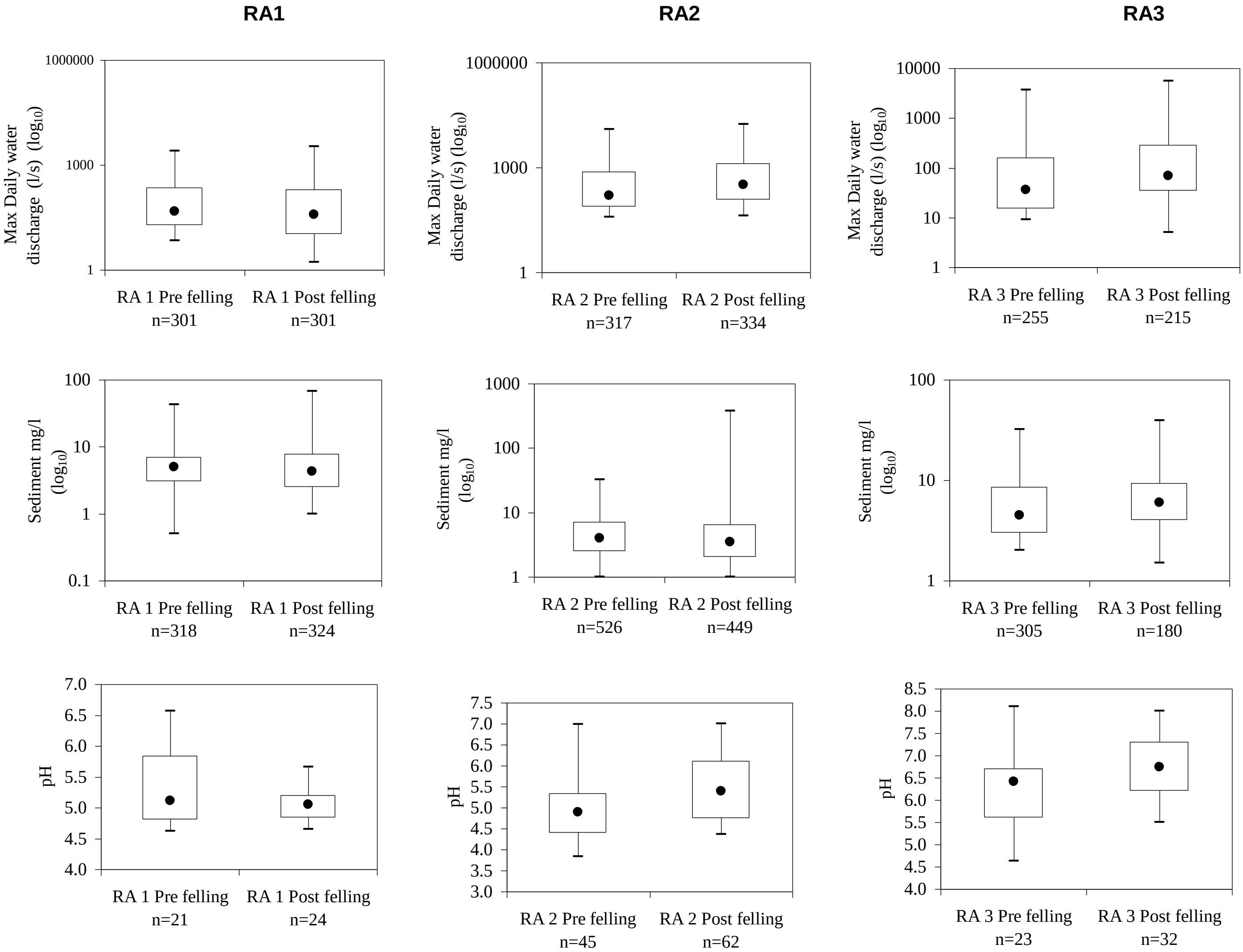

RA1
RA2
RA3
Max Daily water discharge (l/s) (log10)
1000000
1000
1
RA 1 Pre felling n=301
RA 1 Post felling n=301
RA 2 Pre felling n=317
RA 2 Post felling n=334
10000
100
10
RA 3 Pre felling n=255
RA 3 Post felling n=215
Sediment mg/l (log10)
0.1
RA 1 Pre felling n=318
RA 1 Post felling n=324
RA 2 Pre felling n=526
RA 2 Post felling n=449
RA 3 Pre felling n=305
RA 3 Post felling n=180
pH
4.0
4.5
5.0
5.5
6.0
6.5
7.0
7.5
8.0
8.5
3.0
3.5
RA 1 Pre felling n=21
RA 1 Post felling n=24
RA 2 Pre felling n=45
RA 2 Post felling n=62
RA 3 Pre felling n=23
RA 3 Post felling n=32

Figure 3:

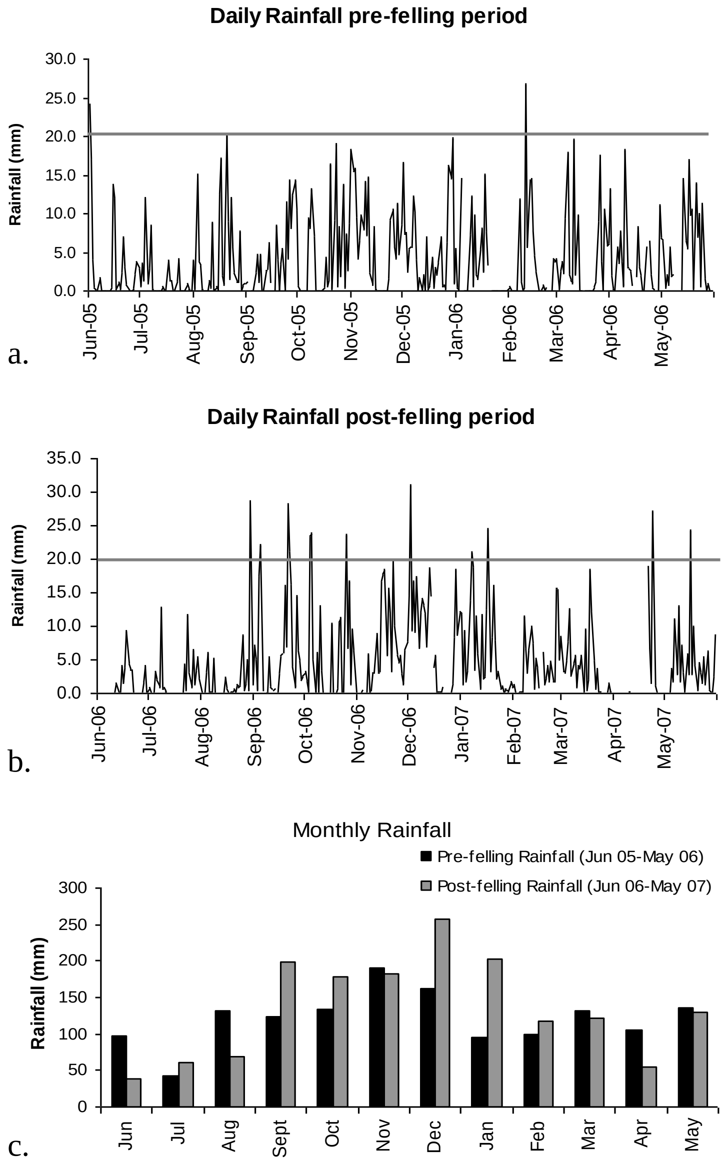
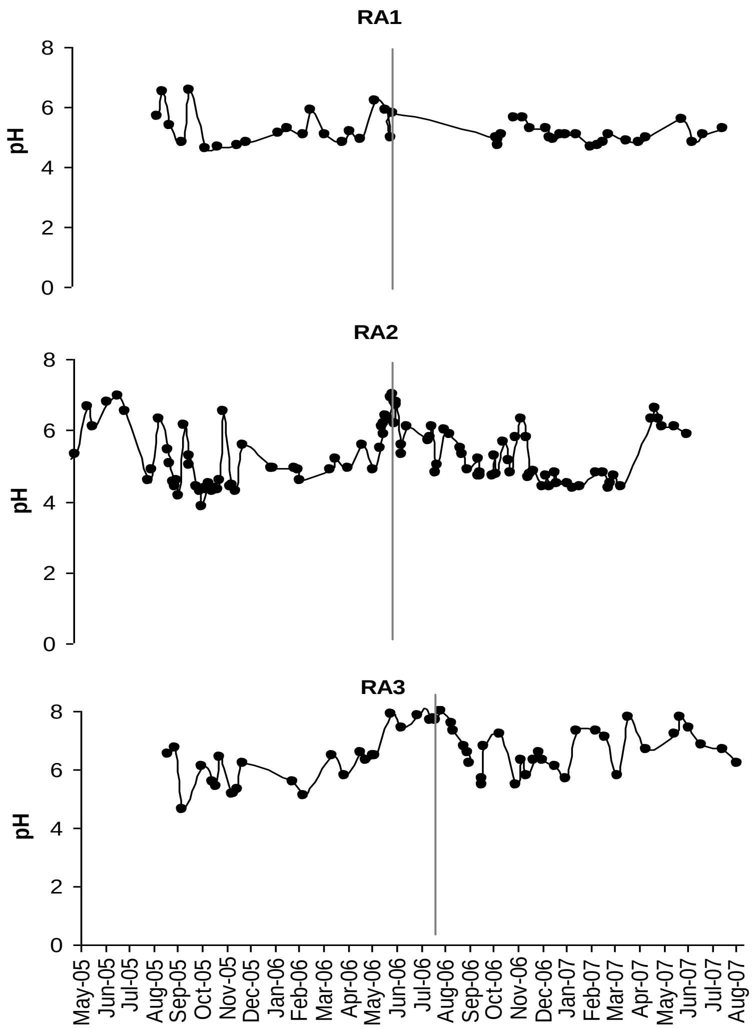

Daily Rainfall pre-felling period
Rainfall (mm)
30.0
25.0
20.0
15.0
10.0
5.0
0.0
Jun-05
Jul-05
Aug-05
Sep-05
Oct-05
Nov-05
Dec-05
Jan-06
Feb-06
Mar-06
Apr-06
May-06
a.
Daily Rainfall post-felling period
Rainfall (mm)
35.0
30.0
25.0
20.0
15.0
10.0
5.0
0.0
Jun-06
Jul-06
Aug-06
Sep-06
Oct-06
Nov-06
Dec-06
Jan-07
Feb-07
Mar-07
Apr-07
May-07
b.
Monthly Rainfall
Pre-felling Rainfall (Jun 05-May 06)
Post-felling Rainfall (Jun 06-May 07)
Rainfall (mm)
300
250
200
150
100
50
0
Jun
Jul
Aug
Sept
Oct
Nov
Dec
Jan
Feb
Mar
Apr
May
c.

Figure 4:



RA1
8
6
4
2
0
pH
RA2
8
6
4
2
0
pH
RA3
8
6
4
2
0
pH
May-05
Jun-05
Jul-05
Aug-05
Sep-05
Oct-05
Nov-05
Dec-05
Jan-06
Feb-06
Mar-06
Apr-06
May-06
Jun-06
Jul-06
Aug-06
Sep-06
Oct-06
Nov-06
Dec-06
Jan-07
Feb-07
Mar-07
Apr-07
May-07
Jun-07
Jul-07
Aug-07

Figure 5:

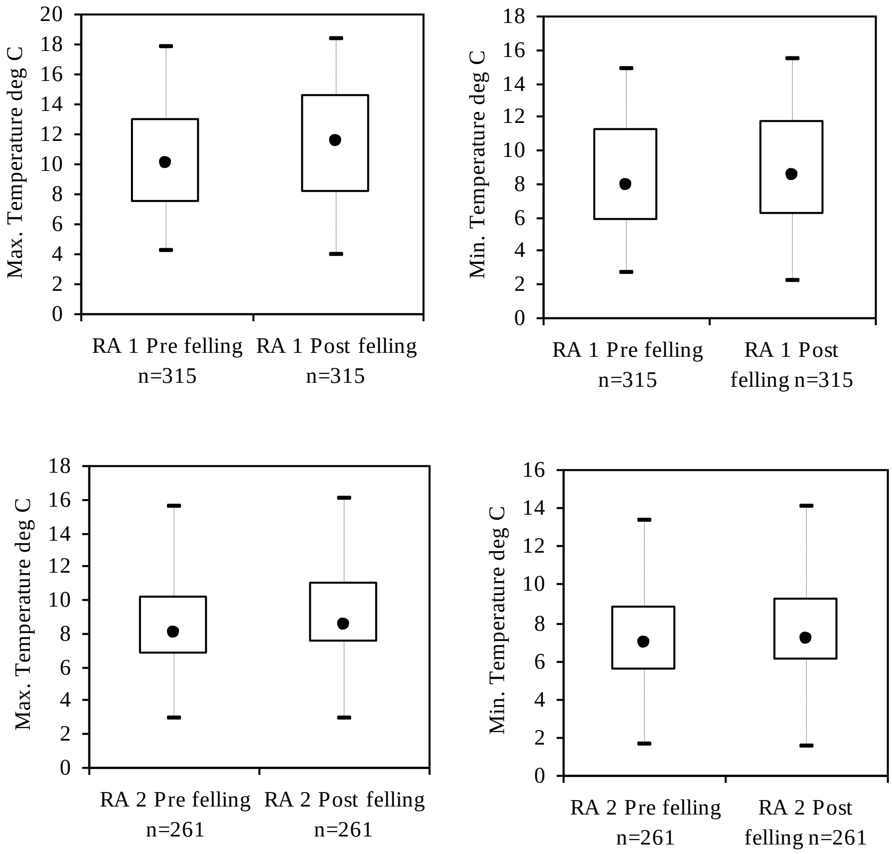

Max. Temperature deg C
20
18
16
14
12
10
8
6
4
2
0
RA 1 Pre felling n=315
RA 1 Post felling n=315
Min. Temperature deg C
18
16
14
12
10
8
6
4
2
0
RA 1 Pre felling n=315
RA 1 Post felling n=315
Max. Temperature deg C
18
16
14
12
10
8
6
4
2
0
RA 2 Pre felling n=261
RA 2 Post felling n=261
Min. Temperature deg C
16
14
12
10
8
6
4
2
0
RA 2 Pre felling n=261
RA 2 Post felling n=261

Figure 6:

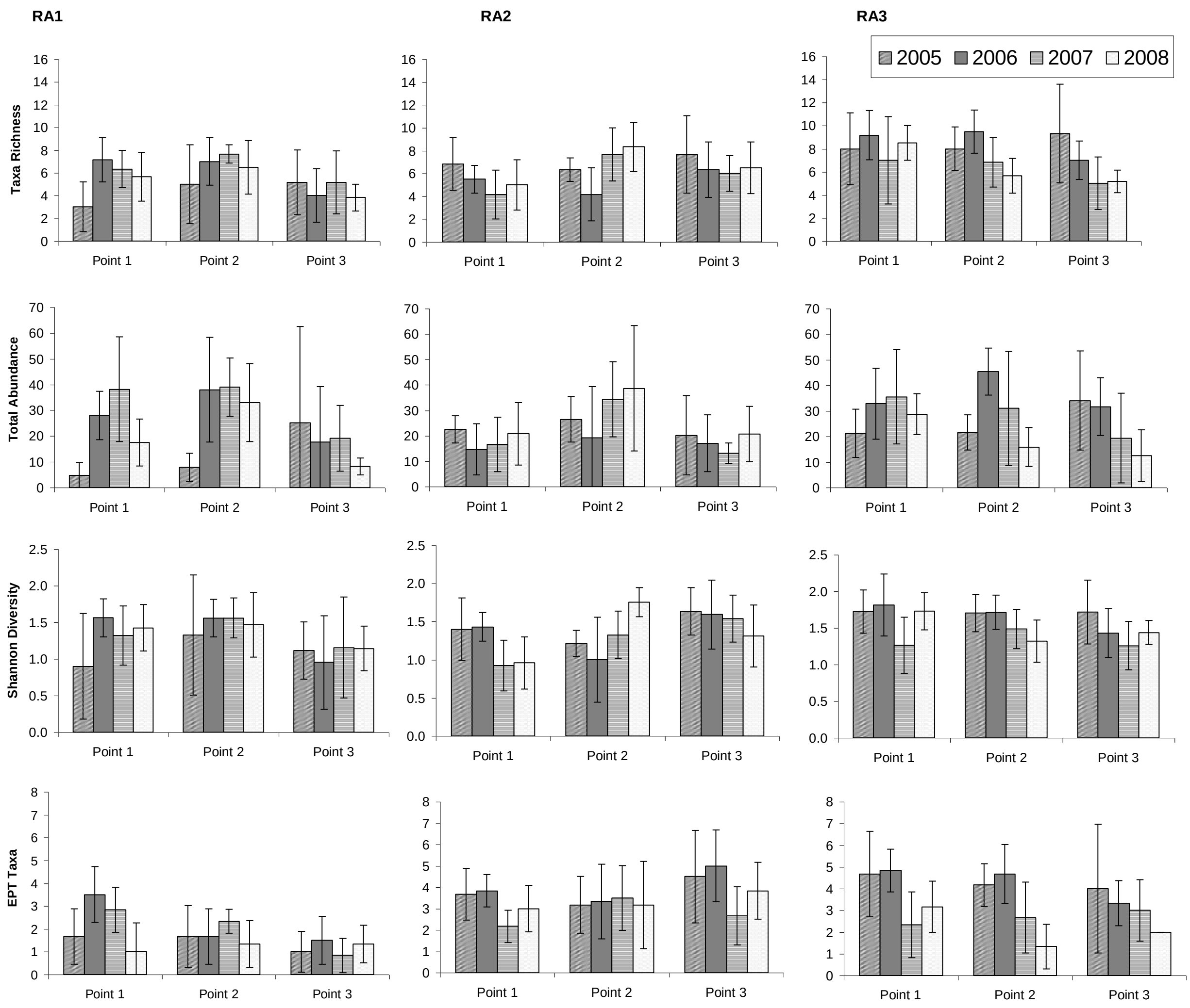

RA1
RA2
RA3
2005
2006
2007
2008
Taxa Richness
Total Abundance
Shannon Diversity
EPT Taxa
Point 1
Point 2
Point 3

Figure 7:

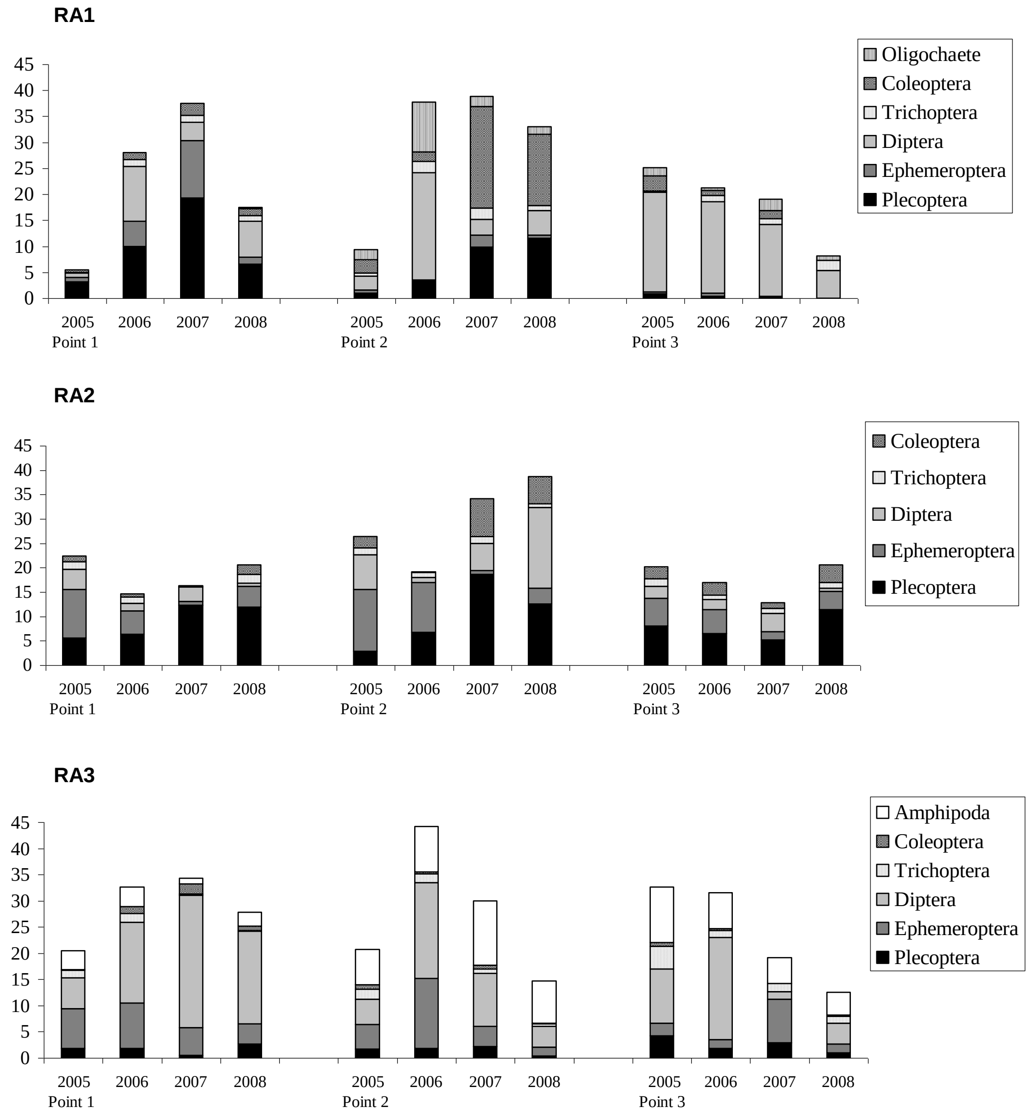

RA1
45
40
35
30
25
20
15
10
5
0
2005
Point 1
2006
2007
2008
2005
Point 2
2006
2007
2008
2005
Point 3
2006
2007
2008
Oligochaete
Coleoptera
Trichoptera
Diptera
Ephemeroptera
Plecoptera
RA2
Coleoptera
Trichoptera
Diptera
Ephemeroptera
Plecoptera
RA3
Amphipoda
Coleoptera
Trichoptera
Diptera
Ephemeroptera
Plecoptera

Figure 8:

Table 1: Physical characteristics of three experimental areas in the west of Ireland where riparian area were cleared of commercial coniferous forestry in order to establish riparian buffer zones.

	RA1	RA2	RA3
sCatchment	Owengarve	Burrishoole	Bangor
Irish National Grid Reference	F 914 012	F 957 093	F 958 272
Total Catchment area (ha)	80.7	416.2	105.3
Total Catchment area forested (ha)	15.2	175.2	105.3
Catchment above study area (ha)	68.7	358.8	87.9
Length of main channel upstream (km)	0.7	3.8	2.1
Riparian zone (pre-felling) (ha)	0.64	0.14	1.06
Riparian zone (post-felling) (ha)	6.3	2.49	1.9
Geology	Quartzite/Schist	Quartzite/Schist/Gneiss	Sandstone
Stream Substratum	Cobble and gravel	Cobble, boulder and gravel	Cobble and gravel
Instream habitat	Pool, high grade riffle	Pool, high grade riffle	Pool, low grade riffle
Land use	Forestry / Commonage	Forestry / Commonage	Forestry
Bank cover	Conifer, silver birch and grass	Conifer, grass, bare soil	Conifer, grass, heather, larch